Environmental Earth Sciences (2019) 78:653
https://doi.org/10.1007/s12665-019-8671-8

ORIGINAL ARTICLE

Effects of restoration age on water conservation function and soil fertility quality of restored woodlands in phosphate mined-out areas

Yong Yuan[1,2,3] · Xiaoying Li[4] · Donghong Xiong[1,2] · Han Wu[1,2,3] · Su Zhang[1,2,3] · Lin Liu[1,2,3] · Wanxin Li[1,2,3]

Received: 29 September 2018 / Accepted: 9 November 2019 / Published online: 19 November 2019

Abstract
The age of vegetation restoration has significant influences on near-surface hydrological function and soil quality in mining areas. This study analyzed the effects of restoration age and tree species on water conservation function (reflected by the water-holding characteristics of litter and soil) and soil fertility quality [reflected by soil bulk density (BD), soil organic carbon (SOC), total N (TN), available N (AN), available P (AP), and available K (AK)] of restored woodlands in phosphate mined-out areas. A primary forest (the control) and six woodlands restored for 4–25 years were selected as test sites. The results showed that total litter storage capacity, litter water-holding capacity, soil effective water-holding capacity, SOC, TN, AN, AP, and the comprehensive soil fertility index (SFI) increased with the restoration age, while small changes were also observed in soil total water-holding capacity, BD, and AK with restoration age. Compared to the control, the litter-modified interception amount, soil total water-holding capacity, and SFI in woodlands restored for 25 years were restored on average by 78%, 77%, and 92%, respectively. Furthermore, the litter water-holding capacities, soil capillary water-holding capacity, and AN in the *Eucalyptus robusta* woodlands restored for 10 years were significantly lower than those in woodlands of *Alnus nepalensis* restored for < 10 years. Our findings highlight that the restoration age has positive impacts on improving the water conservation function and soil fertility quality of restored woodlands, and the influence of tree species on the water conservation function should be specifically considered while carrying out vegetation restoration in phosphate mined-out areas.

Keywords Vegetation restoration · Leaf litter · Water-holding capacity · Soil fertility index · Mined-out area

Introduction

Phosphate mining has been greatly expanding in recent decades in response to the increasing demand for chemical fertilizer. However, large-scale strip-mining operations greatly disturb soil, vegetation, and landscape elements (Shackelford et al. 2018; Yan et al. 2013). This results in serious ecological and environmental consequences as well as health problems (Li 2006; Ye et al. 2009). In recent years, there have been increasing measures towards mitigating, reducing, and offsetting these influences. Among them, vegetation restoration is one of the most effective measures to restore degraded environments and improve the crucial ecosystem processes in mining wastelands (Han et al. 2017;

✉ Donghong Xiong
dhxiong@imde.ac.cn

Yong Yuan
yuanyong@imde.ac.cn

Xiaoying Li
Lxying92@126.com

Han Wu
wuhan@imde.ac.cn

Su Zhang
zhangsu211@foxmail.com

Lin Liu
457259179@qq.com

Wanxin Li
2770786185@qq.com

1 Institute of Mountain Hazards and Environment, Chinese Academy of Sciences and Ministry of Water Resources, Chengdu 610041, China

2 Key Laboratory of Mountain Hazards and Earth Surface Processes, Chinese Academy of Sciences, Chengdu 610041, China

3 University of Chinese Academy of Sciences, Beijing 100049, China

4 College of Ecology and Soil and Water Conservation, Southwest Forestry University, Kunming 650224, China

Wang et al. 2018a, b, c; Zhao et al. 2015). This is because it can transform mining wastelands into woodlands or grasslands; consequently, the ecological landscape and the near soil surface characteristics (plant litters and soil properties) change significantly (Li et al. 2015).

Vegetation restoration causes changes in the characteristics of litter and soil that are closely related to the water conservation function of woodlands. The water conservation function of forest ecosystems is mainly carried out by the canopy layer, litter layer, and soil layer (Bruijnzeel 2004; Edwards et al. 2014). However, the litter layer and soil layer hold more than 80% of the total water conservation capacity of woodlands (Wu et al. 2014). Therefore, investigating the water-holding capacity of litter and soil in restored woodlands has great importance for understanding their water conservation function. Litters from the above-ground plants can cover the soil surface and thus directly regulate surface hydrological processes by increasing surface roughness, reducing raindrop impact, and promoting water infiltration (Xia et al. 2019; Zagyvai-Kiss et al. 2019). Litters can also be incorporated into the soil ecosystem by soil splash and soil fauna activities (Blouin et al. 2013; Sun et al. 2016), and their decomposition alters BD, promotes the formation of soil aggregates, and increases soil capillary porosity (Moreno-de las Heras 2009; Tonks et al. 2017; Upton et al. 2018; Yang et al. 2018). In addition, plant roots can change soil porosity, especially the dead roots, which significantly affect the formation of soil aggregates (Ghidey and Alberts 1997). Any changes in these aspects might result in changes in soil water-holding characteristics. The litter layer is critical for sustaining the forest hydrological function and has a complex and important role in the effective conversion of precipitation into soil layer (Sun et al. 2018a, b; Zagyvai-Kiss et al. 2019). A few studies from Brantley et al. (2013) and Zhang et al. (2019) indicate that the water-holding capacities of litter and soil were significantly affected by woodland age and plant species. Therefore, if the water-holding characteristics of woodlands after restoration get changes either in litter or soil layer, it would change the water conservation function of restored woodlands.

Vegetation restoration that causes changes in soil properties leads to changes in soil fertility quality on degraded lands (Phan Minh et al. 2013). Soil fertility quality can be estimated using various soil fertility indicators (Karlen et al. 2003), such as BD, C, and N (Raiesi 2017; Zhang et al. 2011). Guo et al. (2018) showed that vegetation restoration had significant impacts on soil properties. For instance, vegetation restoration by afforestation promotes the accumulation of litter, and its decomposition directly releases nutrients to the soil system (Xu et al. 2013), which causes changes in soil C and N levels (Deng et al. 2016; Zhang et al. 2016). Vegetation restoration not only accumulates large amounts of C and N (Baddeley et al. 2017; Fu et al. 2010) but also facilitates the storage of P (Garay et al. 2018). In previous studies (Chodak and Niklinska 2010; Jiang et al. 2011), the restoration of soil fertility quality may be affected by the type of afforestation species, because the tree species were strongly related to different variables of soil characteristics (Liu et al. 2018; Urbanova et al. 2014). For example, leguminous plants can boost soil N content faster than other plants due to their large litter N content (Horodecki and Jagodzinski 2017; Mueller et al. 2012), and pure coniferous stands (larch, pine) accumulated less C and N than broadleaved species (Chodak and Niklinska 2010). Another study from Zhong et al. (2017) showed that the restoration age significantly influenced the improvement in soil quality. Alday et al. (2012) suggested that age since restoration was the driving agent of soil changes during the restoration of a coal mine in the short term. Moreover, Guo et al. (2018) found that the improvements in soil quality in the afforested land decreased with restoration age. Overall, the effects of vegetation restoration on soil fertility quality are variable and may depend on both tree species and restoration age (Sun et al. 2018a, 2018b; Zhang et al. 2018). Therefore, a significant spatial and temporal variation likely exists in water conservation function and soil fertility quality due to the variations in restoration age and species of restored woodlands in mining areas.

Most of the current studies on restoration performance of mining wastelands are mostly concentrated on metalliferous mining lands (Shackelford et al. 2018; Wang et al. 2018a, b, c; Wong 2003). However, there are few reports on nonmetalliferous mining lands (Li 2006; Singh and Singh 2001), especially on phosphate mining lands (Ruthrof et al. 2018). He et al. (2013) have studied the effects of vegetation restoration on soil properties in phosphate mined-out areas, but they only focused on the initial stage of restoration (restoration age < 10 years). No other study has been examined on all of the characteristics of water holding and fertility, particularly in a longer restoration age scale (> 10 years). Hence, evaluating the response of water conservation function and soil fertility quality to age since vegetation restoration and tree species could provide an important contribution to scientific ecological restoration in mining areas.

By measuring these potentially interacting variables in relation to the function of water and fertility conservation in restored woodlands, we sought out (1) to quantify the variation in water conservation function (reflected by water-holding characteristics of the litter and topsoil) with restoration age and tree species, (2) to investigate the variation in soil fertility indicators with restoration age and tree species, and (3) to determine the quantitative changes in soil fertility quality with restoration age and tree species using an integrated comprehensive index in a phosphate mined-out area in southwest China.

Materials and methods

Study area and sampling plots

The study was conducted in Kunyang Phosphate Mine (KPM) (102°33′30″E, 24°43′25″N, 15 km^2, 1900–2400 m elevation), which is southwest of the Dianchi watershed and located in Kunming City, Yunnan Province, China (Fig. 1). The KPM is one of the largest open-pit phosphate mines in China; its construction began in 1965, was put into operation in 1972, and had an annual capacity of 4.6 million tons. Due to long-term exploitation, a large amount of vegetation and topsoil had been stripped, resulting in serious soil erosion, eutrophication in Dianchi Lake because of the inflow of N and P leachate with run-off, and other ecological and environmental problems (Guo et al. 2017). Ecological restoration by afforestation has been implemented in phosphate mining lands since the 1980s. In particular, the implementation of a larger scale vegetation restoration project after 2010 transformed large-scale wastelands into woodlands. Most of the existing vegetation in the mining area is due to afforestation after soil reconstruction. The main afforestation species were *Eucalyptus robusta* in the early stage of ecological restoration. More and more species such as *Alnus nepalensis*, *Cupressus funebris,* and *Cupressus torulosa* have been afforested since 2000. Kunming City has a typical subtropical monsoon climate with annual mean precipitation of 907 mm, which mainly occurs from May to September. The annual temperature ranges from − 5.4 to 33 °C with a mean of 15.7 °C. The soil is dominated by yellow soil, which is called an Argosol in Chinese soil taxonomy.

After a detailed field survey, six restored woodlands were selected as test sites. The restoration ages of woodlands were 4, 5, 6, 7, 10, and 25 years. Among them, woodlands restored for 4, 5, 6, and 7 years were mixed *Alnus nepalensis–Cupressus funebris* forests, while those restored for 10 and 25 years were mixed *Eucalyptus robusta–Cupressus funebris* forests. All of them were restored after the mining platforms were covered with soil (80 cm thick). With natural compaction, erosion, and leaching each year, the current soil layer is relatively thin, approximately 30–50 cm. Ideally, the afforestation species of the selected sites should be similar to each other; however, it was difficult to meet this requirement, because the species planted were different at different times. This variety probably influenced the experimental results. However, the slope and direction (south-facing), soil type (Argosol), mode and density of planting, and early management practices of the selected sites were similar. A primary forest of *Pinus armandii* and *Pinus yunnanensis* was selected as the control site. Detailed information for each test site is listed in Table 1.

Litter and soil sampling

The litter and soil samples were collected in August 2014. Each test site consisted of three 20 × 20 m plots approximately 100 m apart, and three sampling sites were set up along the diagonal in each plot. At each sampling site, the litter sample was collected from an area of 1 × 1 m, three soil core samples were collected using a cylindrical core (100 cm^3 volume) at

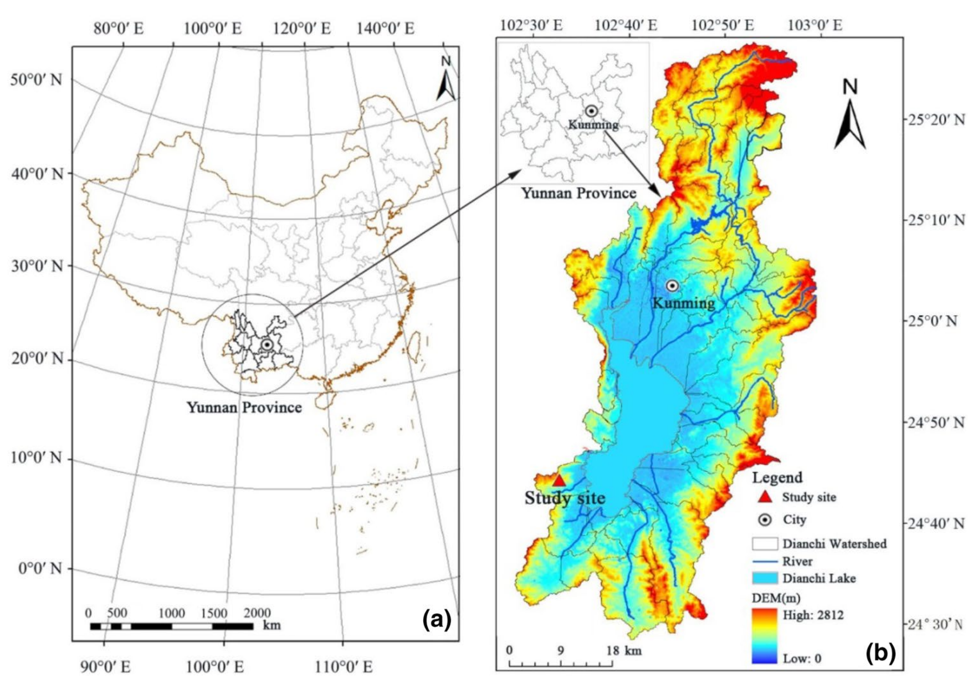

Fig. 1 Location of Yunnan in China **(a)**; location and DEM of the study site (**b**)

Table 1 Basic information for each test site

Site	Elevation (m)	Slope (%)	Vegetation coverage (%)				Litter layer depth (cm)	Afforestation species
			Tree	Shrub	Herb	Total		
RW4	2230	3	25	3	85	93	0.5	*Alnus nepalensis–Cupressus funebris*
RW5	2227	4	52	15	70	82	1.5	*Alnus nepalensis–Cupressus funebris*
RW6	2244	3	45	20	60	85	1.8	*Alnus nepalensis–Cupressus funebris*
RW7	2256	3	73	25	31	85	2.2	*Alnus nepalensis–Cupressus funebris*
RW10	2283	5	75	13	32	80	1.2	*Eucalyptus robusta–Cupressus funebris*
RW25	2220	4	83	8	40	97	4.3	*Eucalyptus robusta–Cupressus funebris*
PF	2200	12	94	22	11	99	4.9	*Pinus armandii–Pinus yunnanensis*

RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest

a depth of 0–20 cm for the measurement of physical properties and always taken more than 50 cm from the rhizosphere of individual plants, and a mixed soil sample (approximately 500 g) was collected at the same depth (0–20 cm) for analyses of chemical properties. All soil samples were collected after the above-ground litters were removed. The field-moist litter samples (M_0) were weighed immediately. The mixed soil samples were air-dried at room temperature and then sieved through a 2 mm mesh for measuring SOC, N, P, and K.

Litter sample analysis

Litter samples were air-dried at room temperature until a constant mass was obtained (14 days). Each air-dried litter sample was weighed, and a subsample (approximately 80 g) of the air-dried leaves was weighed, oven dried (75 °C, 24 h), and then reweighed to obtain a conversion factor between air dry weight and oven dry weight. The oven dried subsample was placed in 20×40 cm litterbags made from 2 mm nylon mesh and then immersed in water, to determine the mass of litter after absorbing water at 0.1, 0.5, 1, 2, 4, 6, 10, and 24 h. Thus, the water-holding capacity and water-holding rate of litter samples in these time periods could be calculated using Eqs. 1–8:

$$R_0 = (M_0 - M_\mathrm{d})/M_\mathrm{d} \times 100\%, \quad (1)$$

$$R_\mathrm{m} = (M_{24} - M_\mathrm{d})/M_\mathrm{d} \times 100\%, \quad (2)$$

$$R_\mathrm{sv} = 0.85R_\mathrm{m} - R_0, \quad (3)$$

$$R_\mathrm{msv} = R_\mathrm{m} - R_0, \quad (4)$$

$$W_\mathrm{m} = M \times R_\mathrm{m}, \quad (5)$$

$$W_\mathrm{sv} = M \times R_\mathrm{sv}, \quad (6)$$

$$W_\mathrm{msv} = M \times R_\mathrm{msv}, \quad (7)$$

$$V_t = (M_t - M_d)/t, \quad (8)$$

where R_0, R_m, R_sv, and R_msv are the natural moisture rate (%), maximum water-holding rate (%), modified interception rate (%), and maximum interception rate (%) of the litter, respectively; M, M_0, M_d, and M_{24} are the total litter storage capacity (t/hm^2), the initial mass of the litter sample in its natural state (g), the mass of the oven dried litter sample (g), and the mass of litter after soaking for 24 h (g), respectively; W_m, W_sv, and W_msv are the maximum water-holding capacity (t/hm^2), modified interception amount (t/hm^2), and maximum interception amount of the litter layer, respectively; and V_t and M_t are the water-absorption rate (t/(hm^2·h)) and the mass (t/hm^2) of litter when soaking for t h. In particular, when the precipitation reaches 20–30 mm, the actual water-holding rate is approximately 85% of the maximum water-holding rate regardless of the water content of the litter layer of any vegetation type. Therefore, to get closer to the actual storage of precipitation, the adjustment coefficient of 0.85 was used to estimate the effective storage rate (modified interception rate) of the litter layer (Wu et al. 2014).

Soil physicochemical analysis

BD and porosity were measured with a cutting ring method (Hossain et al. 2015). Equations 9–11 were used to calculate the soil effective water-holding capacity (W_nc, t/hm^2), the soil capillary water-holding capacity (W_c, t/hm^2), and the soil total water-holding capacity (W_t, t/hm^2):

$$W_\mathrm{nc} = 10000P_\mathrm{nc}h, \quad (9)$$

$$W_\mathrm{c} = 10000P_\mathrm{c}h, \quad (10)$$

$$W_t = W_{nc} + W_c, \quad (11)$$

where P_nc, P_c, and h are the soil noncapillary porosity (%), the soil capillary porosity (%), and the thickness of the soil layer (m), respectively.

SOC was determined by the dichromate oxidation method (Nelson and Sommers 1996). TN was extracted using the Kjeldahl method (Bremner 1996). AN was analyzed by the Alkali diffusion method (Guo et al. 2019). AK was measured in 1 M NH_4OAc extracts by flame photometry (Hou et al. 2019). AP was assayed by the Olsen method (Olsen and Sommers 1982). Each sample was measured with three replicates.

Statistical analysis

All data processing and statistical analysis were performed using Excel 2010 and SPSS 17.0 statistical software. One-way analyses of variances (ANOVAs) were used to evaluate the effect of vegetation restoration age and afforestation species on water conservation function and soil fertility quality. Significance was evaluated at $P < 0.05$ using Duncan multiple range test. Pearson correlation analysis was used to assess the relationships between soil fertility indicators and the soil fertility index. All figures were created using Sigma Plot 10.0 software.

To assess soil fertility quality, a comprehensive soil fertility index (SFI) was produced by a weighted summation method:

$$\mathrm{SFI} = \sum_{i=1}^{n} N_i W_i, \tag{12}$$

where N_i and W_i are the score and the weight of a given soil fertility indicator; n is the number of soil fertility indicators (in this study, it was 6); W_i is the significance of a soil fertility indicator; and N_iW_i is the contribution of the indicator to soil fertility (Sun et al. 2003). BD, SOC, TN, AN, AP, and AK were selected as soil fertility indicators.

The scores of soil fertility indicators are calculated by membership functions related to the indicators of soil fertility. Both ‘S’ (Fig. 2a) and reverse ‘S’ (Fig. 2c) membership functions were adopted in the present study. Soil fertility indicators of PF and RW4 were used as the upper or lower critical values in this study (Table 2).

SOC, TN, AN, AP, and AK are positively correlated with soil fertility quality within a certain range in the ‘S’ curve membership function. An ‘ascending half trapezoid’ (Fig. 2b) can be used to express the ‘S’ curve with the membership function as described by Eq. 13. BD is negatively correlated with soil fertility quality within a certain range in the reverse ‘S’ curve membership function. The reverse ‘S’ curve can be described by a ‘descending half trapezoid’ (Fig. 2d) with the following membership function (Eq. 14):

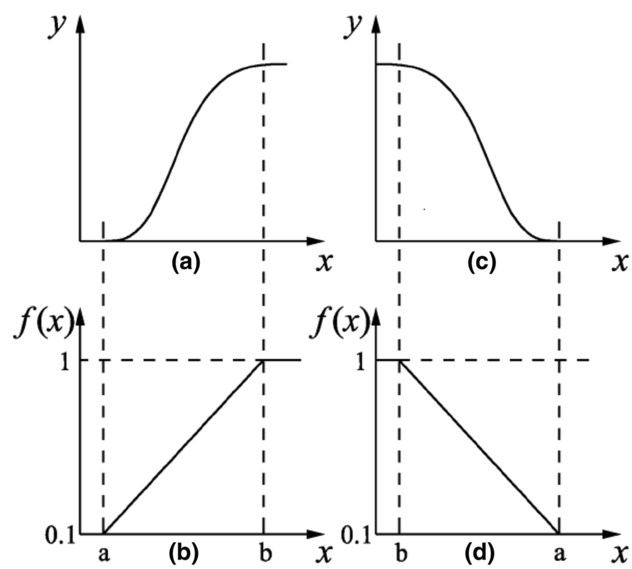

Fig. 2 The membership functions related to soil fertility indicators. Distribution of the ‘S’ curve (**a**), ascending half trapezoid (**b**), reverse ‘S’ curve (**c**), and descending half trapezoid (**d**)

$$f(x) = \begin{cases} 1.0, & (b \le x) \\ 0.9 \times \frac{x-a}{b-a} + 0.1, & (a \le x < b) \\ 0.1, & (x < a) \end{cases} \tag{13}$$

$$f(x) = \begin{cases} 1.0, & (x \le b) \\ 0.9 \times \frac{x-a}{b-a} + 0.1, & (b < x \le a) \\ 0.1, & (a < x) \end{cases}, \tag{14}$$

where $f(x)$ are the membership functions, x is the value of the soil fertility indicator, and a and b are the lower or upper limits of the soil fertility indicator, which have been defined in Table 2. The score is between 0.1 and 1.0. The maximum score of 1.0 indicates that the soil fertility index is completely suitable for plant growth, while the lowest value of 0.1 indicates that the soil fertility index is seriously lacking. Since there is very little soil with no fertility, the minimum score is 0.1; thus, an excessive zero value can be avoided in the calculation.

The weights of soil fertility indicators were evaluated by principle component analysis (PCA). The weight value was the ratio of the communality of each soil fertility indicator over the total communality explained by all six soil fertility indicators (Table 3).

Table 2 Critical values for different indicators in the reverse 'S' and 'S' membership functions

Critical values	Reverse S			S		
	BD (g/cm^3)	SOC (g/kg)	TN (g/kg)	AN (mg/kg)	AP (mg/kg)	AK (mg/kg)
a	1.52	2.69	0.27	15.86	6.91	124.09
b	1.13	15.93	0.87	64.73	22.88	150.93

a and b are the lower and upper critical values for the 'S' membership function, but a and b are the upper and lower critical values for the reverse 'S' membership function. In the present study, the upper or lower critical values are the mean values of soil fertility indicators in PF or RW4; in particular, the lower critical values for AK and AP were from RW5 and RW7

Table 3 Communalities and weights for different soil fertility indicators

Items	BD	SOC	TN	AN	AP	AK
Communalities	0.801	0.931	0.915	0.852	0.944	0.909
Weights	0.150	0.174	0.171	0.159	0.176	0.170

Results

Litter water-holding characteristics

The total litter storage capacities and all litter water-holding capacities in the restored woodlands (except for RW10) increased with restoration age, but all litter water-holding rates decreased with restoration age, as shown in Fig. 3, and all litter water-holding characteristics of RW10 were significantly lower than those of other woodlands. One-way ANOVAs showed that the restoration age had no significant effect on the water-holding capacities and water-holding rate of the litter (restoration age < 10), while afforestation species had a significant influence (restoration age was 10 years). The total litter storage capacity of RW4 was significantly lower than that of the others, but their water-holding rate was significantly higher than that of other woodlands. The total litter storage capacity of RW25 was significantly higher than that of the primary forest, but all their water-holding characteristics were significantly lower than those of PF.

The litter water-holding capacities (rates) of all woodlands increased logarithmically with the water soaking time (Fig. 4). The order of litter water-holding capacities of different woodlands is PF > RW25 > RW4, RW5, RW6 and RW7 > RW10, but the order for litter water-holding rates is RW4 > PF > RW5, RW6 and RW7 > RW25 > RW10. The water-absorption process of litter could be divided into three stages: (1) rapid growth stage (0–1 h), (2) steady growth stage (1–10 h), and (3) stable stage (after 10 h).

The litter water-absorption rate of all woodlands decreased exponentially with the water soaking time (Fig. 5), dropped rapidly in the first h, entered a slow decline stage, and tended to stabilize after 10 h. In the rapid descending stage, the order of litter water-absorption rates in different woodlands is PF > RW25 > RW4, RW5, RW6, RW7 > RW10.

Soil water-holding characteristics

The total soil porosity and soil total water-holding capacity increased gradually with restoration age when the restoration age was ≤ 10 years (except for RW6), but the soil capillary porosity, noncapillary porosity, capillary water-holding capacity, and effective water-holding capacity fluctuated with restoration age (Fig. 6). The soil capillary porosity, total soil porosity, capillary water-holding capacity, and total water-holding capacity of all restored woodlands were significantly lower than those of PF, while the soil noncapillary porosity and effective water-holding capacity in the woodlands restored for ≥ 10 years were higher than that those in PF.

Soil fertility indicators and their temporal variation

Evaluating the characteristics of soil fertility indicators in restored woodlands is important for understanding the restoration of soil fertility quality in mining areas. In our study, the contents of SOC, TN, AN, and AP in restored woodlands increased with restoration age, while BD and AK were less affected by restoration age, as shown in Fig. 7.

Comprehensive soil fertility index and its temporal variation

The soil fertility quality in forestland can be well reflected by a comprehensive soil fertility index (SFI). The SFI of restored woodlands generally varied with restoration age, from 0.3 to 0.9 (Fig. 8). One-way ANOVAs showed that there was no significant difference in SFI among restored woodlands when the restoration age was < 10 years, but the SFI was significantly improved when the restoration age reached a higher level (25 years), which is even closer to the level of PF (Fig. 8).

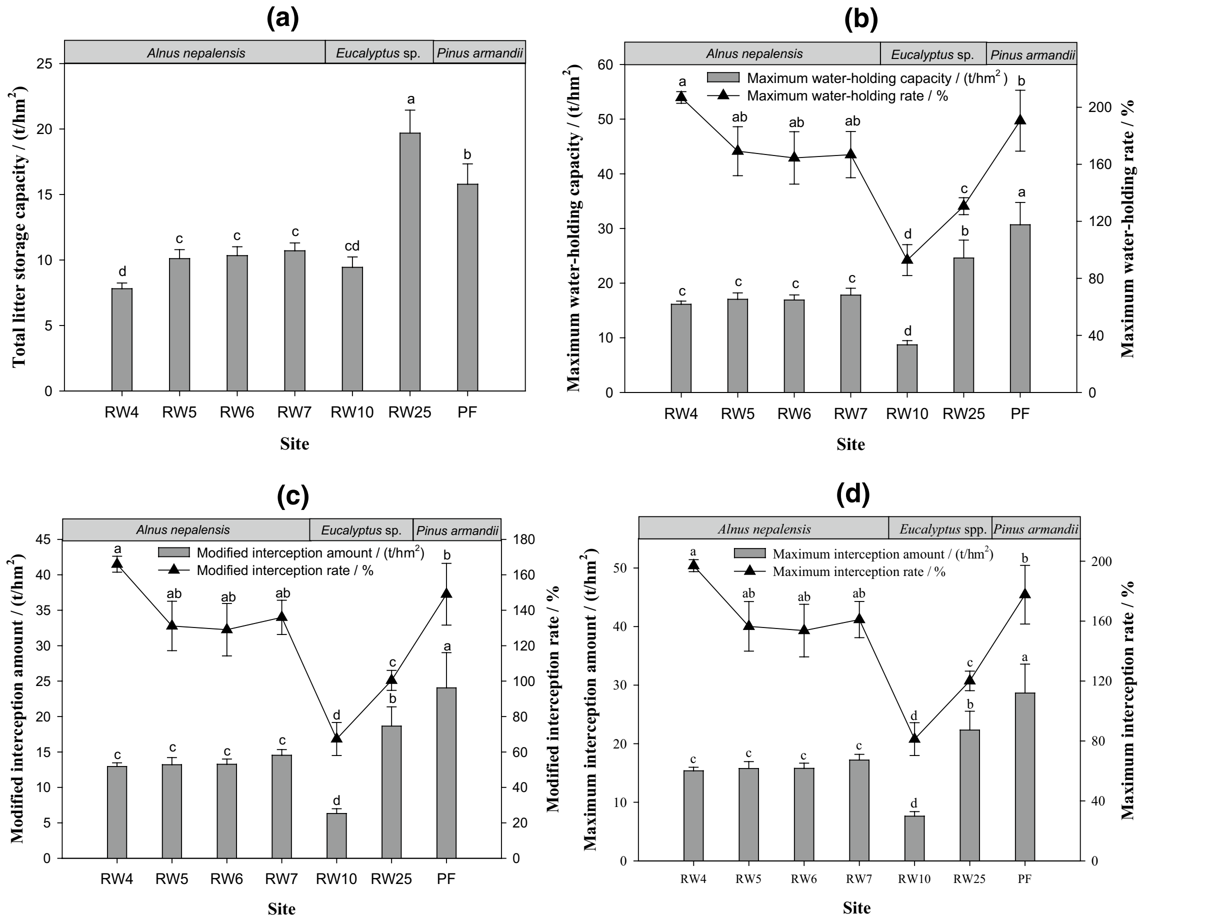

Fig. 3 Total litter storage capacity (**a**), maximum water-holding capacity (rate) (**b**), modified interception amount (rate) (**c**), and maximum interception amount (rate) (**d**) of the litter for each test site. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest. Different letters indicate that significant differences exist between two sites ($P < 0.05$). Repeated letters indicate that no significant difference exists between two sites

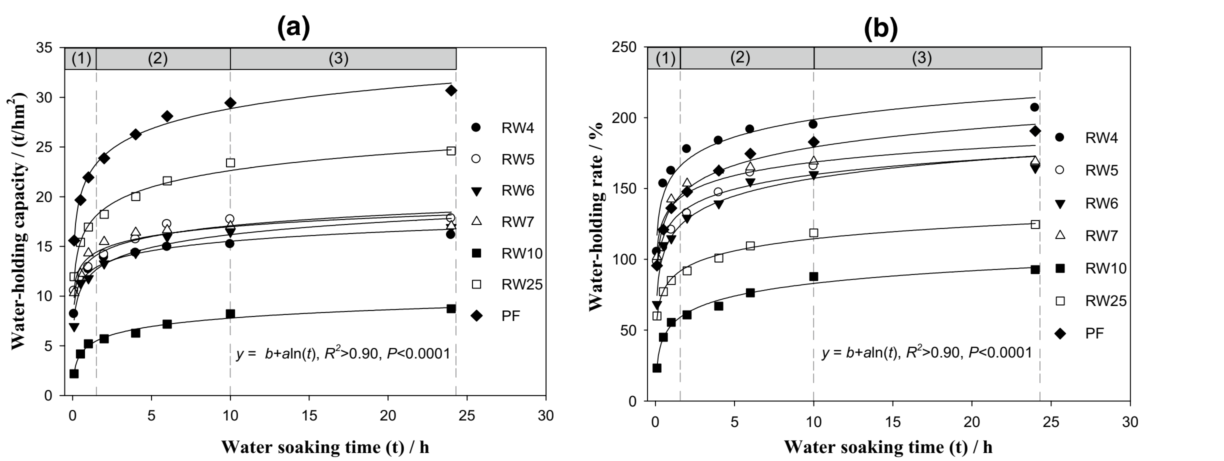

Fig. 4 Variations in litter water-holding capacity (**a**) and litter water-holding rate (**b**) with water soaking time. (1), (2), and (3) represent the water-absorption stage. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest

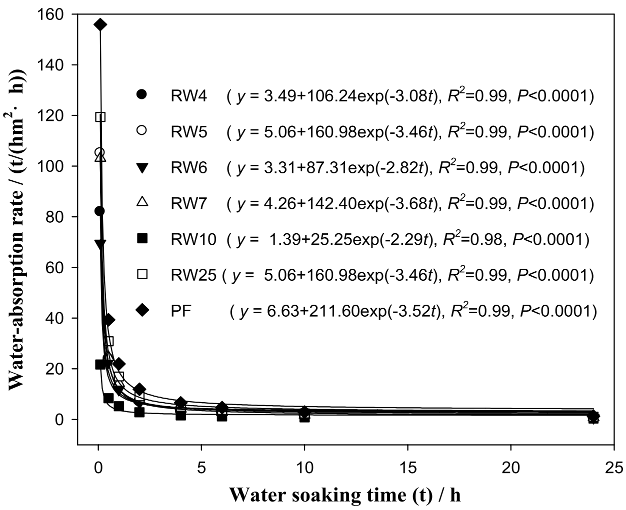

Fig. 5 Variations in litter water-absorption rate with water soaking time. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest

Discussion

Factors influencing the water conservation function

Vegetation restoration generally addresses many stressful conditions on mined wasteland, such as altered hydrological conditions, excess compaction, and low nutrients (Li 2006; Rowland et al. 2009; Singh and Singh 2001). Hydrological characteristics (including the water-holding capacities of litters and near-surface soil) usually determine the restoration of water conservation function.

The water-holding capacity of litter is reflected by its total storage capacity and water-holding rate. These two variables are affected by both vegetation restoration age and the afforestation species, which are shown in Figs. 3, 4, 5 and Table 4. The litter water-holding capacities generally increased with restoration age because of the increase of total storage capacity. However, the above increasing trend will also be abrupt due to tree species differences. For instance, the lowest water-holding capacity was observed in RW10, because the dominant species of RW10 was *Eucalyptus robusta*, which has a fairly low litter water-holding

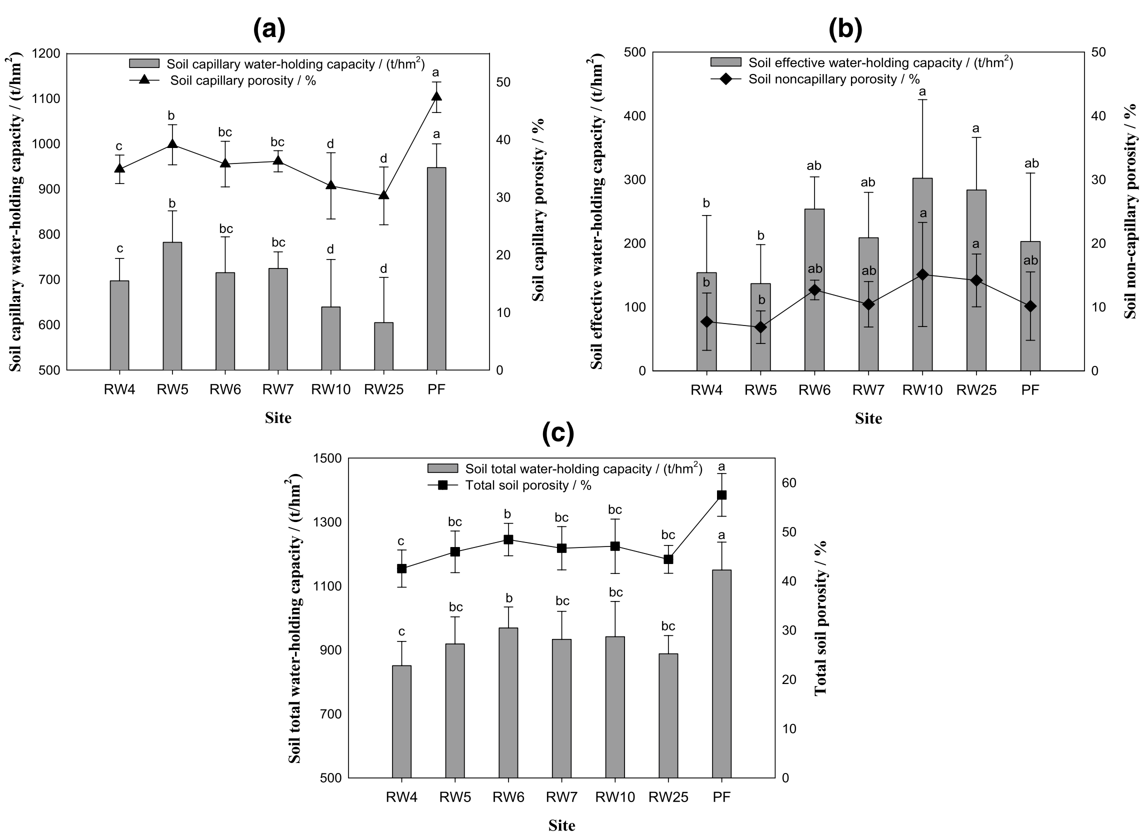

Fig. 6 Soil porosity and water-holding capacities for each test site. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest. Different letters indicate that significant differences exist between two sites ($P < 0.05$). Repeated letters indicate that no significant difference exists between two sites

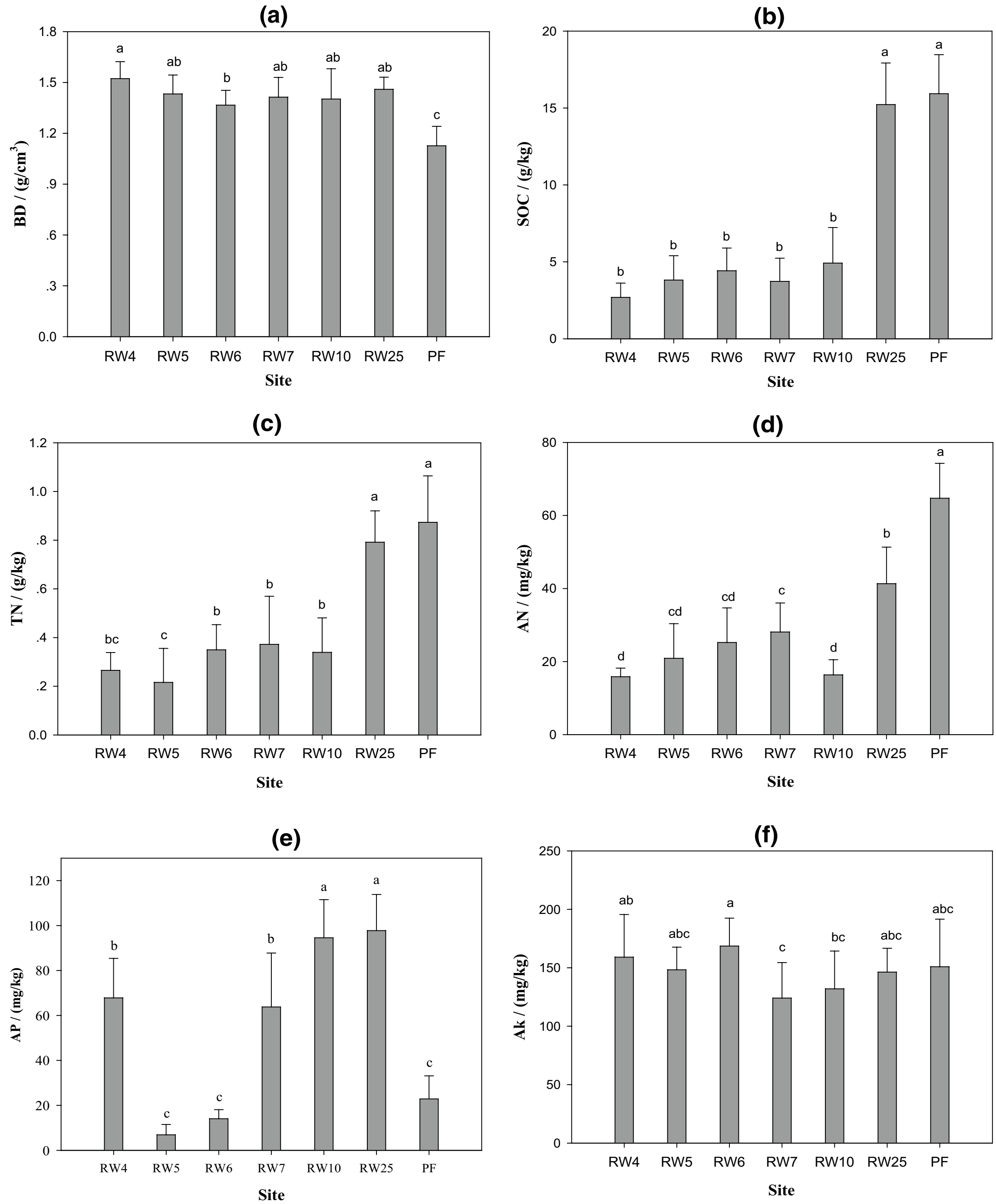

Fig. 7 Variations in BD (**a**), SOC (**b**), TN (**c**), AN (**d**), AP (**e**), and AK (**f**) with restoration age. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest. Error bars represent the standard deviation. Different letters indicate that significant differences exist for those soil fertility indicators between two sites ($P<0.05$). Repeated letters indicate that no significant difference exists for those soil fertility indicators between two sites

rate. Furthermore, the ANOVA results showed no significant difference observed in litter water-holding capacities between *Alnus nepalensis*–*Cupressus funebris* woodlands and *Eucalyptus robusta*–*Cupressus funebris* woodlands. However, the litter water-holding rates in restored woodlands of *Alnus nepalensis*–*Cupressus funebris* were significantly higher than that in restored woodlands of *Eucalyptus robusta*–*Cupressus funebris* with a longer restoration age (Table 4). In addition, the litter water-holding capacity in RW4 with the lowest total litter storage capacity (Fig. 3a)

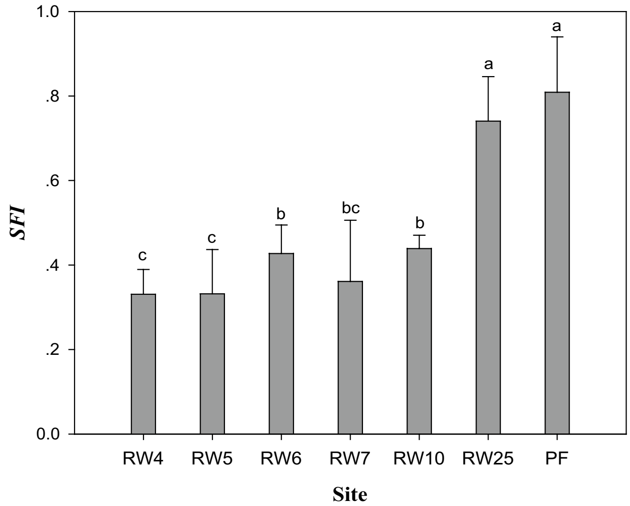

Fig. 8 Variations in soil fertility index (SFI) with restoration age. RW is restored woodland, and the number following it is the restoration age in years. PF is primary forest. Error bars represent the standard deviation. Different letters indicate that significant differences exist for SFI between two sites ($P < 0.05$). Repeated letters indicate that no significant difference exists for SFI between two sites

was close to that in RW5, RW6, and RW7, because RW4 had a higher herb coverage (Table 1) which caused a higher water-holding rate. Our findings indicated that tree species have a significant impact on the litter water-holding rate, which in turn affects the litter water-holding capacity. The current results are supported by the findings by Ruwanza et al. (2013), who found that the *Eucalyptus* species led to water repellency.

In general, the restoration age has a significant effect on litter water-holding capacities, because its significant improvement occurred when the recovery period reached 25 years, even though it cannot reach the level of PF. Previous studies (Zagyvai-Kiss et al. 2019; Zhang et al. 2019) focused on the litter hydrological characteristics of forests with different stand composition. However, a few studies have investigated the changes in litter water-holding properties with restoration age. Our results are beneficial in realizing the influence of vegetation restoration on water conservation function.

Soil is the most important for water storage in forest ecosystems, and its water-holding capacity is determined by soil porosity characteristics (Zhang et al. 2019) as soil moisture is mainly stored in the pores of the soil. A low effective water-holding capacity in the early stages of vegetation restoration was due to high rock/gravel fragments and low SOC concentration in mine soils (Shrestha and Lal 2008). This result is consistent with a study of Ruthrof et al. (2018), reporting that soil water availability in a post-mining substrate was low. Higher soil effective water-holding capacity was observed in the woodlands restored for > 5 years (Fig. 6b), because soil organic matter (SOM) accumulation can improve soil water retention capacity (Alday et al. 2012). This finding indicated that vegetation restoration was conducive to improving soil effective water-holding capacity. This is of great significance to the function of forests in regulating hydrology (Sun et al. 2018a, b; Du et al. 2019) and plays an important role in regulating run-off and reducing peak flow (Putuhena and Cordery 2000).

Soil water-holding capacity was affected by vegetation restoration age and the type of afforestation species used. One of the reasons why the soil capillary water-holding capacity decreased with restoration age is the soil capillary porosity following vegetation restoration was reduced due to capillary pores being blocked by the leaching soil particles. Another reason is the difference in the type of afforestation species used. As shown in Fig. 6a and Table 4, the soil capillary water-holding capacity in restored woodlands of *Eucalyptus robusta–Cupressus funebris* with a longer restoration age was significantly lower than that in restored woodlands of *Alnus nepalensis–Cupressus funebris* with a shorter restoration age, which indicated that *Eucalyptus robusta* forests had a low level of soil water absorption and holding capacity. As reported by Doerr et al. (2000), eucalyptus with a considerable amount of resins, waxes, or aromatic oils commonly associated with water repellency. Nevertheless, the restoration age does have an important impact on the soil effective water-holding capacity, because the soil effective water-holding capacity increased with the restoration age (Fig. 6b) and reached a significant level (Table 4). The above results showed that the function of soil water storage and regulation in restored woodlands increased with restoration age, but the function of soil water retention and supply were affected by the tree species.

Investigating the effects of restoration age and tree species on the water-holding capacity of litter and soil in the restored woodland is conducive in assessing the water conservation effect of vegetation restoration and providing guidance for scientific afforestation models. Our findings indicated that both the vegetation restoration age and the afforestation species have important impacts on the litter water-holding capacity and soil capillaries. Compared with *Eucalyptus robusta–Cupressus funebris* woodlands, the *Alnus nepalensis–Cupressus funebris* woodlands have a better water conservation function.

Factors influencing the soil fertility quality

The restoration age has a significant effect on soil fertility quality in the phosphate mined-out area. The higher contents of C, N, and P in older woodlands of the 25-year chronosequence showed similar trends to those reported by other studies in forest stands of varying age (Bienes et al. 2016). Increasing C and N with restoration age could be ascribed to

Table 4 Water-holding characteristics and soil fertility indicators between restored woodlands with different afforestation species

Items	*Alnus nepalensis–Cupressus funebris* (restored woodland)	*Eucalyptus robusta–Cupressus funebris* (restored woodland)	*Pinus armandii–Pinus yunnanensis* (primary forest)
M (t/hm^2)	9.74 ± 1.29 **b**	14.57 ± 5.75 **a**	15.79 ± 1.55 a
W_m (t/hm^2)	16.98 ± 1.06 b	16.66 ± 8.95 b	30.68 ± 13.07 a
R_m (%)	**176.82 ± 22.30 a**	**108.70 ± 19.10 b**	190.57 ± 61.35 a
W_{sv} (t/hm^2)	13.49 ± 0.93 b	12.49 ± 6.98 b	24.05 ± 10.97 a
R_{sv} (%)	**140.52 ± 18.36 a**	**80.87 ± 16.35 b**	149.04 ± 52.40 a
W_{msv} (t/hm^2)	16.03 ± 1.08 b	14.98 ± 8.32 b	28.65 ± 12.93 a
R_{msv} (%)	**167.05 ± 21.66 a**	**97.17 ± 19.22 b**	177.62 ± 61.60 a
W_t (t/hm^2)	917.86 ± 87.05 b	919.85 ± 150.57 b	1149.92 ± 86.90 a
W_c (t/hm^2)	**729.74 ± 66.69 b**	**605.66 ± 108.81 c**	947.16 ± 52.83 a
W_{nc} (t/hm^2)	188.12 ± 92.72 **b**	314.19 ± 193.16 **a**	202.76 ± 127.28 b
BD (g/cm^3)	1.43 ± 0.12 a	1.43 ± 0.20 a	1.13 ± 0.12 b
SOC (g/kg)	3.67 ± 1.77 **c**	10.07 ± 5.84 **b**	15.93 ± 4.53 a
TN (g/kg)	0.30 ± 0.15 **c**	0.57 ± 0.27 **b**	0.87 ± 0.19 a
AN (mg/kg)	22.53 ± 10.62 b	28.84 ± 14.84 b	64.73 ± 13.54 a
AP (mg/kg)	38.15 ± 50.72 **b**	96.17 ± 49.30 **a**	22.88 ± 10.27 b
AK (mg/kg)	150.02 ± 31.82 a	139.15 ± 27.23 a	150.93 ± 40.71 a
SFI	0.36 ± 0.10 **c**	0.59 ± 0.17 **b**	0.81 ± 0.13 a

Both the data and letters in the same row were shown in bold, representing that the litter or soil variables in the *Alnus nepalensis–Cupressus funebris* woodlands were significantly higher than that in the *Eucalyptus robusta–Cupressus funebris* woodlands, while only the letters were shown in bold indicating that the litter or soil variables in the *Alnus nepalensis–Cupressus* funebris woodlands were significantly lower than that in the *Eucalyptus robusta–Cupressus* funebris woodlands

Data in the table represent the average value ± standard deviation of the litter and soil indicators; different small letters in each row indicate that significant differences exist between two afforestation species ($P < 0.05$); M is the total litter storage capacity (t/hm^2); R_m, R_{sv}, and R_{msv} are maximum water-holding rate (%), modified interception rate (%), and maximum interception rate (%) of the litter, respectively; W_m, W_{sv}, and W_{msv} are the maximum water-holding capacity (t/hm^2), modified interception amount (t/hm^2), and maximum interception amount of the litter layer, respectively; W_c, W_{nc}, and W_t are the soil capillary water-holding capacity (t/hm^2), soil effective water-holding capacity (t/hm^2), and soil total water-holding capacity (t/hm^2), respectively; SFI is the comprehensive soil fertility index

increasing SOM content (Baddeley et al. 2017) as a result of increased above- and below-ground C inputs with restoration age. SOM is mainly of vegetal origin in the form of litter, roots, and exudates (Girona-Garcia et al. 2018). Vegetation restoration provides a large number of material sources for SOM, especially the accumulation of litter, and its further decomposition will release nutrients such as C, N, or P to the soil system. The highest total litter storage capacity (19.69 t/hm^2) was observed in RW25 (Fig. 3a), which indicates that RW25 provided more SOM than other restored woodlands, and this is why, SOC in RW25 (15.22 g/kg) was significantly higher than that in other sites (< 5 g/kg).

Tree species have an impact on the soil fertility of restored woodland, but it is not the most important factor. The dominant species in RW10 was *Eucalyptus robusta*, but that in the restored woodlands with a shorter restoration age (RW4, RW5, RW6, and RW7) was *Alnus nepalensis*, which is a deciduous species with a high amount of litter. Furthermore, the lowest AN content was observed in RW10 (Fig. 7d), while there were no significant differences in the total litter storage capacity between the woodlands restored for ≤ 10 years (Fig. 3a). This suggested that litters from *Alnus nepalensis* had a higher decomposition rate than that from *Eucalyptus robusta* and thus accelerated the N cycling. The result is consistent with the previous studies (Girona-Garcia et al. 2018; Zhong et al. 2017), which suggested that the amount of litter, its decomposition, and its properties are essential factors in the ecosystem N dynamics. However, the AN content in RW25 with *Eucalyptus robusta* was significantly higher than that in RW10 and in other restored woodlands. That is to say, the restoration age has a stronger effect on soil N than tree species. Although the SOC, TN, AP, and *SFI* of the restored woodlands with *Eucalyptus robusta–Cupressus funebris* were significantly higher than that in the restored woodlands with *Alnus nepalensis–Cupressus funebris* (Table 4), that is because the former had a longer restoration age. Therefore, the vegetation restoration age is the most important factor affecting soil fertility quality.

In addition, the mining of phosphate rock caused a significant increase in AP content in restored woodlands when the restoration age was > 6 years. The residual phosphorus-containing minerals in the mining platform become the soil parent material after platforms were covered with soil, and some might also be mixed into the soil system. These minerals would be gradually weathered and released elemental P that is easily absorbed by plants; thus, the AP content was significantly improved after vegetation restoration, especially when the restoration age was more than 6 years. The finding demonstrated that C and N rather than P were the major constraints to the restoration of soil fertility in phosphate mined-out areas. This conclusion is supported by the previous research (Ruthrof et al. 2018), which showed that post-phosphate mining substrates are very poor in AN and can severely limit plant growth. Li et al. (2015) also proposed that P provides the most positive contribution to the restoration of soil elements in phosphorus mining areas. In contrast, a different conclusion has been reported for other mine types (Nikolic et al. 2011; Wang et al. 2018a, b, c), who found that P deficiency is a major limiting factor for plant growth in copper mine tailings. However, excessive P content in the later stages of vegetation restoration (> 7 years) warns that appropriate soil and water conservation measures to prevent phosphorus loss are necessary, because the large loss of elemental P will be a major threat to the water ecological environment of Dianchi Lake in the eastern KPM.

Moreover, both the restoration age and tree species had no significant effect on BD and AK. The soil of restored lands in mine areas is tight and that their improvement is difficult in short term restoration, which was reflected by the higher BD in restored woodlands than that of the PF. The main reason for the small change in AK with restoration age is that soil K is mainly derived from the soil parent material. Higher AK content in the early stage indicates that there is no need to apply excessive K fertilizer during this period. This is of great practical significance to the management of vegetation restoration in mining areas.

Vegetation restoration plays an important role in improving soil fertility quality because of the increase of SFI in restored woodlands with the restoration age. Even though the soil fertility quality was relatively low in the first decade of restoration (Fig. 8), it was close to that of the PF when the restoration age was 25 years. This was congruent with the findings reported by Guo et al. (2018), who found that soil fertility quality significantly increased with increasing restoration age up to 27 years. In our study, SFI showed significant correlations ($P < 0.01$) with BD, SOC, TN, and AN, with correlation coefficients of −0.441, 0.907, 0.882, and 0.804, respectively. This suggests that SOC, TN, and AN are the main contributors to soil fertility in the phosphate mined-out area. This result can be explained by similar trends of SFI, SOC, TN, and AN with restoration age. Besides, even if SFI of RW25 is close to that of PF, it will still be a long process to achieve complete restoration in phosphate mined-out areas.

The above results indicate that vegetation restoration plays an important role in the restoration of soil C, N, and P in mining areas. Afforestation species have an important impact on soil N, but the effect on soil comprehensive fertility quality is not significant. The restoration age rather than the tree species is the most important factor affecting soil fertility quality. These findings suggest that vegetation restoration by afforestation alters litter and soil properties, thereby resulting in changes in nutrients that may improve soil fertility quality of restored woodland, and this effect is more pronounced as the increase of restoration age.

Conclusions

The impacts of restoration age and tree species on water conservation function and soil fertility quality of restored woodlands were examined in a phosphate mined-out area of southwest China. The results showed that litter water-holding capacities (maximum water-holding capacity, modified interception amount and maximum interception amount) and soil effective water-holding capacity increased with restoration age, suggesting that the restoration age is an important driver for the improvement of water conservation function. SOC, TN, AN, AP, and SFI increased with restoration age, and SFI had significant positive correlations with SOC, TN, and AN, which indicates that soil C and N are highly susceptible to restoration age and can be used as sensitive indicators to indicate the restoration of soil fertility in mining areas. There was a significant increase in AP content after vegetation restoration in the phosphate mining area; this shows that it is necessary to strengthen soil and water conservation measures to reduce P migration and reduce the threat to surrounding water bodies, especially to the heavily polluted Dianchi Lake. The litter water-holding capacities, soil capillary water-holding capacity, and AN in RW10 were significantly lower than in those restored for < 10 years; this suggests that plant species plays a major role in governing water conservation function, and soil N cycling and planting *Alnus nepalensis* rather than *Eucalyptus robusta* on the phosphate mining wastelands is more suitable for the restoration of water and soil function of forest ecosystems. Compared to PF, litter water-holding capacities, soil total water-holding capacity, and SFI of RW25 were still lower than those of PF, indicating that the full restoration of water conservation function and soil fertility is slow (> 25 years). Based on these results, it is clear that both the vegetation restoration age and the afforestation species have important impacts on the water conservation function of the restored woodlands in phosphate mined-out areas. However, the main factor affecting soil fertility quality is the restoration age rather than the type of afforestation species used. Furthermore, the risk of P loss after vegetation restoration should be specifically considered when evaluating the impact of revegetation on soil fertility in phosphate mine areas adjacent to lakes. Our results provide insight into vegetation restoration in areas with degraded lands.

Acknowledgements We thank Dahal Nirmal Mani for his help with editing. This study was supported by projects XDA20020401 (the Strategic Priority Research Program of the Chinese Academy of Sciences), S2012FZ0557 (the Applied Basic Research Program of Yunnan), and 2017YFC0505102 (the National Key Research and Development Program of China). This support is gratefully acknowledged.

References

Alday JG, Marrs RH, Martinez-Ruiz C (2012) Soil and vegetation development during early succession on restored coal wastes: a six-year permanent plot study. Plant Soil 353:305–320

Baddeley JA, Edwards AC, Watson CA (2017) Changes in soil C and N stocks and C:N stoichiometry 21 years after land use change on an arable mineral topsoil. Geoderma 303:19–26

Bienes R, Marques MJ, Sastre B, Garcia-Diaz A, Ruiz-Colmenero M (2016) Eleven years after shrub revegetation in semiarid eroded soils. Influence in soil properties. Geoderma 273:106–114

Blouin M, Hodson ME, Delgado EA, Baker G, Brussaard L, Butt KR, Dai J, Dendooven L, Peres G, Tondoh JE, Cluzeau D, Brun JJ (2013) A review of earthworm impact on soil function and ecosystem services. Eur J Soil Sci 64:161–182

Brantley S, Ford CR, Vose JM (2013) Future species composition will affect forest water use after loss of eastern hemlock from southern Appalachian forests. Ecol Appl 23:777–790

Bremner JM (1996) Nitrogen-total. In: Sparks DL, Page AL, Helmke PA, Loeppert RH (eds) Methods of soil analysis, Part 3—chemical methods, vol 5. SSSA Book Series. Soil Science Society of America, American Society of Agronomy, Madison, pp 1085–1121

Bruijnzeel LA (2004) Hydrological functions of tropical forests: not seeing the soil for the trees? Agr Ecosyst Environ 104:185–228

Chodak M, Niklinska M (2010) The effect of different tree species on the chemical and microbial properties of reclaimed mine soils. Biol Fert Soils 46:555–566

Deng Q, Cheng X, Hui D, Zhang Q, Li M, Zhang Q (2016) Soil microbial community and its interaction with soil carbon and nitrogen dynamics following afforestation in central China. Sci Total Environ 541:230–237

Doerr SH, Shakesby RA, Walsh RPD (2000) Soil water repellency: its causes, characteristics and hydro-geomorphological significance. Earth Sci Rev 51:33–65

Du J, Niu J, Gao Z, Chen X, Zhang L, Li X, Van Doorn NS, Luo Z, Zhu Z (2019) Effects of rainfall intensity and slope on interception and precipitation partitioning by forest litter layer. CATENA 172:711–718

Edwards DP, Tobias JA, Sheil D, Meijaard E, Laurance WF (2014) Maintaining ecosystem function and services in logged tropical forests. Trends Ecol Evol 29:511–520

Fu X, Shao M, Wei X, Horton R (2010) Soil organic carbon and total nitrogen as affected by vegetation types in Northern Loess Plateau of China. Geoderma 155:31–35

Garay M, Amiotti N, Zalba P (2018) Response of phosphorus pools in Mollisols to afforestation with Pinus radiata D. Don in the Argentinean Pampa. For Ecol Manag 422:33–40

Ghidey F, Alberts EE (1997) Plant root effects on soil erodibility, splash detachment, soil strength, and aggregate stability. Trans Am Soc Agric Eng 40:129–135

Girona-Garcia A, Badia-Villas D, Jimenez-Morillo NT, de la Rosa JM, Gonzalez-Perez JA (2018) Soil C and N isotope composition after a centennial Scots pine afforestation in podzols of native European beech forests in NE-Spain. CATENA 165:434–441

Guo W, Yang F, Li Y, Wang S (2017) New insights into the source of decadal increase in chemical oxygen demand associated with dissolved organic carbon in Dianchi Lake. Sci Total Environ 603:699–708

Guo S, Han X, Li H, Wang T, Tong X, Ren G, Feng Y, Yang G (2018) Evaluation of soil quality along two revegetation chronosequences on the Loess Hilly Region of China. Sci Total Environ 633:808–815

Guo A, Zhao Z, Zhang P, Yang Q, Li Y, Wang G (2019) Linkage between soil nutrient and microbial characteristic in an opencast mine, China. Sci Total Environ 671:905–913

Han X, Zhao F, Tong X, Deng J, Yang G, Chen L, Kang D (2017) Understanding soil carbon sequestration following the afforestation of former arable land by physical fractionation. CATENA 150:317–327

He L, Fang X, Meng G, Li G, Shao J, Chai Y, Kong J (2013) Effect of *Alnus nepalensis* cultivation on soil biological and physicochemical properties during restoration near a phosphate smelter in Kunyang, Yunnan Province, SW China. J Soil Sci Plant Nut 13:355–366

Horodecki P, Jagodzinski AM (2017) Tree species effects on litter decomposition in pure stands on afforested post-mining sites. For Ecol Manag 406:1–11

Hossain MF, Chen W, Zhang Y (2015) Bulk density of mineral and organic soils in the Canada's arctic and sub-arctic. Inf Process Agric 2:183–190

Hou X, Han H, Tigabu M, Cai L, Meng F, Liu A, Ma X (2019) Changes in soil physico-chemical properties following vegetation restoration mediate bacterial community composition and diversity in Changting, China. Ecol Eng 138:171–179

Jiang L, Han X, Dong N, Wang Y, Kardol P (2011) Plant species effects on soil carbon and nitrogen dynamics in a temperate steppe of northern China. Plant Soil 346:331–347

Karlen DL, Ditzler CA, Andrews SS (2003) Soil quality: why and how? Geoderma 114:145–156

Li MS (2006) Ecological restoration of mineland with particular reference to the metalliferous mine wasteland in China: a review of research and practice. Sci Total Environ 357:38–53

Li G-X, Zhang J, Shao J-P, Zhou B, Bi B, Xie K-M, Fang X-J, Wang Y-Z (2015) Chemical properties of soil layers of restoration sites in phosphate mining area, China. Environ Earth Sci 73:2027–2030

Liu S, Hou X, Yang M, Cheng F, Coxixo A, Wu X, Zhang Y (2018) Factors driving the relationships between vegetation and soil properties in the Yellow River Delta, China. CATENA 165:279–285

Moreno-de las Heras M (2009) Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean–Continental environment. Geoderma 149:249–256

Mueller KE, Eissenstat DM, Hobbie SE, Oleksyn J, Jagodzinski AM, Reich PB, Chadwick OA, Chorover J (2012) Tree species effects on coupled cycles of carbon, nitrogen, and acidity in mineral soils at a common garden experiment. Biogeochemistry 111:601–614

Nelson DW, Sommers LE (1996) Total carbon, organic carbon, and organic matter. In: Sparks DL, Page AL, Helmke PA, Loeppert RH (eds) Methods of soil analysis, Part 3—chemical methods,

vol 5.3. SSSA Book Series. Soil Science Society of America, American Society of Agronomy, Madison, pp 961–1010
Nikolic N, Kostic L, Djordjevic A, Nikolic M (2011) Phosphorus deficiency is the major limiting factor for wheat on alluvium polluted by the copper mine pyrite tailings: a black box approach. Plant Soil 339:485–498
Olsen SR, Sommers LE (1982) Phosphorus. In: Page AL (ed) Methods of soil analysis. Part 2. Chemical and microbiological properties. Agronomy monograph, vol 9.2. American Society of Agronomy, Soil Science Society of America, Madison, pp 403–430
Phan Minh S, Lamb D, Bonner M, Schmidt S (2013) Carbon sequestration and soil fertility of tropical tree plantations and secondary forest established on degraded land. Plant Soil 362:187–200
Putuhena WM, Cordery I (2000) Some hydrological effects of changing forest cover from eucalypts to Pinus radiata. Agr For Meteorol 100:59–72
Raiesi F (2017) A minimum data set and soil quality index to quantify the effect of land use conversion on soil quality and degradation in native rangelands of upland arid and semiarid regions. Ecol Indic 75:307–320
Rowland SM, Prescott CE, Grayston SJ, Quideau SA, Bradfield GE (2009) Recreating a functioning forest soil in reclaimed oil sands in Northern Alberta: an approach for measuring success in ecological restoration. J Environ Qual 38:1580–1590
Ruthrof KX, Steel E, Misra S, McComb J, O'Hara G, Hardy GESJ, Howieson J (2018) Transitioning from phosphate mining to agriculture: responses to urea and slow release fertilizers for Sorghum bicolor. Sci Total Environ 625:1–7
Ruwanza S, Gaertner M, Richardson DM, Esler KJ (2013) Soil water repellency in riparian systems invaded by *Eucalyptus camaldulensis*: a restoration perspective from the Western Cape Province, South Africa. Geoderma 200:9–17
Shackelford N, Miller BP, Erickson TE (2018) Restoration of open-cut mining in semi-arid systems: a synthesis of long-term monitoring data and implications for management. Land Degrad Dev 29:994–1004
Shrestha RK, Lal R (2008) Land use impacts on physical properties of 28 years old reclaimed mine soils in Ohio. Plant Soil 306:249–260
Singh A, Singh JS (2001) Comparative growth behaviour and leaf nutrient status of native trees planted on mine spoil with and without nutrient amendment. Ann Bot-London 87:777–787
Sun B, Zhou S, Zhao Q (2003) Evaluation of spatial and temporal changes of soil quality based on geostatistical analysis in the hill region of subtropical China. Geoderma 115:85–99
Sun L, Zhang G-h, Luan L-l, Liu F (2016) Temporal variation in soil resistance to flowing water erosion for soil incorporated with plant litters in the Loess Plateau of China. CATENA 145:239–245
Sun D, Zhang W, Lin Y, Liu Z, Shen W, Zhou L, Rao X, Liu S, Cai X, He D, Fu S (2018a) Soil erosion and water retention varies with plantation type and age. For Ecol Manag 422:1–10
Sun J, Yu X, Wang H, Jia G, Zhao Y, Tu Z, Deng W, Jia J, Chen J (2018b) Effects of forest structure on hydrological processes in China. J Hydrol 561:187–199
Tonks AJ, Aplin P, Beriro DJ, Cooper H, Evers S, Vane CH, Sjogersten S (2017) Impacts of conversion of tropical peat swamp forest to oil palm plantation on peat organic chemistry, physical properties and carbon stocks. Geoderma 289:36–45
Upton A, Vane CH, Girkin N, Turner BL, Sjogersten S (2018) Does litter input determine carbon storage and peat organic chemistry in tropical peatlands? Geoderma 326:76–87
Urbanova M, Snajdr J, Brabcova V, Merhautova V, Dobiasova P, Cajthaml T, Vanek D, Frouz J, Santruckova H, Baldrian P (2014) Litter decomposition along a primary post-mining chronosequence. Biol Fertil Soils 50:827–837
Wang D, Zhang B, Zhu L, Yang Y, Li M (2018a) Soil and vegetation development along a 10-year restoration chronosequence in tailing dams in the Xiaoqinling gold region of Central China. CATENA 167:250–256
Wang J, Luo X, Zhang Y, Huang Y, Rajendran M, Xue S (2018b) Plant species diversity for vegetation restoration in manganese tailing wasteland. Environ Sci Pollut R 25:24101–24110
Wang Y, Shen Z, Zhang Z (2018c) Phosphorus speciation and nutrient stoichiometry in the soil-plant system during primary ecological restoration of copper mine tailings. Pedosphere 28:530–541
Wong MH (2003) Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils. Chemosphere 50:775–780
Wu D, Xin X-b, Zhao M-y, Pei S-x, Kong Y (2014) Hydrological effects of litters and soil under different forests in Jiulongshan of Beijing. For Res 27:417–422 **(in Chinese)**
Xia L, Song X, Fu N, Cui S, Li L, Li H, Li Y (2019) Effects of forest litter cover on hydrological response of hillslopes in the Loess Plateau of China. CATENA 181:104076
Xu S, Liu LL, Sayer EJ (2013) Variability of above-ground litter inputs alters soil physicochemical and biological processes: a meta-analysis of litterfall-manipulation experiments. Biogeosciences 10:7537
Yan D, Zhao F, Sun OJ (2013) Assessment of Vegetation Establishment on Tailings Dam at an Iron Ore Mining Site of Suburban Beijing, China, 7 Years After Reclamation with Contrasting Site Treatment Methods. Environ Manag 52:748–757
Yang B, Zhang W, Xu H, Wang S, Xu X, Fan H, Chen HYH, Ruan H (2018) Effects of soil fauna on leaf litter decomposition under different land uses in eastern coast of China. J For Res 29:973–982
Ye M, Li JT, Tian SN, Hu M, Yi S, Liao B (2009) Biogeochemical studies of metallophytes from four copper-enriched sites along the Yangtze River, China. Environ Geol 56:1313–1322
Zagyvai-Kiss KA, Kalicz P, Szilágyi J, Gribovszki Z (2019) On the specific water holding capacity of litter for three forest ecosystems in the eastern foothills of the Alps. Agr For Meteorol 278:107656
Zhang C, Xue S, Liu G-B, Song Z-L (2011) A comparison of soil qualities of different revegetation types in the Loess Plateau, China. Plant Soil 347:163–178
Zhang X, Liu Z, Zhu B, Bing Y, Nhu Trung L, Du L, Zhu Z (2016) Impacts of mixed litter decomposition from Robinia pseudoacacia and other tree species on C loss and nutrient release in the Loess Plateau of China. J For Res 27:525–532
Zhang W, Gao D, Chen Z, Li H, Deng J, Qiao W, Han X, Yang G, Feng Y, Huang J (2018) Substrate quality and soil environmental conditions predict litter decomposition and drive soil nutrient dynamics following afforestation on the Loess Plateau of China. Geoderma 325:152–161
Zhang B, Xu Q, Gao D, Jiang C, Liu F, Jiang J, Ma Y (2019) Higher soil capacity of intercepting heavy rainfall in mixed stands than in pure stands in riparian forests. Sci Total Environ 658:1514–1522
Zhao F, Kang D, Han X, Yang G, Yang G, Feng Y, Ren G (2015) Soil stoichiometry and carbon storage in long-term afforestation soil affected by understory vegetation diversity. Ecol Eng 74:415–422
Zhong Y, Yan W, Wang R, Shangguan Z (2017) Differential responses of litter decomposition to nutrient addition and soil water availability with long-term vegetation recovery. Biol Fertil Soils 53:939–949

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.