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Vegetation and soil restoration in refuse dumps from open pit coal mines

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ABSTRACT

Because the exploitation of mineral resources has resulted in the destruction of vast amounts of land and caused serious environmental problems, ecological restoration and mine reclamation have become important components of the sustainable development strategies of many countries. In this study, the changes in plant species diversity and succession of soil physicochemical properties were studied in revegetated refuse dumps of varying ages (1995, 1998, 2003, 2005 and 2014) and an undisturbed reference site of the Heidaigou opencast coal mine, which is located in the Inner Mongolia Autonomous Region, China. In addition, based on the space-for-time substitution approach, the rate and extent of recovery of the vegetation and soil characteristics in the above long-term chronosequence were estimated. The results showed that opencast mining revegetation have a positive effect on environmental restoration. The total species number and the richness, coverage and biomass of herbaceous species increased significantly with increasing site age. However, for shrub species, an initial increase was observed over a period of 10 years and then followed by a subsequent decrease. The mean soil water content in the oldest vegetated areas (1995 site, 2.3–5.9%) was significantly greater than in the younger vegetated areas (e.g., 2014 site, 1.1–2.5%) in the shallow layer (0–60 cm); however, in the deeper layers (60–200 cm), the mean soil water content in the oldest vegetated areas (4.2–8.1%) was significantly lower than in the younger vegetated areas (5.1–12.5%). The proportions of silt and clay, depth of topsoil and biological soil crusts, and concentrations of soil organic C, K, total N, and total P increased with the years since revegetation. The estimated time required for plant traits and soil properties in the revegetated site to reach the same levels as in the reference site was between 23 and 25 years. These results suggest that soil recovery and vegetation recovery are equally important in eco-restoration activities in semiarid areas, and the conservation of soil and vegetation habitat is therefore a crucial issue for land managers.

1. Introduction

Coal is the primary source of energy in China (Lam, 2005) and has made great contributions to local economic development for years (Liu and Diamond, 2005) as well as the economic and social development of China (Miao and Marrs, 2000). However, opencast mining involves the displacement of large amounts of excess material (mine waste) from coal mining activities (Bradshaw, 1997; Josa et al., 2012). This anthropogenic change in soil structure exerts a profound effect on the sustainable development of ecosystems over large areas, especially in transition zones between arid and semiarid regions with fragile environments (Vasssilis and Wyseure, 1998; Wu et al., 2011; Li et al., 2012). In opencast mining areas, approximately 0.22 hm^2 of land can be destroyed per ten thousand tons of coal production (Miao et al., 2000). To date, 8.84×10^3 hm^2 of disturbed land and 1.63×10^4 hm^2 of waste dumps have been produced in China, and these areas are increasing at a ratio of 8–9% annually (Xia, 2004; Li et al., 2013). These dumps are physically, nutritionally and biologically poor in nature (Li, 2006). Furthermore, natural succession on these lands requires more time (Dutta and Agrawal, 2003). For example, Srivastava et al. (1989) estimated that it might take approximately 200 years of natural succession on a mine spoil for the total nitrogen pool to recover to the level

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of native forest soil. Therefore, a series of vegetation restoration projects, including tree planting, agricultural reclamation, botanical garden construction and other ventures, were implemented in these coal mine refuse dumps (Ezeaku, 2012; Li et al., 2013). Similar examples of spontaneous and assisted site recovery are underway in more than 100 mining sites in eastern Germany (e.g., Wanner and Dunger, 2002; Tischew and Kirmer, 2007; Krummelbein et al., 2010), the Teruel coalfield in central-eastern Spain (Moreno-de las Heras et al., 2008), the Singrauli coalfield of India (Singh et al., 2000), the open-cast coal mining area near Sokolov in northwestern Bohemia (Pižl, 2001), and the Pingshuo Antaibao opencast mining area on the Chinese Loess Plateau (Li et al., 2012). Site recovery has played a large role in the control of soil erosion and land degradation and accelerates the natural recovery processes to restore soil fertility and enhance biological diversity (Singh and Singh, 2006; Hendrychova, 2008). Dutta and Agrawal (2003) showed that shoot biomass was increased 2.5–3 times in 7-year-old plantations of different plant species compared with 4-year-old plantations. Revegetation with *Casuarina equisetifolia* and *Anacardium occidentale* at a serpentine-mined area located at Moa in northeastern Cuba increased soil aggregate stability by approximately 66 and 16% and decreased soil bulk density sharply 4 years after planting, from 1.49 to 1.26 and 1.36 g cm^{-3}, respectively (Izquierdo et al., 2005). However, much of the research in this field addresses restoration over a short period. Without systematic monitoring and long-term studies, there is little information on the long-term feedback influences of vegetation on the environment, making it impossible to scientifically evaluate the patterns and processes of ecological restoration through revegetation, and particularly the recovery rate of plant and soil characteristics.

Attempting to account for such mechanisms is a challenge for ecologists due to the difficulty of monitoring a number of plant and soil parameters over time. For this reason, a chronosequence approach, in which sites of different ages are assumed to represent points in time in the development of individual sites, is attractive. The underlying method is "space-for-time substitution" (SFT), which is an approach that has been successful, in which general trends are to be generated by extrapolation of a temporal trend from a series of different-aged samples (Pickett, 1989). Chronosequence approaches have provided valuable insight into patterns of forest succession and soil development (Lichter, 1998; Yanai et al., 2003), long-term changes in vegetation (Johnson and Miyanishi, 2008), decision making for ecosystem management (Larsen et al., 2001; Lindegren et al., 2009), and the impacts of wildfire on the global carbon cycle (e.g., Wardle et al., 2003) and climate changes on the range and distribution of species (Beale et al., 2008; Copeland et al., 2010; Mbogga et al., 2010).

The Heidaigou opencast coal mine is located in the middle of the Zhungeer coal field in the Inner Mongolia Autonomous Region of China and is the largest opencast coal mine in China, with an annual output of 20 million tons (Liu and Fan, 2002). Refuse dumps that have been stabilized by revegetation at different times offer an ideal opportunity to study vegetation succession in extreme environments, as soil conditions before revegetation are largely driven by geomorphological processes and vegetation succession commences after revegetation. The aim of this work was to investigate vegetation and soil restoration in the refuse dumps of open pit coal mines. Our particular objectives were as follows: (1) to examine changes in the species composition, plant species cover and biomass and the effects of soil variation over a 25-year period after revegetation and (2) to estimate the recovery rates of vegetation and soil characteristics in a chronosequence of revegetation sites compared with native vegetation sites. This study will contribute to our understanding of vegetation and soil restoration in opencast coal mining regions.

2. Materials and methods

2.1. Study area

The Heidaigou open pit coal mine is located in the middle of the Zhungeer coal field in the Inner Mongolia Autonomous Region of China, and it ranks as the third largest coal mine based on its reserves. The altitude at the site is 1025–1302 m, and the geographical coordinates are 39°43′–39°49′N and 111°13′–111°20′E. The Heidaigou coal mine occupies a total area of 5,124 ha, and waste dumps occupy more than 629 ha of the total area. The site is located in a temperate continental arid climate zone that is cold in winter, dry and windy in spring, hot in summer, and mild and pleasant in autumn. The annual average temperature is 7.2 °C; the average annual rainfall is 426.3 mm, with decreased rainfall predominantly occurring between June and September; and the precipitation during this period accounts for approximately 60% to 70% of the total. The average annual evaporation is 1,943.6 mm, and the relative humidity is 58%. The wind is mostly calm, flowing in a north-northwest direction at an average speed of 2.2 m/s. The coal mine is situated in a typical steppe ecological region, with the main vegetation consisting of *Stipa* species.

The five refuse dumps (northern, eastern, western, abandoned and new refuse dumps at the Heidaigou open pit coal mine) began vegetation reconstruction in 1995, 1998, 2003, 2005 and 2014. These dumps are located at a site away from the coal-bearing area. The soils of overburdened dumps are physically, nutritionally, and biologically poor. They usually consist of a mixture of fine-grained to coarse-grained particles and rock fragments, which cause geotechnical and environmental problems on disposal. As seen in Fig. 1, due to the destruction of mining activities for the mine surface, the original physical and chemical properties of the soil of mine district make fundamental changes, which are reflected mainly in severely damaged soil aggregates, loose soil, the topography of the area is undulating. The reclamation and revegetation of surface coal-mine disturbances have established sustainable and healthy arable-land ecosystems from bare mounds, although the natural succession occurs through a very slow process. The weathered materials were stabilized by vegetation succession and mixed with organic materials, resulting in improved physical properties of the materials and more favorable conditions for plant growth. A range of reclamation techniques are available for coal mine reclamation. The most frequently planted tree species (such as *Robinia pseudoacacia, Pinus tabulaeformis, Populus simonii*), shrubs (such as *Tamarix chinensis, Caragana microphylla, Hippophae rhamnoides, Artemisia gmelinii, Armeniaca sibirica, Artemisia giraldii*) and grasses (such as *Stipa bungeana, Bothriochloa ischaemum, Pennisetum centrasiaticum, Lespedeza potaniniiv, Corispermum tylocarpum, Astragalus meliltoides, Thermopsis lanceolata, Leymus secalinus, Medicago sativa, Calamagrostis epigejos, Cleistogenes squarrosa*) were selected as the main revegetated plants. In this mixed plantation configuration, inter-planting grasses were around 1 m away from shrubs and trees at the densities of 1 m × 2 m, 1 m × 3 m, respectively. Different refuse dumps with different ages were revegetated using very similar approaches, including planting the same species of trees and shrubs with the same density, they can represent the different successional stages of reclamation. The basic characteristics of the six sites are shown in Table 1.

2.2. Sampling method and data collection

For the assessment of plant species, three quadrats were established at each of the six experimental sites at the Heidaigou open pit coal mine, totaling 18 quadrats. The shrubland survey area was 10 m × 10 m, whereas the herb plot was 1 m × 1 m. The species richness (number of species per quadrat), shrub cover, herbaceous

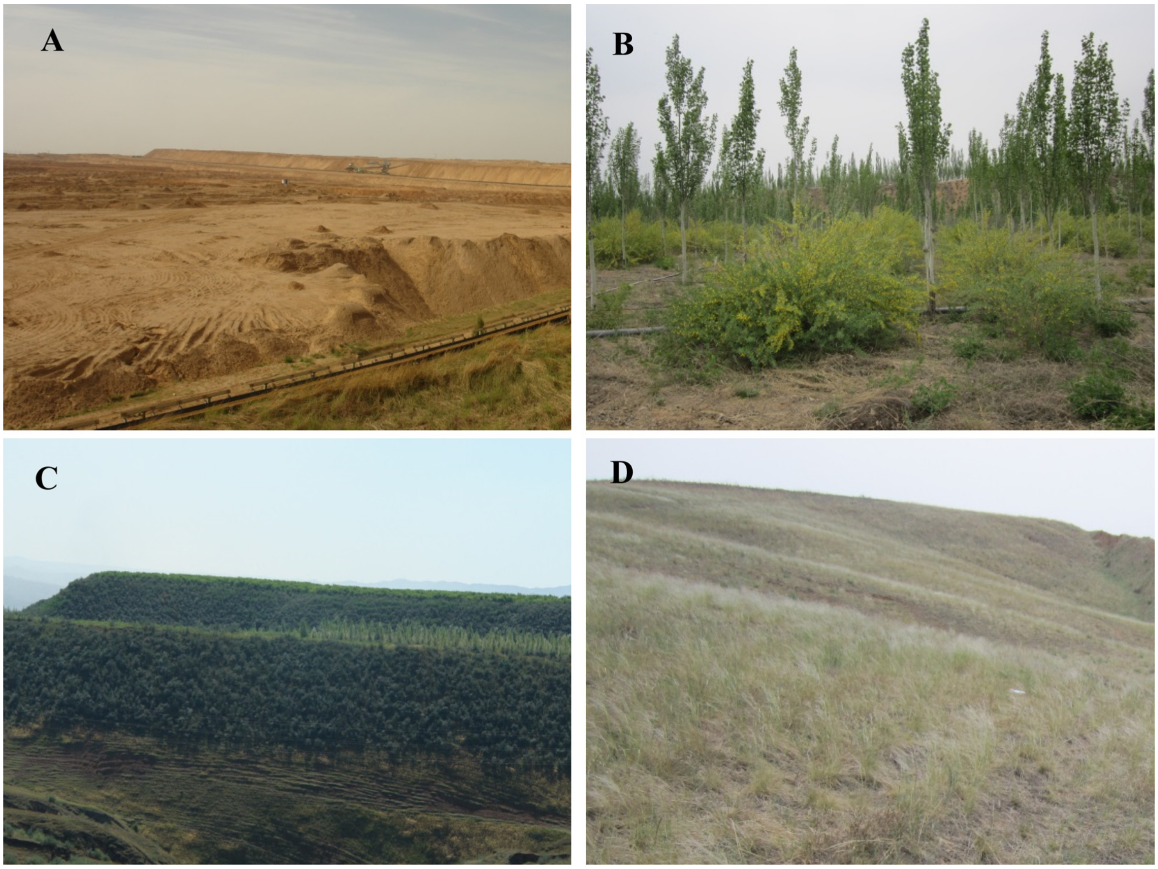

Fig. 1. View of the refuse dump (A), revegetaion (B), the final re-vegetation system (C) and a reference site with native vegetation (D) at the Heidaigou Opencast Coal Mine.

Table 1
Basic information about the sample plots.

Site	Northern	Eastern	Western	Abandoned land	New	Natural vegetation community
The elevation of the plane (m)	1275	1275	1260	1275	1275	1172
Area of occupied land (hm^2)	197	210	170	15	180	–
Year reclamation began	1995	1998	2003	2005	2014	–

cover, and the biomass and height of each species in the artificial vegetation communities were measured in the refuse dumps, the abandoned land site and a control site with natural vegetation. The soil sampling sites were the same sites where the plant observation plots were set up, and three replicate soil samples were collected from the sites of different ages at a depth of 0–50 cm during the 2013 growing season. Air-dried soil samples were sieved through a 2 mm screen and used for further analysis. Particle size was assessed with a pipette method, and soil bulk density was determined by inserting a metallic core (0.05 m in depth and diameter) into the soil. The maximum water-holding capacity (WHC) was also estimated using the same cutting ring in intact soil. The soil cutting ring was closed at one end with a fine mesh and open at the other end and was saturated with water for 6 h. Then, the surplus water was absorbed using a sand bed, and the remaining water in the soil represented the maximum WHC (Öhilinger, 1996). In the sampled plots, soil water contents were measured using the oven-drying method (Nanjing Institute of Soil Research, 1980). Soil samples were taken from core samples and dried at 105 °C for 24 h. Then, soil samples were obtained at 16 different depths: 10, 20, 40, 60, 80, 100, 120, 140, 160, 180, 200, 220, 240, 260, 280 and 300 cm, with 3 replicates for each depth.

Soil organic carbon (SOC) was determined through the dichromate oxidation method of Walkley–Black (Nelson and Sommers, 1982). Soil soluble salts were analyzed using methods described by the Nanjing Institute of Soil Research, Chinese Academy of Sciences (1980). Total N was measured using a Kjeltec system with a 1026 Distilling Unit (Tecator AB, Höganäs, Sweden). Soil phosphorus (P) and potassium (K) were measured using standard methods for observation and analysis developed by the Chinese Ecosystem Research Network (CERN) (Liu, 1996). The Scheibler calcimeter method was employed to measure the $CaCO_3$ content of the soil (Rowell, 1994).

In this study, the Simpson index (D; Simpson 1949), Shannon–Wiener index (H′; Peet 1974), and Pielou evenness index (E; Pielou 1975) were used to measure plant diversity ($D = 1 - \sum P_i^2$; $H' = -\sum P_i \ln P_i$; and $E = (-\sum P_i \ln P_i)/\ln S$, where p_i is the relative importance value (IV) of species I (relative height/relative coverage), and S is the total number of species i in the quadrat, i.e., an abundance index). Beta (β)-diversity refers to species replacement with environmental gradient changes. Some scholars refer to this as the species turnover rate, species replacement rate, or rate of biotic change. This rate can be described using binary attribute data or quantitative data. In this study, well-defined binary attribute data in the form of the Sorenson index (C_J) and two widely used quantitative variables, C_N and C_{MH}, were employed for these calculations (Whittaker 1972). In the Sorenson index, $C_J = j/(a+b)$, where j is the number of species shared by two communities, and a and b are the species numbers in community A and community B, respectively. In the Bray–Curtis index (Bray and Curtis, 1957), $C_N = 2jN/(aN + bN)$, where aN is the species number in sample plot A, bN is the species number in sample plot B, and jN is the sum of the smallest individual number of species shared by sample plots A (jaN) and B (jbN). In this calculation, the IV of the species was used to replace individual numbers according to the Morisita–Horn index (Li, 2001). In the Morisita-Horn index, $C_{MH} = 2\sum(an_i{\cdot}bn_i)/[(da + db)\,aN{\cdot}bN]$, where aN and bN are the same as in the above formula, an_i and bn_i are the individual numbers of species in sample plots A and B, respectively (replaced by IV), and $da = \sum an_i^2/aN^2$; $db = \sum bn_i^2/bN^2$.

The recovery rates of the key parameters of the soil characteristics were modeled using linear relationships or a polynomial model. The recovery time was obtained by estimating the number of years required to undergo a 100% change to reach the level of the asymptote. The correlations between the vegetative features and the soil parameters at the different revegetated sites were determined through Pearson's correlation analysis. The statistical analyses were performed using SPSS 13.0 (SPSS Inc., Chicago, IL, USA) and Origin 7.0 (OriginLab, Northampton, MA, USA) software.

3. Results

3.1. Temporal and spatial variations of vegetation

The number of plant species present increased linearly with time (Table 2), and 12 species were recorded after 20 years of revegetation. In addition, the probability of new plant species being observed gradually diminished, and the species composition gradually tended toward a relatively balanced state. A total of 14 plant species were identified in the natural vegetation communities, and the main plant life forms consisted of perennial species (71.4%, including *Astragalus meliltoides, Thermopsis lanceolata, Pennisetum centrasiaticum, Bothriochloa ischaemum, Lespedeza potaniniiv, Medicago sativa, Artemisia giraldii, Calamagrostis epigejos, Setaria viridis* and *Leymus secalinus*), annual plants (21.4%, including *Corispermum tylocarpum, Eragrostis poaeoides* and *Salsola ruthenica*), and a few xerophytic semi-shrubs (7.2%), such as *Hippophae rhamnoides*. In the initial stages, the planted shrubs grew rapidly and became established on the refuse dumps. They reached the highest mean species richness of 5.85 ± 1.21 in the abandoned land over a period of 10 years, which then gradually decreased to 1.97 ± 0.52 in the northern refuse dumps established in 1995a. However, the maximum coverage of $25.05 \pm 1.41\%$ occurred over a period of 17 years, and the coverage then gradually decreased. After 20 years, the coverage of planted shrubs was reduced from the highest level to $8.64 \pm 1.30\%$. As shown in Table 1, there was a linear increasing tendency of the species richness and coverage of herbaceous species with time of revegetation. In general, no significant difference in shrub and herbaceous species richness was observed between the abandoned land (2005a) and the western (2003a), eastern (1998a) and northern (1995a) refuse dumps ($P > 0.05$). However, significant differences were observed between the early revegetated sites (1995a and 1998a) and the later groups (2003a and 2005 sites) ($P < 0.05$). In contrast, a significant difference in shrub and herbaceous species coverage was detected between the different sites ($P < 0.05$). Likewise, differences in biomass occurred at the different sites. The biomass of naturally established shrubs increased from $0.15 \pm 0.03\ kg\ m^{-2}$ to $1.55 \pm 0.12\ kg\ m^{-2}$ in 17-year-old revegetation and then gradually decreased to $0.75 \pm 0.11\ kg\ m^{-2}$. An apparent increase in the biomass of herbaceous species occurred with time; however, no significant difference was observed among the different sites ($P > 0.05$).

Measurement of the β-diversity of the different-aged vegetation showed that after 20 years of succession (planted in 1995), plant species diversity reached a relatively high level (D = 0.814–0.886, H′ = 1.792–2.393), whereas the vegetation community in the new refuse dumps exhibited lower diversity (D = 0.206–0.327, H′ = 0.309–0.474) (Table 3). However, the planted vegetation in the refuse dumps still presented a lower diversity index, suggesting that the structure of the planted vegetation was relatively simple and that community stability was weak and susceptible to disturbance. Table 2 also lists the Sorenson, Bray–Curtis, and Morisita–Horn indices used to measure the β-diversity of vegetation planted in the five different periods. Although the first index is calculated from a formula for binary data and the latter two are the calculated using quantitative data, the comparative results are the same. This quantitative method is an approach that is used to study the developing dynamics of species diversity in the rehabilitation process of the degraded ecosystem through time-space mutual replacement. The results show that the β-diversity indices between each of the restored refuse dumps planted at different times are gradually increasing, indicating that the restoration plants are developing toward the natural vegetation type in the rehabilitation process. The difference in the species composition was greatest in 1998a (CMH = 1.108, CN = 0.719, CJ = 1.052), meaning that the species turnover rate was highest, followed by the b-diversity indices for the 1995a vegetation. Therefore, during the 20 years of vegetation succession, the time interval of 15–20 years showed the highest species turnover rate; thus, elements of community structure, such as the species composition and structure layer, exhibited the greatest change in these stages.

3.2. Changes in soil properties and the spatial distribution of the soil water content

The variations in soil texture, nutrient states, and other features observed at the different sites were remarkable (Table 4). Twenty years after the vegetation had been planted, the proportion of sand-sized particles in the topsoil (0–20 cm depth) had decreased from 81.6% to 69.6% in the vegetated areas, and the proportion of silt and clay had increased from 10.4% and 8.0% to 18.6% and 11.9%, respectively. For bulk density, no significant differences were found among the different sites ($P > 0.05$); however, the bulk density was reduced after vegetation reclamation. The establishment of vegetation enhanced the WHC. The maximum WHC increased from 9.26% to 18.0% at the 20-year-old site. The maximum WHC at the 20-year-old site was 75.8% of the value at the reference site. In agreement with the WHC results, the topsoil water content increased with this successional gradient defined by the 0- to 20-year-old revegetated sites and the non-degraded reference site with native vegetation. The soil water content at the 20-year-old site was 76.8% of that in the reference site. SOM, total N, P, K, $CaCO_3$, total salt, EC and pH increased significantly in the period following stabilization. Furthermore, significant differences were found among the six sites ($P < 0.05$). However, the levels of these parameters were lower than at the reference site (natural vegetation communities) by 17.7–74.8%. In general, the incremental rates of these parameters were markedly higher in the younger sites than in the older sites.

The spatial distribution of the soil water content in the soil profile also changed with the time since stabilization (Fig. 2). The mean soil water content in relation to depth was significantly higher in the oldest vegetated areas (1995 site, 2.3–5.9%) than in the younger vegetated areas (e.g., 2014 site, 1.1–2.5%) in the shallow layer (0–60 cm); however, in the deeper layers (60–200 cm), the mean soil water content in the oldest vegetated areas (4.2–8.1%) was significantly lower than in the younger vegetated areas (5.1–12.5%).

Table 2
Changes in total species number and the richness, coverage and biomass of shrub and herbaceous species after revegetation on refuse dumps at each different-aged site.

Year Stabilized	Northern (1995)	Eastern (1998)	Western (2003)	Abandoned land (2005)	New (2014)	Natural
Total species number	12.17 ± 1.61^{a}	7.38 ± 0.84^{b}	6.15 ± 1.03^{b}	5.09 ± 1.18^{bc}	2.27 ± 0.66^{c}	14.51 ± 1.47^{a}
Richness of shrub species	1.97 ± 0.52^{bc}	3.02 ± 0.95^{ab}	4.05 ± 1.19^{ab}	5.85 ± 1.21^{a}	1.06 ± 0.14^{c}	1.03 ± 0.20^{c}
Richness of herbaceous species	10.63 ± 1.02^{b}	4.07 ± 1.14^{c}	3.33 ± 0.67^{cd}	2.39 ± 0.46^{cd}	1.29 ± 0.18^{d}	13.27 ± 1.38^{a}
Shrub cover (%)	8.64 ± 1.30^{d}	25.05 ± 1.41^{a}	16.04 ± 0.69^{b}	13.26 ± 0.51^{c}	3.46 ± 0.54^{e}	2.51 ± 0.84^{e}
Herb cover (%)	75.65 ± 1.28^{b}	50.24 ± 1.10^{c}	30.42 ± 1.63^{d}	20.52 ± 1.21^{e}	5.03 ± 1.39^{f}	87.18 ± 1.45^{b}
Shrub biomass ($kg\,m^{-2}$)	0.75 ± 0.11^{c}	1.55 ± 0.12^{a}	1.07 ± 0.05^{b}	0.85 ± 0.14^{bc}	0.15 ± 0.03^{d}	0.25 ± 0.10^{d}
Herb biomass ($kg\,m^{-2}$)	0.69 ± 0.16^{ab}	0.46 ± 0.11^{bc}	0.25 ± 0.07^{cd}	0.15 ± 0.04^{d}	0.05 ± 0.01^{d}	0.84 ± 0.14^{a}

Values represents means ± SE. Different small letters denote significant difference among different-aged site ($P < 0.05$).

Table 3
The comparison of plant diversity and the measurement of b-diversity after revegetation on refuse dumps at each different-aged site.

Year Stabilized	Northern (1995)	Eastern (1998)	Western (2003)	Abandoned land (2005)	New (2014)	Natural
Simpson index (D)	0.814–0.886	0.792–0.833	0.713–0.792	0.694–0.715	0.206–0.327	0.896–0.921
Mean	0.855	0.822	0.792	0.714	0.267	0.939
Shannon-Wiener index	1.792–2.393	1.833–2.015	1.752–1.874	1.479–1.622	0.309–0.474	2.115–2.673
Mean	2.192	1.933	1.876	1.630	0.392	2.591
Pielou evenness	0.724–0.936	0.692–0.853	0.593–0.767	0.557–0.704	0.104–0.211	0.907–0.982
Mean	0.873	0.795	0.696	0.645	0.158	0.954
Sorenson index		1.108	0.581	0.545	0.339	0.948
Bray-Curtis index		0.719	0.196	0.154	0.132	0.634
Morisita Horn index		1.052	0.391	0.226	0.208	0.864

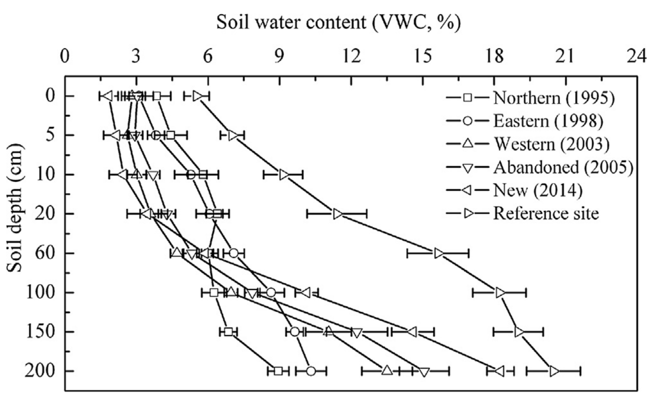

Fig. 2. The spatial variation of soil water content in different-aged sites.

3.3. *Recovery rate of topsoil and vegetation characteristics*

The recovery rates of the six key physical soil properties (sand, silt and clay contents, bulk density, WHC, and topsoil water content) and eight key chemical properties (pH, total N, P, and K, organic C, total salt, $CaCO_3$ and EC) in the chronosequence of revegetated sites at the refuse dumps of the Heidaigou open pit coal mine were modeled with linear or polynomial equations (see Appendix A and B in the Supplementary material). The proportion of sand in the soil texture was estimated to require 23 years to reach the value at the reference site (Table 5). The proportions of clay and silt increased with site age and would require 19 and 25 years, respectively, to reach 100% of the values at the reference site. The bulk density was slow to recover and could take over 59 years to reach the value at the reference site. The topsoil water content was estimated to require 34 years to reach the value at the reference site, while WHC would require 32 years. For soil chemical properties such as soil pH, total N, P and K, at least 21, 39, 30 and 24 years, respectively, would be required for recovery. The times estimated for the recovery of organic C, total salt and $CaCO_3$ were nearly the same, at 24, 26 and 26 years, respectively. The recovery of EC took place more rapidly, requiring 21 years to reach the value at the reference site. The vegetation traits exhibited clear relationships with the recovery chronosequence (see Appendix C Supplementary material). As shown in Table 5, the recovery of the total species number would take 25 years to reach 100% of the value at the reference site. Furthermore, the recovery of shrub and herbaceous species richness was estimated to require 21 and 23 years, respectively, whereas the recovery of shrub and herbaceous species coverage and biomass would require 23 and 29 years and 22 and 23 years, respectively, to attain the values at the reference site.

4. Discussion

Vegetative stabilization is a widely accepted technique for controlling soil erosion and stabilizing the dump slope of coal mine wastes (Nicolau, 2003; Singh and Singh, 2006; Tischew and Kirmer, 2007; Hendrychova, 2008). The objective of restoration was to achieve recovery corresponding to the pre-mining ecosystem, and the ecosystem characteristics that are typically measured to assess recovery are related to the composition, structure, and pattern of vegetation (Hobbs, 1999; Cooke and Johnson, 2002). In this study, with the succession of planted vegetation, the plant community structure gradually changed shifted a primarily shrub composition to a complex structure dominated by herbaceous species as the coverage and biomass of herbaceous plants increased over time after the initial establishment of the vegetation. Although the biomass and the coverage of shrub species increased in the first 16 years, the shrub vegetation then decreased, and a relatively stable state was achieved. As the surface soil conditions changed, the formation of biological soil crusts and an increasing thickness of litter prevented water from infiltrating into the deeper soil layers (Li et al., 2002; Wang et al., 2007), where shrubs establish deep root systems (Zhang et al., 2009; Huang and Zhang, 2015). Therefore, a decline in the soil water content occurs in deeper soil layers, as shown in Fig. 1. Thus, the deep-rooted shrubs are gradually eliminated from the community and replaced by annual and perennial herbaceous species, such as *Stipa bungeana, Bothriochloa ischaemum, Pennisetum centrasiaticum*. The measurement of b-diversity demonstrated that there was an important species turnover stage in the succession of planted vegetation during the first 20 years, which occurred between 15 and 20 years after the establishment of reclamation vegetation. Planted trees and shrubs gradually declined during vegetation succession, and herbs (whose coverage and richness were greater than those of shrubs) became dominant, making the community structure more complex. Hence, temporally dynamic

Table 4
Changes in texture, nutrient, physicochemical properties of topsoil, and its habitat after revegetation on refuse dumps at each different-aged site.

Year Stabilized	Partial Size (%)			Nutrient Contents (g/kg)			OrganicC (%)	pH	Total Salt (%)	$CaCO_3$	EC	WHC	Bulk Density (number/km^2)	Topsoil water content (%)
	Sand	Silt	Clay	N	P	K								
Northern (1995)	69.57 ± 1.14^{d}	18.55 ± 1.36^{ab}	11.88 ± 1.12^{b}	0.40 ± 0.03^{ab}	5.98 ± 0.32^{b}	10.63 ± 0.21^{b}	6.51 ± 0.22^{b}	8.65 ± 0.26^{a}	1.53 ± 0.24^{bc}	2.22 ± 0.09^{b}	0.60 ± 0.06^{a}	18.00 ± 0.85^{b}	1.29 ± 0.26^{b}	31.13 ± 0.81^{b}
Eastern (1998)	72.16 ± 1.39^{c}	17.74 ± 1.10^{b}	10.10 ± 1.42^{cd}	0.28 ± 0.05^{bc}	5.32 ± 0.28^{b}	8.13 ± 0.30^{c}	4.39 ± 0.33^{c}	8.22 ± 0.21^{ab}	1.26 ± 0.27^{abc}	1.93 ± 0.04^{c}	0.55 ± 0.05^{ab}	15.39 ± 1.28^{bc}	1.46 ± 0.29^{a}	27.11 ± 1.21^{c}
Western (2003)	75.07 ± 1.35^{b}	15.09 ± 1.40^{c}	9.84 ± 1.64^{bc}	0.19 ± 0.06^{c}	4.21 ± 0.35^{c}	6.22 ± 0.23^{d}	3.52 ± 0.42^{d}	7.83 ± 0.24^{bc}	1.01 ± 0.21^{bc}	1.48 ± 0.06^{d}	0.43 ± 0.04^{bc}	13.22 ± 1.74^{cd}	1.54 ± 0.13^{a}	21.34 ± 1.17^{d}
Abandoned land (2005)	79.22 ± 1.36^{a}	12.56 ± 1.12^{cd}	8.22 ± 1.47^{d}	0.17 ± 0.04^{c}	3.67 ± 0.22^{c}	5.24 ± 0.22^{e}	3.17 ± 0.25^{d}	7.70 ± 0.29^{bc}	0.80 ± 0.25^{c}	1.20 ± 0.05^{e}	0.32 ± 0.04^{cd}	11.92 ± 1.05^{de}	1.68 ± 0.29^{a}	17.46 ± 1.66^{e}
New (2014)	81.63 ± 1.32^{a}	10.39 ± 1.15^{d}	7.98 ± 1.01^{d}	0.15 ± 0.04^{c}	2.68 ± 0.32^{d}	3.19 ± 0.29^{f}	2.18 ± 0.30^{e}	7.22 ± 0.23^{c}	0.70 ± 0.21^{c}	0.87 ± 0.05^{f}	0.25 ± 0.06^{d}	9.26 ± 1.06^{e}	1.80 ± 0.23^{a}	11.78 ± 1.03^{f}
Reference site	62.76 ± 1.29^{e}	19.15 ± 1.03^{a}	18.09 ± 1.02^{a}	0.51 ± 0.07^{a}	7.43 ± 0.22^{a}	12.68 ± 0.34^{a}	7.55 ± 0.30^{a}	8.77 ± 0.20^{a}	1.91 ± 0.27^{a}	2.79 ± 0.05^{a}	0.67 ± 0.06^{a}	23.74 ± 0.83^{a}	2.12 ± 0.21^{a}	40.53 ± 1.65^{a}

Values represents means ± SE. Different small letters denote significant difference among different-aged site ($P < 0.05$).

changes in plant diversity can reflect the successional features of vegetation to some extent (Li et al., 2014). This finding may provide basic knowledge for the management of planted vegetation in the refuse dumps of the Heidaigou open pit coal mine.

Changes in vegetation have always been associated with soil properties (Chodak and Niklinska, 2010), as demonstrated by the marked changes in vegetation and soil properties after the establishment of plants in the refuse dumps. With an increase in revegetation age, the percentage of silt and clay particles in the soil significantly increased, while the content of coarse sand particles decreased, and the texture of the soil tended to gradually become more similar to that of the soil in the natural vegetation zone. A fine soil texture facilitates the formation of soil aggregates and increases soil porosity (Li et al., 2007), in addition to improving the topsoil's water-holding capacity, which in turn enhances the water content of the topsoil and, ultimately, the overall productivity of the soil. Furthermore, fine-textured soil supports the growth of herbaceous species better than coarse-textured soil, which facilitates the growth of xerophytic shrubs (Brown, 1997). The increasing clay and silt contents in the soil texture over time after revegetation are also positively correlated with the improvement of most soil parameters, such as increases in soil organic matter and soil nutrient status (Parton et al., 1988; Burke et al., 1990; Brown, 1997). Our results revealed that organic C, $CaCO_3$ and EC increased with an increasing recovery time. In contrast, pH decreased with an increasing recovery time. Most of the soil parameters measured in this study were strongly related to site age (Table 3), suggesting that a revegetative approach is effective for enhancing topsoil recovery in opencast mining regions. Furthermore, complex interactions exist among the examined parameters. For example, SOC plays an important role in plant colonization and establishment through acting as a reservoir of essential elements, particularly N and P. SOC is a source of cation exchange capacity and soil pH buffering and enhances soil porosity, improving the movement of oxygen through the soil (Bohn et al., 2001; Li et al., 2004, 2007).

The recovery of vegetation and soil properties determines how quickly a pre-mining ecosystem can be restored and whether such a restoration goal is ecologically possible and sustainable (Cooke and Johnson, 2002; Boyer and Wratten, 2010). Based on the recovery rates of soil physicochemical properties and vegetation characters observed in revegetated refuse dumps of different ages (1995, 1998, 2003, 2005 and 2014) at the Heidaigou opencast coal mine, the recovery of topsoil and vegetation characteristics were estimated to require between 19 and 59 years (Table 4). Recovery was slowest for the bulk density (59 years). This feature of recovery has been confirmed by other researchers (Li et al., 2007). Among the examined soil characteristics, the recovery of the clay content, topsoil water content, depth of the soil and crust, and total salt content were determined to require approximately 20–30 years to reach the levels in the native ecosystem (reference site). The revegetated area provides sufficient amounts of fine particles and litter for accumulation on the stabilized surface in the study region (Li et al., 2004, 2007). Additionally, the development of biological soil crusts facilitates soil formation on the stabilized refuse dump surface, and the increases in clay content and soil depth in turn enhance the topsoil water-holding capacity and increase the water content in the topsoil layer. Over a 20-year period, we found that the recovery of most soil characteristics was more rapid in early stages than in later stages, which has also been noted in some reports on the recovery of landslip sites in moist tropical forest ecosystems (e.g., Singh et al., 2001) and in the dry hill county of New Zealand after landslip erosion (Sparling et al., 2003). The recovery of the vegetation was the same as that of the soil properties, although the vegetation was more susceptible to meteorological factors. Overall, the findings of the present study indicated that the maximum recovery of some

Table 5
The recovery rate of soil and vegetation characteristics on refuse dumps at each different-aged site.

parameters		Rate parameter	Asymptote	Years of recovery
Soil properties	Sand (%)	0.070	92.140	22.890
	Silt (%)	0.050	3.970	25.060
	Clay (%)	0.030	4.010	19.310
	Bulk Density (%)	0.030	1.890	59.000
	WHC (%)	0.480	8.250	32.270
	Topsoil water content (%)	0.850	11.420	34.250
	pH	0.080	7.060	21.380
	Total N (g kg^{-1})	0.011	0.086	38.550
	Total P (g kg^{-1})	0.170	2.410	29.530
	Total K (g kg^{-1})	0.014	3.230	23.540
	Organic C (g kg^{-1})	0.010	2.250	24.040
	Total Salt (g kg^{-1})	0.001	0.592	25.560
	CaCO3 (g kg^{-1})	0.098	0.592	25.760
	EC (m s^{-1})	0.001	0.240	21.310
Vegetative traits	Total species number	0.015	2.314	25.490
	Richness of shrub species	0.033	1.062	21.470
	Richness of herbaceous species	0.031	1.296	23.480
	Shrub cover (%)	0.107	3.448	23.020
	Herb cover (%)	0.177	5.032	21.920
	Shrub biomass (kg m^{-2})	0.004	0.148	29.390
	Herb biomass (kg m^{-2})	0.002	0.049	22.540

topsoil properties and vegetation properties could reach the levels in the native ecosystem in 30 years (Table 2).

This result was consistent with some research from other regions (Gao et al., 1998; Singh et al., 2001; Sparling et al., 2003; Pensa et al., 2004). However, the recovery rate observed in this study was slower than that reported in revegetated desert ecosystems (Li et al., 2007). This disparity can be explained by regional heterogeneity, which mainly reflects the influence of rainfall levels. For example, in arid areas where the rainfall is less than 200 mm, Li et al. (2007) suggested that soil recovery is a slow process, potentially requiring between 70 and 245 years to recover to the level of the native desert steppe. Thus, conservation of the soil habitat is the primary strategy for land managers. Based on this theory, many researchers have suggested the use of imported soil as cover material as a quick and simple solution (Whitbread-Abrutat, 1997; Li, 2006; Zhang and Chu, 2010), especially in extreme regions where the rainfall levels are less than 100 mm (Li, 2005). In addition, the physical conditions of overburden materials (usually amended with metallic waste) are often unfavorable for plant growth because they are bare and exhibit an inadequate particle size distribution and high gravel content (Bradshaw and Chadwick, 1980). However, the costs involved in topsoil removal, storage and protection are high. Actually, at least 50 cm of loessal soil or mixtures of topsoil, subsoil and loessal soil (named as raw mixed-loess soil) must be directly spread over surface-mined waste (Li et al., 2013). The alternative is to use benign materials, which may also be waste products, that are often locally obtainable as complete cover (Bradshaw, 2000), followed by assisted reclamation. Therefore, the application of new soil surface repair technologies, such as the use of biological soil crusts, is in full swing at some important energy bases, roads, airports and other sites in arid desert areas (Li et al., 2014). These methods have been effective in addressing sand erosion and land degeneration to some extent, and more importantly, they are inexpensive and do not harm the environment. However, in semiarid areas where rainfall ranges from 450 to 600 mm, such as the area examined in this study, the recovery rate of most soil characteristics is generally the same as the recovery rate of vegetation properties. Thus, soil recovery and vegetation recovery are equally important in eco-restoration activities, and in these areas, the key steps to successful reclamation therefore involve suitable substrate amendment or the selection of appropriate plant species for revegetation. If the two approaches are combined, the recovery rate should improve, explaining why the recovery time required to attain levels similar to those in native ecosystems was estimated to be less than 50 years. In the Heidaigou opencast coal mine, the topsoil was removed first, followed by the subsoil, as separate layers, which were then stored until closure of the mine, when the abandoned open pit was backfilled, and trees, shrubs and grass were then planted in the soil (Huang et al., 2015). When rainfall is greater than 600 mm, the use of manual accelerated soil-reconstruction approaches, such as coverage with imported soil or biological soil crusts, is not necessary. Instead, the recovery of plant community characteristics is especially important, and the key steps for successful reclamation involve the selection of appropriate plant species (Dowarah et al., 2009; Evans et al., 2013). Restoration in opencast mining is mostly carried out with fast-growing herbaceous species to control the erosion of embankments during rainfall, especially immediately after their construction. For the artificial introduction of plants, a successful approach to reclamation is to select species that are well adapted to the local environment (Tordoff et al., 2000; Dutta and Agrawal, 2003). In the successful afforestation on the Amarkantak mine area, *Gravillea pteridifolia, Eucalyptus camaldulensis, Pinus roxburghii* and *Pongamia pinnata* were observed to be the most suitable species on the basis of growth performance (Chaturvedi, 1983). In the coal mine spoils of Kansas, Rogers (1983) tested the use of 13 species (both native and exotic) for 22 years and found that *Platanus occidentalis, Juniperus virginiana, Quercus macrocarpa, Pinus taeda* and *P. echinata* showed good growth and recommended these species for reclamation purposes. Furthermore, some of these plants have evolved biological mechanisms to resist, tolerate or thrive on toxic metalliferous substrates and are known as metallophytes (Whiting et al., 2004). Ideal choices include pioneer species such as *Cynodon dactylon, Festuca rubra, Agrostis tenuis, Agrostis stolonifera, Typha latifolia* and *Phragmites australis.* Gramineous grasses and legumes are also favored options because of their adaptation to nutrient-deficient soil and their rapid growth. In addition, plants that provide economic benefits might also be considered for planting in these areas, such as vegetables, fruit trees or other economically useful crops. For example, medical herbs are commonly grown on reclaimed lands, which is a practice that is directly encouraged by local governments and should continue to be encouraged (Miao and Marrs, 2000; Li, 2006).

5. Conclusion

Revegetation activities performed at the refuse dumps of the Heidaigou open pit coal mine improved the soil environment for the colonization and establishment of plant species. Plant coverage, biodiversity and biomass increased significantly with increasing site age. The mean soil water content in relation to depth was significantly higher in the oldest vegetated areas (1995 site, 2.3–5.9%) than in the younger vegetated areas (e.g., 2014 site, 1.1–2.5%) in the shallow layer (0–60 cm); however, in the deeper layers (60–200 cm), the mean soil water content in the oldest vegetated areas (4.2–8.1%) was significantly lower than in the younger vegetated areas (5.1–12.5%). The proportions of silt and clay, depth of the topsoil and biological soil crusts, and concentrations of soil organic C, K, total N and total P increased with the years since revegetation. The estimated time required for both plant traits and soil properties in the revegetated site to reach the same levels as in the reference site was between 23 and 25 years. These results suggest that soil recovery and vegetation recovery are equally important in eco-restoration activities in these semiarid areas, and the restoration of soil and vegetation is therefore a crucial issue for land managers.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2016.06.108.

Reference:

Öhilinger, R., 1996. Maximum water-holding capacity. In: Schinner, F., Öhlinger, R., Kandeler, E., Margesin, R. (Eds.), Methods in Soil Biology. Springer, Berlin.
Beale, C.M., Lennon, J.J., Gimona, A., 2008. Opening the climate envelope reveals no macroscale associations with climate in European birds. Proc. Natl. Acad. Sci. U. S. A. 105, 14908–14912.
Bohn, H.L., McNeal, B.L., O'Connor, G.A., 2001. Soil Chemistry. Wiley, New York, pp. 155–171.
Boyer, S., Wratten, S.D., 2010. The potential of earthworms to restore ecosystem services after opencast mining—a review. Basic Appl. Ecol. 11, 196–203.
Bray, J.R., Curtis, J.D., 1957. An ordination of upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.
Bradshaw, A., Chadwick, M.J., 1980. The Restoration of Land. Blackwell Scientific Publications, Oxford.
Bradshaw, A., 1997. Restoration of mined lands-using natural processes. Ecol. Eng. 8 (4), 255–269.
Bradshaw, A., 2000. The use of natural processes in reclamation—advantages and difficulties. Landsc. Urban Plan 51, 89–100.
Brown, J.F., 1997. Effect of experimental burial on survival, growth, and resource allocation of three species of dunes plants. J. Ecol. 85, 151–158.
Burke, I.C., Schimel, D.S., Yonker, C.M., Parton, W.J., Joyce, L.A., Lauenroth, W.K., 1990. Regional modeling of grassland biogeochemistry using GIS. Landsc. Ecol. 4, 45–54.
Chaturvedi, J.K., 1983. Afforestation of bauxite mined area in central India. Indian For. 105 (7), 465–484.
Chodak, M., Niklinska, M., 2010. The effect of different tree species on the chemical and microbial properties of reclaimed mine soils. Biol. Fertil. Soils 46, 555–566.
Cooke, J.A., Johnson, M.S., 2002. Ecological restoration of land with particular reference to the mining of metals and industrial minerals: a review of theory and practice. Environ. Rev. 10, 41–71.
Copeland, J.P., McKelvey, K.S., Aubry, K.B., Landa, A., Persson, J., Inman, R.M., Lofroth, E., Golden, H., Squires, J.R., Magoun, A., Schwartz, M.K., Wilmot, J., Copeland, C.L., Yates, R.E., Kojola, I., May, R., 2010. The bioclimatic envelope of the wolverine (*Gulo gulo*): do climatic constraints limit its geographic distribution? Can. J. Zool. 88, 233–246.
Dowarah, J., Deka Boruah, H.P., Gogoi, J., Pathak, N., Saikia, N., Handique, A.K., 2009. Eco-restoration of a high-sulphur coal mine over burden dumping site in northeast India: a case study. J. Earth Syst. Sci. 118, 597–608.
Dutta, R.K., Agrawal, M., 2003. Restoration of opencast coal mine spoil by planting exotic tree species: a case study in dry tropical region. Ecol. Eng. 21, 143–151.
Evans, D.M., Zipper, C.E., Burger, J.A., Strahm, B.D., Villamagna, A.M., 2013. Reforestation practice for enhancement of ecosystem services on a compacted surface mine: path toward ecosystem recovery. Ecol. Eng. 51, 16–23.
Ezeaku, P.I., 2012. Evaluating the influence of open cast mining of solid minerals on soil, landuse and livelihood systems in selected areas of Nasarawa State, North-Central Nigeria. J. Ecol. Nat. Environ. 4, 62–70.
Gao, L., Miao, Z., Bai, Z., Zhou, X., Zhao, J., Zhu, Y., 1998. A case study of ecological restoration at the Xiaoyi Bauxite Mine, Shanxi Province, China. Ecol. Eng. 11, 221–229.
Hendrychova, M., 2008. Reclamation success in post-mining landscapes in the Czech Republic: a review of pedological and biological studies. J. Landsc. Stud. 1, 63–78.
Hobbs, R.J., 1999. Restoration of disturbed ecosystems. In: Walker, L. (Ed.), Ecosystems of the World 16: Ecosystems of Disturbed Ground. Elsevier, Amsterdam.
Huang, L., Zhang, Z.S., 2015. Stable isotopic analysis on water utilization sources of two xerophytic shrubs in a revegetated desert area: Tengger Desert, China. Water 7, 1030–1045.
Huang, L., Zhang, P., Hu, Y.G., Zhao, Y., 2015. Vegetation succession and soil infiltration characteristics under different restoration models on refuse dumps at the Heidaigou Opencast Coal Mine. Global Ecol. Conserv. 4, 255–263.
Izquierdo, I., Caravaca, F., Alguacil, M.M., Hernández, G., Roldán, A., 2005. Use of microbiological indicators for evaluating success in soil restoration after revegetation of a mining area under subtropical conditions. Appl. Soil Ecol. 30, 3–10.
Johnson, E.A., Miyanishi, K., 2008. Testing the assumptions of chronosequences in succession. Ecol. Lett. 11, 419–431.
Josa, R., Jorbab, M., Vallejo, V.R., 2012. Opencast mine restoration in a Mediterranean semi-arid environment: failure of some common practices. Ecol. Eng. 42, 183–191.
Krummelbein, J., Horn, R., Raab, T., Bens, O., Hüttl, R., 2010. Soil physical parameters of a recently established agricultural recultivation site after brown coal mining in Eastern Germany. Soil Tillage Res. 111 (1), 19–25.
Lam, P., 2005. Energy in China: development and prospects. China Perspect. 59, 14–25.
Larsen, D.P., Kincaid, T.M., Jacobs, S.E., Urquhart, N.S., 2001. Designs for evaluating local and regional scale trends. Bioscience 51, 1069–1078.
Li, X.R., Wang, X.P., Li, T., Zhang, J.G., 2002. Microbiotic soil crust and its effect on vegetation and habitat on artificially stabilized desert dunes in Tengger Desert, North China. Biol. Fertil. Soils 35, 147–154.
Li, X.R., Xiao, H.L., Zhang, J.G., Wang, X.P., 2004. Long-term ecosystem effects of sandbinding vegetation in the Tengger Desert, northern China. Restor. Ecol. 12, 376–390.
Li, X.R., He, M.Z., Duan, Z.H., Xiao, H.L., Jia, X.H., 2007. Recovery of topsoil physiochemical properties in revegetated sites in the sand-burial ecosystems of the Tengger Desert, northern China. Geomorph 88, 254–265.
Li, H., Shao, H., Li, W., Bai, Z., 2012. Improving soil enzyme activities and related quality properties of reclaimed soil by applying weathered coal in opencast mining areas of the Chinese Loess Plateau. Clean-Soil Air Water 40, 233–238.
Li, S.Q., Di, X.Y., Wu, D.M., Zhang, J.T., 2013. Effects of sewage sludge and nitrogen fertilizer on herbage growth and soil fertility improvement in restoration of the abandoned opencast mining areas in Shanxi, China. Environ. Earth Sci. 70, 3323–3333.
Li, X.R., Zhang, Z.S., Tan, H.J., Gao, Y.H., Liu, L.C., Wang, X.P., 2014. Ecological restoration and recovery in the wind-blown sand hazard areas of northern China: relationship between soil water and carrying capacity for vegetation in the Tengger Desert. Sci. China (C) 57, 539–548.
Li, X.R., 2001. Study on shrub community diversity of Ordos Plateau, Inner Mongolia, Northern China. J. Arid Environ. 47, 271–279.
Li, X.R., 2005. Influence of variation of soil spatial heterogeneity on vegetation restoration. Sci. China Ser. D 48, 2020–2031.
Li, M.S., 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.
Lichter, J., 1998. Primary succession and forest development on coastal Lake Michigan sand dunes. Ecol. Monogr. 68, 487–510.
Lindegren, M., Möllmann, C., Nielsen, A., Stenseth, N.C., 2009. Preventing the collapse of the Baltic cod stock through an ecosystem-based management approach. Proc. Natl Acad. Sci. U. S. A. 106, 14722–14727.
Liu, J., Diamond, J., 2005. China's environment in a globalizing world. Nature 435, 1179–1186.
Liu, Z.B., Fan, J.F., 2002. The application of the ecological succession principle on the land reclaimation In the open coal mines. Opencast Min. Technol. 5, 27–30.
Liu, G.S., 1996. Soil Physical and Chemical Analysis and Description of Soil Profiles. Chinese Standard Press, Beijing.
Mbogga, M.S., Wang, X., Hamann, A., 2010. Bioclimate envelope model predictions for natural resource management: dealing with uncertainty. J. Appl. Ecol. 47, 731–740.
Miao, Z., Marrs, R., 2000. Ecological restoration and land reclamation in open-cast mines in Shanxi Province China. J. Environ. Manag. 59, 205–215.
Miao, Z.W., Bai, Z.K., Gao, L., 2000. Ecological rebuilding and land reclamation in surface mines in Shaxi province China. J. Environ. Sci. 12 (4), 486–497.

Moreno-de las Heras, M., Nicolau, J.M., Espigares, T., 2008. Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecol. Eng. 34 (2), 168–178.

Nanjing Institute of Soil Research, Chinese Academy of Sciences, 1980. Analysis of Soil Physic-chemical Features. Shanghai Science and Technology Press, Shanghai, China.

Nelson, D.W., Sommers, L.E., 1982. Total carbon organic carbon and organic matter. In: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis. , 2nd ed. American Society of Agronomy, Madison, Wisconsin, pp. 539–577.

Nicolau, J.M., 2003. Trends in relief design and construction in opencast mining reclamation. Land Degrad. Dev. 14, 215–226.

Parton, W.J., Stewart, J., Cole, V.C., 1988. Dynamics of C, N, P and S in grassland soils: a model. Biogeochemistry 5, 109–131.

Peet, R.H., 1974. The measurement of species diversity. Annu. Rev. Ecol. Evol. Syst. 5, 285–307.

Pensa, M., Sellin, A., Luud, A., Valgma, I., 2004. An analysis of vegetation restoration on opencast oil shale mines in Estonia. Restor. Ecol. 12 (2), 200–206.

Pižl, V., 2001. Earthworm succession in afforested colliery spoil heaps in the Sokolov region, Czech Republic. Rest. Ecol. 9, 359–364.

Pickett, S.T.A., 1989. Space-for-time substitution as an alternative to long-term studies. In: Likens, E. (Ed.), Long-term Studies in Ecology: Approaches and Alternatives. Springer, New York.

Pielou, E.C., 1975. Ecological Diversity. John Wiley and Sons Inc., New York.

Rogers, R.S., 1983. Annual variability in community organization of forest herbs: effects of an extremely warm and dry early spring. Ecology 64, 1086–1091.

Rowell, D.L., 1994. Soil Science: Methods and Applications. Longman and Scientific Technical, New York.

Simpson, E.H., 1949. Measurement of diversity. Nature 163, 688.

Singh, A.N., Singh, J.S., 2006. Experiments on ecological restoration of coal mine spoil using native trees in a dry tropical environment, India: a synthesis. New For. 31, 25–39.

Singh, A., Jha, A.K., Singh, J.S., 2000. Effect of nutrient enrichment on native tropical trees planted on Singrauli Coalfields, India. Restor. Ecol. 8, 80–86.

Singh, K.P., Mandal, T.N., Tripathi, S.K., 2001. Patterns of restoration of soil physicochemical properties and microbial biomass in different landslide sites in the sal forest ecosystem of Nepal Himalaya. Ecol. Eng. 17, 385–401.

Sparling, G., Ross, D., Trustrum, N., Arnold, G., West, A., Speir, T., Schipper, L., 2003. Recovery topsoil characteristics after landslip erosion in dry hill country of New Zealand, and a test of the spacefor-time hypothesis. Soil Biol. Biochem. 35, 1575–1586.

Srivastava, S.C., Jha, A.K., Singh, J.S., 1989. Changes with time in soil biomass C, N and P of mine spoils in a dry tropical environment. Can. J. Soil Sci. 69, 849–855.

Tischew, S., Kirmer, A., 2007. Implementation of basic studies in the ecological restoration of surface-mined land. Restor. Ecol. 15, 321–325.

Tordoff, G.M., Baker, A.J.M., Willis, A.J., 2000. Current approaches to the revegetation and reclamation of metalliferous mine wastes. Chemosphere 41, 219–228.

Vasssilis, Z.A., Wyseure, G.C.L., 1998. Modeling of water and nitrogen dynamics on an undisturbed soil and a restored soil after open-cast mining. Agricult. Water Manage. 37, 21–40.

Wang, X.P., Li, X.R., Xiao, H.L., Berndtsson, R., Pan, Y.X., 2007. Effects of surface characteristics on infiltration patterns in an arid shrub desert. Hydrol. Proc. 21, 72–79.

Wanner, M., Dunger, W., 2002. Primary immigration and succession of soil organisms on reclaimed opencast coal mining areas in eastern Germany. Eur. J. Soil Biol. 38, 137–143.

Wardle, D.A., Hornberg, G., Zackrisson, O., Kalela-Brundin, M., Coomes, D.A., 2003. Long-term effects of wildfire on ecosystem properties across an island area gradient. Science 300, 972–975.

Whitbread-Abrutat, P.H., 1997. The potential of some soil amendments to improve tree growth on metalliferous mine wastes. Plant Soil 192, 199–217.

Whiting, S.N., Reeves, R.D., Richards, D., Johnson, M.S., Cooke, J.A., Malaisse, F., Paton, A., Smith, J.A.C., Angle, J.S., Chaney, R.L., Ginocchio, R., Jaffre, T., Johns, R., McIntyre, T., Purvis, O.W., Salt, D.E., Schat, H., Zhao, F.J., Baker, A.J.M., 2004. Research priorities for conservation of metallophyte biodiversity and their potential for restoration and site remediation. Restor. Ecol. 12, 106–116.

Whittaker, R.H., 1972. Evolution and measurement of species diversity. Taxon 21, 213–251.

Wu, X., Hu, Z.Q., Li, J., Zhang, H.Y., Hu, J.L., 2011. A study of land reclamation and ecological restoration in a resource-exhausted city—a case study of Huaibei in China. Int. J. Min. Reclam. Environ. 25, 332–341.

Xia, H.P., 2004. Ecological rehabilitation and phytoremediation with four grasses in oil shale mined land. Chemosphere 54 (3), 345–353.

Yanai, R.D., Currie, W.S., Goodale, C.L., 2003. Soil carbon dynamics after forest harvest: an ecosystem paradigm reconsidered. Ecosystems 6, 197–212.

Zhang, H., Chu, L., 2010. Plant community structure, soil properties and microbial characteristics in revegetated quarries. Ecol. Eng. 37 (8), 1104–1111.

Zhang, Z.S., Li, X.R., Liu, L.C., Jia, R.L., Zhang, J.G., Wang, T., 2009. Distribution, biomass, and dynamics of roots in a revegetated stand of *Caragana korshinskii* in the Tengger Desert, northwestern China. J. Plant Res. 122, 109–119.