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The ecological impacts of residues from the heap leaching of ion-adsorption rare earth clays

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Abstract

Heap leaching for rare earth elements poses a serious long-term threat to the adjacent ecological systems in mining areas. The purpose of this research is to thoroughly study the environmental effects of heap leaching in ion-adsorption rare earth element mine tailings after restoration by ecological measures. Soil samples were collected from four heap leaching rare earth element mine tailings and a control site to investigate their effects on topsoil chemical properties and the results of vegetation restoration. The findings revealed that the effects of heap leaching operations increase soil acidity; the soil's sand content was more than twice that of the control site, while the silt content was much lower. Natural and artificial restoration are difficult to establish in mining areas. Plant growth was hampered in the early stages of rehabilitation due to a lack of organic carbon. Then, because of the lack of nitrogen, the plants' succession did not take place over time. Therefore, carbon fertilizers are recommended at the early stage of rehabilitation and soil tumbling to increase the efficiency of nitrogen uptaking. Although heap leaching is no longer permitted, the historic environmental problem at ion-adsorption mine tailings persists. As a result, more research is needed to develop strategies for contamination management in these areas.

Introduction

Southern China is rich in ion-adsorption rare earth element (REE) resources, which account for more than 80% of the world's heavy rare earth element (HREE) production supply (Dushyantha et al. 2020; Estrade et al. 2019; Huang et al. 2015; Kynicky et al. 2012; Wall 2013). They are formed

by in-situ weathering of host rocks (granitic or igneous) in a humid and warm climate, resulting in the development of kaolin, halloysite, and montmorillonite clays, which can adsorb REE ions (Huang et al. 2015; Moldoveanu and Papangelakis 2012; Xiao et al. 2016). Ionic REE mining techniques have been dominated by hydrometallurgy techniques since their discovery in Jiangxi Province, China, in 1969, which applies leachate (eg. $(NH_4)_2SO_4$) to extract physisorbed REE^{3+} ions from the clay surface (Feng et al. 2021; Huang et al. 2015; Li et al. 2019). The extraction following the reaction mechanism, $Clay\text{-}REE^{3+} + 3NH_4^+ \rightarrow Clay\text{-}(NH_4^+)_3 + REE^{3+}$. REE mining has resulted in severe soil erosion, permanent ecosystem and biodiversity loss, and air pollution in mining areas because of the unique combination of ionic REE deposits and mining techniques (Liu et al. 2017). For instance, in Ganzhou, Jiangxi province, one REE resource center, mining has created 191 million tons of tailings, destroyed 153 km^2 of forests, and abandoned 302 untreated mines (Yang et al. 2013; Liu et al. 2015). Although tank and heap leaching has been replaced by the less environmentally damaging technique of in-situ leaching for ion-adsorption REE mining (Huang et al. 2015; Liu et al.

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2019), residual ecological hazards by heap leaching, such as deforestation and soil deconstruction, remain distinct.

The environmental issues caused by ion-adsorbed rare earth mining have drawn increasing attention; however, there is currently very little research on ion-type rare earth mine tailings that are related, and those that do exist are insufficiently thorough and have inconsistent results. For instance, in terms of soil nitrogen content, Wen (2012) stated that the soil alkaline hydrolyzable nitrogen content in the tailings was negligible, whereas Xu et al., (1999) suggested that it was larger than that of the surrounding normal vegetation. Additionally, there haven't been many studies done on the biological aspects of tailings sites, such as the quality of the vegetation and soil microbes. The heap leaching technique (Fig. 1) completely strips the topsoil in the mining area, causing physical problems with the territory's geography, such as landslides and the loss of vegetation cover, resulting in soil erosion, crop production damage, and biodiversity loss (Barnes 2020; Liang et al. 2014; Yang et al. 2013). Experimental studies in these areas suggested that the soil fertility and physiochemical properties in mine tailing remained very poor even after years of vegetation restoration (Zhou et al. 2015; Liu et al. 2015). Additionally, the majority of ion-adsorption REE mines are located on fragile granite hills which makes ecological recovery more challenging (Estrade et al. 2019; Liang et al. 2014; Ou et al. 2020; Wall 2013; Wen et al. 2013).

The significance of REE in a variety of contemporary applications demonstrates the rising need for the green economic in future (Balaram 2019; Barnes 2020; Charalampides et al. 2015). Thus, there is a need to comprehensively study the effects of REE mining technologies on the environment in mining areas, which will provide scientific knowledge on the future REE exploitation and ecological conservation in mining areas. This study aims to evaluate
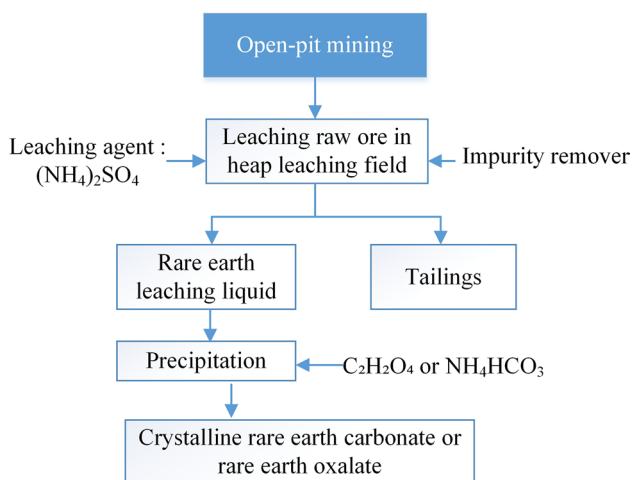
the soil physiochemical properties and vegetation recovery at Changting County, Fujian, which is one of the REE supplying centers (Kynicky et al. 2012). It was carried out in 2017, through investigation of several parameters we aim to provide the following information: 1) the effects of REE leap leaching mining on topsoil pH and texture; 2) an analysis of total organic carbon (TOC), total nitrogen (TN), total phosphorus (TP), and total potassium (TK) changes in topsoil; and 3) four indicators (the Shannon diversity index (H), the Pielou evenness index (E), the Simpson dominance index (D), and the Margalef Richness Index (R)) (Margalef 1951; Pielou 1966; Shannon and Weaver 1949; Simpson 1949)) were calculated to evaluate the effects of restoration in the vegetation community and diversity.

Materials and methods

Sampling sites

The sampling locations are in Changting County (25°18′–26°02′N, 116°01′–116°39′E), Fujian, China, which has a district area of 3099.59 km² and a mid-subtropical humid monsoon climate (Fig. 2). According to the Changting County People's Government Network, the annual average temperature is 18.3 °C, and there is 1700 mm of rain on average per year. The terrain consists of hilly regions with streams and an advanced water system. The soil types are laterite and degraded laterite, which are both capable of retaining little moisture and fertility. They also have weak erosion resistance (Chen et al. 2019a, b). Secondary subtropical evergreen broad-leaved forests make up the majority of the vegetation since anthropogenic activities like farming and REE mining have devastated the traditional forest.

Heap leaching has been mainly applied in Changting County for REE exploration since the 1970s. After 2008, it was prohibited in Fujian province, but mine tailings remained in those areas (Chen et al. 2019a, b). However, heap leaching contaminated spots remain in the mine tailings in this area. In this study, five sampling sites in Changting County were chosen to investigate the environmental impacts of the heap leaching of REE. Four REE mining sites (S1: Niushitang, S2: Xiakeng, S3: Santongbazhou, S4: Huangnikeng) with different management years (8 years, 6 years, 3 years, 0 years) and an unmined control site (CK: Longjing). The management year represents the period of the artificial vegetation restoration, from the start of the restoration time till the sampling time.

Soil sampling and analysis

We set up three subsample points at each sampling site and used an S-shaped point placement and multi-point

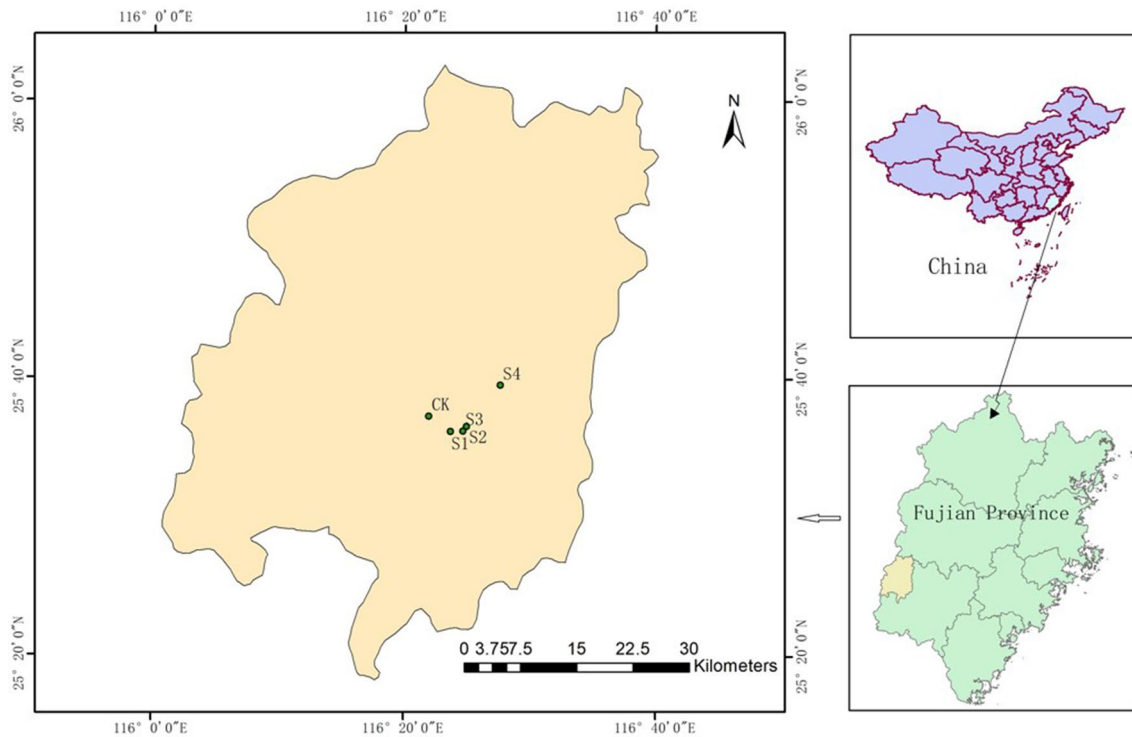**Fig. 1** Process flow of heap leaching

Fig. 2 Location of the sampling site

sampling method to collect soil from all sampling sites. Because there is still a lack of vegetation cover in heap leaching mined areas, some zones account for only 5–15% of the total. Herbaceous recovery is the primary step and dominant in the topsoil layer. Therefore, five soil samples from each point were collected from 0–10 cm and 10–20 cm in all sampling plots. Each soil sample was carefully placed into a plastic bag and delivered to our laboratory.

All the soil samples were placed in a cool and ventilated place in the laboratory and air dried naturally. After removing impurities, such as rocks and roots, the soil samples were first grounded in a mortar, then passed through a 100-mesh (0.149 mm) sieve. All the soil parameter determinations followed the protocol of the agricultural soil chemical analysis method proposed by Lu (2000). The methods are as follows: The pH value was measured by potentiometers after extraction by soil–water suspension (Soil: Water = 1:2.5); the soil texture was evaluated by a Malvern particle size analyzer (Masterrizer 2000, UK); TOC and TN were investigated by a carbon and nitrogen element analyzer (Elementar Vario MAX, Germany); TP and TK were extracted by the $HClO_4$–H_2SO_4 and HF-$HClO_4$ method, determined by a Flow Analyzer (SKalar san++) and a flame photometer (FP640), respectively.

Plant survey and vegetation parameters

We applied a sample survey method to investigate the vegetation diversity in the sampling areas. The landscape map of the sampling sites is listed in the supplementary materials Fig. S1. Three subsample plots with an area of 20×20 m were set up on each site. Three 5×5 m areas for shrub plots along the diagonal line of each sample plot were selected. Within each shrub plot, two 2×2 m herb plots along the diagonal line were examined. The number of plants, their species name, family name, height, height under branches, ground diameter, crown width, and coverage were noted for various tree and shrub species. (those with a diameter ≥ 4.5 cm at 1.3 m height were recorded as a tree). While for the herbaceous, we documented their species name, family name, height, sub-coverage, and number of clumps. To evaluate the stability and diversity of their vegetative systems, biodiversity metrics were determined.

Biodiversity usually has four aspects: genetic diversity, species diversity, ecosystem diversity, and landscape diversity. The calculated biodiversity indices are listed as follows: the Shannon diversity index (H), the Pielou evenness index (E), the Simpson dominance index (D), and the Margalef richness index (R) (Margalef 1951; Pielou 1966; Shannon and Weaver 1949; Simpson 1949).

The calculation formulas as follow:

The Shannon diversity index (H)

$$H = -\sum_{i=1}^{s} P_i \ln P_i \tag{1}$$

Among the species, the more even the distribution, the greater the H value. When all individuals are different species, the diversity index is the largest; when all individuals belong to the same species, the diversity index is the smallest.

Pielou evenness index (E)

$$E = \frac{H}{\ln S} \tag{2}$$

E varies between 0 and 1. The lower the E value, the less evenness in a community between species.

Simpson dominance index (D)

$$D = -\sum_{i=1}^{S} \frac{N_i(N_i - 1)}{N(N-1)} \tag{3}$$

A large D value indicates that some species are dominant and the diversity is small; on the contrary, a smaller value demonstrates that the number of individuals in each species is uniform and there is large plant diversity.

Margalef richness Index (R)

$$R = \frac{S-1}{\ln N} \tag{4}$$

R value increases as the number of species increases.

In the above formulas $P_i = N_i/N P_i = N_i/N P_i$ represents the proportion of species i of the total individuals, which is the relative abundance of the i species. N is the observed total number of individuals, N_i is the number of individuals in the plot i, and S is the number of species observed in the sample.

Statistical analysis

All experimental data were analyzed using Microsoft Excel and IBM SPSS 25.0. The dataset was firstly processed by the normality test. The normality of various variables was accepted by the Shapiro–Wilk or Kolmogorov–Smirnov test ($p > 0.05$). The homogeneity of variances was tested by the Levene test. One-way analysis of variance (One-Way ANOVA) with the least significant difference (LSD) was used to test the differences in TOC, TN, TP, and TK. Follow up with the Pearson correlation analysis method for correlation analysis.

Results and discussion

Soil physiochemical properties

The soil pH and texture

The pH value and texture are decisive properties of soil that impact the availability of nutrients and can indicate the soil quality (Bronick and Lal 2005; Li et al. 2020; Zhang et al. 2020a, b). The soil pH is related to the parent material and stays stable under natural conditions. It fluctuates mainly due to external drivers, such as farming and resource extraction (Li et al. 2020). In our study, the soil pH in all areas varied between 4.31 and 5.22 (Table 1). The soil samples from mined areas all showed acidity, and visible piles suggest that the heap leaching process significantly increased soil acidity, which is also stated

Table 1 Soil pH and texture by heap leaching

Sampling site	Restoration period/Year	Sampling depth/cm	pH	Particle size distribution/%		
				Sand	Silt	Clay
S1	8	0–10	4.64 ± 0.15	51.44 ± 5.19aA	39.13 ± 3.43aB	9.43 ± 1.99aC
		10–20	5.22 ± 0.17	51.10 ± 12.02aA	39.15 ± 8.31aB	9.75 ± 3.73aB
S2	6	0–10	4.35 ± 0.12	44.97 ± 10.59bB	44.89 ± 8.72aB	10.13 ± 2.03aC
		10–20	4.44 ± 0.32	52.83 ± 3.83aA	38.64 ± 3.59aB	8.53 ± 0.26aB
S3	3	0–10	4.72 ± 0.03	53.61 ± 4.40aA	38.42 ± 3.52aB	8.05 ± 1.24aC
		10–20	4.58 ± 0.15	54.42 ± 5.52aA	37.36 ± 4.92aB	8.23 ± 0.74aB
S4	0 (abandoned)	0–10	4.31 ± 0.01	53.63 ± 2.60aA	27.35 ± 3.74aC	19.02 ± 1.14aA
		10–20	4.42 ± 0.01	54.54 ± 1.29aA	26.78 ± 2.81aC	18.58 ± 4.11aA
CK	0 (unmined)	0–10	4.31 ± 0.14	23.93 ± 7.25aC	61.07 ± 7.45aA	15.01 ± 2.04bB
		10–20	4.60 ± 0.15	20.56 ± 6.42aB	60.11 ± 3.86aA	19.63 ± 2.68aA

S1, Niushitang; S2, Xiakeng; S3, Santongbazhou; S4, Huangnikeng; CK, Longjing

Low case letters indicate a significant difference between sampling depths in the same sampling site

Capital letters indicate a significant difference between sampling sites at the same depth ($P < 0.05$)

by other studies (Liang et al. 2014; Wei et al. 2001; Yu et al. 2009). Only the pH of the 10–20 cm of soil at S1 was greater than 5, this might due to the prolonged period of vegetation restoration. In contrast, S4, where restoration was absent, displayed the highest acidity in both soil layers, with pH readings that were less than 4.5. This corresponds to Zhang et al., (2018) and Zhang et al., (2020a, b)'s study. With increasing years of restoration, the soil pH changes from strong acidity to slight acidity. The 0–10 cm soil pH of CK was lower than 4.5, which may be a result of the acidity of the subtropical red soil itself and the growth of the native dominant fern *Dicranopteris pedata*. Because its root system concentrates on the surface layer and produces organic acid, which can cause the soil pH to decrease (Chen et al. 2016). Furthermore, ammonia leachate accelerates soil acidification (Zhang et al. 2008, 2020a, b). The soil remains acidic after phytoremediation (Liu et al. 2020a, b). Therefore, mining activities by heap leaching increase soil acidity.

There were no significant differences in soil texture at the same site between 0–10 cm and 10–20 cm at all locations, except for the sand content in S2 and the clay content in the CK. This implies that heap leaching influences on 0–20 cm of topsoil are similar. While in the comparison between sample plots, soil texture varied significantly in the leap leaching sites compared to the CK. The sand content in mining areas was significantly higher than that in the CK ($P < 0.05$), 2 times higher, and the silt and clay content were significantly reduced in the mined sites. The higher sand content in all mined areas demonstrates that heap leaching mining has resulted in soil destruction, which has not improved after years of restoration. Liu et al., (2015) also conclude that soil structures did not improve in the long term after mining. Compared to S4, the restoration sites increased in silt content. It is mainly

because the growth of plants improves the soil structure (Liu et al. 2015; Zhang et al. 2018).

Soil nutrient characteristics

Soil nutrients determine various soil functions and play an important role in the formation of soil structure, nutrient release, and microbial activities (Brevik et al. 2015; Bronick and Lal 2005). TOC, TN, TP, and TK are important parameters of soil fertility, which are also key nutrient elements for plant growth (Liu et al. 2020a, b; Ma et al., 2013). The contents of TOC, TN, TP, and TK after heap leaching are shown in Table 2. The TOC (15.7 g•kg^{-1}) at the CK's 0–10 cm soil was not significantly different from the Fujian Province red soil background value (22.1 g•kg^{-1}) (Chen et al. 1992). There is no significant difference between soil layers at the same sampling site. The TOC (0.69–15.7 g•kg^{-1}) was significantly lower in the sampling sites than in the CK at different depths ($P < 0.05$), although they improved after restoration. The highest content was found at 0–10 cm at S1 (3.76 g•kg^{-1}) and the lowest at S2, they are 4 times and 17.84 times lower than at the CK, respectively. Because fertilizer was administered earlier to promote the growth of the planted vegetation, TOC was higher at S3 than at S2. However, TOC reduced as a result of the plantations disappearing in the harsh environments of the mining sites. Then, as some indigenous fern species, such as *D. pedata*, slowly encroached on the area, the TOC increased. This demonstrates that the recovery of soil organic carbon content in mining areas is a slow process. The result is similar to Zhang's study (2018). Their study also showed that the TOC of 10–20 cm soil was still low (about 5 g•kg^{-1}) after 30 years of treatment.

The content of TN ranged from 0.21 to 1.21 g•kg^{-1}. In the 10–20 cm soil of S1 and S4 were significantly lower

Table 2 The features of soil carbon, nitrogen and phosphorus contents

Sampling site	Restoration period/Year	Sampling depth/cm	Organic carbon/(g·kg^{-1})	Total nitrogen/(g·kg^{-1})	Total phosphorus/(g·kg^{-1})	Total potassium/(g·kg^{-1})
S1	8	0–10	3.76 ± 2.17aB	0.53 ± 0.29aA	0.06 ± 0.02aB	9.37 ± 1.66aA
		10–20	0.70 ± 0.49aB	0.26 ± 0.12aB	0.05 ± 0.01aB	8.95 ± 4.26aA
S2	6	0–10	0.88 ± 0.19aB	0.33 ± 0.11aA	0.08 ± 0.02aB	9.53 ± 1.16aA
		10–20	0.69 ± 0.24aB	0.32 ± 0.03aA	0.07 ± 0.02aA	7.84 ± 0.32aA
S3	3	0–10	3.17 ± 1.93aB	0.93 ± 0.64aA	0.07 ± 0.01aB	8.42 ± 1.50aA
		10–20	1.02 ± 0.74aB	1.21 ± 0.95aA	0.06 ± 0.01aB	7.90 ± 0.18aA
S4	0 (abandoned)	0–10	15.70 ± 6.92aA	1.01 ± 0.40aA	0.11 ± 0.02aA	6.37 ± 4.14aA
		10–20	4.19 ± 0.14aA	0.37 ± 0.18aA	0.08 ± 0.02aA	6.79 ± 3.65aA
CK	0 (unmined)	0–10	0.89 ± 0.08aB	0.21 ± 0.01aB	0.05 ± 0.01aB	4.22 ± 0.16aB
		10–20	0.72 ± 0.03aB	0.21 ± 0.01aB	0.05 ± 0.01aB	4.49 ± 0.14aB

Low case letters indicate a significant difference between sampling depths in the same sampling plot. Capital letters indicate a significant difference between sampling plots at the same depth ($p < 0.05$)

than the other three plots ($P < 0.05$). The highest concentration of TN appeared in S3. TN of 10–20 cm soil reached 1.21 g•kg^{-1}, while this was only 0.6 g•kg^{-1} at the CK, which demonstrated the residual leachate $(NH_4)_2SO_4$ after the mining area was abandoned increased TN. It is agreed with (Zhang et al. 2020a, b), the presence of chemical residues leads to the TN content at the mined site higher than the unmined soil. However, the TN was similar at S1, S2, and S4. This indicates that plants are difficult to effectively uptake residual nitrogen from the soil. A further decrease in nitrogen at those sampling plots was because the residual $(NH_4)_2SO_4$ in the soil was easily washed away by precipitation, which resulted in the total nitrogen content at S2 being about 0.3 g•kg^{-1} in both soil layers. Zhang et al. (2020a, b) study proofed that ammonia of surface layer at ionic REE mine tailing was lower than that of the deeper soil.

All sites had extremely low levels of TP, ranging from 0.05–0.11 g•kg^{-1}. This is because soil phosphorus mainly comes from the parent rock. And in southern China, subtropical soils are deficient in phosphorus, with TP of 0–20 cm in the unmined red soil background value is only 0.15 g•kg^{-1} (Soil Surveying Office in Fujian 1991). At the CK site, TP content of 0–10 cm soil was similar to that of uneroded laterite, but at 10–20 cm was also low. It was significantly higher in the CK than in the sampling sites at different depths, except for the 10–20 cm soil layer of S2. TP in the mining areas was all lower than 0.08 g•kg^{-1}. Acidic soil pH has made managing phosphorus for crops a major challenge (Brevik et al. 2015). The lack of phosphorus limited the growth of soil microorganisms, which made it difficult to improve the soil properties in the mining area and further restricted plant growth (Hou et al. 2015). The changing trend of TP in the four mining areas increased first and then decreased with the extension of the rehabilitation time. Because plants absorbed a certain amount of phosphorus at the beginning of the restoration, TP in the S3 soil was relatively low. With the death of plants, phosphorus was released into the soil, so TP increased at S2. Then it decreased as new plants invaded and absorbed phosphorus.

The content of TK varied from 4.22–9.37 g•kg^{-1}. S4 was significantly lower than that of other sites ($P < 0.05$) at both 0–10 cm and 10–20 cm. At the S1, S2, and S3, TK at 0–10 cm was higher than at 10–20 cm, whereas the CK and S4 showed the opposite trend. TK contents at the mined sites were all less than 10 g•kg^{-1} with the lowest at the S4 of both layers, indicating the heap leaching mining method caused potassium loss. While no significant difference between S1, S2, S3, and CK showed that restoration increases the TK content.

Overall, the mined sites had lower TOC, TN, TP, and TK values than the CK. While compared to the S4, the soil nutrients gradually improved with time of restoration. Although all of the contents were lower than at the CK,

S1 shares a similar profile as the CK. Zhang et al., (2018) also suggest that TOC, TN, TP and TK were significantly increased after 5 years of restoration. This is also proved by Ma et al., (2013), who also discovered that TOC, TN, TP, and TK were increased 408.9%, 391.2%, 240.6%, and 330.5%, after 10 years of restoration, respectively. Because the soil in this region also has a low phosphorus content, the potassium content has increased significantly with the restoration. As a result, organic carbon and nitrogen are the restricted parameters of vegetation hardly succession in the mine tailings.

Correlation between total organic carbon and total nitrogen

We further compared the correlation between the TOC and total TN, their equating relationship is shown in Fig. 3. The correlation between TOC and TN after heap leaching was very weak ($R^2 = 0.213$), which is significant different as most types of soil, R^2 reaches 0.95 at a significant level of 0.01 (Bai et al. 2020). Additionally, the C:N values of S1, S2, S3, S4, and CK are 7.02, 2.74, 2.17, 3.51, 4.17, 15.42 (0–10 cm) and 3.02, 2.17, 0.89, 3.43, 11.04 (10–20 cm), respectively. The C:N of control site is within the range of Fujian soil (11.7–18.2), while all the mined sited are lower. Mining activities destroyed the balancing of carbon and nitrogen nutrients. The lack of TOC and additional ammonia nitrogen at the early stage after mining aggravated the unbalancing. Currently, $(NH4)_2SO_4$ is used in the ion-adsorption REE exploitation at concentrations ranging from 0.5 to 2% and up to 5% (Qiu et al. 2014; Tang et al. 2018; Zhang et al. 2020a,
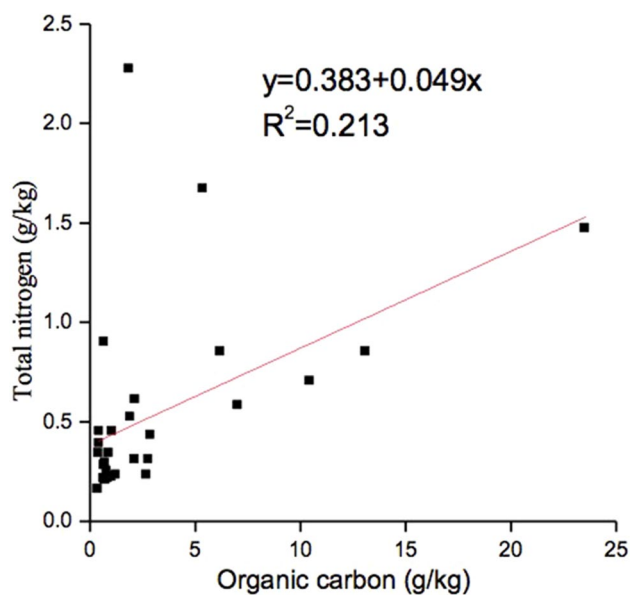
Fig. 3 Relationship between soil organic carbon and total nitrogen

b). To produce a ton of raw REE requires 7–10 tons of $(NH_4)_2SO_4$ and the leachate residues, NH_4^+ and SO_4^{2-}, contaminate the downstream, which can reach 3500–4000 mg/L after REE precipitation in surrounding rivers (Åström, 2001; Gao and Zhou 2011; Xiao et al. 2015; Zhang et al. 2020a, b). Furthermore, the presence of ammonium nitrogen can also easily cause soil mineral depletion because it changes the ion composition and strength of the soil solution, which affects metal ion fixation and release, thus intensifying the acidification process (Zhou et al., 2014). As a result, we recommend that carbon fertilizer be applied early in the restoration process to aid plant establishment. With the residual $(NH_4)_2SO_4$ in the soil washed away by precipitation and moved to the deeper soil layer, nitrogen became the limited nutrient for plant growth. Therefore, soil tumbling will improve the utilization of residual nitrogen.

Plant diversity

The vegetation at all sampling sites are shown in Table 3. The dominant species included *Camellia oleifera, Dicranopteris pedata, Eulalia speciosa, Lespedeza bicolor, Liquidambar formosana, Miscanthus floridulus, Paspalum wetsfeteini, Pinus massoniana,* and *Schima superba.* The vegetation cover rate in each site is 97%, 90%, 50%, 25%, and 100%, respectively.

Table 4 shows that, except for CK, none of the mine tailings grew trees. The vegetation was devastatingly destroyed after heap leaching mining. No trees have grown in any of the sampling sites despite the manual vegetation restoration. Although there were trees, such as *P. massoniana,* that appeared in some areas, they did not meet the criteria for trees and can only be registered as shrubs. Therefore, we can conclude that it takes a long time to grow into trees in REE heap leaching tailings. The result corresponds to Li et al.'s (2020) study. They used remote sensing monitoring to determine sequential images of land damage and restoration in six countries in southern Jiangxi province. The land destruction remained extensive, with 38.10 km^2 of vegetation loss after two decades of restoration.

The diversity index showed that the herb layer decreased first and then increased. This is because the planted herbaceous *P. wettsteinii* increased the surface cover at the early stage of restoration, then gradually died, but the wild native *D. pedata* slowly took over the *P. wettsteinii* and became the dominant species. In terms of the shrub layer, it slightly increased with the increasing time of management. It is because, at the restoration sites, the planted *S. superba* was similar at different sites. Then it showed an increased rate with the increase in restoration years because of the growth of young *P. massoniana.* The evenness index showed a similar change to the diversity index, with the shrub layer showing an increasing trend and the herb layer decreasing first

Table 3 The location and dominant species of sampling sites

Sampling site	S1	S2	S3	S4	CK
Restoration year	8	6	3	0 (abandoned)	0 (unmined)
Longitude	116°26′08″E	116°24′41″E	116°24′42″E	116°16′52″E	116°27′37″E
Latitude	25°36′52″N	25°36′29″N	25°36′13″N	25°31′49″N	25°39′32″N
Dominant species	*Dicranopteris pedata, Schima superba, Lespedeza bicolor, Liquidambar formosana, Eulalia speciosa*	*Dicranopteris pedata, Lespedeza bicolor, Liquidambar formosana, Paspalum wetsfeteini, Schima superba*	*Dicranopteris pedata, Paspalum wetsfeteini, Lespedeza bicolor, Liquidambar formosana, Camellia oleifera*	*Dicranopteris pedata, Pinus massoniana, Lespedeza bicolor, Liquidambar formosana, Miscanthus floridulus*	*Dicranopteris pedata, Pinus massoniana, Schima superba, Miscanthus floridulus*

Dominant species are ordered according to the proportion of species

Table 4 Diversity of plant community after heap leaching

Sampling site	Restoration period/Year	Shannon diversity index (E)			Pielou evenness index (E)			Simpson dominance index (D)			Margalef richness index (R)		
		Herb	Shrub	Tree	Herb	Shrub	Tree	Herb	Shrub	Tree	Herb	Shrub	Tree
S1	8	1.09	1.49	–	0.61	0.93	–	0.49	0.20	–	0.92	1.38	–
S2	6	0.83	1.43	–	0.51	0.89	–	0.55	0.21	–	0.71	1.52	–
S3	3	1.17	1.36	–	0.85	0.84	–	0.36	0.24	–	0.55	1.74	–
S4	0 (abandoned)	0.67	0.46	–	0.96	0.42	–	0.50	0.75	–	0.35	0.55	–
CK	0 (unmined)	0.47	0.89	0.27	0.34	0.64	0.39	0.77	0.46	0.86	0.55	0.77	0.19

and then increasing, which further proves that the restoration first reflected the changes in herbaceous species. Zhang et al., (2018) also suggested that herb diversity and evenness decreased at the early stage of restoration, and then increased after 6 years of management. The plant communities overall tend to be stable after 5–7 years of restoration. The dominance index showed that in the three management areas there were no obvious dominant shrub species. Whereas the herb layer showed a trend of growing first, then decreasing because the dominance of *D. pedata* species became obvious with time. The highest dominance index in the untreated site also suggested the shrub species *P. massoniana* and herbaceous species *D. pedata's* important in natural restoration. The richness index shows more obvious changes, with an increasing trend in the herbaceous layer and a decreasing trend in the shrub layer as the management years increase.

In summary, revegetation can restore the previous vegetation in a degraded ecosystem, enhance soil fertility and structure, encourage the recovery of animal and microbial species in impacted areas, and eventually make it easier to recover and rebuild the structure and function of the entire ecosystem (Liu et al. 2017). After years of rehabilitation, the plant community was not in a good diversity stage. The number of species was small because manually planted species mostly died out. Only a few dominant species, such as *P. massoniana* in shrub form and herbaceous species like *D. pedata* can adapt to the extreme environment in the mine tailing. Thus, the improvement of restoration is a critical factor in the rehabilitation of the mine tailing's ecological environment, and the use of soil amendments is an effective way of addressing this issue (Zhang et al. 2020a, b). Luo et al., (2018) explored the application of biochar amendment in the rehabilitation of the ion-adsorption mine tailings, results revealed that alfalfa revegetation and sludge biochar improved soil physiochemical properties. (Zhang et al. 2020a, b) stated that applying bamboo biochar and coal ash significantly increased soil pH, organic carbon, and cation exchange capacity. On the other hand, the phytoextraction potentials of hyperaccumulating plants (eg. *D. pedata* and *Phytolacca Americana*) in the mine tailings have also been discussed (Chour et al. 2018; Liu et al. 2020a, b; Qin et al. 2019; Yuan et al. 2018). However, systematically restoration in a large scale is still rare. Additional concerns should be addressed in the environmental management of ion-adsorption REE mine tailings.

Conclusion

Ion-adsorption REE are unique resources that restrict the world's supply of HREE. The increasing demand for REE intensifies mining activities which will lead to environmental degradation in the mining area. Mining activities further lead to vegetation loss, soil structural destruction, and nutrient loss, resulting in a fragile ecosystem even after years of restoration. This paper systematically analyzed the soil properties and vegetation recovery in different heap leaching mine tailings, which have been neglected. The results showed that even after eight years of soil nutrient improvement, no tree had been established. Therefore, further studies on the heap leaching mine tailing are still needed. Nevertheless, mining activities also cause REE pollution, and ammonia leachate [$(NH_4)_2SO_4$] that might lead to severe ecological pollution over the long term and is potentially harmful to the surrounding biota. In order to ensure the sustainability of ion-adsorption mining, research should concentrate on the geochemical behaviors of ammonia leachate and REE's ecological consequences on the local population.

Data availability All data generated or analyzed during this study are included in this article.

Declarations

Conflict of interest The author declare that they have no conflicts of interest.

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