



# Determination of Bioavailability, Potential Ecological and Human Health Risks, and Biomonitoring of Potential Toxic Elements in Gold Mine Tailings from Four Areas of Antioquia, Colombia

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**Abstract** Gold mining has helped many countries and regions grow and prosper. Yet, because of the significant environmental effects produced, waste management (mine tailings) has emerged as a crucial issue for the sector. In this study, the concentrations of several potential toxic elements (PTEs) (Fe, Zn, Cu, As, Hg, Pb, Cr, Cd, and Ni) in mine tailings from four gold mining operations (Yalí, Puerto Berrio, Buriticá 1, and Buriticá 2,) were quantified. A biomonitoring system has also been suggested, along

with two environmental risk indexes for ecosystems and human health. To do this, composite samples of tailings were collected from the four regions. A PTEs analysis using an ICP-OES as well as physicochemical characterization (pH, electrical conductivity, cation exchange capacity, organic matter) have been performed. Indicators such as antioxidant activity and H<sub>2</sub>O<sub>2</sub> content in *Phaseolus vulgaris* plants have been used to estimate the bioavailability of PTEs. In contrast to the pseudo-total concentration, the data showed that the bioavailable concentration of PTEs correlated with the risk. The ecological system and public health are more at risk from the mine tailings in Buriticá and Puerto Berrio. It has been demonstrated that *P. vulgaris* is a quick and effective technique to assess the danger to the environment and public health connected with mining regions, according to the suggested biomonitoring system. The findings of this study can assist the relevant authorities in taking quick action to improve environmental protection and risk management of hazardous waste in gold mining areas.

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## 1 Introduction

Gold mining has contributed significantly to the socio-economic development of nations around the world (Betancur-Corredor et al., 2018; Buch et al., 2021; Jiang et al., 2021; Sauerwein, 2020). Gold processing activities historically been associated with the production of large volumes of hazardous waste, like mine tailings, which are a fine-grained wastes produced during the extraction of gold ore (Akoto & Anning, 2021; Orimoloye & Ololade, 2020; Zhu et al., 2022). It is reported that for every ton of metal extracted from mine, approximately 2–12 ton mine tailings yield (Jiang et al., 2021) and it is estimated that worldwide the production of tailings exceeds 10 billion tons (Adiansyah et al., 2015; Kodirov et al., 2018; Kwon et al., 2015). The massive production of this waste are considered one of the main causes of environmental health degradation produced by mining, mainly due to the leaching of potentially toxic elements (PTEs) like Cu, Pb, Hg, Cd, Ni, Zn, Cr, and As (Bueno et al., 2009; Wang et al., 2017; Barcelos et al., 2020; Orimoloye & Ololade, 2020; Suppes & Heuss-Abbichler, 2021). Thus, PTEs accumulation rates are faster than under natural conditions, affecting their integrity for decades (Ngole-Jeme & Fantke, 2017; Teixeira et al., 2019).

Gold mining is one of the fastest growing industries in Colombia becoming the 5th largest gold producer in Latin America (Bustamante et al., 2016). In Colombia, studies have been conducted associating gold mining, specifically Hg contamination, with ecosystem and human health risks. In these studies, artisanal gold mining has been found to cause significant environmental contamination. In addition, these same studies have shown that Hg has been associated with human health problems, as well as that Hg has bioaccumulated in the population surrounding the mining activity (Gutiérrez-Mosquera et al., 2018b; Marrugo-Negrete et al., 2017; Olivero-Verbel et al., 2015; Pinedo-Hernández et al., 2015). To our knowledge few studies have focused on determining the concentration of other PTEs such as Fe, Zn, Cu, As, Pb, Cr, Cd y Ni in tailings from gold mining activity in this country, even though these elements represent high risks to human health and the environment (Adamo & Zampella, 2008; Alloway, 2013). For example, in Chocho, Colombia, a significant ecological risk associated with Pb enrichment of sediments because of gold

mining activity has been reported (Gutiérrez-Mosquera et al., 2018a). As well as in Córdoba, Colombia, where gold mining has affected agricultural soils due to the concentration of elements such as Ni, Cd, As, and Pb (Becerra-Agudelo et al., 2022; Marrugo-Negrete et al., 2017). The department of Antioquia (northwest Colombia) accounts for 40% of Colombia's gold mining production (Betancur-Corredor et al., 2018). However, no reports were found regarding the determination of the concentration of PTEs in mining tailings in the department of Antioquia. In addition, there is an absence of studies estimating the potential risk to human health and ecosystems in this region associated with the concentration of PTEs in mining tailings.

Besides, current reports for Colombia estimate the pseudo-total concentration of PTEs and not the bioavailable concentration. It has been established that pseudo-total concentrations are not a good indicator of potential environmental risk, since they do not reflect the fraction of the contaminant that can interact with biological systems (Adamo & Zampella, 2008; Becerra-Agudelo et al., 2022; Jorge Mendoza et al., 2017). The bioavailability of PTEs can be determined by indirect methods like chemical extractions (Becerra-Agudelo et al., 2022; Rodríguez-Seijo et al., 2020; Salazar et al., 2013). However, these methods do not reflect the interaction of the metabolism of living organisms with PTEs, so it is recommended to complement bioavailability determinations of PTEs with direct measurements, e.g., bioaccumulation in plants (bioindicators) (Acosta-Luque et al., 2023; López et al., 2023; Rodríguez-Seijo et al., 2020; Serrano et al., 2023). The use of a plant as a bioindicator of metal bioaccumulation may become a strategy to determine the potential risk of bioavailability of PTEs in mining tailings. In this context, it would be necessary to develop a simple biomonitoring system with a fast-growing and low-maintenance plant, which would help to have threshold values for bioaccumulation in tissues (e.g., leaves and/or roots) that can be correlated with the potential ecological and human health risks. Previous studies suggest the use of methodologies that allow the estimation of harmful effects on human health and ecosystems based on bioavailable concentrations of PTEs can be important tools for environmental zoning of the territory. Examples of these methodologies are the indices of potential risk to human health and risk to the ecosystem (Becerra-Agudelo et al., 2022; Castañeda-Restrepo et al., 2022). For instance the ecotoxicological risk of accumulation of PTEs in the environment has

been studied using indicators such as human health risk assessment (Becerra-Agudelo et al., 2022). According to our knowledge, the application of these methodologies is limited in mining areas in Colombia, although their use could be an important tool for the environmental management of contaminated lands. Because, in Colombia, current regulations (Decree 1076 of 2015 and Resolution 2206 of 2016) require tests for tailings acidification potential (acid mine drainage) and leaching tests to determine the capability of the tailings to release PTEs into the environment. Despite the potential risks associated with occupational or environmental exposure to PTEs, there are no tests being requested to quantify their impact on the population's health. However, no tests are requested to quantify the risk associated with the population's health due to occupational or environmental exposure to PTEs. Currently, the regulations in Colombia for the mining sector are being updated, and in the short term, these types of tests will also be required. Therefore, in the medium and long terms, these environmental risk indicators will become necessary.

Therefore, the objectives of this study are i) to determine the bioavailable concentrations of PTEs (Fe, Zn, Cu, As, Hg, Pb, Cr, Cd, and Ni) in the gold mining tailings, ii), to calculate the indexes of potential risk to the ecosystem and human health, iii) to calculate the pollution indices, potential ecological risk index, and potential human health risk assessment, and iv) to propose a biomonitoring system for

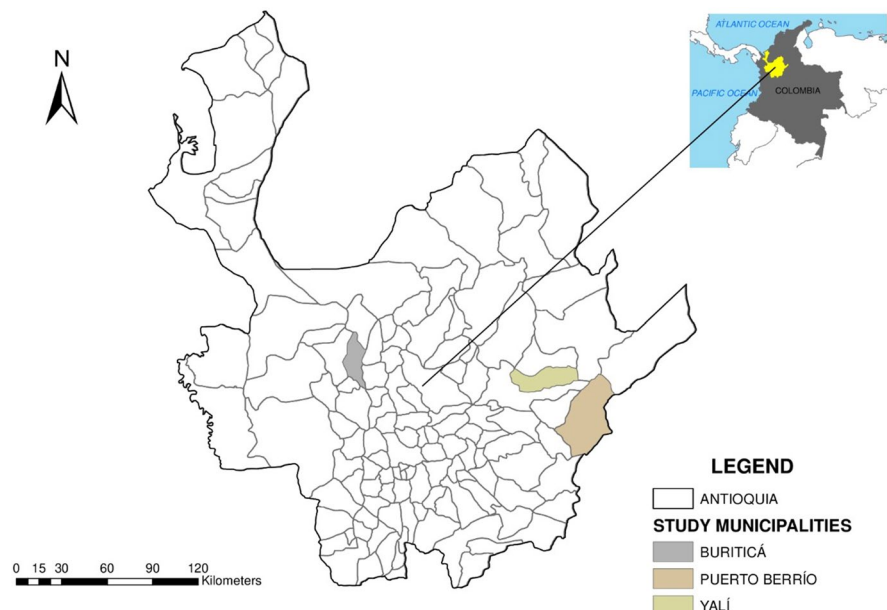
PTEs (Fe, Zn, Cu, As, Hg, Pb, Cr, Cd, and Ni) in mine tailings.

## 2 Materials and Methods

### 2.1 Study Areas and Sampling of Mining Tailings

Three active gold mines were selected for this study, located in the municipalities of Buriticá, Puerto Berrió, and Yalí (Antioquia, Colombia) (Fig. 1). All municipalities being investigated have their metropolitan centers located within a 5 km radius of the mines being examined. Samples were taken in the direct area of influence of the mines, where gold is often processed and mine tailings are accumulated in structures known as dams. Agro-industrial land use, which combines agriculture and the gold mining sector, is prevalent in all municipalities, in addition to residential use. These municipalities represent areas with an accelerated growth of gold mining activity in Colombia. Buriticá: municipality located on the eastern margin of the western mountain range. The gold mineralization is found within the so-called Buriticá Tonalite associated with the Sabanalarga Batholith of reported Mesozoic age. This unit forms N-S elongated bodies across the Cauca River. The structural control is N-S associated with the Cauca and Tonusco faults (Rodríguez-Seijo et al., 2020). Puerto Berrió: located in the

**Fig. 1** Location of study areas. Buriticá (B1 and B2), Yalí (YA), and Puerto Berrió (PB)



lower part of the eastern slope of the central mountain range of Colombia. Tailings samples were taken from the sector known as Minas el Vapor. Locally, three lithological units can be distinguished: Neisfeldspathic, the Segovia batholith and the East Segovia sedimentite of Mesozoic age (Restrepo-Sánchez et al., 2015). Structurally, the gold mining area is affected by the Otú, Palestina, Cimitarra and El Bagre faults, locally in the Minas del Vapor sector the faults have a predominant NW trend; the mineralization of veiniform orogenic gold type. Yalí: located on the eastern slope of the central mountain range. The gold mineralization is associated with the geological unit known as Batolito Antioqueño with variable composition of Tonalite, Granodiorite and Diorite, all Mesozoic age. This geological body has structural controls to the east by the Palestina fault system and to the west by the Romeral fault system. Gold mineralization is in the form of veinlets.

Mining tailings samples were collected from the surface of tailing piles at heights ranging from 1 to 3 m above ground and at the base of tailing pile. The samples were obtained from mining projects located in the department of Antioquia, Colombia. One project in the municipality of Puerto Berrío (PB), another mining project located in the municipality of Yalí (YA), and two mining projects located in the municipality of Buriticá (BU1 and BU2). At each site (PB, YA, BU1, and BU2), three composite samples were taken. Each composite sample consisted of twelve subsamples, which were collected with plastic shovels, deposited in clean plastic containers (1 L), and homogenized using the same plastic shovels. On average, each composite sample weighed between 500 and 800 g.

## 2.2 Physicochemical Characteristics of Mining Tailings and Pseudo-Total Concentration and Bioavailability of PTEs

All samples were air-dried, homogenized, and sieved (2-mm mesh). pH and electrical conductivity (EC) were determined using a 1:2 ratio (mine tailings: deionized water). Cation exchange capacity (CEC) was measured using ammonium acetate extraction (pH 7.0). Organic matter (OM) content was quantified using the loss on ignition method. For the determination of pseudo-total concentration of PTEs 1.0 g of

mining tailings sample was subjected to acid digestion ( $\text{HNO}_3\text{:HCl}$  1:3) in an Ethos One Milestone helping Chemist microwave. In brief, 10 ml of  $\text{HNO}_3$  and 3 ml of HCl were added. The temperature of the samples was raised to 175 °C in approximately 5.5 min and remained at 175 °C for an additional 4.5 min, following US EPA method 3051A.

After digestion, the samples were diluted with ultrapure water (DW) and passed through a 0.45 µm membrane filter, subsequently the filtrates were used for the determination of metals by inductively coupled optical emission spectrometry using an Agilent Technologies 5100 ICP-OES equipment (ICP-OES). To determine the bioavailability of PTEs, the mining tailings samples were subjected to the first step of BCR extraction method, which was used to establish the bioavailable fraction of PTEs (Salazar et al., 2013). Once the extracts were obtained, filtered through a 0.45 µm membrane filter and the filtrates were used for the determination of metals by ICP-OES.

## 2.3 Geochemical Index

### 2.3.1 Risk Assessment Code

The Risk Assessment Code (RAC) was used as an indicator of risk due to the bioavailability of PTEs. The RAC index was calculated from the results of the concentration of PTEs of the BCR extraction (PTEs water-soluble + PTEs exchangeable + PTEs carbonates) (Jiang et al., 2021). This include the exchangeable geochemical fractions and associated with carbonates, and correspond to those of greater bioavailability (Favas et al., 2011).

$$\text{RAC} = \left( \frac{C_{av}}{C} \right) * 100 \quad (1)$$

where  $C_{av}$  is the available PTEs concentration from first step of BCR extraction (mg/kg) and  $C$  is the pseudo-total PTEs concentration (mg/kg). According to the values of the RAC, the level of risk is classified in:  $\text{RAC} \leq 1\%$ , it does not represent any risk;  $1\% < \text{RAC} \leq 10\%$ , low risk;  $10\% < \text{RAC} \leq 30\%$ , moderate risk;  $30\% < \text{RAC} \leq 50\%$ , high risk and,  $50\% < \text{RAC}$ , very high risk.

### 2.3.2 Geochemical Abundance Index

The geochemical abundance index (GAI) corresponds to a relative indicator that compares the pseudo-total concentration of the metal, with the concentration of the same metal in a reference material (Table S1).

$$GAI = \log_2 \left[ \frac{C}{1.5 * S} \right] \quad (2)$$

where C is the pseudo-total PTEs concentration (mg/kg), and S is the reference concentration of the metal (mg/kg).  $GAI \leq 0$  indicates concentrations of metals below the average content in soil,  $GAI = 1$ , represents 3 to 6 times above average content in soil,  $GAI = 2$ , means 6 to 12 times more,  $GAI = 3$ , means 12 to 24 times more,  $GAI = 4$ , it means 24 to 48 times more,  $GAI = 5$ , means 48 to 96 times more and  $GAI \geq 6$ , means more than 96 times the average concentration. Generally, from a GAI value of 3, it is considered a significant increase which requires a more detailed examination.

## 2.4 Potential Risk

### 2.4.1 Potential Ecological Risk Index

To assess the biological relevance of the PTEs the potential ecological risk index was calculated as the sum of the single hazard factors (Hakanson, 1980):

$$RI = \sum E_r^i = \sum T_r^i(C/S) \quad (3)$$

where RI Represents the total risk index,  $E_r^i$  is the potential ecological risk index of PTEs contamination on an individual form,  $T_r^i$  is the toxic response factor for contamination by each PTE (1.0 for Zn, 2.0 for Cr, 5.0 for Cu and Pb, 10 for As, and 30 for Cd) (Islam et al., 2017), C is the pseudo-total PTEs concentration (mg/kg), and S is the reference concentration of PTEs (mg/kg).

According to the values of  $E_r^i$  the ecological risk is classified in: Low risk,  $E_r^i < 40$ ; Moderate risk,  $40 \leq E_r^i < 80$ ; Considerable risk,  $80 \leq E_r^i < 160$ ; High risk,  $160 \leq E_r^i < 320$  and very high risk,  $E_r^i \geq 320$ . The values of the RI risk index are interpreted as: low risk,  $RI < 50$ ; Moderate risk,  $50 \leq RI < 200$ ; Considerable risk,  $200 \leq RI < 300$  and high risk,  $RI \geq 300$  (Hakanson, 1980).

### 2.4.2 Potential Human Health Risk Assessment

**Exposure Assessment** Human exposure to PTEs was calculated by estimating the chronic daily intake (CDI) (mg/kg/day). Three main routes of exposure were used: ingestion ( $CDI_{Ingest}$ ), inhalation ( $CDI_{Inhale}$ ), and dermal absorption ( $CDI_{Dermal}$ ) (Barcelos et al., 2020). The available concentration (first step of BCR extraction) of PTEs was used to calculate the risk to human health, since the use of pseudo-total concentrations generates an overestimation of the risk (Becerra-Agudelo et al., 2022; Jorge Mendoza et al., 2017):

$$CDI_{Ingest} = [(C_{av} * IngR * EF * ED)/(BW * AT)] * CF \quad (4)$$

$$CDI_{Inhale} = (C_{av} * InhR * EF * ED)/(PEF * BW * AT) \quad (5)$$

$$CDI_{Dermal} = [(C_{av} * SA * AF_{soil} * EF * ED)/(BW * AT)] * CF \quad (6)$$

where  $C_{av}$  is the available concentration of PTEs (mg/kg),  $IngR$  indicates the ingestion rate of the soil (mg/day),  $EF$  is the exposure frequency (days/year),  $ED$  is the exposure duration (years),  $BW$  is the average body weight (kg),  $AT$  is the averaging time (days),  $CF$  is the conversion factor (kg/mg),  $InhR$  is the inhalation rate (m<sup>3</sup>/cm),  $PEF$  is the particle emission factor (m<sup>3</sup>/kg),  $SA$  is the surface area of the skin that is in contact with the soil (cm<sup>2</sup>),  $AF_{soil}$  is the skin adherence factor for soil (mg/cm<sup>2</sup>),  $ABS$  is the dermal absorption factor. The exposure factors used for the estimation of CDI according to the values recommended by the USEPA, 2012 (Table S2).

**Non-Carcinogenic Risk Assessment** The non-carcinogenic risk for an individual PTEs was characterized by the target hazard quotient (HQ). Additionally, a hazard index (HI) was calculated by the sum of the HQ value of each PTEs for each CDI ( $CDI_{Ingest}$ ,  $CDI_{Inhale}$ , and  $CDI_{Dermal}$ ):

$$HQ = CDI/Rfd \quad (7)$$

$$HI = \sum HQ = HQ_{Ingest} + HQ_{Inhale} + HQ_{Dermal} \quad (8)$$

where: Rfd is the reference dose (0.0371, 0.7, 0.3, 0.02, 0.0003, 0.001, 0.0035, and 0.003 mg/kg-day for Cu, Fe, Zn, Ni, As, Cd, Pb, and Cr respectively)

(USEPA, 2012). Based on the HI values, no significant risk of non-carcinogenic effects is anticipated if the values are less than one ( $HI < 1$ ). However, in the case of HI value exceeds one ( $HI > 1$ ), there is a probability that non-carcinogenic risk effects may occur which tends to increase with the increment of HI value.

**Carcinogenic Risk Assessment** The carcinogenic risk is the probability of developing a type of cancer in the entire lifetime of a human as a result of the exposure to a potential carcinogen PTEs (Xiao et al., 2017). The risk to cancer (CR) for each CDI ( $CDI_{Ingest}$ ,  $CDI_{Inhale}$ , and  $CDI_{Dermal}$ ) was estimated using the cancer slope factor for each potential carcinogen PTEs. The total life-time cancer risk (LCR) was calculated as the sum of the CR (Ingest, Inhale, and Dermal). The acceptable threshold value of the LCR used was  $1.0E^{-04}$  (USEPA, 2012).

$$CR = CDI * CSF \quad (9)$$

$$LCR = CR_{Ingest} + CR_{Inhale} + CR_{Dermal} \quad (10)$$

where CSF is the cancer slope factor (6.3, 0.5, 0.0085, and 1.5 mg/kg-day for Cd, Cr, Pb, and As, respectively) (USEPA, 2012).

## 2.5 Biomonitoring

An in vivo greenhouse bioassay was conducted for biomonitoring. For this purpose, was selected *Phaseolus vulgaris* as it is a widely used biological model (Meers et al., 2007; Restrepo-Sánchez et al., 2015). Certified seeds of *P. vulgaris* (Agrosemillas®) were acquired. After, five seeds (one per pot) were transferred to plastic pots containing gold mining tailings samples from BU1, BU2, YA, and PB. Then, 4.0 g of a NPK (18–18–18) commercial fertilizer (Monómeros®) was added to all pots. Deionized water (DW) was used to maintaining the water holding capacity (WHC) to 80%. After one week, seed germination was determined by counting the number of germinated seeds. Seed germination percentage (SG) was calculated using the following equation.

$$SG = (GS/TS) * 100 \quad (11)$$

where GS is the number of seeds germinated and TS is the total number of seeds sown for each mine tailings.

After 3 months, three plants were taken, one plant from each pot, each plant corresponded to a sample to analyze the response variables (biochemical markers, biometric parameters and PTEs concentration), a total of 3 samples were analyzed for each mining tailings in each municipality. Leaf subsamples from each plant were taken and immediately frozen in liquid nitrogen and stored at  $-80\text{ }^{\circ}\text{C}$  until used for biochemical marker analysis. The antioxidant activity (Aa) was determined by 2,7'-dichlorodihydrofluorescein diacetate (DCFH) probe, which reacts indiscriminately with reactive oxygen species (ROS) and reactive nitrogen species (RNS) generated by the compound azo, 2,2'-diazobis (2-amidinopropane dihydrochloride) (AAPH) in an aqueous medium and forms the fluorescent compound 2,7-dichlorofluorescein (DCF). The antioxidants in the samples capture ROS and RNS and reduce the fluorescence emitted by DCF. Briefly, 50  $\mu\text{l}$  of a 0.3 M AAPH solution, 50  $\mu\text{l}$  of a 2.4 mM DCFH ethanolic solution, 2850  $\mu\text{l}$  of 75 mM phosphate buffer (pH 7.4), and 50  $\mu\text{l}$  of the biomass extract were mixed. The biomass extract was obtained by mixing 0.3 g of macerated fresh biomass using liquid nitrogen and mortar, with 2 ml of 10 mM phosphate buffer (pH 7.0). The intensity of fluorescence emitted during the first 10 min was read and compared with the intensity emitted in the absence of the sample ( $\lambda$  excitation: 326 nm, a  $\lambda$  emission: 432 nm and 10 nm slit). The results are expressed as  $\mu\text{mol}$  of Trolox Equivalents (TE) per g of fresh biomass (FB) by constructing a standard curve using different concentrations of TROLOX® (Saldarriaga et al., 2020). The determination of intracellular H<sub>2</sub>O<sub>2</sub> concentration in the plant material was performed by mixing 0.2 g of macerated plant sample using liquid nitrogen and mortar, then 0.1% w/v trichloroacetic acid was added, followed by centrifugation at 13000 rpm for 10 min. Subsequently, 0.5 mL of supernatant was taken and 0.5 mL of 10 mM phosphate buffer at pH 7.0 and 1 mL of 1 M KI were added. The reaction mixture was incubated for 10 min at 25°C after which the absorbance was read at a wavelength of 390 nm using a GENESYSTEM 10S double-has bench-top UV visible spectrophotometer. The blank was

prepared by replacing the amount of sample with 10 mM potassium phosphate buffer at pH 7.0 (Chong et al., 2005). Shoot height (cm) was the distance between the roots collar and the apex of the plant (Gagnon et al., 2020). Root length (cm) was measured from the root collar to the most distal root apex (Morales-Ruano et al., 2019). Plant biomass (roots, stem, leaves, and beans) was dried at room temperature. Subsequently, the dry biomass was used to determine the total concentration of PTEs. Briefly, 1.0 g of dry biomass sample was subjected to acid digestion and subsequent quantification of elements using ICP-OES. In brief, dried tissues samples were ground and sieved to determine PTSs. The samples were digested with boiling 5.0 ml of HNO<sub>3</sub> at 80°C for 6 h using an open digestion system (López et al., 2022).

## 2.6 Quality Assurance and Quality Control

Four certified synthetic solutions for trace elements ((Millipore Sigma) were used for ICP-OES\* calibration. Calibration curve was accepted at  $R^2 > 0.9996$ . The LOD (mg L<sup>-1</sup>) for Fe, Pb, Ni, Zn, Cu, As, Cr, and Cd were 0.026, 0.089, 1.12, 0.20, 0.028, 0.041, and 0.007, respectively. The LOQ (mg L<sup>-1</sup>) for Fe, Pb, Ni, Zn, Cu, As, Cr, and Cd were 0.075, 0.015, 0.029, 0.40, 0.06, 0.09, 0.013, 0.003, respectively. Certificated reference material (CRM) BCR® 701 (lake sediment) and NIST SMR 1573a (Tomato leaves) were included in each digestion/extraction batch for quality assurance. For mining tailings samples the recoveries of PTEs calculated relative to the certified concentration (CRM) ranged between 90 and 103%. The specific values for each PTEs ranged from 90 to 102% for Fe, 96–100% for Pb, 90–103% for Ni, 99–100% for Zn, 96–101% for Cu, and 91–102% for As, 93–100% for Cr, and 91–101% for Cd. For plant samples the recoveries for the PTEs ranged from 93 to 99%. The specific values for each PTEs ranged from 93 to 95% for Fe, 91–98% for Pb, 94–96% for Ni, 98–99% for Zn, 91–97% for Cu, and 97–99% for As, 95–97% for Cr, and 93–97% for Cd. Duplicate samples were included to evaluate reproducibility. The coefficients of variation for each set of duplicate reference samples ranges from 0.8 to 9.8% (average 5.3%). Certified synthetic solutions for ICP-OES and blank (DW) were included in each batch for quality assurance. All reagents were of analytical grade,

glassware was soaked in acid (3% HNO<sub>3</sub>) bath overnight prior use. DW (18.2 OMEGA cm<sup>-1</sup>, Mili-Q) was used.

## 2.7 Data Analysis

Microsoft Excel® and Statgraphics® Centurion XVIII were used for the statistical processing of the data. The data shown represent the mean of 3 replicates. Thus, 3 samples were analyzed to determine the pseudo-total concentration of PTEs in the mine tailings of each municipality, 3 samples were analyzed to determine the bioavailable concentration of PTEs in the mine tailings of each municipality, and 3 plant samples were analyzed to determine the response variables (biochemical markers, biometric parameters and PTEs concentration) for the plants sowed in the mine tailings of each municipality. The coefficient of variation (CV) was obtained as  $CV = (\text{standard deviation}/\text{Mean}) * 100$ . ANOVA and Post hoc tests (Tukey) were applied to calculate differences among groups (BU1, BU2, YA, and PB). For all the analyses, statistical differences were set at 5%. To explore the possible correlations between variables, multivariate statistics were performed. The map was generated using ArcMap.

## 3 Results and Discussion

### 3.1 Pseudo-Total and Bioavailable Concentration of PTEs

The mean of pseudo-total concentrations of PTEs in samples are listed in Table 1. The PTEs concentration in samples was as follows: Fe > Zn > As > Cu > Pb > Cd > Cr > Ni in BU1 sample, Fe > Zn > Cu > As > Pb > Ni > Cr > Cd in BU2 sample, Fe > Cu > Cr > Zn > Pb > As > Ni > Cd in YA sample, and Fe > Zn > Pb > Cu > As > Cr > Cd > Ni in PB sample. The highest content of Zn, Cd, Pb, Cr, and Ni were found in the PB sample. These values were on average 6-, 7-, 13-, 4-, and threefold higher than those found for BU1, BU2, and YA samples. While the highest concentrations of Fe, Cu, and As were found in mining tailings from BU1. Which on average were 1-, 2-, and 329-fold higher, with respect to the concentration for these elements in BU2, PB and YA samples. YA sample presented the lowest concentrations of all PTEs, except

**Table 1** Pseudo-total and bioavailable concentration of PTEs (mg/kg) in mine tailings from Puerto Berrío (PB), one from Yalí (YA), and two from Buriticá (BU1 and BU2)

	Metal	BU1	CV	BU2	CV	YA	CV	PB	CV
Pseudo-total	Fe	189,064	1.2%	188,861	1.7%	57,473	4.7%	167,844	7.7%
	Zn	6069	1.5%	1876	3.6%	90	0.8%	27,700	1.5%
	Cu	2765	1.1%	1440	4.7%	514	1.8%	1578	8.1%
	As	3555	2.6%	90	4.4%	27	0.5%	869	8.7%
	Cd	14.3	2.4%	9.4	1.0%	3.4	1.0%	65.2	8.6%
	Pb	2017	2.2%	37	4.6%	80	3.3%	9726	1.0%
	Cr	73	6.9%	13	3.1%	104	5.0%	281	6.5%
	Ni	22	1.0%	31	1.3%	6.5	5.6%	51	0.9%
Bioavailable	Fe	1740	0.3%	1034	2.6%	467	2.6%	1820	1.0%
	Zn	750	5.4%	39	0.1%	5.1	1.0%	10,180	1.0%
	Cu	225	1.6%	29.4	0.1%	7.8	4.3%	184	1.0%
	As	97.6	3.5%	1.2	0.2%	0.6	1.0%	1.6	1.0%
	Cd	1.8	1.6%	0.4	0.10%	0.4	0.1%	28.8	1.0%
	Pb	50	4.5%	2.2	0.1%	1.1	5.2%	405	1.0%
	Cr	1.2	0.1%	0.4	0.1%	1.4	1.0%	3.2	1.0%
	Ni	2.8	1.0%	0.81	0.1%	1.2	0.1%	40	1.0%
Average values of Soil Clean-up Standards (SCSs) proposed by Provoost et al. (2006)									
Metal			As	Cd	Cr	Cu	Pb	Ni	Zn
SCSs for residential land use (mg/kg)			34	15	10,175	643	419	282	4546
SCSs for industrial land use (mg/kg)			173	254	14,298	7207	1150	3488	20,952

\*Values represent the mean ( $n=3$ ). CV: coefficients of variation (%)

for Pb and Cr. Gold mining is a major source of metal and metalloid emissions into the environment. Tailings from this activity are generally characterized with higher concentrations of PTEs like Co, Cr, Cu, Pb, Zn, Ni, As, and Mn (Ngole-Jeme & Fantke, 2017).

To better understand the environmental impacts caused by the presence of PTEs in mining tailings, these values were compared with the standards of soil clean-up standards (SCSs) for residential and industrial land use for various countries (mg/kg) (Table 1) (Provoost et al., 2006). In the samples BU1, BU2, and YA neither of the PTEs exceeded the SCSs value for industrial land use. The PTEs like Zn, As, and Pb exceeded the SCSs values for industrial land use in the PB sample. For BU1 the concentrations of Zn, Cu, As, and Pb and for BU2 As and Cu exceeded the SCSs for residential land use, respectively. PB samples, the pseudo-total concentration of Zn, Cu, As, Cd, and Pb are above the SCSs limits for residential land use. The YA sample did not exceed the SCSs for residential use. According to the results of this study, the PET content of the tailings in the study areas may

represent a potential environmental risk to the surrounding population.

The results for the bioavailable concentration of PTEs in the mining tailings are shown in Table 1 and the RAC values are summarized in Table 2. Based on RAC, the bioavailability profiles of the studied PTEs in tailings samples were as follows: Ni > Cd > Zn > Cu > As > Pb > 1.6 > Fe in BU1 sample, Pb > Cd > Cr > Ni > Zn > Cu > As > Fe in BU2 sample, Ni > Pb > Cd > Zn > As > Cu > Fe in YA sample, and Ni > Cd > Zn > Cu > Pb > Fe > As in PB sample. According to RAC values, the PTEs (except Ni, Cd, and Zn in the PB tailing) were distributed in lower proportion in the bioavailable fraction, indicating that a higher percentage of the mass of these elements are associated with less bioavailable fractions or even a residual fraction. In line with these findings, in mine tailings from Brazil, a high percentage of bioavailability (47–99%) was reported for PTEs such as Mn and Pb, compared to other elements studied, which were mainly distributed in the residual fraction according to the BCR procedure (Barcelos et al., 2020).



**Table 2** Risk Assessment Code (RAC) (%) values for PTEs in mine

Mine tailings	Fe	Zn	Cu	As	Cd	Pb	Cr	Ni	
	RAC (%)								
BU1	0.9	12.4	8.1	2.7	12.6	2.5	1.6	12.7	
BU2	0.5	2.1	2.0	1.3	4.3	5.9	3.1	2.6	
YA	0.8	5.7	1.5	2.2	11.8	12.5	1.3	18.5	
PB	1.0	36.8	11.7	0.2	44.2	4.2	1.1	78.4	

Index were calculated using the average bioavailable concentrations of each PTE

According to RAC values the risk for Fe was not representative. For tailings BU2 and YA Zn presented a low risk, while for BU1 and PB this element presented a moderate and high risk, respectively. The RAC for Cu in tailings BU1, BU2, and YA was low, but in tailings PB it was moderate. In the PB tailings the RAC for As was not representative, and for the BU1, BU2 and YA tailings it was low. Cd presented low risk values in tailings BU2, moderate risk in tailings BU1 and YA, and high risk in tailings PB. A moderate risk was found for Pb in sample YA, and tailings BU1, BU2, and PB presented low risk due to availability. For all tailings Cr was low risk. Finally, a low risk for Ni was detected in tailings BU2. In contrast, tailings BU1 and YA presented moderate risk, and a very high risk for this element was found in tailings from PB.

No direct relationship was found between the pseudo-total concentration and the RAC values for PTEs (Fig. 1). Therefore, a higher total concentration does not necessarily indicate higher bioavailability. So, from an environmental point of view, PTEs risk assessment in mine tailings cannot be based solely on pseudo-total concentrations. The dynamics of PTEs in the environment involves much more than the pseudo-total concentration, since physicochemical characteristics such as pH, OM contents, cation exchange capacity, among others, govern the distribution of these pollutants in the less and higher available fractions (Smolders & Mertens, 2013). PB tailings showed the highest RAC values for Zn, Cu, Cd, and Ni, possibly associated with lower pH and OM values compared to the other tailings (Table 3). The bioavailability of these elements is favored under conditions of acidity and poor organic colloid content (Smolders & Mertens, 2013). On the other hand, in this same tailing As had the lowest RAC values. This can be explained by the fact that As is normally present as an anion in the bioavailable phase, therefore the low pH conditions favor the appearance of positive charges on the surface of the Fe and Al oxides, facilitating its adsorption on the surface of these

**Table 3** Physicochemical characteristics of mining tailings from Puerto Berrío (PB), one from Yalí (YA), and two from Buriticá (BU1 and BU2)

Mining tailings	pH	EC ( $\mu\text{S}/\text{cm}$ )	CEC (meq/100 g)	OM (%)
BU1	$7.4 \pm 0.5$	$1059 \pm 34$	$6.23 \pm 0.4$	$6.9 \pm 0.3$
BU2	$8.2 \pm 0.3$	$4282 \pm 21$	$14.21 \pm 0.5$	$7.5 \pm 0.4$
YA	$7.7 \pm 0.2$	$1173 \pm 12$	$6.28 \pm 0.7$	$4.6 \pm 0.1$
PB	$5.4 \pm 0.24$	$266 \pm 14$	$6.85 \pm 0.1$	$7.9 \pm 0.3$

\*EC: electrical conductivity, CEC: Cationic exchange capacity, OM: Organic matter. Values represent the mean ( $n=3$ )  $\pm$  SD

(López et al., 2022; Sposito, 2008). Surprisingly, Pb did not present a highest RAC in the PB tailing, possibly because this element is generally not very bioavailable when the pH of the medium is higher than 4.0, as it is easily adsorbed on OM (Castañeda-Restrepo et al., 2022; Smolders & Mertens, 2013). Thus, a lower OM content in the YA sample would explain the high bioavailability of Pb found in this tailing.

### 3.2 Comparison of Pseudo-Total Concentration of PTEs of the Studied Mine Tailings with Those in Other Global Mining Areas

Table 4 compares the results of pseudo-total concentrations of PTEs obtained with the concentrations reported in gold mining tailings by other studies worldwide. The average concentrations values (mg/kg) of Fe indicates a fivefold larger than those reported for mining tailings in Nigeria, Namibia, and Ghana. With respect to Zn, it was sevenfold higher than the average concentration for other countries (1913 mg/kg). Cd was threefold higher when compared to the mean concentration value in other gold mining tailings (27 mg/kg). The amount of Pb exceeded twofold the average concentration for the other reports (2720 mg/kg). The average Cu and Cr content found in this study was like the average of other studies, 982 and 137 mg/kg, respectively. Whereas the average As and

**Table 4** Pseudo-total concentration of PTEs from mine tailings worldwide

Location	Ore	Fe	Zn	Cu	As	Cd	Pb	Cr	Ni	Reference
Colombia	Au	57473–192073 (124773)	90–27700 (13805)	514–1440 (977)	27–3555 (191) (65.6)	3.39–65.2 (4881)	13–281 (147)	6.5–31 (18.7)	-	This study
Kongo	Au	-	-	-	1787–4629 (3208)	-	-	-	-	Nyarko (2001)
Korea	Au, Ag	-	580–7541 (3059)	30–749 (385)	143.8–3584 (51.4)	2.2–20 (9.4) (20.3)	125–50.8 (20.3)	-	-	Lim et al. (2008)
Cuba	Au	-	-	-	1085–8650 (5410)	-	-	-	-	Toujaguez et al. (2013)
China	Au	-	358–839 (370)	7.36–468 (141)	0.15–2.28 (0.89)	2.39–16 (7)	599–2105 (599)	-	-	Xiao et al. (2017)
Finland <sup>a</sup>	Au	-	71	120	1520	<0.3	4.3	74	100	Kiventerä et al. (2018)
Egypt	Au	-	56–67 (63.5)	58–46 (50.5)	2643–3183 (2936.3)	-	11.8–16 (9)	224–244 (237)	129–161 (149.5)	Redwan and Bamoussa (2019)
Chile	Au, Cu	-	96.2–338 (217)	1982–8740 (5361)	10.1–119 (64.5)	-	15–272 (143)	-	14.2–86.7 (50.4)	Medina Tripodi et al. (2019)
Canada	Au	-	-	44.9–479 (277.5)	57.5–766.4 (350.4)	-	7.05–31.31 (17.2)	1.52–4.36 (2.7)	-	Gagnon et al. (2020)
Nigeria <sup>b</sup>	Au	49700	119.2	-	4.79	0.68	2802	96.9	80.6	Adewumi and Lamuyan (2020)
Brazil <sup>c</sup>	Au	-	13059 ± 638	-	413 ± 4	143 ± 4	2337 ± 4256	413 ± 4	-	Barcelos et al. (2020)
Brazil <sup>d</sup>	Au	-	207.4 ± 99.05	266.67 ± 107.66	-	-	168.7 ± 69.31	82.0 ± 52.62	126.6 ± 69.78	Teixeira et al. (2019)
Namibia	Au, Cu	8745–33623 (21184)	23–86 (54.5)	364–4014 (2189)	-	-	19–183 (101)	10–95 (52.5)	25–58 (41.5)	Ugwanga and Kgabi (2020)
Ghana <sup>e</sup>	Au	6000	-	50	19	3.1	14	-	-	Akoto and Anning (2021)

<sup>a,b</sup>Total amount of PTEs. <sup>c,d</sup>Total amount of PTEs ± SD. <sup>e</sup> Approximate amount of PTEs. Values are presented as Minimum–Maximum (Mean)

Ni concentration were 6- and fivefold less compared to that reported for mine tailings in other countries, 1270 and 91 mg/kg, respectively. However, it is important to highlight that the concentration of As in BU1 was higher than the same average value. The PTEs distribution and concentration in mining tailings are mainly influenced by the host rock composition of the extracted ore (Redwan & Bamousa, 2019). Elevated levels of Pb, As (for BU1), Fe, and Cd are geological, can be attributed to the lithology of the study areas. Elements like Pb is most likely associated with different minerals types (e.g., galena PbS) occurred in BU1, BU2, and PB. Elements like As and Fe are typically associated with the weathering of arsenopyrite (FeAsS) and pyrite (FeS), present in the epithermal deposits of massive sulphides characteristic of this zone. Cd concentrations may have been favored by the sedimentary lithology of the area as sedimentary rocks have been shown to contain higher levels of Cd than igneous rocks (Smolders & Mertens, 2013). This comparative analysis suggests that the waste materials from gold mining activities at BU1, BU2, YA, and PB generate a significant input of PTEs compared to other gold mines for which similar studies have been conducted. Other studies conducted in Colombia in mining areas have shown that the pseudo-total concentration of PTEs, such as Cu, Zn, V, Co, Cr, and Pb, is higher than the lithological concentration. This is due to the release of these elements from mine tailings (Gutiérrez-Mosquera et al., 2018a). Similarly, Marrugo-Negrete et al. (2017) found Ni enrichment in agricultural soils affected by ferro-nickel mining. The same authors also reported an increase in Hg concentrations in soils downstream of gold mining areas in northern Colombia. An enrichment of metals like Pb and Cd was discovered as a result of mining activities in a coal mining region

in the department of Antioquia (Castañeda-Restrepo et al., 2022). Overall, as shown in Table 4, the pseudo-total concentration of PETs in the studied areas is significantly higher than that in other studies conducted globally. This could be the result of several factors. On the one hand, as explained above, the geology of the deposit. On the other hand, the contributions of illegal mining processes, which are made using bad practices, which release a greater amount of PTEs into the environment. In addition, the authorities do not have complete control of the land, which results in the extraction processes being out of control and the environmental consequences growing because of excessive extraction.

### 3.3 Geochemical Abundance Index

The results of the GAI are shown in Table 5. For the PTEs Fe, Cr and Ni, the GAI results for all samples were less than zero, indicating that the values are below the reference concentrations. Similar results for the enrichment of these metals were found by Kusin et al. (2018). The tailing of BU1 is where more values are presented  $GAI > 3.0$  (significant increase), especially As and Cd. For the BU2 sample, there is a considerable reduction in the GAI values with respect to the BU1 tailings in the metals analyzed, except for Cu whose GAI value is the highest of the four tailings analyzed. As for YA Cu, and for PB Cd, Zn, and Pb. The enrichment of PTEs suggests a strong influence of ore deposits (Kusin et al., 2018). Thus, the GAI values in tailings could be attributed to the primary minerals of gold mining, arsenopyrite, pyrite, chalcopyrite, and galene. However, the GAI also contemplates enrichment processes due to anthropogenic activities, which

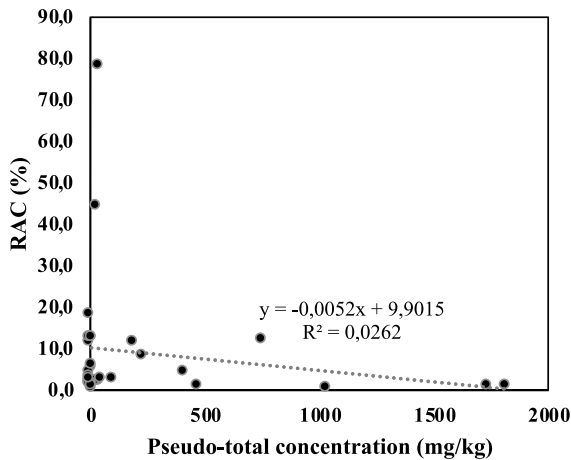
**Table 5** Geochemical Abundance Index (GAI), single hazard factors (Eir), and ecological risk index (RI) values for PTEs

Mine tailings	Fe	Zn	Cu	As	Cd	Pb	Cr	Ni	RI	
	GAI									
BU1	-2.1	5.5	5.9	8.6	8.1	5.3	-0.5	-1.8	-	
BU2	-2.1	3.8	8.3	3.3	4.2	-0.5	-3.0	-1.3	-	
YA	-3.8	-0.6	3.5	1.6	2.7	0.6	0.0	-3.5	-	
PB	-2.3	7.7	5.1	6.6	10.3	7.5	1.4	-0.5	-	
	Eir									
BU1	-	67.4	460	5925	1229	288	2.10	2.60	7974	
BU2	-	20.9	240	150	805	5.30	0.40	3.80	1227	
YA	-	1.00	85.7	45.9	290	11.5	3.00	0.80	438	
PB	-	307	263	1449	5589	1389	8.10	6.20	9013	

\*Index were calculated using the average pseudo-total concentrations of each PTE

could also favor the increase in PTEs concentrations found in this study.

The results of the GAI suggest more attention to be paid to environmental risks in the PB area. Since, it presents at the same time high values of

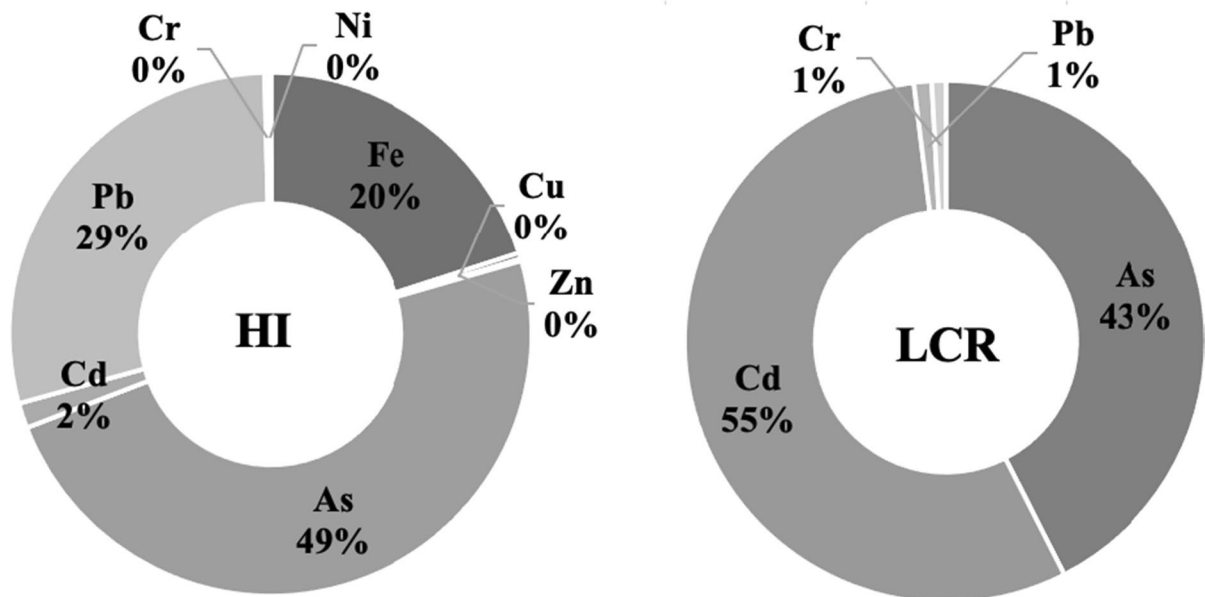


**Fig. 2** Relationship between the pseudo-total concentration and risk assessment code (RAC%). The graph plots the pseudo total concentration of the PTEs (X-axis) and the RAC% for each PTEs (Y-axis)

availability (RAC), as well as of enrichment (GAI). Therefore, the increase in the concentrations of PTEs, especially Cd, Zn and Cu, will be distributed to the most available fractions. These results indicate that it is important to evaluate the results of the total metal concentrations in a tailing versus the reference values and not to analyze the environmental impact with the pseudo-total values only; in this case although in the tailings of BU1 and PB predominate total metals such as Zn and Cu, the greatest increase in GAI values is for metals like Cd, As, and Pb (Figs. 2 and 3).

### 3.4 Ecological Risk

The results of the calculation of the potential ecological risk are shown in Table 5. Cr and Ni presented low ecological risk in all tailings. BU1 presented moderate risk for Zn, high risk for Pb and very high risk for Cu, As, and Cd. In BUI2 a low risk was found for Zn and Pb, considerable risk for Cu, high risk for As and very high risk for Cd. Pb and Zn represent a low risk in the YA tailings. Zn and Cu presented high risk and As, Cd, and Pb very high risk. Regarding the RI, all tailings fall into the high-risk classification.



**Fig. 3** Potential non-carcinogenic (HI) and carcinogenic (LCR) risk distribution for the mine tailings collected from the municipalities of Buriticá (BU1, BU2), Yalí (YA), and Puerto

Berrio (PB). The figures show the calculated risk for all tailing samples, both adults and children

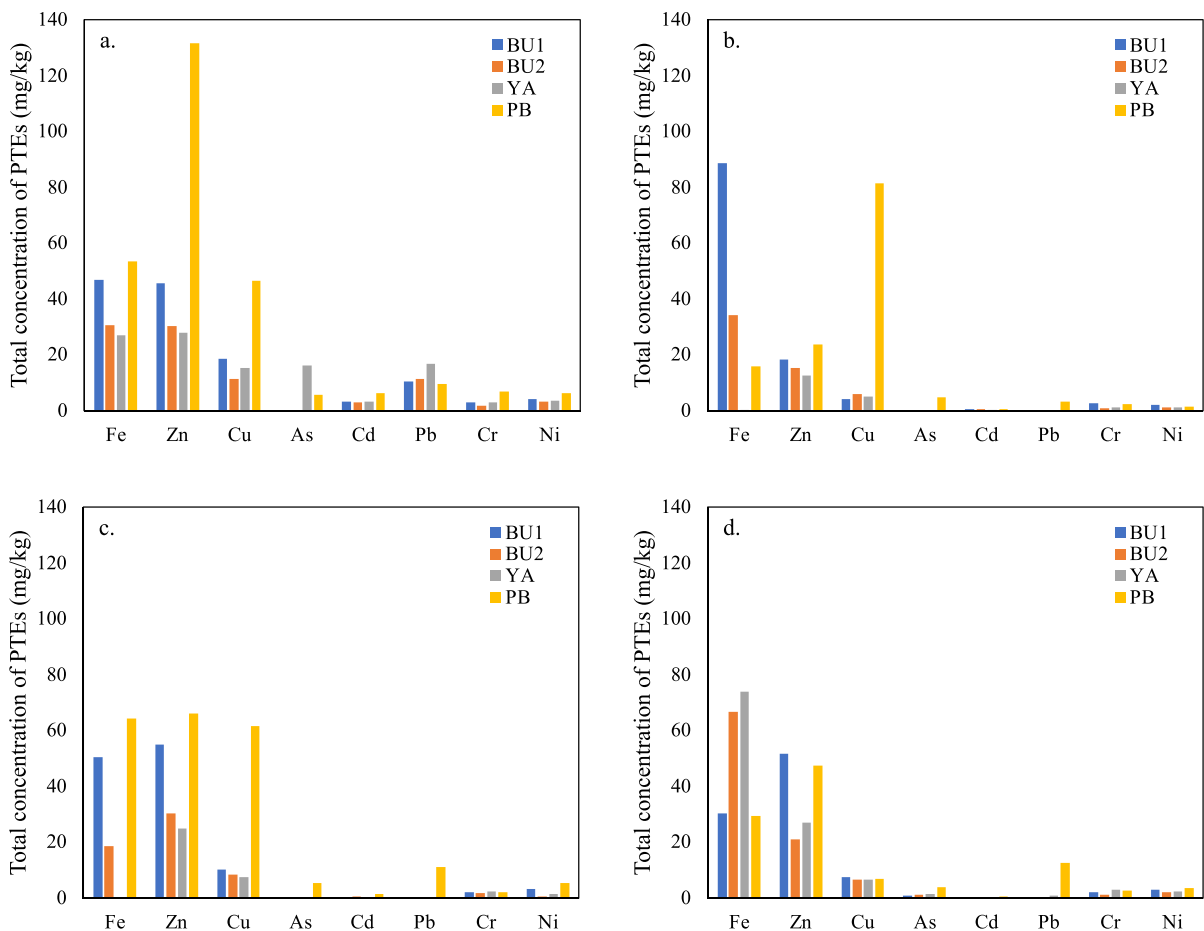
The largest contribution to RI was made by As in tailings BU1 (Fig. 4a) and by Cd for tailings BU2, YA, and PB (Fig. 4b, c, d). When conducting a RI analysis considering the sum of Eir for all PTEs in all mine tailings samples, As, Cd and Pb are shown to be the most significant PTEs (Fig. 4e).

Although the availability values (RAC) for Zn and Ni were high and very high, the Eir results (Fig. 4) did not show a significant contribution of these metals to the ecological risk. In contrast, metals such as As and Pb, which had low RAC values, were classified as highly hazardous to the ecosystem. Then, the ecological risk will not only be conditioned by the concentration, but also by the toxicity of each metal. In the case of this study, As and Pb are more toxic than

Zn and Ni for biological systems (Hakanson, 1980). For example, has a high level of toxicity even at the lowest levels, and is considered a priority element in terms of its impact on living organisms (Arias Espana et al., 2018). Cd has been considered one of the most toxic elements contributed by mining activities, and one of the main indicators of ecosystemic risk (Kusin et al., 2018).

### 3.5 Human Health Risk

Pollution caused by heavy metal toxicity would have an impact on human health and the environment. Metals such as As, Cr and Pb have a high level of toxicity, and have been grouped into priority metals that are



**Fig. 4** Pseudo-total concentration of PTEs (mg/kg) in *P. vulgaris*, a) root, b) stem, c) leaf, and d) beans after 3 months of sowing in mine tailings. Reference value for leaf PTEs concentrations (mg/kg) in plants from uncontaminated environment

(Remon et al., 2013): Zn (40.60 mg/kg), Cu (8.90 mg/kg), Cd (0.08 mg/kg), Pb (0.84 mg/kg), and Cr and Ni (1.50 mg/kg). Values represent the mean (n=3)

**Table 6** Hazard quotient (HQ) and cumulative hazard index (HI) for non-carcinogenic risk. HQ was calculated using the average available concentrations of each PTE. The HI values in bold exceed the threshold of 1.0 (USEPA, 2012). Bold numbers indicate potential risk to human health

	PTEs							
	Fe	Zn	Cu	As	Cd	Pb	Cr	Ni
BU1								
Adult								
HQ-Ingest	6.4E-02	1.5E-03	1.0E-03	4.5E-01	2.5E-03	2.0E-02	5.5E-04	1.9E-04
HQI-Inhale	9.5E-06	2.2E-07	1.5E-07	6.6E-05	3.6E-07	2.9E-06	8.1E-08	2.8E-08
HQ-Dermal	2.6E-04	5.9E-06	4.1E-06	1.8E-03	9.8E-06	7.8E-05	2.2E-06	7.7E-07
HI	6.5E-02	1.5E-03	1.0E-03	4.5E-01	2.5E-03	2.0E-02	5.5E-04	1.9E-04
Children								
HQ-Ingest	6.0E-01	1.4E-02	9.6E-03	4.2E+00	2.3E-02	1.8E-01	5.1E-03	1.8E-03
HQI-Inhale	4.4E-05	1.0E-06	7.1E-07	3.1E-04	1.7E-06	1.3E-05	3.8E-07	1.3E-07
HQ-Dermal	1.2E-03	2.7E-05	1.9E-05	8.3E-03	4.6E-05	3.6E-04	1.0E-05	3.6E-06
HI	6.0E-01	1.4E-02	9.6E-03	<b>4.2E+00</b>	2.3E-02	1.8E-01	5.1E-03	1.8E-03
BU2								
Adult								
HQ-Ingest	3.8E-02	7.6E-05	1.3E-04	5.5E-03	5.5E-04	8.6E-04	1.8E-04	5.5E-05
HQI-Inhale	5.6E-06	1.1E-08	2.0E-08	8.1E-07	8.1E-07	1.3E-07	2.7E-08	8.1E-09
HQ-Dermal	1.5E-04	3.0E-07	5.4E-07	2.2E-05	2.2E-05	3.4E-06	7.3E-07	2.2E-07
HI	3.8E-02	7.6E-05	1.3E-04	5.5E-03	5.5E-03	8.6E-04	1.8E-04	5.5E-05
Children								
HQ-Ingest	3.6E-01	7.1E-04	1.3E-03	5.1E-02	5.1E-03	8.0E-03	1.7E-03	5.1E-04
HQI-Inhale	2.6E-05	5.2E-08	9.2E-08	3.8E-06	3.8E-07	5.9E-07	1.3E-07	3.8E-08
HQ-Dermal	7.1E-04	1.4E-06	2.5E-06	1.0E-04	1.0E-05	1.6E-05	3.4E-06	1.0E-06
HI	3.6E-01	7.1E-04	1.3E-03	5.1E-02	5.1E-03	8.1E-03	1.7E-03	5.1E-04
YA								
Adult								
HQ-Ingest	1.7E-02	1.0E-05	3.6E-05	2.7E-03	5.5E-04	7.0E-04	6.4E-04	8.2E-05
HQI-Inhale	2.5E-06	1.5E-09	5.3E-09	4.0E-07	8.1E-08	1.0E-07	9.4E-08	1.2E-08
HQ-Dermal	6.9E-05	4.0E-08	1.4E-07	1.1E-05	2.2E-06	2.8E-06	2.6E-06	3.3E-07
HI	1.7E-02	1.0E-05	3.6E-05	2.8E-03	5.5E-04	7.1E-04	6.4E-04	8.3E-05
Children								
HQ-Ingest	1.6E-01	9.3E-05	3.3E-04	2.6E-02	5.1E-03	6.6E-03	6.0E-03	7.7E-04
HQI-Inhale	1.2E-05	6.8E-09	2.5E-08	1.9E-06	3.8E-07	4.8E-07	4.4E-07	5.6E-08
HQ-Dermal	3.2E-04	1.9E-07	6.7E-07	5.1E-05	1.0E-05	1.3E-05	1.2E-05	1.5E-06
HI	1.6E-01	9.3E-05	3.3E-04	2.6E-02	5.1E-03	6.6E-03	6.0E-03	7.7E-04
PB								
Adult								
HQ-Ingest	6.7E-02	2.0E-03	8.4E-05	2.7E-04	1.1E-02	2.5E-01	1.3E-03	7.3E-04
HQ-Inhale	9.9E-06	2.9E-07	1.2E-08	4.0E-08	1.6E-06	3.7E-05	2.0E-07	1.1E-07
HQ-Dermal	2.7E-04	7.9E-06	3.4E-07	1.1E-06	4.4E-05	9.9E-04	5.4E-06	2.9E-06
HI	6.8E-02	2.0E-03	8.4E-05	2.8E-04	1.1E-02	2.5E-01	1.4E-03	6.8E-03
Children								
HQ-Ingest	6.3E-01	1.9E-02	7.8E-04	2.6E-03	1.0E-01	2.3E+00	1.3E-02	6.8E-03
HQI-Inhale	4.6E-05	1.4E-06	5.8E-08	1.9E-07	7.5E-06	1.7E-04	9.3E-07	5.0E-07
HQ-Dermal	1.3E-03	3.7E-05	1.6E-06	5.1E-06	2.0E-04	4.6E-03	2.5E-05	1.4E-05
HI	6.3E-01	1.9E-02	7.9E-04	2.6E-03	1.0E-01	<b>2.3E+00</b>	1.3E-02	6.8E-03

**Table 7** Carcinogenic risk for different exposure pathways for adults and children. CR and was calculated using the average available concentrations of each PTE. The LCR values in bold exceed the threshold of 1.0E-04 (USEPA, 2012)

	PTEs			
	As	Cd	Pb	Cr
<b>BU1</b>				
Adult				
CR-Ingest	2.0E-04	1.6E-05	5.8E-07	8.2E-07
CR-Inhale	2.9E-08	2.3E-09	8.6E-11	1.2E-10
CR-Dermal	8.0E-07	6.2E-08	2.3E-09	3.3E-09
LCR	2.0E-04	1.6E-05	5.8E-07	8.3E-07
Children				
CR-Ingest	1.9E-03	1.4E-04	5.4E-06	7.7E-06
CR-Inhale	1.4E-07	1.1E-08	4.0E-10	5.6E-10
CR-Dermal	3.7E-06	2.9E-07	1.1E-08	1.5E-08
LCR	1.9E-03	1.5E-04	5.4E-06	7.7E-06
<b>BU2</b>				
Adult				
CR-Ingest	2.5E-06	3.5E-06	2.6E-08	2.7E-07
CR-Inhale	3.6E-10	5.1E-10	3.8E-12	4.0E-11
CR-Dermal	9.8E-09	1.4E-08	1.0E-10	1.1E-09
LCR	2.5E-06	3.5E-06	2.6E-08	2.8E-07
Children				
CR-Ingest	2.3E-05	3.2E-05	2.4E-07	2.6E-06
CR-Inhale	1.7E-09	2.4E-09	1.8E-11	1.9E-10
CR-Dermal	4.6E-08	6.4E-08	4.8E-10	5.1E-09
LCR	2.3E-05	3.2E-05	2.4E-07	2.6E-06
<b>YA</b>				
Adult				
CR-Ingest	1.2E-06	3.5E-06	2.1E-08	9.6E-07
CR-Inhale	1.8E-10	5.1E-10	3.1E-12	1.4E-10
CR-Dermal	4.9E-09	1.4E-08	8.4E-11	3.8E-09
LCR	1.2E-06	3.5E-06	2.1E-08	9.6E-07
Children				
CR-Ingest	1.2E-05	3.2E-05	2.0E-07	8.9E-06
CR-Inhale	8.5E-10	2.4E-09	1.4E-11	6.6E-10
CR-Dermal	2.3E-08	6.4E-08	3.9E-10	1.8E-08
LCR	1.2E-05	3.2E-05	2.0E-07	9.0E-06
<b>PB</b>				
Adult				
CR-Ingest	3.3E-06	2.5E-04	4.7E-06	2.2E-06
CR-Inhale	4.8E-10	3.7E-08	6.9E-10	3.2E-10
CR-Dermal	1.3E-08	9.9E-07	1.9E-08	8.7E-09
LCR	3.3E-06	2.5E-04	4.7E-06	2.2E-06
Children				
CR-Ingest	3.1E-05	2.3E-03	4.4E-05	2.0E-05
CR-Inhale	2.3E-09	1.7E-07	3.2E-09	1.5E-09

**Table 7** (continued)

	PTEs			
CR-Dermal	6.1E-08	4.6E-06	8.8E-08	4.1E-08
LCR	3.1E-05	2.3E-03	4.4E-05	2.0E-05

important to public health. Even at the lowest levels of vulnerability, they are considered toxic that would influence multiple organ damage and these metallic elements were categorized as human carcinogens.

The CDI values for ingestion, inhalation and dermal contact for children and adults can be seen in Table S3. The HQ and HI values are shown in Table 6, and the CR and LCR in Table 7. All the calculated HI values were not greater than the recommended threshold in tailings BU2 and YA. However, the HI value for As in tailing BU1 exceeded the recommended threshold for non-carcinogenic risk in children by 2.0-fold, as well as Pb in the PB tailing exceeded the recommended value by 2.3-fold. As for the carcinogenic risk, determined for As, Cd, Pb, and Cr, in BU1 sample the LCR shows that As exceeds on average 2.1- and 19-fold the recommended threshold value for both adults and children, respectively. In the same tailings, Cd exceeds 1.5-fold the recommended value for children. In the PB tailings, the LCR for Cd is 2.5- and 23-fold above the recommended threshold. When comparing the LCR values for As and Cd in both tailings with the other PTEs, they differ on average by an order of magnitude of 1E+05. In the BU2 and YA tailings the LCR values did not exceed the threshold for any PTE. The results for non-carcinogenic and carcinogenic risk are in line with the findings for ecological risk. Both analyses suggest that As, Cd and Pb are highly impactful PTEs to the environment and public health and should be considered as priority targets for monitoring and treatment (Fig. 3). Regardless of the carcinogenic or non-carcinogenic risk, children were found to be more susceptible to the potential health risk due to the PTEs concentration in tailings mines.

### 3.6 Biomonitoring of PTEs in Mine Tailings

Table 8 shows the values of germination percentage, antioxidant activity, hydrogen peroxide concentration, root length and shoot height of *P. vulgaris* planted in mine tailings of BU1, BU2, YA, and PB. Germination

**Table 8** Values for germination percentage, antioxidant activity, hydrogen peroxide concentration, root length, and shoot height of *P. vulgaris*. Mine tailings from Buritica (BU1, BU2), Yalí (YA), and Puerto Berrio (PB)

Mine tailings	SG (%)	Aa ( $\mu\text{mol TE / g FB}$ )	H <sub>2</sub> O <sub>2</sub> (ng / mL)	Root length (cm)	Shoot height (cm)
BU1	60b	76.3a	233a	8.1b	10.8b
CV	1.0%	1.0%	3.1%	1.5%	0.3%
BU2	100a	49.7b	117b	15.8a	15.4a
CV	2.5%	0.9%	3.6%	2.0%	1.0%
YA	100a	50.4b	121b	12.4a	18.9a
CV	0.0%	1.2%	1.6%	0.4%	2.2%
PB	40b	60.8a	239a	4.6b	8.3b
CV	1.5%	2.0%	2.4%	1.0%	0.5%

Values represent the mean ( $n=3$ ). CV: coefficients of variation (%). SG: germination percentage. Aa: Antioxidant activity. H<sub>2</sub>O<sub>2</sub>: hydrogen peroxide. FB: fresh biomass. Different letters indicate statistical significance at  $P < 0.05$  (Tukey's test)

percentage, root length and shoot height were significantly reduced in plants planted in BU1 and PB tailings compared to plants planted in BU2 and YA. As for biochemical markers, the concentration of H<sub>2</sub>O<sub>2</sub> increased significantly in plants planted in the BU1 and PB tailings. Likewise, the Aa increased for these plants. Plants corresponding to the BU2 and YA tailings showed lower H<sub>2</sub>O<sub>2</sub> and Aa values. Changes in Aa and H<sub>2</sub>O<sub>2</sub> concentration are related to the abiotic stress in *P. vulgaris* plants, since PTEs are associated with oxidative stress in plants, PTEs modifies the concentration of reactive oxygen species and antioxidant machinery in plants (Gill & Tuteja, 2010; Saldarriaga et al., 2020). Also, the concentration of PTEs could affect the growth of *P. vulgaris* plants (Leudo et al., 2020). Biochemical markers (H<sub>2</sub>O<sub>2</sub> and Aa) were significantly correlated with biometric parameters (root length and shoot height) (Table 9). Abiotic stress in plants caused by PTEs can be reflected in plant morphological changes, such as root length (Arroyave et al., 2013). The

correlation found in this study (Table 9) suggests that the biometric parameters of *P. vulgaris* plants are an easily measured indicator of the abiotic stress caused by PTEs present in mine tailings. Based on the results, it is suggested that the SG, root length, and shoot height of the *P. vulgaris* plant can be used to monitor the concentration of PTEs in mine tailings. This type of monitoring methodology could be implemented by the authorities and/or the population close to the mining activity.

Also, LCR values for children and adults were significantly correlated with SG, root length, and shoot length (Table 10). The use of selected plants as biomonitors facilitates the assessment of a contaminant's bioavailability and comparison among sites. The LCR were calculated using the bioavailable concentration values of the PTEs. The correlation found indicates that the bioavailability determined by the BCR method can be related to the bioavailability of PTEs for *P. vulgaris* plants. A higher bioavailability of PTEs in PB and BU1 is related to the trends found for biomarkers Aa and H<sub>2</sub>O<sub>2</sub> and the root length and shoot length values found. These results support the idea that biometric parameters can be used to monitor the risk to human health associated with PTEs in mine tailings.

On the other hand, the bioaccumulation of metals in the *P. vulgaris* plant can also be used as a tool for monitoring PTEs in mine tailings. Figure 4 shows the concentrations of PTEs in the *P. vulgaris* plants sown in the different mine tailings. The plants that were sown in PB mine exceeded the PTEs values in leaves proposed as a guideline for uncontaminated soils (Remon et al., 2013). This can be related to the lower values of SG, root length, shoot length and

**Table 9** Pearson's correlation analysis between SG, Aa, H<sub>2</sub>O<sub>2</sub>, root length, and shoot height

	SG	Aa	H <sub>2</sub> O <sub>2</sub>	Root length	Shoot height
SG		-0.69*	-0.97*	0-89*	0.94*
Aa			0.77*	-0.62*	-0.56*
H <sub>2</sub> O <sub>2</sub>				-0.87*	-0.93*
Root length					0.77*
Shoot height					

SG: germination percentage. Aa: Antioxidant activity. H<sub>2</sub>O<sub>2</sub>: hydrogen peroxide. Statistical significance is represented by \*  $P < 0.05$



**Table 10** Pearson's correlation analysis between potential carcinogenic risk (LCR), SG, root length, and shoot height

	SG	Root length	Shoot height	As LCR adult	Cd LCR adult	Pb LCR adult	Cr LCR adult	As LCR children	Cd LCR children	Pb LCR children	Cr LCR children
SG											
Root length	0.89*		0.94*	-0.79*	-0.98*	-0.85*	-0.59*	-0.75*	-0.86*	-0.85*	-0.59*
Shoot height		0.77*		-0.64*	-0.91*	-0.83*	-0.64*	-0.63*	-0.81*	-0.83*	-0.64*
As LCR adult					-0.92*	-0.85*	-0.52*	-0.78*	-0.85*	-0.85*	-0.52*
Cd LCR adult						-0.15	-0.56*	0.99*	-0.21	-0.16	-0.51*
Pb LCR adult						0.84*	0.54*	0.31	0.82*	0.84*	0.54*
Cr LCR adult							0.85*	-0.21	0.98*	0.99*	0.84*
As LCR children								-0.55*	0.88*	0.84*	0.99*
Cd LCR children									-0.27	-0.21	-0.55*
Pb LCR children										0.98*	0.88*
Cr LCR children											0.84*

SG: germination percentage. Statistical significance is represented by \*  $P < 0.05$

increased values of  $H_2O_2$  and Aa in the *P. vulgaris* plants that were planted in PB mine tailing (Table 8). In the same line, plants sown in the tailings of BU1, exceeded the concentration of Cu, Cr and Ni in leaves for uncontaminated soils. In terms of biometric and biochemical parameters, these plants had a similar trend to the plants sown in the PB tailings (Table 8).

#### 4 Conclusions

This is the first report of bioavailable concentrations of PTEs in mine tailings from the department of Antioquia, Colombia. In addition, a methodology for biomonitoring PTEs is proposed, which can also be correlated with potential risks to human health and the ecosystem in the region. The results obtained in this study can help the competent authorities to take immediate measures focused on better environmental and risk management of hazardous waste in gold mining areas.

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**Data availability** All data generated or analyzed during this study are included in this published article [and its supplementary information files.

#### Declarations

**Conflict of Interest** The authors declare that they have no conflict of interest.

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#### References

- Acosta-Luque, M. P., López, J. E., Henao, N., Zapata, D., Giraldo, J. C., & Saldarriaga, J. F. (2023). Remediation of Pb-contaminated soil using biochar-based slow-release P fertilizer and biomonitoring employing bioindicators. *Science and Reports*, *13*, 1657. <https://doi.org/10.1038/s41598-022-27043-8>
- Adamo, P., Zampella, M., 2008. Chemical speciation to assess potentially toxic metals’ (PTMs’) bioavailability and geochemical forms in polluted soils, in: *Environmental Geochemistry*. Elsevier, 175–212. <https://doi.org/10.1016/B978-0-444-53159-9.00009-7>
- Adewumi, A. J., & Laniyan, T. A. (2020). Contamination, sources and risk assessments of metals in media from Anka artisanal gold mining area Northwest Nigeria. *Science of the Total Environment*, *718*, 137235. <https://doi.org/10.1016/j.scitotenv.2020.137235>
- Adiansyah, J. S., Rosano, M., Vink, S., & Keir, G. (2015). A framework for a sustainable approach to mine tailings management: Disposal strategies. *Journal of Cleaner Production*, *108*, 1050–1062. <https://doi.org/10.1016/j.jclepro.2015.07.139>
- Akoto, R., & Anning, A. K. (2021). Heavy metal enrichment and potential ecological risks from different solid mine wastes at a mine site in Ghana. *Environment Advance*, *3*, 100028. <https://doi.org/10.1016/j.envadv.2020.100028>
- Alloway, B.J. (Ed.), 2013. *Heavy Metals in Soils*, Environmental Pollution. Springer Netherlands, Dordrecht. <https://doi.org/10.1007/978-94-007-4470-7>
- Arias Espana, V. A., Rodriguez Pinilla, A. R., Bardos, P., & Naidu, R. (2018). Contaminated land in Colombia: {A critical review of current status and future approach for the management of contaminated sites. *Science of the Total Environment*, *618*, 199–209. <https://doi.org/10.1016/j.scitotenv.2017.10.245>
- Arroyave, C., Tolrà, R., Thuy, T., Barceló, J., & Poschenrieder, C. (2013). Differential aluminum resistance in *Brachiaria* species. *Environmental and Experimental Botany*, *89*, 11–18. <https://doi.org/10.1016/j.envexpbot.2013.01.001>
- Barcelos, D. A., Pontes, F. V. M., da Silva, F. A. N. G., Castro, D. C., dos Anjos, N. O. A., & Castilhos, Z. C. (2020). Gold mining tailing: Environmental availability of metals and human health risk assessment. *Journal of Hazardous Materials*, *397*, 122721. <https://doi.org/10.1016/j.jhazmat.2020.122721>
- Becerra-Agudelo, E., López, J. E., Betancur-García, H., Carbal-Guerra, J., Torres-Hernández, M., & Saldarriaga, J. F. (2022). Assessment of the application of two amendments (lime and biochar) on the acidification and bioavailability of Ni in a Ni-contaminated agricultural soils of northern Colombia. *Heliyon*, *8*, e10221. <https://doi.org/10.1016/j.heliyon.2022.e10221>
- Betancur-Corredor, B., Loaiza-Usuga, J. C., Denich, M., & Borgemeister, C. (2018). Gold mining as a potential driver of development in Colombia: Challenges and

- opportunities. *Journal of Cleaner Production*, 199, 538–553. <https://doi.org/10.1016/j.jclepro.2018.07.142>
- Buch, A. C., Niemeyer, J. C., Marques, E. D., & Silva-Filho, E. V. (2021). Ecological risk assessment of trace metals in soils affected by mine tailings. *Journal of Hazardous Materials*, 403, 123852. <https://doi.org/10.1016/j.jhazmat.2020.123852>
- Bueno, P. C., Bellido, E., Rubí, J. A. M., & Ballesta, R. J. (2009). Concentration and spatial variability of mercury and other heavy metals in surface soil samples of periurban waste mine tailing along a transect in the Almadén mining district (Spain). *Environmental Geology*, 56, 815–824. <https://doi.org/10.1007/s00254-007-1182-z>
- Bustamante, N., Danoucaras, N., McIntyre, N., Díaz-Martínez, J.C., Restrepo-Baena, O.J., 2016. Review of improving the water management for the informal gold mining in Colombia. *Revista Facultad de Ingeniería Universidad de Antioquia*. <https://doi.org/10.17533/udea.redin.n79a16>
- Castañeda-Restrepo, I., Salzar Giraldo, J.P., López, J.E., 2022. Disponibilidad y niveles de contaminación de cadmio y plomo en cuatro suelos colombianos. Un análisis del riesgo a la salud humana y al ecosistema. *Respuestas* 26. <https://doi.org/10.22463/0122820X.3029>
- Chong, T. M., Abdullah, M. A., Fadzillah, N. M., Lai, O. M., & Lajis, N. H. (2005). Jasmonic acid elicitation of anthraquinones with some associated enzymic and non-enzymic antioxidant responses in *Morinda elliptica*. *Enzyme and Microbial Technology*, 36, 469–477. <https://doi.org/10.1016/j.enzmictec.2004.11.002>
- Favas, P. J. C., Pratas, J., Gomes, M. E. P., & Cala, V. (2011). Selective chemical extraction of heavy metals in tailings and soils contaminated by mining activity: Environmental implications. *J. Geochemical Explor.*, 111, 160–171. <https://doi.org/10.1016/j.gexplo.2011.04.009>
- Gagnon, V., Rodrigue-Morin, M., Migneault, M., Tardif, A., Garneau, L., Lalonde, S., Shipley, B., Greer, C. W., Bellenger, J.-P., & Roy, S. (2020). Survival, growth and element translocation by 4 plant species growing on acidogenic gold mine tailings in Québec. *Ecological Engineering*, 151, 105855. <https://doi.org/10.1016/j.ecoleng.2020.105855>
- Gill, S. S., & Tuteja, N. (2010). Reactive oxygen species and antioxidant machinery in abiotic stress tolerance in crop plants. *Plant Physiology and Biochemistry*, 48, 909–930. <https://doi.org/10.1016/j.plaphy.2010.08.016>
- Gutiérrez-Mosquera, H., Shruti, V. C., Jonathan, M. P., Roy, P. D., & Rivera-Rivera, D. M. (2018a). Metal concentrations in the beach sediments of Bahía Solano and Nuquí along the Pacific coast of Chocó, Colombia: A baseline study. *Marine Pollution Bulletin*, 135, 1–8. <https://doi.org/10.1016/j.marpolbul.2018.06.060>
- Gutiérrez-Mosquera, H., Sujitha, S. B., Jonathan, M. P., Sarkar, S. K., Medina-Mosquera, F., Ayala-Mosquera, H., Morales-Mira, G., & Arreola-Mendoza, L. (2018b). Mercury levels in human population from a mining district in Western Colombia. *Journal of Environmental Sciences*, 68, 83–90. <https://doi.org/10.1016/j.jes.2017.12.007>
- Hakanson, L. (1980). An ecological risk index for aquatic pollution control: a sedimentological approach. *Water Research*, 14, 975–1001. [https://doi.org/10.1016/0043-1354\(80\)90143-8](https://doi.org/10.1016/0043-1354(80)90143-8)
- Islam, M. S., Ahmed, M. K., Raknuzzaman, M., Habibullah-Al-Mamun, M., & Kundu, G. K. (2017). Heavy metals in the industrial sludge and their ecological risk: A case study for a developing country. *J. Geochemical Explor.*, 172, 41–49. <https://doi.org/10.1016/j.gexplo.2016.09.006>
- Jiang, L., Sun, H., Peng, T., Ding, W., Liu, B., & Liu, Q. (2021). Comprehensive evaluation of environmental availability, pollution level and leaching heavy metals behavior in non-ferrous metal tailings. *Journal of Environmental Management*, 290, 112639. <https://doi.org/10.1016/j.jenvman.2021.112639>
- Jorge Mendoza, C., Tatiana Garrido, R., Cristian Quilodrán, R., Matías Segovia, C., & José Parada, A. (2017). Evaluation of the bioaccessible gastric and intestinal fractions of heavy metals in contaminated soils by means of a simple bioaccessibility extraction test. *Chemosphere*, 176, 81–88. <https://doi.org/10.1016/j.chemosphere.2017.02.066>
- Kiventerä, J., Sreenivasan, H., Cheeseman, C., Kinnunen, P., & Illikainen, M. (2018). Immobilization of sulfates and heavy metals in gold mine tailings by sodium silicate and hydrated lime. *Journal of Environmental Chemical Engineering*, 6, 6530–6536. <https://doi.org/10.1016/j.jece.2018.10.012>
- Kodirov, O., Kersten, M., Shukurov, N., & Martín Peinado, F. J. (2018). Trace metal(loid) mobility in waste deposits and soils around Chadak mining area. *Uzbekistan. Sci. Total Environ.*, 622–623, 1658–1667. <https://doi.org/10.1016/j.scitotenv.2017.10.049>
- Kusin, F. M., Azani, N. N. M., Hasan, S. N. M. S., & Sulong, N. A. (2018). Distribution of heavy metals and metalloid in surface sediments of heavily-mined area for bauxite ore in Pengerang, Malaysia and associated risk assessment. *CATENA*, 165, 454–464. <https://doi.org/10.1016/j.catena.2018.02.029>
- Kwon, M. J., Yang, J.-S., Lee, S., Lee, G., Ham, B., Boyanov, M. I., Kemner, K. M., & O’Loughlin, E. J. (2015). Geochemical characteristics and microbial community composition in toxic metal-rich sediments contaminated with Au–Ag mine tailings. *Journal of Hazardous Materials*, 296, 147–157. <https://doi.org/10.1016/j.jhazmat.2015.04.035>
- Leudo, A. M., Cruz, Y., Montoya-Ruiz, C., Delgado, M. D., & Saldarriaga, J. F. (2020). Mercury Phytoremediation with *Lolium perenne-Mycorrhizae* in Contaminated Soils. *Sustainability*, 12, 3795. <https://doi.org/10.3390/su12093795>
- Lim, H. S., Lee, J. S., Chon, H. T., & Sager, M. (2008). Heavy metal contamination and health risk assessment in the vicinity of the abandoned Songcheon Au–Ag mine in Korea. *Journal of Geochemical Exploration*, 96, 223–230. <https://doi.org/10.1016/j.gexplo.2007.04.008>
- López, J. E., Arroyave, C., Aristizábal, A., Almeida, B., Builes, S., & Chávez, E. (2022). Reducing cadmium bioaccumulation in *Theobroma cacao* using biochar: basis for scaling-up to field. *Heliyon*, 8(6), 09790. <https://doi.org/10.1016/j.heliyon.2022.e09790>
- López, J. E., Zapata, D., & Saldarriaga, J. F. (2023). Evaluation of different composting systems on an industrial scale as a contribution to the circular economy and its impact on human health. *J Air Waste Manage Assoc*, 73, 679–694. <https://doi.org/10.1080/10962247.2023.2235299>
- Marrugo-Negrete, J., Pinedo-Hernández, J., & Díez, S. (2017). Assessment of heavy metal pollution, spatial distribution and origin in agricultural soils along the Sinú {River}

- {Basin}, {Colombia}. *Environmental Research*, 154, 380–388. <https://doi.org/10.1016/j.envres.2017.01.021>
- Meers, E., Samson, R., Tack, F. M. G., Ruttens, A., Vandegheuchte, M., Vangronsveld, J., & Verloo, M. G. (2007). Phytoavailability assessment of heavy metals in soils by single extractions and accumulation by *Phaseolus vulgaris*. *Environmental and Experimental Botany*, 60, 385–396. <https://doi.org/10.1016/j.envexpbot.2006.12.010>
- Medina Tripodi, E. E., Gamboa Rueda, J. A., Aguirre Céspedes, C., Delgado Vega, J., & Collao Gómez, C. (2019). Characterization and geostatistical modelling of contaminants and added value metals from an abandoned Cu–Au tailing dam in Taltal (Chile). *Journal of South American Earth Sciences*, 93, 183–202. <https://doi.org/10.1016/j.jsames.2019.05.001>
- Morales Ruano, S., Martín-Peinado, F. J., Estepa Molina, C. M., & Bagur-González, M. G. (2019). A quick methodology for the evaluation of preliminary toxicity levels in soil samples associated to a potentially heavy-metal pollution in an abandoned ore mining site. *Chemosphere*, 222, 345–354. <https://doi.org/10.1016/j.chemosphere.2019.01.123>
- Ngole-Jeme, V. M., & Fantke, P. (2017). Ecological and human health risks associated with abandoned gold mine tailings contaminated soil. *PLoS ONE*, 12, e0172517. <https://doi.org/10.1371/journal.pone.0172517>
- Nyarko, B. J. B. (2001). Determination of arsenic in some water bodies, untreated ore and tailing samples at Konongo in the Ashanti region of Ghana and its surrounding towns and villages by instrumental neutron activation analysis. *Journal of Radioanalytical and Nuclear Chemistry*, 249, 581–585. <https://doi.org/10.1023/A:1013246231569>
- Olivero-Verbel, J., Caballero-Gallardo, K., & Turizo-Tapia, A. (2015). Mercury in the gold mining district of San Martín de Loba, South of Bolívar (Colombia). *Environmental Science and Pollution Research*, 22, 5895–5907. <https://doi.org/10.1007/s11356-014-3724-8>
- Orimoloye, I. R., & Ololade, O. O. (2020). Potential implications of gold-mining activities on some environmental components: A global assessment (1990 to 2018). *J. King Saud Univ. - Sci.*, 32, 2432–2438. <https://doi.org/10.1016/j.jksus.2020.03.033>
- Pinedo-Hernández, J., Marrugo-Negrete, J., & Díez, S. (2015). Speciation and bioavailability of mercury in sediments impacted by gold mining in {Colombia}. *Chemosphere*, 119, 1289–1295. <https://doi.org/10.1016/j.chemosphere.2014.09.044>
- Provoost, J., Cornelis, C., & Swartjes, F. (2006). Comparison of soil clean-up standards for trace elements between countries: Why do they differ? *Journal of Soils and Sediments*, 6, 173–181. <https://doi.org/10.1065/jss2006.07.169>
- Redwan, M., & Bamousa, A. O. (2019). Characterization and environmental impact assessment of gold mine tailings in arid regions: A case study of Barramiya gold mine area, Eastern Desert. *Egypt Journal. African Earth Science*, 160, 103644. <https://doi.org/10.1016/j.jafrearsci.2019.103644>
- Remon, E., Bouchardon, J.-L., Le Guédard, M., Bessoule, J.-J., Conord, C., & Faure, O. (2013). Are plants useful as accumulation indicators of metal bioavailability? *Environmental Pollution*, 175, 1–7. <https://doi.org/10.1016/j.envpol.2012.12.015>
- Restrepo-Sánchez, N. E., Acevedo-Betancourth, L., Henao-Murillo, B., & Peláez-Jaramillo, C. (2015). Remediation effect of compost on soluble mercury transfer in a crop of *Phaseolus vulgaris*. *Journal of Environmental Sciences*, 31, 61–67. <https://doi.org/10.1016/j.jes.2014.09.038>
- Rodríguez-Seijo, A., Lourenço, J., Arenas-Lago, D., Mendo, S., Vega, F. A., & Pereira, R. (2020). Chemical availability versus bioavailability of potentially toxic elements in mining and quarry soils. *Chemosphere*, 251, 126421. <https://doi.org/10.1016/j.chemosphere.2020.126421>
- Salazar, G. J. P., Alfaro-De la Torre, M. C., Aguirre, R. N. J., Briones-Gallardo, R., Cedeño, C. J., & Peñuela, M. G. A. (2013). Geochemical fractionation of manganese in the Riogrande II reservoir, Antioquia. *Colombia Environmental Earth Science*, 69, 197–208. <https://doi.org/10.1007/s12665-012-1947-x>
- Saldarriaga, J. F., Cruz, Y., & López, J. E. (2020). Preliminary study of the production of metabolites from in vitro cultures of *C. ensiformis*. *BMC Biotechnology*, 20, 49. <https://doi.org/10.1186/s12896-020-00642-x>
- Sauerwein, T. (2020). Gold mining and development in Côte d'Ivoire: Trajectories, opportunities and oversights. *Land Use Policy*, 91, 104323. <https://doi.org/10.1016/j.landusepol.2019.104323>
- Serrano, M. F., López, J. E., & Saldarriaga, J. F. (2023). Use of activated rice husk biochar for the removal of metals and microorganisms from treated leachates from landfills. *Journal of Material Cycles and Waste Management*, 25, 3414–3424. <https://doi.org/10.1007/s10163-023-01762-0>
- Smolders, E., Mertens, J., 2013. Cadmium, in: Alloway, B.J. (Ed.), Heavy {Metals} in {Soils}. Springer Netherlands, Dordrecht, 283–311. [https://doi.org/10.1007/978-94-007-4470-7\\_10](https://doi.org/10.1007/978-94-007-4470-7_10)
- Sposito, G. (2008). *The chemistry of soils* (2nd ed.). Oxford University Press, Oxford.
- Suppes, R., & Heuss-Abbichler, S. (2021). Resource potential of mine wastes: A conventional and sustainable perspective on a case study tailings mining project. *Journal of Cleaner Production*, 297, 126446. <https://doi.org/10.1016/j.jclepro.2021.126446>
- Teixeira, R. A., de Souza, E. S., de Lima, M. W., Dias, Y. N., da Silveira Pereira, W. V., & Fernandes, A. R. (2019). Index of geoaccumulation and spatial distribution of potentially toxic elements in the Serra Pelada gold mine. *Journal of Soils and Sediments*, 19, 2934–2945. <https://doi.org/10.1007/s11368-019-02257-y>
- Toujaguez, R., Ono, F. B., Martins, V., Cabrera, P. P., Blanco, A. V., Bundschuh, J., & Guilherme, L. R. G. (2013). Arsenic bioaccessibility in gold mine tailings of Delita. *Journal of Hazardous Materials*, 262, 1004–1013. <https://doi.org/10.1016/j.jhazmat.2013.01.045>
- USEPA, 2012, Integrated Risk Information System of the US Environmental Protection Agency.
- Uugwanga, M. N., & Kgabi, N. A. (2020). Assessment of metals pollution in sediments and tailings of Klein Aub and Oamites mine sites Namibia. *Environment Advance*, 2, 100006. <https://doi.org/10.1016/j.envadv.2020.100006>
- Wang, L., Ji, B., Hu, Y., Liu, R., & Sun, W. (2017). A review on in situ phytoremediation of mine tailings. *Chemosphere*, 184, 594–600. <https://doi.org/10.1016/j.chemosphere.2017.06.025>
- Xiao, R., Wang, S., Li, R., Wang, J. J., & Zhang, Z. (2017). Soil heavy metal contamination and health risks associated

with artisanal gold mining in Tongguan, Shaanxi. *China Ecotoxicology Environment Safety*, 141, 17–24. <https://doi.org/10.1016/j.ecoenv.2017.03.002>

Zhu, Y., Wang, Z., Li, Z., & Yu, H. (2022). Experimental research on the utilization of gold mine tailings in magnesium potassium phosphate cement. *Journal Building Engineering.*, 45, 103313. <https://doi.org/10.1016/j.jobee.2021.103313>

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