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RESEARCH ARTICLE

# Assessing the Ecological Success of Restoration by Afforestation on the Chinese Loess Plateau

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## Abstract

**Afforestation has been accepted as a key measure for preventing soil erosion on the Chinese Loess Plateau for 40 years. In this study, we assessed the ecological success of afforestation by comparing afforested with pre-afforested (croplands) and natural recovery sites in a typical watershed on the Loess Plateau. We evaluated the ecosystem response in terms of vegetation structure, plant diversity, and several key ecological processes of soil moisture, soil nutrients, and soil anti-erodibility. Compared with the croplands, we found that the following indexes were significantly enhanced in afforested sites: vegetation structure and species diversity (species richness, Margalef index, Shannon–Wiener index, and Sorensen's similarity index), soil nutrients (organic carbon, total nitrogen, extractable ammonium nitrogen, available potassium, and available phosphorous), and soil anti-erodibility indexes (water-stable soil aggregates, mean weight diameter, and the ratio of soil structure dispersion). Afforestation offered few additional advantages when compared with natural recovery sites. More importantly, afforestation had significant negative effects on soil desiccation, with negative impacts on the long-term sustainability of these ecosystems. In order to develop self-sustaining and functional ecosystems, our results suggest that natural revegetation offers a more adaptive and appropriate method of ecological restoration on the Loess Plateau.**

**Key words: afforestation, plant diversity, soil anti-erodibility, soil desiccation, soil nutrients, vegetation structure.**

## Introduction

Soil erosion ranks as one of the most serious environmental problems in the world because soil erosion from land areas is widespread and adversely affects all natural and human-managed ecosystems, including agriculture and forestry (Pimentel & Kounang 1998). China is one of the most seriously soil erosion-affected regions of the world, especially the Loess Plateau (Shi & Shao 2000). Soil erosion also causes siltation of rivers and reservoirs off-site (Hessel et al. 2003) increasing the flooding risk along the lower reaches of the Yellow River basin (McVicar et al. 2002). Previous studies have concluded that artificially accelerated soil erosion caused by deforestation is the primary cause (Zheng 2006). Therefore, tree planting and afforestation have been considered a key technique for soil and water conservation to minimize erosion on the Loess Plateau in the past 40 years (Li et al. 2008).

Since 1949, afforestation on the Loess Plateau has been widely implemented. Fast growing tree and shrub species are usually planted for vegetation restoration. These trees and shrubs grow well in the early stages after planting, but their rate of growth often declines once the initial water supply is exhausted, leading to soil desiccation and dried soil layers (Chen et al. 2007; McVicar et al. 2007*a*). Long-term soil water deficit and soil desiccation are increasing threats to the normal growth of afforestation on the Loess Plateau (Cao et al. 2007; Shangguan 2007), together with a decrease of streamflow in different scale catchments (McVicar et al. 2007*a*). Eventually, this threat results in forest degradation, low productive "small but old tree" forests, and tree death in drought years (Chen et al. 2008). As a result, soil erosion is not effectively controlled. To further control soil erosion, the Chinese government implemented the "Grain for Green" project in March 2000. This is a large-scale project that requires farmers to reserve a part of their sloping farmland for trees, shrubs, or grasses (McVicar et al. 2007*a*; Zhou et al. 2009). The government agencies in charge of the project issued a decree that "ecological forests" (i.e. trees for soil and water conservation, e.g. locust and buckthorn) must account for 80% of the restored lands and the rest should be "economic forests" (i.e. horticultural tree crops e.g. apples and dates). In this way, the drive for vegetation recovery on the P1ateau has turned into a single-issue movement of afforestation. However, there are controversies over the appropriateness of afforestation in this area. Based on the historical documents, as relatively dense forests had grown over a vast area of the Loess Plateau before devastation by human activities, some researchers recommend

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planting trees on the Loess Plateau (Shi 1991). However, others suggest that future soil water conservation in this area should focus on the planting of grasses and herbs, according to the evidence from spore-pollen and fossil records (Jiang & Ding 2005). Others suggest “successional” re-planting to mimic soil evolution processes by first re-planting grasses and herbs (to improve soil water holding capacity), before planting shrubs and trees 5–10 years after the initial revegetation (McVicar et al. 2010). A successful revegetation program is likely to require planting a mix of grasses, shrubs, and trees down the hillsides (Fu & Chen 2000).

To help resolve these controversies, studying existing afforested stands is very important, as there are practical references for evaluating the suitability and sustainability of afforestation. Knowledge about these stands is essential for ecologically successful restoration on the Loess Plateau. The success of restoration can be viewed as a continuous process from establishment to successfully developing those attributes that ensure a self-sustaining naturally and functioning ecosystem (Reay & Norton 1999). To evaluate this success, it is important to know whether key ecosystem processes have been restored and whether the restored system is ecologically sustainable with a potential for biodiversity conservation (Ryder & Miller 2005). Three key ecosystem metrics are relevant here. Vegetation structure provides information on habitat suitability, ecosystem productivity, and succession pathways (Wang et al. 2004). Species diversity provides information on susceptibility of invasions and trophic structure and ecosystem resilience (Nichols & Nichols 2003). And ecosystem processes provide information on biogeochemical cycles and nutrient cycling necessary for the long-term ecosystem stability (Herrick 2000). Therefore, vegetation structure, species diversity, and ecosystem processes have been identified as the measures of restoration success (Ruiz-Jaén & Aide 2005). On the Loess Plateau, studies on afforestation have focused on the following: soil moisture and desiccation (Chen et al. 2008; Li et al. 2008), soil erosion control (Zheng 2006), soil properties (Wang et al. 2003), soil microbial biomass (An et al. 2009), plant biodiversity (Jiang et al. 2003), tree growth and survival (Cao et al. 2007, 2008), and reductions in water yield from afforested catchments (McVicar et al. 2007*a*). However, no comprehensive studies have reported afforestation success as a whole.

In order to assess the restoration success of afforestation on the Loess Plateau, we addressed the following questions: (1) How does the vegetation structure and species diversity of the afforested sites differ from pre-planting sites (croplands) or natural recovery sites?; (2) How do the main process-related ecosystem components of the afforested sites differ from pre-planting or natural recovery sites?; and (3) Has the restoration by afforestation been ecologically successful in this area?

## Methods

### Study Site

The study site is the 8.27 km$^2$ Zhifanggou watershed, located in Yan River basin of Ansai County, North Shaanxi Province. The area ranges from longitude 109°14′09″ E to 109°16′01″ E and latitude 36°43′11″ N to 36°46′25″ N, and the elevation ranges from 1,425 to 1,040 m with the average of 1,200 m. The soil type is monotonous yellow-brown earth, which is prone to soil erosion (Shi & Shao 2000) and gully erosion is well developed with gully density of 8.1 km/km$^2$ (Zhang et al. 2004). The climate is the transitional zone from semi-humid warm climate to semiarid climate with an average annual precipitation of 504 mm (1970–2006). The rainfall usually falls during June–September, accounting for approximately 74% of the whole year (McVicar et al. 2007*b*). The watershed is located in the ecotone of forest and grassland, and there are 48 families and 162 species of higher plants (Zhang et al. 2007). Due to long-term human activities, most natural vegetation has been destroyed. Current land uses include cropland, woodland, grassland, shrubland, orchards, and residential areas in a mosaic rural landscape pattern. Major crops are potatoes (*Solanum tuberosum*), beans (*Phaseolus vulgaris*), maize (*Zea mays*), and millet (*Panicum miliaceum*). The woods are dominated by planted *Robinia pseudoacacia*. The steppe vegetation mainly comprises *Artemisia gmelinii*, *Artemisia giraldii*, *Lespedeza davurica*, and *Stipa bungeana*. The shrubland is dominated by afforested *Caragana korshinskii*, *Hippophae rhamnoides*, and the native species *Sophora viciifolia* (Fu et al. 2006).

The Zhifanggou watershed has been an experimental site of the Institute of Soil and Water Conservation (CAS) since 1973, and an experimental area of national key scientific and technical significance since 1986. After more than 30 years of comprehensive management, the watershed shows significant improvements in the ecological environment (Zhang et al. 2007) and economic metrics (Fu et al. 2006), soil erosion has been significantly mitigated, and Zhifanggou watershed is a state-level model watershed of the “Grain for Green” project (Zhang et al. 2004).

### Data Collection

Field data was collected in September 2005 and five transects were established across the watershed from the head to the outlet. Sample plots were chosen according to the variation of vegetation along the five transects. We used landowner interviews to determine the time as cropping land management practices were abandoned in favor of some form of restoration (i.e. natural recovery or active afforestation) of the sample plots. A total of 74 plots were measured (Fig. 1).

The sample size was 1 × 10 m for grasslands, 5 × 5 m for shrublands, and 10 × 10 m for tree-dominated sites. The coverage of each species was estimated visually by two observers working together. The height, diameter at breast height (DBH), and the planar projected diameter of canopy of trees and shrubs were also recorded. Slope, aspect, elevation, and geographical position (latitude/longitude) of plots were measured by a global positioning system and a slope gradient meter.

In each plot, soil samples were taken from six points in an S-shaped pattern to a depth of 0–20 cm. Soil was analyzed by the methods described by the Agricultural Chemistry

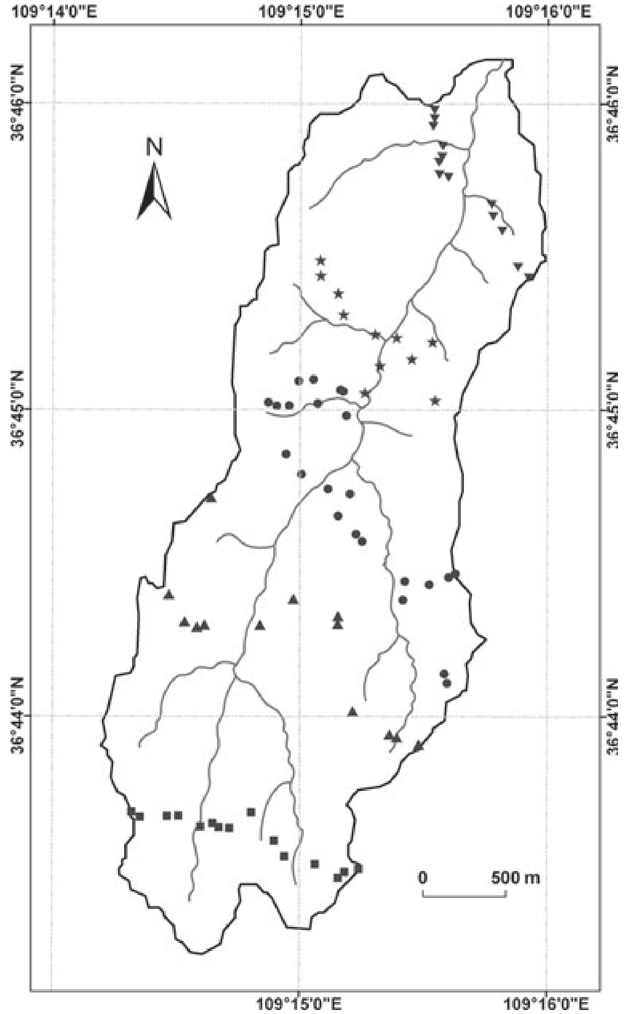


Figure 1. The distribution of sample plots in Zhifanggou watershed. ■, ▲, ●, ★, and ▼ are sample plots in the five transects, respectively.

Committee of Chinese Soil Academy (1984). Organic matter was measured by the $K_2Cr_2O_7$ method, total N was measured by the $K_2SO_4$-$CuSO_4$-Se distillation method, and total P was measured by the $HClO_4$-$H_2SO_4$ colorimetric method. Extractable ammonium N (alkali extractable N) was measured by the diffusion method with 10 mL of 1 N NaOH, 2% $H_3BO_3$ and 0.005 N HCl at 40°C for 24 hours. Available P was measured by the colorimetric method with 0.5M $NaHCO_3$ extraction (1:20), and available K was measured by the atomic absorption spectrum method with 1M $NH_4OAC$ extraction (1:20, pH = 7). In order to determine the gravimetric water content of the soil, soil samples were also taken to a depth of 0–500 cm with 20 cm depth intervals. Then, samples were heated at 105–110°C for 10 hours and the weight of pre- and post-oven drying gives the gravimetric soil moisture content. Soil bulk density was measured by the ring (100 $cm^3$) method. Soil mechanical composition and soil micro-aggregates were analyzed using Mastersizer 2000 (Malvern, U.K.), and soil aggregate structure was measured by dry-sieving and wet-sieving methods.

### Data Analysis

To analyze ecosystem functional characteristics in restored sites, it is necessary to compare equivalent metrics from pre-restored and reference sites (Hobbs & Norton 1996). Of the 74 plots in the Zhifanggou watershed, croplands were chosen as the pre-afforested sites (three plots). The main herbaceous (14 plots) and shrublands (four plots) were chosen as the reference sites (these 18 plots had been naturally restored for over 20 years). The afforested sites contained *C. korshinskii* (six plots) and *R. pseudoacacia* (eight plots) that had been planted over 20 years ago.

The measures of vegetation structure included vegetation cover, height, DBH, canopy diameter, vertical stratification, and dominant species. Species richness, Shannon-Wiener index (Shannon & Weaver 1949), Margalef index (Margalef 1958), Pielou index (Pielou 1969), and Sorensen's similarity index (Sorensen 1948) were used as the measures of plant diversity. Soil nutrient levels, soil water content, and indices related to soil anti-erodibility (such as water-stable soil aggregates, mean weight diameter [MWD], ratio of soil structure dispersion, and soil bulk density) were used as measures to assess ecological processes. The differences among afforestated sites, pre-afforestated sites, and reference sites were compared by analysis of variance (ANOVA) and least-significant difference (LSD) test using SPSS V13.0. The equations used to calculate Shannon-Wiener index, Margalef index, Pielou index, and Sorensen's similarity index follow:

$$\text{Shannon-Wiener index: } H' = -\sum_{i=1}^{S} (P_i \ln P_i)$$

$$\text{Margalef index: } d_{\mathrm{Ma}} = (S-1)/\ln N$$

$$\text{Pielou index: } J_{\mathrm{sw}} = H/\ln S$$

$$\text{Sorensen's similarity index: } C_{\mathrm{s}} = 2C/(S_1+S_2)$$

where $S$ is the number of species, $P_i$ is the proportion on individuals or the abundance of the $i$th species expressed as a proportion of the total in the community, ln is the log base-e, $N$ is the total individual number of all the species, $S_1$ is the total number of species recorded in the first community, $S_2$ is the total number of species recorded in the second community, and $C$ is the number of species common to both communities.

## Results

### Vegetation Structure

Table 1 shows the vegetation structure of two main afforested sites. The density of *Robinia pseudoacacia* varied from 300 to 4,500 individuals/ha with average height between 5 and 15 m, DBH between 18 and 60 cm and canopy diameter between 1.2 and 5 m. In the shrub layer, one to four shrub species occurred with the average height between 0.2 and 2.3 m; in the herb layer, the main species were *Artemisia giraldii*, *Stipa bungeana*, *Cleistogenes caespitosa*, and *Lespedeza davurica*, with the average vegetation height from 0.23 to 0.45 m, and

**Table 1.** The vegetation structure in *Robinia pseudoacacia* and *Caragana korshinskii* sites (mean ± standard deviation).

| | | Afforested Layer | | | | | Shrub Layer | | | Herbaceous Layer | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Site | Species Richness | Cover (%) | Density (individuals/ha) | Height (m) | DBH (cm) | Canopy Diameter (m) | Cover (%) | Height (m) | Main Species | Cover (%) | Height (m) | Main Species |
| Rps | 16.3 ± 5.4 | 35.0 ± 17.5 | 1,587.5 ± 1,504.7 | 9.3 ± 3.7 | 31.5 ± 15.7 | 2.9 ± 1.3 | 16.1 ± 20.0 | 1.0 ± 0.7 | Ps, Sv, Ck, Ba, Rpa, Cf, Rps, Fs, Rpar, Bp, Pb, Rx, Cm, As, Ps | 31.9 ± 29.4 | 0.33 ± 0.08 | Sb, Cc, Ph, Ct, Agi, Agm, Ld, Pbi, Ls, Am, Dc, Cl, Mr, Pc |
| Ck | 15.2 ± 4.8 | 44.2 ± 6.7 | 5,866.7 ± 2,243.8 | 1.3 ± 0.1 | | 1.1 ± 0.2 | Just Pu and Bp appeared in one plot, with height of 0.88–1.4 m and canopy diameter of 0.45–0.5 m | | | 31.7 ± 10.8 | 0.45 ± 0.17 | Agm, Sb, Ld, Agi, Ha, Pbi, Cc, Tp, Pt |

Here vegetation cover means vegetation (green leaves, yellow leaves, and branches) cover fraction (Lu et al. 2003). Rps—*Robinia pseudoacacia*, Ck—*Caragana korshinskii*, Ps—*Periploca sepium*, Sv—*Sophora viciifolia*, Ba—*Buddleya alternifolia*, Rpa—*Rubus parvifolius*, Cf—*Clematis fruticosa*, Fs—*Forsythia suspense*, Rpar—*R. parvifolius*, Bp—*Berberis purdomii*, Pb—*Pyrus betulaefolia*, Rx—*Rosa xanthina*, Cm—*Cotoneaster multiflorus*, As—*Armeniaca sibirica*, Ps—*P. sepium*, Sb—*Stipa bungeana*, Cc—*Cleistogenes caespitosa*, Ph—*Patrinia heterophylla*, Ct—*Cynanchum thesioides*, Agi—*Artemisia giraldii*, Agm—*Artemisia gmelinii*, Ld—*Lespedeza davurica*, Pbi—*Potentilla bifurca*, Ls—*Leymus secalinus*, Am—*Artemisia mongolica*, Dc—*Dendranthema chanetii*, Cl—*Carex lancifolia*, Mr—*Melica radula*, Pc—*Phragmites communis*, Ha—*Heteropappus altaicus*, Tp—*Thalictrum petaloideum*, Pt—*Potentilla tanacetifolia*, Pu—*Prinsepia uniflora*.

the total vegetation cover of *R. pseudoacacia* sites was from 65 to 110% with a tree layer of 15–60% and an herb layer of 5–80%.

The density of *Caragana korshinskii* was 3,200–8,800 individuals/ha with a height of 1.19–1.40 m and canopy diameter of 0.82–1.33 m; other shrubs like *Prinsepia uniflora* and *Berberis purdomii* occurred in only one site. In the herb layer, the main species were *A. giraldii*, *S. bungeana*, *L. davurica*, *Artemisia gmelinii*, *Heteropappus altaicus*, the average vegetation height ranged from 0.30–0.69 m; and the total vegetation cover of the *C. korshinskii* sites ranged from 50 to 90% with a shrub layer of 35–50% and an herb layer of 15–45%.

Compared with the reference sites (natural recovery sites), afforested sites had vertical canopy structure layers, but the shrub layer of *R. pseudoacacia* sites was simple with few shrub plants except for two plots with abundant planted *Forsythia suspensa* (60% cover) and *C. korshinskii* (20% cover), respectively. A higher *R. pseudoacacia* cover (50–60%) was associated with very low herb cover (5–10%), whereas a higher herb cover (60–80%) was associated with low *R. pseudoacacia* cover (20–30%). The cover of *C. korshinskii* was 35–50% with an herb cover of 15–45%, whereas cover varied from 40 to 85% in the natural restoration sites. The average vegetation cover of afforested sites was 1.8–2.0 times that of pre-afforested site ($p = 0.008$) and 1.6–1.7 times that of natural herb sites ($p = 0.000$), but no significant difference ($p = 0.832$) from the natural shrub sites.

### Plant Diversity

Table 2 shows the plant diversity indices of afforested and reference sites. As expected, the species richness, Margalef index, Shannon-Wiener index of afforested sites and reference sites (natural recovery) were much higher than those of the pre-afforested sites (slope croplands). Comparing afforested sites with pre-afforested sites, the species richness changed from 5.33 to 12.5–19.0 (a 135–257% increase), Margalef index increased from 1.31 to 2.10–3.33 (a 60–154% increase), Shannon-Wiener index changed from 1.15 to 1.48–2.22 (a 29–93% increase), and Pielou index changed from 0.58 to 0.76 (a 31% increase). However, the afforested sites did not show higher species richness, Margalef index, or Shannon-Wiener index, that is, afforestation offered few additional advantages of plant diversity compared with the same metrics for the natural recovery sites ($p > 0.05$).

However, the Sorensen's similarity index showed low similarity of species content between the afforested sites and natural recovery sites. The Sorensen's similarity index between the afforested sites and natural recovery sites varied from 0.26 to 0.62 with an average of 0.43. The Sorensen's similarity index between cropland and all the afforested and natural recovery sites was very low (0–0.14), whereas the maximum Sorensen's similarity index (0.68) was between the natural herb recover sites in the north-facing and south-facing slopes. The Sorensen's similarity index between the afforested sites varied form 0.34 to 0.54, and between the natural herb and shrub recover sites was 0.29–0.35 (Table 3).

**Table 2.** The plant diversity indexes of vegetation sites (mean ± standard deviation).

| *Vegetation Sites* | *Margalef Index* | *Shannon-Wiener Index* | *Pielou Index* | *Species Richness* |
|---|---|---|---|---|
| Pre-afforested | | | | |
| CL | $1.31 \pm 0.15^{b}$ | $1.15 \pm 0.25^{c}$ | $0.69 \pm 0.06^{ab}$ | $5.33 \pm 1.15^{b}$ |
| Afforested | | | | |
| CKNF | $2.83 \pm 0.88^{a}$ | $1.81 \pm 0.19^{ab}$ | $0.66 \pm 0.03^{ab}$ | $16.50 \pm 4.77^{a}$ |
| CKSF | $2.27 \pm 0.24^{ab}$ | $1.71 \pm 0.01^{abc}$ | $0.68 \pm 0.01^{ab}$ | $12.50 \pm 0.71^{ab}$ |
| RPNF | $3.33 \pm 1.16^{a}$ | $2.22 \pm 0.47^{a}$ | $0.76 \pm 0.12^{a}$ | $19.00 \pm 5.94^{a}$ |
| RPSF | $2.10 \pm 0.38^{ab}$ | $1.48 \pm 0.39^{bc}$ | $0.58 \pm 0.17^{b}$ | $13.50 \pm 3.42^{ab}$ |
| Reference | | | | |
| NHNF | $3.24 \pm 0.78^{a}$ | $2.18 \pm 0.22^{a}$ | $0.75 \pm 0.05^{a}$ | $19.33 \pm 4.80^{a}$ |
| NSNF | $3.14 \pm 1.47^{a}$ | $1.98 \pm 0.53^{a}$ | $0.69 \pm 0.09^{ab}$ | $18.20 \pm 8.53^{a}$ |
| NHSF | $1.89 \pm 0.42^{ab}$ | $1.71 \pm 0.23^{abc}$ | $0.74 \pm 0.16^{ab}$ | $10.50 \pm 2.12^{ab}$ |
| NSSF | $2.94 \pm 1.30^{a}$ | $1.72 \pm 0.30^{abc}$ | $0.64 \pm 0.03^{ab}$ | $16.50 \pm 9.19^{a}$ |

CL, cropland; CKNF, *Caragana korshinskii* in north-facing slope; CKSF, *C. korshinskii* in south-facing slope; NHNF, natural herb in north-facing slope; NHSF, natural herb in south-facing slope; NSNF, natural shrub in north-facing slope; NSSF, natural shrub in south-facing slope; RPNF, *Robinia pseudoacacia* in north-facing slope; RPSF, *R. pseudoacacia* in south-facing slope. Same letters in the same column indicate no significant differences between vegetation sites (e.g. a, ab, and abc means no significant differences among Shannon-Wiener index of RPNF, CKSF, and CKNF), and no same letters in the same column indicate significant differences between vegetation sites (e.g. a and bc means significant differences between Shannon-Wiener index in RPNF and RPSF) based on LSD test ($p < 0.05$).

**Table 3.** The Sorensen's similarity index between vegetation sites.

| | *CL* | *CKNF* | *CKSF* | *RPNF* | *RPSF* | *NHNF* | *NSNF* | *NHSF* | *NSSF* |
|---|---|---|---|---|---|---|---|---|---|
| CL | 1 | | | | | | | | |
| CKNF | 0.13 | 1 | | | | | | | |
| CKSF | 0.13 | 0.49 | 1 | | | | | | |
| RPNF | 0.12 | 0.45 | 0.34 | 1 | | | | | |
| RPSF | 0.10 | 0.54 | 0.52 | 0.49 | 1 | | | | |
| NHNF | 0.14 | 0.54 | 0.35 | 0.56 | 0.42 | 1 | | | |
| NSNF | 0.05 | 0.26 | 0.32 | 0.38 | 0.30 | 0.31 | 1 | | |
| NHSF | 0.10 | 0.62 | 0.39 | 0.53 | 0.47 | 0.68 | 0.29 | 1 | |
| NSSF | 0.00 | 0.50 | 0.45 | 0.30 | 0.54 | 0.29 | 0.21 | 0.35 | 1 |

CL, cropland; CKNF, *Caragana korshinskii* in north-facing slope; CKSF, *C. korshinskii* in south-facing slope; NHNF, natural herb in north-facing slope; NHSF, natural herb in south-facing slope; NSNF, natural shrub in north-facing slope; NSSF, natural shrub in south-facing slope; RPNF, *Robinia pseudoacacia* in north-facing slope; RPSF, *R. pseudoacacia* in south-facing slope.

### Soil Water

As expected, the water content in the 0–80 cm soil layer was affected by rainfall during the sampling period. The variation was not significant ($p = 0.782$) among pre-afforested sites, afforested sites, and reference sites, but it was very significant ($p = 0.000$) in the 80–500 cm soil layer. The soil water content in the 80–500 cm layer ranged from 4.55 to 7.79% with the averages of 6.34, 6.31, 6.60, and 5.66% in *C. korshinskii* both in the north-facing and south-facing slopes, *R. pseudoacacia* both in the north-facing and south-facing slopes, respectively. The values of soil water content were above 10.37% with the averages of 12.91, 12.09, and 16.38% of natural herb sites both in the north-facing and south-facing slopes, and croplands, respectively. The loess soil layers in the natural shrub sites were shallower (about 200–220 cm, rock underlying). The soil water content in 80–220 cm of natural shrub sites ranged from 5.28 to 7.82% with an average of 6.37% in the south-facing slope, and it ranged from 6.02 to 11.80% with an average of 8.48% in the north-facing slope. Furthermore, the soil water content of natural shrub sites in both slopes was higher than that in the afforested sites ranging from 5.04 to 6.12% in the same soil layers (Fig. 2).

### Soil Nutrients

Soil organic carbon, total N, extractable ammonium N, available K, and available P of afforested sites were much higher than pre-afforested sites, except for available P of *C. korshinskii* in the north-facing slope (which was a little lower but not significant, $p = 0.361$). However, soil organic carbon and extractable ammonium N of planted sites were not significantly different from those of natural reference sites ($p = 0.093$ and 0.069), and those of natural shrub sites were even higher. Available P of planted sites was not significantly higher than that of natural herb sites ($p = 0.072$), whereas available K of planted sites was not significantly higher than that of natural shrub sites ($p = 0.488$) but significantly higher than that of natural herb sites ($p = 0.001$). The variation of total P among all the sites was not significant ($p = 0.179$), neither was that of total N ($p = 0.013$) (Fig. 3).

### Soil Anti-erodibility

Table 4 shows the indices of soil anti-erodibility of the nine sites. The variation of greater than 5 mm soil water-stable aggregates was only significant when comparing pre-afforested sites with *R. pseudoacacia* in south-facing ($p = 0.011$) and shrub in the south-facing slope ($p = 0.015$). The variation of 1–5 mm soil water-stable aggregates was significant when comparing natural shrub in the north-facing slope with natural herb in the north-facing slope ($p = 0.016$), *R. pseudoacacia* in the south-facing slope ($p = 0.026$), *C. korshinskii* in both slopes ($p = 0.002$) and croplands ($p = 0.008$). The variation of greater than 0.25 mm soil water-stable aggregates of pre-afforested sites, *C. korshinskii* (in both slopes) and natural herb in the north-facing slope was significantly lower than that of natural shrub in the south-facing slope sites ($p = 0.013$).

**Table 4.** The indices related to soil anti-erodibility of the vegetation sites (mean ± standard deviation).

| Vegetation Sites | Soil Water-Stable Aggregates >5 mm (%) | 5–1 mm (%) | >0.25 mm (%) | MWD (mm) | Ratio of Soil Structure Dispersion (%) | Soil Bulk Density ($g/cm^3$) |
|---|---|---|---|---|---|---|
| Pre-afforested | | | | | | |
| CL | $8.53 \pm 7.78^{c}$ | $4.30 \pm 2.46^{d}$ | $19.47 \pm 10.80^{c}$ | $0.57 \pm 0.43^{c}$ | $84.27 \pm 7.92^{a}$ | $1.33 \pm 0.03^{a}$ |
| Afforested | | | | | | |
| CKNF | $23.00 \pm 11.89^{bc}$ | $7.38 \pm 1.52^{cd}$ | $35.98 \pm 11.36^{bc}$ | $1.38 \pm 0.62^{bc}$ | $70.15 \pm 10.38^{ab}$ | $1.26 \pm 0.13^{ab}$ |
| CKSF | $21.70 \pm 8.06^{bc}$ | $9.25 \pm 1.06^{bcd}$ | $37.15 \pm 7.14^{bc}$ | $1.37 \pm 0.38^{bc}$ | $68.30 \pm 9.97^{abc}$ | $1.20 \pm 0.05^{ab}$ |
| RPNF | $34.58 \pm 16.66^{abc}$ | $13.35 \pm 4.58^{ab}$ | $58.10 \pm 17.10^{a}$ | $2.15 \pm 0.85^{ab}$ | $57.09 \pm 9.37^{bc}$ | $1.36 \pm 0.12^{a}$ |
| RPSF | $31.33 \pm 6.42^{ab}$ | $8.73 \pm 2.56^{bcd}$ | $47.85 \pm 9.40^{ab}$ | $1.84 \pm 0.38^{ab}$ | $61.71 \pm 6.00^{bc}$ | $1.34 \pm 0.06^{a}$ |
| Reference | | | | | | |
| NHNF | $22.84 \pm 11.78^{bc}$ | $10.33 \pm 2.79^{bc}$ | $40.11 \pm 13.56^{b}$ | $1.47 \pm 0.62^{bc}$ | $66.69 \pm 11.60^{bc}$ | $1.30 \pm 0.09^{a}$ |
| NSNF | $25.10 \pm 20.36^{abc}$ | $16.45 \pm 0.78^{a}$ | $53.70 \pm 22.20^{ab}$ | $1.77 \pm 1.07^{abc}$ | $50.67 \pm 10.50^{c}$ | $0.97 \pm 0.05^{b}$ |
| NHSF | $31.12 \pm 15.42^{abc}$ | $11.84 \pm 2.22^{ab}$ | $51.90 \pm 15.00^{ab}$ | $1.93 \pm 0.81^{ab}$ | $55.09 \pm 12.70^{c}$ | $1.31 \pm 0.08^{a}$ |
| NSSF | $47.20 \pm 9.76^{a}$ | $13.70 \pm 3.39^{ab}$ | $67.35 \pm 5.59^{a}$ | $2.80 \pm 0.37^{a}$ | $46.85 \pm 5.75^{c}$ | $1.12 \pm 0.17^{b}$ |

CL, cropland; CKNF, *Caragana korshinskii* in north-facing slope; CKSF, *C. korshinskii* in south-facing slope; RPNF, *Robinia pseudoacacia* in north-facing slope; RPSF, *R. pseudoacacia* in south-facing slope; NHNF, natural herb in north-facing slope; NHSF, natural herb in south-facing slope; NSNF, natural shrub in north-facing slope; NSSF, natural shrub in south-facing slope. Same letters in the same column indicate no significant differences between vegetation sites (e.g. a, ab, and abc means no significant differences among MWD of NSSF, NHSF, and NSNF), and no same letters in the same column indicate significant differences between vegetation sites (e.g. a and bc means significant differences between MWD in NSSF and NHNF) based on LSD test ($p < 0.05$).

The MWD of pre-afforested sites was significantly different from those of *R. pseudoacacia* in both slopes ($p = 0.042$), natural herb and shrub in the south-facing slope ($p = 0.020$), whereas others were similar ($p = 0.292$). The soil structure dispersion ratios of *R. pseudoacacia* sites and four natural reference sites were significantly lower than those of pre-afforested sites ($p = 0.007$), whereas the soil bulk density of natural shrub sites was significantly lower than those of afforested sites ($p = 0.026$), natural herb sites ($p = 0.001$) and pre-afforested sites ($p = 0.027$). On the whole, soil water-stable aggregates and MWD of afforested sites were much higher, while the ratio of soil structure dispersion was lower than that of croplands. However, there was no significant ($p > 0.05$) difference between afforested sites and natural reference sites in greater than 5 mm soil water-stable aggregate and MWD (except natural shrub in the south-facing slope was much higher) and in greater than 0.25 mm soil water-stable aggregate (except natural shrub in the south-facing slope and *R. pseudoacacia* in the north-facing slope were much higher), whereas the soil structure dispersion ratio of natural sites was even lower than that of afforested sites.

## Discussion

*Question 1:* How does the vegetation structure and species diversity of the afforested sites differ from pre-planting sites (croplands) or natural recovery sites? Undoubtedly, compared with the pre-afforested sites (i.e. croplands), vegetation structure and species diversity of afforested sites have been improved. However, compared with the reference natural recovery sites, afforested sites had vertical canopy structure layers, but only a few plants in the shrub layer. Furthermore, the afforested sites did not show much higher species richness, Margalef index, and Shannon-Wiener index compared with natural recovery sites, and the Sorensen's similarity index between the afforested sites and natural recovery sites was not high. The vegetation structure and plant diversity of afforested sites in our study contrast with those of Ziwuling forest area, an ecologically stable and functional ecosystem (Zheng 2006). Li and Shao (2003) found in Ziwuling area where the species richness of main five forests sites was 48–63, while vegetation cover was 160–170% of the land area with a tree layer of 49.9–100%, a shrub layer of 18.8–88.9%, and an herb layer of 15.9–100%. Although in a temperate New Zealand forest, species richness of canopy species was 8 and 12 and of regenerating tree species was 22 and 30 in the 30- and 35-year-old planting restoration sites, respectively (Reay & Norton 1999), and much higher than in our afforested sites. Taken together, afforestation contributed to the tree layer or shrub layer compared with natural regeneration, however, it offered few additional advantages in species diversity. Even declining, dead branches and "small-aged trees" can be found in the *Robinia pseudoacacia* and *Caragana korshinskii* sites.

*Question 2:* How do the main process-related ecosystem components of the afforested sites differ from pre-planting or natural recovery sites? On the Loess Plateau, soil water is one key factor limiting vegetation development and recovery (Cao et al. 2007; Shangguan 2007; Chen et al. 2008). The average soil water content in the 80–500 cm layer of natural herb sites was twice as that of afforested sites, and the average soil water content in the 80–200 cm layer of natural shrub sites were 1.3 times higher than that of afforested sites in the same soil layer. This soil water condition of afforested sites belongs to the middle dried layer (soil moisture 6–9%) and severely dried soil layers (soil moisture <6%), which negatively affects tree growth resulting in low productivity (Wang et al. 2000). Research in 23 types of tree and shrub forestlands in seven locations in three vegetation zones by Li et al. (2008) indicated that soil desiccation is very common in the Loess Plateau

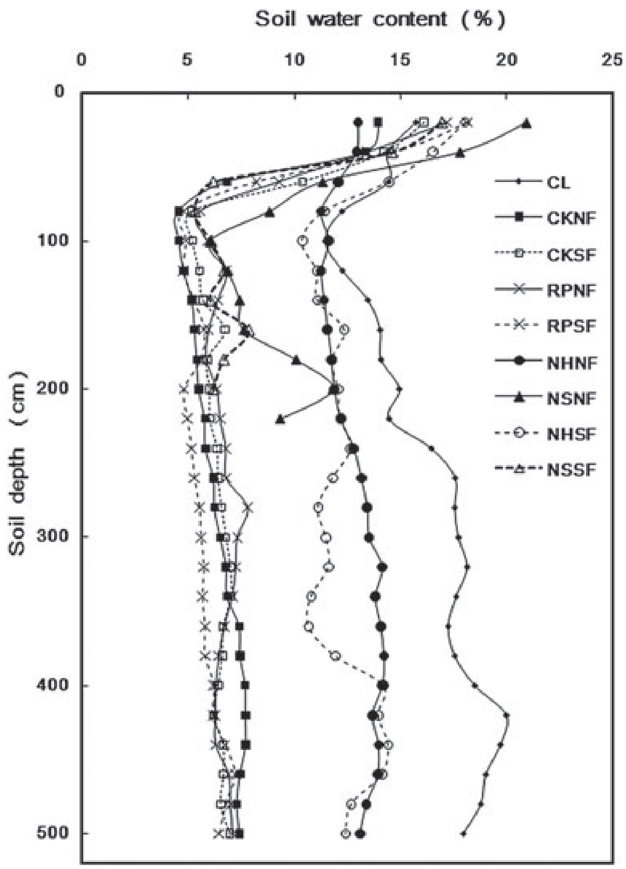


Figure 2. Soil water content of the vegetation sites in the 0–500 cm layers. CL, cropland; CKNF, *Caragana korshinskii* in north-facing slope; CKSF, *C. korshinskii* in south-facing slope; RPNF, *Robinia pseudoacacia* in north-facing slope; RPSF, *R. pseudoacacia* in south-facing slope; NHNF, natural herb in north-facing slope; NSNF, natural shrub in north-facing slope; NHSF, natural herb in south-facing slope; NSSF, natural shrub in south-facing slope. NSNF and NSSF plots had soil only 220 and 200 cm deep with rocks below.

region. Therefore, soil desiccation is one type of particular hydrological phenomenon during the process of vegetation restoration on the Loess Plateau of China. It negatively affects the water cycle in soils, which often cuts off water pathways to supplement groundwater, heavily influences the growth and natural succession of vegetation, and requires a very long time to recover (Chen et al. 2008). Current evidence suggests that afforestation typically reduces local average water yield as well as low flows (McVicar et al. 2007*a*), so afforestation is increasingly considered as a land use activity that threatens water resources security (van Dijk & Keenan 2007).

Different soil nutrients limit biomass production, species composition, and diversity (Critchley et al. 2002). Our study showed that soil organic carbon, total N, extractable ammonium N, available K, and available P of afforested sites were much higher than those of pre-afforested sites. However, on average soil organic matter and extractable ammonium N of afforested sites were not significantly ($p > 0.05$) different from those in the natural reference sites, and the variation of total P among all sites was not significant ($p > 0.05$). Wang et al. (2003) also showed that the soil organic matter, total N, extractable ammonium N, and total P have little difference between grassland and woodland based on 94 sampling sites including six transects in the Danangou catchment in An'sai. Thus, we conclude that there was little overall difference in the soil nutrient improvement by afforestation and natural recovery. Similar results have been found in Puerto Rico by Ruiz-Jaén and Aide (2005), but the soil nutrients there are much higher than in our study site.

For soil anti-erodibility, our study showed that there were few differences between afforested sites and natural reference sites in soil water-stable aggregate composition, and the ratio of soil structure dispersion of natural sites were even lower than that of afforested sites. This result has also been found in the Mediterranean region where the lowest-growing plant cover tends to reduce soil erosion and run-off more effectively than the taller and open medium-sized shrubs (Durán Zuazo et al. 2006). Some of the shrub species were found to produce higher sediment yields than the grass species in the dryland environment of Jornada, New Mexico, U.S.A (Michaelides et al. 2009). The magnitude of canopy closure is not the critical condition of an anti-erosive effective community. Species-rich communities can resist environmental perturbations through complementarity and redundancy, and their highly diverse rooting structures are of great importance for slope stabilization (Naeem 1998). The litter layer plays an important role in preventing soil detachment and providing surface roughness that minimizes soil particle movement down the slope and reduces run-off velocity (Hartanto et al. 2003). It has also been shown that high canopy closure with low understory coverage often experience severe soil erosion in the hilly red soil region of southern China (Zheng et al. 2008). Therefore, the development of vegetation cover close to ground is the key for controlling soil erosion, and the exact types of vegetation are less critical.

From the above discussion, the changes in soil water, soil nutrients, and soil anti-erodibility among afforested, pre-afforested, and reference sites suggest that natural vegetation recovery is a better management option than afforestation on the Loess Plateau.

*Question 3:* Has the restoration of afforestation been ecologically successful in this area? Afforestation has resulted in soil desiccation, even dried soil layers, and has offered no additional advantages in terms of species diversity, soil nutrients, and soil anti-erodibility compared with natural recovery. However, soil desiccation is a major issue limiting development and sustainability of forest vegetation on the Loess Plateau (Shangguan 2007). Cao et al. (2007) found that large-scale afforestation in loess soils potentially increases the severity of soil water shortages, degrades the natural environment, and increases the risks of desertification and serious economic losses because of over-consumed soil moisture. Soil desiccation and dried soil layers obviously lead to soil degradation, slowing plant growth rate, and can cause community failure and death over large areas, poor natural regeneration, greater difficulty in afforestation after decline and drought

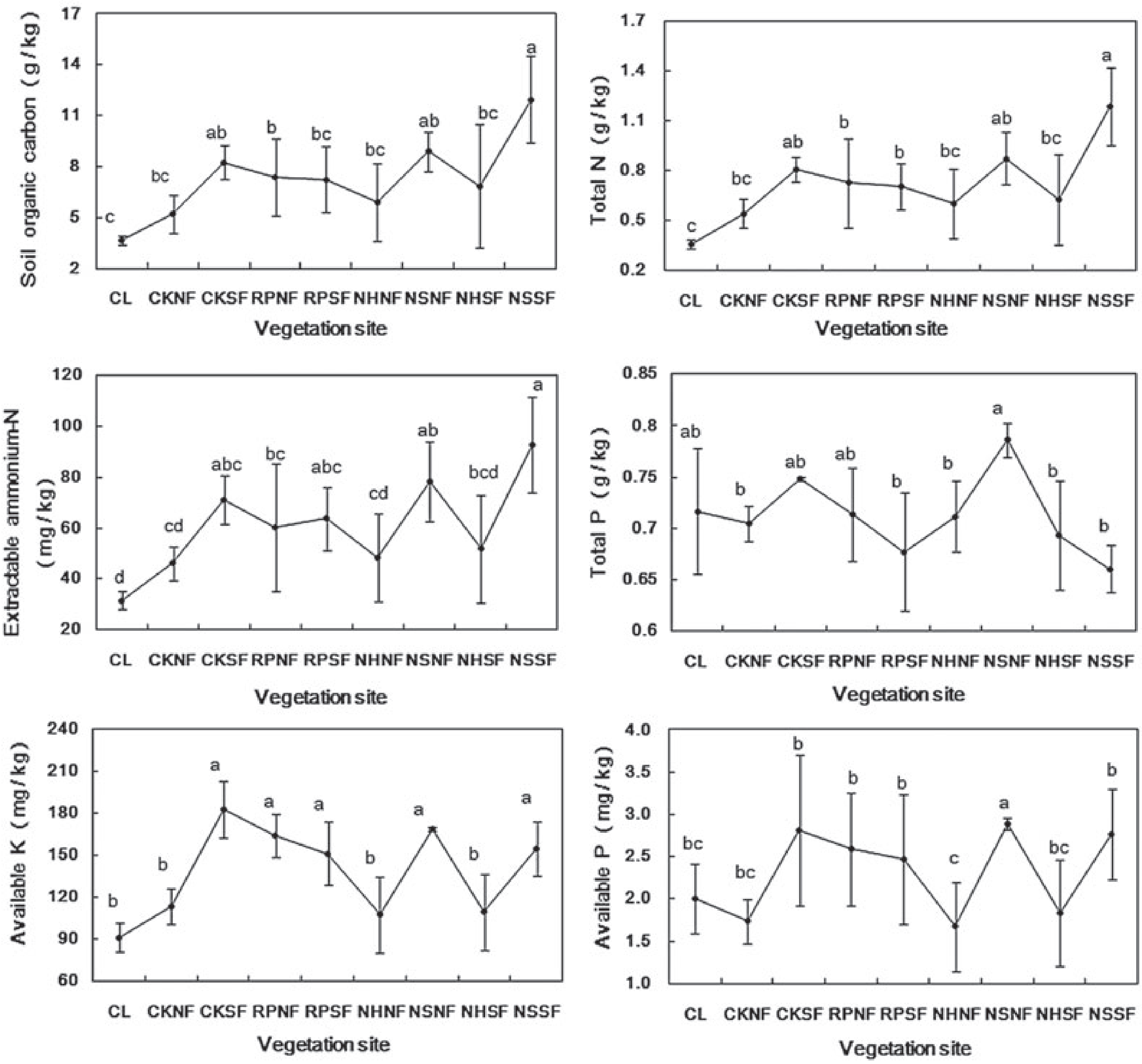


Figure 3. The soil nutrient content in the 0–20 cm layers of the vegetation sites. CL, cropland; CKNF, *Caragana korshinskii* in north-facing slope; CKSF, *C. korshinskii* in south-facing slope; RPNF, *Robinia pseudoacacia* in north-facing slope; RPSF, *R. pseudoacacia* in south-facing slope; NHNF, natural herb in north-facing slope; NSNF, natural shrub in north-facing slope; NHSF, natural herb in south-facing slope; NSSF, natural shrub in south-facing slope. Same letters in the same column indicate no significant differences between vegetation sites (e.g. a, ab, and abc means no significant differences among extractable ammonium N of NSSF, NSNF, and CKSF sites), and no same letters indicate significant differences between vegetation sites (e.g. a and bc means significant differences between soil organic carbon of NSSF and NHSF sites) based on LSD test ($p < 0.05$).

trends in microclimate (McVicar et al. 2007*a*; Shangguan 2007; Chen et al. 2008; Li et al. 2008). Moreover, evidence from the pollen record (Jiang & Ding 2005), phytolith (Lü et al. 1999), carbon isotopes (Liu et al. 2005), and paleosols (Guo et al. 1998) shows that the main body of the Loess Plateau has been covered by grassland vegetation during the Holocene epoch with more monsoon precipitation, even in the southern humid part of the plateau. The explanation for dense forest vegetation is probably associated with hilly areas and gully areas which have experienced more favorable water conditions (Liu et al. 1996). Moreover, by the end of 2005, the "Grain for Green" program has planted about 400–600 million trees in an area of 87,000 km$^2$ in Shaanxi Province (Zhou et al. 2009), but a small proportion of farmers consider planting trees (8.9%) or forage species (2.2%) to be a priority, and 37.2% of farmers plan to return to cultivation in forested areas and grassland once the project subsidies end in 2018 (Cao et al. 2009). This suggests that if afforestation has no evident ecological and economical effects, much of the restored vegetation will be converted back into farmland or rangeland again, which compromises the sustainability of environmental achievements (Cao et al. 2009).

Taken together, we can conclude that natural revegetation without intensive human interference (such as farming and grazing) could offer a more adaptive and appropriate restoration approach, rather than relying on large-scale afforestation as the primary way of implementing ecological restoration in the Loess Plateau region.

### Implications for Practice

In the degraded and arid ecosystem such as the Chinese Loess Plateau:

- Soil moisture is the indispensable measure to assess ecological success of afforestation, and natural recovery should be taken as references.
- Long-term sustainable ecological restoration should not rely on large-scale afforestation.
- Natural revegetation without intensive human interference should be considered as a more adaptive and appropriate type for ecological restoration.
- More work is needed to determine the suitable places and allocation mode for afforestation, and the technological approaches for accelerating natural revegetation.

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## LITERATURE CITED

An, S. S., A. Mentler, V. Acosta-Martínez, and E. H. Blum Winfried. 2009. Soil microbial parameters and stability of soil aggregate fractions under different grassland communities on the Loess Plateau, China. Biologia **64:**424–427.

Cao, S. X., L. Chen, Z. D. Liu, and G. Wang. 2008. A new tree-planting technique to improve tree survival and growth on steep and arid land in the Loess Plateau of China. Journal of Arid Environments **72:**1374–1382.

Cao, S. X., L. Chen, C. G. Xu, and Z. D. Liu. 2007. Impact of three soil types on afforestation in China's Loess Plateau: growth and survival of six tree species and their effects on soil properties. Landscape and Urban Planning **83:**208–217.

Cao, S. X., C. G. Xu, L. Chen, and X. Q. Wang. 2009. Attitudes of farmers in China's northern Shaanxi Province towards the land-use changes required under the Grain for Green Project, and implications for the project's success. Land Use Policy **26:**1182–1194.

Chen, H. S., M. A. Shao, and Y. Y. Li. 2008. Soil desiccation in the Loess Plateau of China. Geoderma **143:**91–100.

Chen, L. D., Z. L. Huang, J. Gong, B. J. Fu, and Y. L. Huang. 2007. The effect of land cover/vegetation on soil water dynamic in the hilly area of the loess plateau, China. Catena **70:**200–208.

Critchley, C. N. R., B. J. Chambers, J. A. Fowbert, R. A. Sanderson, A. Bhogal, and S. C. Rose. 2002. Association between lowland grassland plant communities and soil properties. Biological Conservation **105:**199–215.

Durán Zuazo, V. H., J. R. Francia Martínez, C. R. Rodríguez Pleguezuelo, A. Martínez Raya, and B. Carcéles Rodríguez. 2006. Soil-erosion and runoff prevention by plant covers in a mountainous area (SE Spain): implications for sustainable agriculture. Environmentalist **26:**309–319.

Fu, B. J., and L. D. Chen. 2000. Agricultural landscape spatial pattern analysis in the semiarid hill area of the Loess Plateau, China. Journal of Arid Environments **44:**291–303.

Fu, B. J., Q. J. Zhang, L. D. Chen, W. W. Zhao, H. Gulinck, G. B. Liu, Q. K. Yang, and Y. G. Zhu. 2006. Temporal change in land use and its relationship to slope degree and soil type in a small catchment on the Loess Plateau of China. Catena **65:**41–48.

Guo, Z. T., T. S. Liu, N. Fedoroff, L. Y. Wei, Z. L. Ding, N. Q. Wu, H. Y. Lü, W. Y. Jiang, and Z. S. An. 1998. Climate extremes in loess of China coupled with the strength of deep-water formation in the North Atlantic. Global and Planetary Change **18:**113–128.

Hartanto, H., R. Prabhu, A. S. E. Widayat, and C. Asdak. 2003. Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. Forest Ecology and Management **180:**361–374.

Herrick, J. E. 2000. Soil quality: indicator of sustainable land management? Applied Soil Ecology **15:**75–83.

Hessel, R., I. Messing, L. D. Chen, C. Ritsema, and J. Stolte. 2003. Soil erosion simulations of land use scenarios for a small Loess Plateau catchment. Catena **54:**289–302.

Hobbs R. J., and D. A. Norton. 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology **4:**93–110.

Jiang, H. C., and Z. L. Ding. 2005. Temporal and spatial changes of vegetation cover on the Chinese Loess Plateau through the last glacial cycle: evidence from spore-pollen records. Review of Palaeobotany and Palynology **133:**23–37.

Jiang, Y., M. Y. Kang, Q. Z. Gao, L. H. He, M. Xiong, Z. B. Jia, and Z. P. Jin. 2003. Impact of land use on plant biodiversity and measures for biodiversity conservation in the Loess Plateau in China—a case study in a hilly-gully region of the Northern Loess Plateau. Biodiversity and Conservation **12:**2121–2133.

Li, J., B. Chen, X. F. Li, Y. J. Zhao, Y. J. Ciren, B. Jiang, W. Hu, J. M. Cheng, and M. A. Shao. 2008. Effects of deep soil desiccation on artificial forestlands in different vegetation zones on the Loess Plateau of China. Acta Ecologica Sinica **28:**1429–1445.

Li, Y. Y., and M. A. Shao. 2003. Natural succession and evolution of structural characteristics of forest community in Ziwuling area on the Loess Plateau (in Chinese with English abstract). Acta Botanica Boreali-occidentalia Sinica **23:**693–699.

Liu, D. S., Z. T. Guo, N. Q. Wu, and H. Y. Lü. 1996. Prehistoric vegetation on the Loess Plateau: steppe or forest? Journal of Southeast Asian Earth Science **13:**341–346.

Liu, W. G., H. Yang, Y. N. Cao, Y. F. Ning, L. Li, J. Zhou, and Z. S. An. 2005. Did an extensive forest ever develop on the Chinese Loess Plateau during the past 130 ka?: a test using soil carbon isotopic signatures. Applied Geochemistry **20:**519–527.

Lu, H., M. R. Raupach, T. R. McVicar, and D. J. Barrett. 2003. Decomposition of vegetation cover into woody and herbaceous components using AVHRR NDVI time series. Remote Sensing of Environment **86:**1–18.

Lü, H. Y., D. S. Liu, N. Q. Wu, J. M. Han, and Z. T. Guo. 1999. Phytolith record of vegetation succession in the southern Loess Plateau since late Pleistocene (in Chinese with English abstract). Quaternary Science **4:**336–349.

Margalef, R. 1958. Information theory in ecology. General Systems **3:**36–71.

McVicar, T. R., L. T. Li, T. G. Van Niel, L. Zhang, R. Li, Q. K. Yang, et al. 2007*a*. Developing a decision support tool for China's re-vegetation program: simulating regional impacts of afforestation on average annual streamflow in the Loess Plateau. Forest Ecology and Management **251:**65–81.

McVicar, T. R., T. G. Van Niel, L. Li, M. F. Hutchinson, X. Mu, and Z. Liu. 2007*b*. Spatially distributing monthly reference evapotranspiration and pan evaporation considering topographic influences. Journal of Hydrology **338:**196–220.

McVicar, T. R., T. G. Van Niel, L. T. Li, Z. M. Wen, Q. K. Yang, R. Li, and F. Jiao. 2010. Parsimoniously modelling perennial vegetation suitability and identifying priority areas to support China's re-vegetation program in the Loess Plateau: matching model complexity to data availability. Forest Ecology and Management **259:**1277–1290.

McVicar, T. R., G. Zhang, A. S. Bradford, H. Wang, W. R. Dawes, L. Zhang, and L. T. Li. 2002. Monitoring regional agricultural water use efficiency for Hebei Province on the North China Plain. Australian Journal of Agricultural Research **53:**55–76.

Michaelides, K., D. Lister, J. Wainwright, and A. J. Parsons. 2009. Vegetation controls on small-scale runoff and erosion dynamics in a degrading dryland environment. Hydrological Processes **23:**1617–1630.

Naeem, S. 1998. Species redundancy and ecosystem reliability. Conservation Biology **12:**39–45.

Nichols, O. G., and F. M. Nichols. 2003. Long-term trends in faunal recolonization after bauxite mining in the jarrah forest of southwestern Australia. Restoration Ecology **11:**261–272.

Pielou, E. C. 1969. An introduction to mathematical ecology. John Wiley, New York.

Pimentel, D., and N. Kounang. 1998. Ecology of soil erosion in ecosystems. Ecosystems **1:**416–426.

Reay, S. D., and D. A. Norton. 1999. Assessing the success of restoration plantings in a temperate New Zealand Forest. Restoration Ecology **7:**298–308.

Ruiz-Jaén, M. C., and T. M. Aide. 2005. Vegetation structure, species diversity, and ecosystem processes as measures of restoration success. Forest Ecology and Management **218:**159–173.

Ryder, D. S., and W. Miller. 2005. Setting goals and measuring success: linking patterns and processes in stream restoration. Hydrobiologia **552:**147–158.

Shangguan, Z. P. 2007. Soil desiccation occurrence an its impact on forest vegetation in the Loess Plateau of China. The International Journal of Sustainable Development and World Ecology **14:**299–306.

Shannon, C. E., and W. Weaver. 1949. The mathematical theory of communication. University of Illinois Press, Urbana.

Shi, H., and M. A. Shao. 2000. Soil and water loss from the Loess Plateau in China. Journal of Arid Environments **45:**9–20.

Shi, N. H. 1991. Comments on the historical distribution and vicissitudes of the national vegetation of China (in Chinese with English abstract). Collections of Essays On Chinese Historical Geography **3:**43–73.

Sorensen, T. A. 1948. A method of establishing groups of equal amplitude in plant sociology based on similarity of species content, and its application to analyses of the vegetation on Danish commons. Kongelige Danske Videnskabernes Selskabs Biologiske Skrifter **5:**1–34.

van Dijk, A. I. J. M., and R. J. Keenan. 2007. Planted forests and water in perspective. Forest Ecology and Management **251:**1–9.

Wang, J., A. C. Borsboom, and G. C. Smith. 2004. Flora diversity of farm forestry plantations in southeast Queensland. Ecological Management and Restoration **5:**43–51.

Wang, J., B. J. Fu, Y. Qiu, and L. D. Chen. 2003. Analysis on soil nutrient characteristics for sustainable land use in Danangou catchment of the Loess Plateau, China. Catena **54:**17–29.

Wang, L., M. A. Shao, and Q. C. Hou. 2000. Preliminary research on measured indexes of dried soil layer (in Chinese with English abstract). Journal of Soil Water Conservation **4:**87–90.

Zhang, F., S. L. Zhang, Z. J. Cheng, and H. Y. Zhao. 2007. Time structure and dynamics of the insect communities in bush vegetation restoration areas of Zhifanggou watershed in Loess hilly region. Acta Ecologica Sinica **27:**4555–4562.

Zhang, Q. J., B. J. Fu, L. D. Chen, W. W. Zhao, Q. K. Yang, G. B. Liu, and H. Gulinck. 2004. Dynamics and driving factors of agricultural landscape in the semiarid hilly area of the Loess Plateau, China. Agriculture, Ecosystems and Environment **103:**535–543.

Zheng, F. L. 2006. Effect of vegetation changes on soil erosion on the Loess Plateau. Pedosphere **16:**420–427.

Zheng, H., F. L. Chen, Z. Y. Ouyang, N. M. Tu, W. H. Xu, X. K. Wang, H. Miao, X. Q. Li, and Y. X. Tian. 2008. Impacts of reforestation approaches on runoff control in the hilly red soil region of Southern China. Journal of Hydrology **356:**174–184.

Zhou, H. J., A. Van Rompaey, and J. A. Wang. 2009. Detecting the impact of the "Grain for Green" program on the mean annual vegetation cover in the Shaanxi province, China using SPOT-VGT NDVI data. Land Use Policy **26:**954–960.