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Vegetation community and soil characteristics of abandoned agricultural land and pine plantation in the Qinling Mountains, China

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ABSTRACT

In order to determine the changes in the characteristics of the vegetation and soil following agricultural abandonment and compare the effects of different restoration approaches on ecosystem recovery, we studied the vegetation community and soil characteristics (nutrients, bulk density, water content and pH) of *Pinus tabulaeformis* plantations and abandoned croplands in different successional stages in the Foping National Nature Reserve, located in the Qinling Mountains, northwest China. The results indicated that natural vegetation and habitat could be restored via natural regeneration. These spontaneous restoration forests were characterized by high diversity, high soil fertility and rich unique species. The soil organic matter, total carbon, total nitrogen, available potassium, community cover, depth of litter, depth of humus and soil water content increased significantly with years after abandonment, while the total potassium, total phosphorus, available phosphorus, soil pH did not seem to change significantly with abandonment time. The soil mineral nitrogen (NH_4-N + NO_3-N) tended to increase during the first 50 years after abandonment and then decreased. The pine plantations tended to show a low level of biodiversity in tree and herb layer, but the shrub layer (including sapling) composition and diversity were similar to secondary forests. Although the pine plantation showed lower soil fertility, they did not seem to result in the habitat-degradation.

1. Introduction

Forest restoration has been increasingly concerned by governments and ecologists since deforestation has result in serious environmental issues and the loss of ecological services (Foley et al., 2005; Lamb et al., 2005; Chazdon, 2008). In some countries (*e.g.*, China, America, etc.) forest cover has begun to expand because of afforestation and natural regeneration (Fang et al., 2001; Chazdon, 2008). Agricultural abandonment and tree planting for commercial or restoration purposes are two major driving forces for the forest restoration (Guariguata and Ostertag, 2001; Rey Benayas et al., 2007; Chazdon, 2008).

Restoration approach is crucial for the structure and function of the newly established forest (Ashton et al., 2001; Lamb et al., 2005; Jones et al., 2009). Appropriate approach to restore forest ecosystems depends on the severity of damage to the land resource, the goals of rehabilitation and the availability of resources for repairing the damage (Lugo, 1997). Current restoration approaches have focused on the recovery of composition, structure, natural habitat, ecosystem process and services (Lee et al., 2002; Ruiz-Jaén and Aide, 2005; Lamb et al., 2005; Maestre et al., 2006; Marcos et al., 2007; Chazdon, 2008; Robin and Chazdon, 2008).

Many studies demonstrated that traditional timber plantations had made only minor contribution to improve ecosystem services and enhance biodiversity conservation (Lamb et al., 2005), and might have generated soil degradation and negative impacts on biodiversity (Liu et al., 1998; Jiang et al., 2003; Paritsis and Aizen, 2008). A few studies argued that some plantations did not cause neither a persistent site degradation nor degeneration of the community (Czerepko, 2004), and could play a role in conserving biodiversity and work as a nurse for establishing late-successional tree species (Ashton et al., 1997; Lugo, 1997; Hartley, 2002). These studies indicated that some tree plantations could act as an effective tool for arresting site degradation and a catalyst for land and forest vegetation rehabilitation (Lugo, 1997; Otsamo, 2000).

Alternative of forest restoration is to protect and facilitate natural regeneration in abandoned lands, where the degraded system has not crossed an ecological threshold and reached a new steady state condition (Lamb et al., 2005). Natural regeneration has increasingly received attention for its higher natural value and lower cost (Prach and Pyšek, 2001; Prach, 2003; Jiao et al., 2007). Knowledge of secondary succession in abandoned agricul-

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tural lands yields insights into conservation and restoration of degraded lands (Lee et al., 2002; Bonet, 2004; Jiao et al., 2007).

China's 'Grain-for-Green' project (*i.e.*, conversion of cropland to forest and grassland), launched in 1999, is one of the world's most ambitious ecological restoration programs. The project has ambition to increase area of forest and grassland to 32×10^6 ha by 2010, including 14.667×10^6 ha former croplands and 17.33×10^6 ha former barren lands. Abandonment for natural recovery and tree plantation are two major approaches for forest rehabilitation. Previous studies on the development of vegetation and soil qualities in abandoned croplands have provided significant information for forest restoration (Liu et al., 2002; Li and Shao, 2006; Jiao et al., 2007). However, further studies, which focus on the response of vegetation structure, species richness and soil properties to restoration approaches, are still essential, especially in biodiversity hotspots. Great efforts for restoring ecosystems have been made in China's biodiversity hotspots (*e.g.*, establishing nature reserve, reafforestation), but in some protected areas, the rate of loss of the high-quality habitat was much higher than before the effort was made (Liu et al., 2001). Choosing appropriate approach to restore ecosystems in biodiversity hotspots is extremely important.

In the present study, we investigated the vegetation community and soil characteristics of pine plantations and abandoned croplands in different succession stages in the Foping National Nature Reserve (FNNR), located in Qinling Mountains, China. The region is a typical mountainous zone of China and famous for its high biodiversity. FNNR is a member of the UNESCO/MAB World Network of Biosphere Reserves, and provides important habitat for many endangered animals, *e.g.*, giant panda (*Ailuropoda melanoleuca*), golden monkey (*Rhinopithecus roxellana*) and golden takin (*Budorcas taxicolor bedfordi*). However, FNNR is located in state-level poverty-stricken county, where the human and environmental resources clash sharply. The objectives of the current study are to: (1) examine the possibility of the natural recovery of species-rich vegetation and habitat on arable fields after abandonment in the biodiversity hotspots; (2) determine the changes in the characteristics of the vegetation and soil following abandonment; and (3) compare the effects of natural regeneration and plantation on ecosystem recovery.

2. Materials and methods

2.1. Study area

This study was conducted in the FNNR, south aspect of the Qinling Mountains, China (Fig. 1). The region belongs to a subtropical humid region, and annual precipitation ranges from 950 to 1200 mm, most of which falls between July and September. Mean annual temperature is 11.8 to -0.3 °C in January and 21.9 °C in July. Frost free days average 220 per year (Wang et al., 2006).

Natural vegetation of the study area is deciduous broad-leaved forests with coverage of over 70%. The soil is characterized by slight acid yellow brown soil developed form granitic gneiss. The agriculture practice had been intensive in history, but after the Guangxu period of Qing dynasty (1875–1908), large numbers of population had emigrated from this region (Ren et al., 1998). In 1978, the FNNR was built to protect the endangered giant pandas, followed by the "Grain-for-Green" project launched in 1999. Consequently, previous agriculture lands were abandoned and natural regeneration communities are at different secondary succession stages with little human activities.

2.2. Field work and laboratory analysis

The sampling was carried out within 4 km in a small watershed (Fig. 1). In the sampling region, a comprehensive survey of vegetation and soil was conducted to ensure the representativeness of the sampling sites. To minimize variability, near sites were selected with similar soil type, parent material and topography (slope: 20–40°) (Table 1). Fourteen typical abandoned croplands (including 3 grasslands; 5 shrublands; 6 secondary forests) were selected to represent different periods of natural succession after abandonment. We inferred that the sites derived from cropland based on the abandoned irrigation ditch and ridge (Ren et al., 1998). The abandonment time was inquired or estimated from the tree-rings of oldest pioneer trees in stands. Four pine (*Pinus tabulaeformis*) plantations established on previous croplands were selected. Because the pine plantations were planted as ecological forests, there were no fertilization, cutting and desinsection. All the sampling sites were located within the nature reserve with little disturbance and management practice (Table 1). Three croplands were selected nearby the abandoned croplands. Corn was cultivated and had been harvested before investigation in the croplands. Because the undisturbed old forest was absent in the sampling area, we have to compare the stands with old forests in other area of the FNNR (Ren et al., 1998).

The altitude of all the sampling sites varied from 1075 to 1301 m a.s.l. (Table 1). We considered the vegetation types in the region did not change along this altitude variation based on our comprehensive field survey and previous study about the vertical zone spectrum of vegetation in FNNR (Yue et al., 2000). The sampling sites were located in the oak forest zone (Yue et al., 2000). The vegetation, especially in the understory and the remnant natural vegetation nearby the sampling sites, was characterized by *Quercus variabilis*, *Castanea mollissima*, *Bashania fargesii*, etc., and the composition of vegetation communities was similar. Prior land use may have created differences among sites that could confound our chronosequence approach, but no detailed information was available to allow more detailed testing of the chronosequence assumption in some sites. Although there were a few obvious limitations in space-for-time substitution, the chronosequence approach could provided significant insights into the patterns and mechanisms of plant and soil evolvement (Foster and Tilman, 2000; Davidson et al., 2007).

Field data collection was carried out in August 2007. Three 1 m × 1 m herb quadrats, three 5 m × 5 m shrub quadrats (including shrub and sapling), and four 10 m × 10 m tree quadrats were investigated in each stand, while they were applicable. The cover of each species, community cover of each layer (*i.e.*, herb, shrub and tree), herb abundance (Braun-Blanquet) and tree height were estimated visually by the same pair of observers working together (so that bias, if it exists, is similar in all the stands). Diameters at breast height (DBH) and base were measured. In each sampling site, five plots were selected randomly for measure the depth of litter and humus layer.

Since soil properties change mainly occurred in the surface soil after land use conversion from cropland to natural vegetation or plantation (Liu et al., 2002; Li and Shao, 2006; Chen et al., 2007; Zhang et al., 2010), so three soil samples were taken randomly at a depth of 0–20 cm in each stand. The soil samples were air-dried, and passed through 1.0 and 0.25 mm sieves. Soil organic matter content (SOM) (by the $K_2Cr_2O_7$ titration method after digestion), total phosphorus (TP) (molybdenum blue colorimetry), total potassium (TK) (flame photometry), available phosphorus (AP) (by 0.5 M $NaHCO_3$ extraction (1:20) colorimetric method), and available potassium (AK) (by 1 M NH_4OAC extraction (1:20) flame photometry), were determined. Soil pH was determined in 1:2.5 soil–water slurry using a combination glass electrode. NO_3-N was extracted with pure water and measured by phenol disulfonic acid spectriphotometric method. NH_4-N was extracted from fresh soil samples with a 2 M KCl solution and concentration was determined by indophenol blue colorimetry. Total nitrogen (TN) and total carbon (TC) content

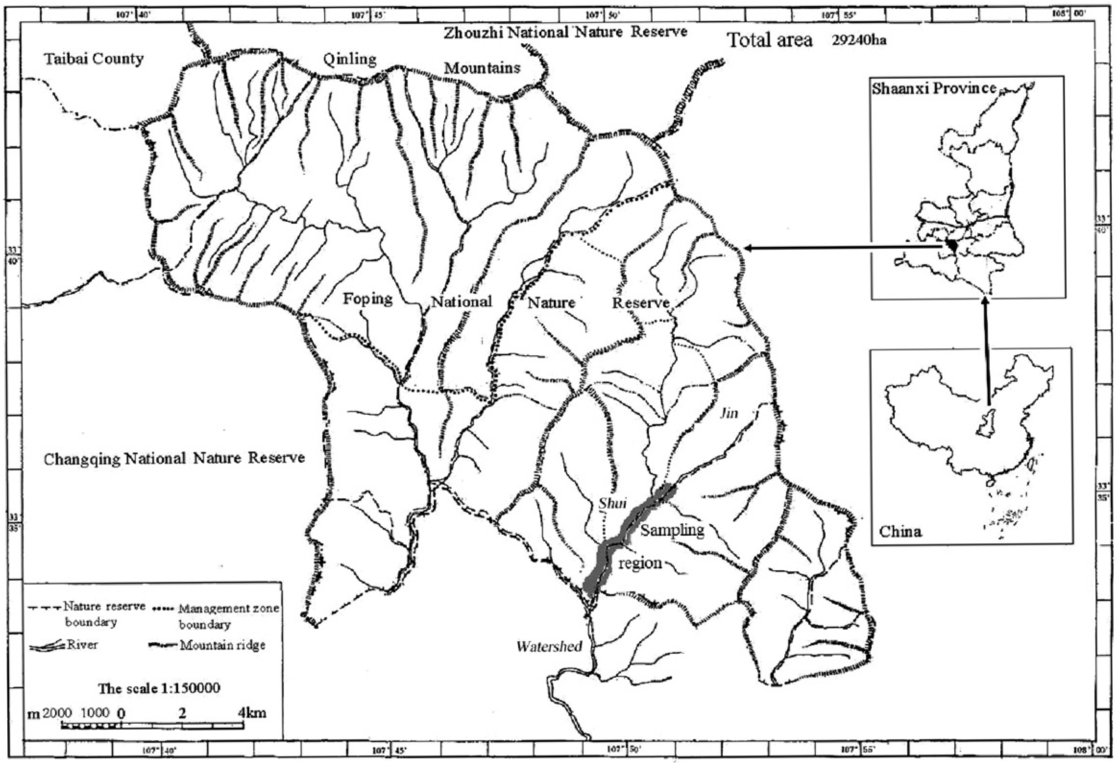

Fig. 1. Location of the study area.

Table 1
Description of sampling sites.

Sites	Land use	Age (year)	Cover (%)	Slop (°)	Altitude (m)	Latitude (N)	Longitude (E)	Prior land use	Source of disturbances	Management practices
F1	Cropland	0	50	25	1075	33°32′52.1″	107°49′6.4″	Corn field	Farmer, widelife	Tillage, fertilization
F2	Cropland	0	35	21	1100	33°33′8.4″	107°49′15″	Corn field	Farmer, widelife	Tillage, fertilization
F3	Cropland	0	40	25	1202	33°35′2.3″	107°50′35.5″	Corn field	Farmer, widelife	Tillage, fertilization
H1	Secondary herb	1	80	25	1252	33°35′4″	107°50′40.5″	Corn field	Widelife	Abandonment with no management
H2	Secondary herb	5	80	30	1233	33°34′54.8″	107°50′31.1″	Corn field	Widelife	Abandonment with no management
H3	Secondary herb	4	30	20	1130	33°33′42″	107°49′31″	Corn field	Widelife	Abandonment with no management
S1	Secondary shrub	5	54	32	1098	33°33′12.4″	107°49′21.3″	Corn field	Widelife	Abandonment with no management
S2	Secondary shrub	6	50	21	1290	33°35′16.1″	107°51′2.6″	Corn field	Widelife	Abandonment with no management
S3	Secondary shrub	7	85	25	1251	33°35′4.5″	107°50′39.9″	Corn field	Widelife	Abandonment with no management
S4	Secondary shrub	7	45	20	1092	33°33′8.5″	107°49′15″	Corn field	Widelife	Abandonment with no management
S5	Secondary shrub	12	60	20	1269	33°35′10.8″	107°50′59.3″	Corn field	Widelife	Abandonment with no management
N1	Secondary forest	15	70	26	1280	33°34′54.1″	107°50′31.1″	Corn field	Widelife	Abandonment with no management
N2	Secondary forest	20	95	22	1140	33°33′44.7″	107°49′32.4″	Corn field	Widelife	Abandonment with no management
N3	Secondary forest	≈45	80	30	1301	33°35′21.5″	107°51′12.6″	Cropland	Widelife	Abandonment with no management
N4	Secondary forest	≈50	85	31	1298	33°35′7.9″	107°50′59.8″	Cropland	Widelife	Abandonment with no management
N5	Secondary forest	≈70	95	20	1276	33°35′19.7″	107°51′7.2″	Cropland	Widelife	Abandonment with no management
N6	Secondary forest	≈100	90	27	1257	33°32′21.2″	107°49′13.7″	Cropland	Widelife	Abandonment with no management
P1	Pine plantation	10	45	25	1095	33°33′8.5″	107°49′15″	Corn field	Widelife	No fertilization, cutting and desinsection
P2	Pine plantation	15	85	30	1170	33°33′44″	107°49′33.2″	Corn field	Widelife	No fertilization, cutting and desinsection
P3	Pine plantation	30	95	40	1101	33°32′52.1″	107°49′6.3″	Cropland	Widelife	No fertilization, cutting and desinsection
P4	Pine plantation	≈45	89	30	1121	33°33′8.5″	107°49′14.9″	Cropland	Widelife	No fertilization, cutting and desinsection

(%) were measured using a Nitrogen/Carbon Analyzer (NA-1500-NC Series 2) with Eager 200 software (Fisons Instruments, Beverly, MA). To measure bulk density and soil water content, three intact soil cores were taken by core sampler and dried at 105 °C for 48 h.

2.3. Statistical analysis

The species importance values (IV) were calculated by the relative density, relative frequency and relative dominance (relative dominance of herb and shrub were estimated from cover). Five diversity indices were used to assess the vegetation community, and they were Shannon–Wiener heterogeneity index (*H*) (Shannon & Weaver, 1949), Simpson heterogeneity index (*D*) (Simpson, 1949), Pielou evenness index (*E*) (Pielou, 1966), Margalef richness index (*Ma*) (Margalef, 1958), Patrick richness index (*S*) (Patrick, 1949).

We used the multi-response permutation procedures (MRPP) in PC-ORD Version 4.0 (McCune and Mefford, 1999) to test for differences in species composition. One-way analysis of variance (ANOVA) and non-parametric Mann–Whitney's *U*-test were used to determine whether there were significant differences between land use types in terms of diversity indices. Pearson correlation analysis was performed between the environmental variables and diversity indices. Species indicator analysis, carried out in PC-ORD Version 4.0 (McCune and Mefford, 1999), was used to indicate which species were unique to particular land uses (Dufendre and Legendre, 1997).

In order to study the relationship between the vegetation and soil variables, Cannonical Correspondence Analysis (CCA) was carried out using the ordination program CANOCO (version 4.5, ter Braak and Smilauer, 2002). Montecarlo test (1000 permutations) was employed to determine the significance of the obtained canonical axes. In each site, top ten species (with importance value) of each layer (*i.e.*, herb, shrub, tree) were selected as the representative species. Species occurring at least one stand under this selection criterion were considered in CCA. Ten soil variables of all the studied environmental variables were selected by forward selection procedure: TC, TN, TK, AK, AP, C:N, pH, BD, the depth of litter (DL) and humus (DH) layer.

To compare the effect of different approaches on the species richness and habitat variables, ANOVA was used to determine whether there were significant differences between plantations and secondary forests. Two 'relative old' secondary forests (~70 years and ~100 years) were eliminated in order to minimize the effect of time.

In order to analyze changes in the environmental variables during the successional process, the relationships between environmental variables and the abandonment time were estimated using curve estimation and the best regression model was chosen. ANOVA, regression and Pearson's correlation analysis were performed using program SPSS 13.0.

3. Results

3.1. Floristic composition

A total of 47 tree species, 87 herbaceous species and 90 shrub species (including 26 saplings) were recorded in all the sampling sites. Forty-five species occurred only at one site. The secondary forests contained 6 unique herbaceous species and 28 unique shrub species, while the plantations contained 5 and 14, respectively. A few endangered and rare species including *Gastrodia elata*, *Glycine soja*, *Sargentodoxa cuneata*, presented in shrublands and secondary forests.

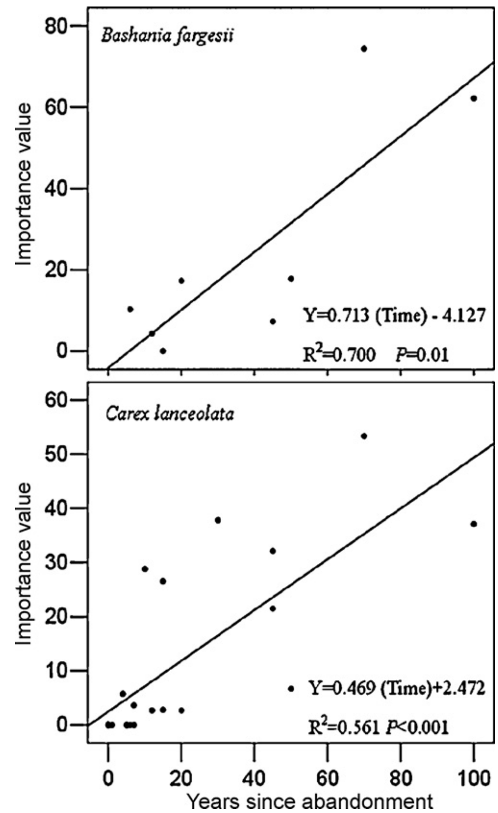

Fig. 2. Relationship between years since abandonment and the IV of two indictor species.

Results from multi-response permutation procedures (MRPP) showed that the overstory (tree layer) of secondary forests was different from that of the plantations, while no differences were found in the understory (*i.e.*, herb layer and shrub layer) between them. The herb compositions of three pairings (farmland with grassland, grassland with shrubland, plantation with secondary forest) were similar, and other 7 pairings were different.

Indicator species analysis showed that 30 species had indicator values ≥50% (Table 2). Annual herbs were major indictor species for the farmland and grassland. *Rubus flosculosus* and *Salix sinopurpurea* were indictor species for shrubland, where they were also dominant species. The major indictor species for the plantations included *P. tabulaeformis*, *Rubus coreanus*, *Trachycarpus fortunei*, *Q. variabilis*, *Pteridium aquilinum* var. *latiusculum*, *Corylus heterophylla*, *Lysimachia stenosepala* var. *stenosepala* and *Carex lanceolata*, while half of them occurred in secondary forests. *C. lanceolata* was also one of the most common species in the secondary forests. Most indicator species (7 of the 11) of the secondary forest were also present in plantations. The IV of *C. lanceolata* increased significantly with years after abandonment (Fig. 2). *B. fargesii*, appearing in shrubland about 6 years after abandonment, was dominant in the shrub layer of the relative old secondary forests and its IV also increased significantly with abandonment time (Fig. 2).

3.2. Species diversity

For the herb layer, richness and heterogeneity differed between shrubland (S = 14.8, H = 2.40) and plantation (S = 10.75, H = 2.08), and the heterogeneity (*H*) of shrubland were also significantly higher than that of grassland (H = 2.18) (Table 3). Herbaceous species richness was significantly negatively correlated with the abandonment time, DH and soil pH ($p < 0.05$), and the heterogeneity (*D*, *H* indices) was significantly negatively correlated with the

Table 2
Species indicator analysis.

Species	Land use type					p-Value
	Cropland	Grassland	Shrubland	Pine plantation	Secondary forest	
Mazus gracilis	100	0	0	0	0	0.002
Myosoton aquaticum	100	0	0	0	0	0.002
Capsella bursa-pastoris	67	0	0	0	0	0.026
Viola philippica	67	0	0	0	0	0.022
Oxalis acetosella	61	0	2	0	0	0.019
Sedum lineare	59	0	2	0	0	0.025
Gnaphalium affine	55	0	4	0	0	0.025
Equisetum ramosissimum	0	67	0	0	0	0.036
Poa annua	0	67	0	0	0	0.036
Rubus flosculosus	0	0	71	3	0	0.009
Salix sinopurpurea	0	0	60	0	0	0.04
Pinus tabulaeformis	0	0	0	97	1	0.001
Rubus coreanus	0	0	0	75	0	0.005
Trachycarpus fortunei	0	0	0	75	0	0.005
Quercus variabilis	0	0	0	71	2	0.018
Pteridium aquilinum var. *latiusculum*	0	0	1	70	0	0.007
Corylus heterophylla	0	0	0	58	11	0.025
Lysimachia stenosepala var. *stenosepala*	0	0	0	58	0	0.005
Carex lanceolata	0	1	1	57	38	0.017
Abelia engleriana	0	0	0	4	70	0.018
Schisandra sphenanthera	0	0	0	5	68	0.017
Smilax scobinicaulis	0	0	2	3	67	0.005
Carpinus turczaninowii	0	0	0	15	67	0.003
Viburnum lobophyllum	0	0	0	0	67	0.014
Acer erianthum	0	0	0	0	67	0.018
Acer davidii	0	0	0	0	67	0.01
Bashania fargesii	0	0	3	4	63	0.02
Rubus amabilis	0	0	5	0	62	0.028
Toxicodendron verniciflunm	0	0	1	2	58	0.034
Arthraxon hispidus	5	1	2	18	54	0.037

Table 3
Mean value and standard error of diversity indices (*H*, Shannon–Wiener heterogeneity index; *D*, Simpson heterogeneity index; *E*, Pielou evenness index; *Ma*, Margalef richness index; *S*, Patrick richness index).

	Land use type	*S*	*D*	*H*	*E*	*Ma*
Herb layer	Cropland	13.00(1.15)	0.87(0.01)	2.31(0.04)	0.90(0.02)	2.68(0.32)
	Grassland	14.00(1.73)	0.83(0.03)	2.18(0.01)	0.84(0.04)	2.82(0.38)
	Shrubland	14.80(0.66)	0.88(0.01)	2.40(0.05)	0.89(0.02)	3.00(0.14)
	Secondary forest	13.17(3.13)	0.80(0.07)	2.08(0.33)	0.91(0.02)	2.64(0.68)
	Pine plantation	10.75(1.25)	0.83(0.02)	2.08(0.11)	0.89(0.02)	2.12(0.27)
Shrub layer	Shrubland	9.20(3.02)	0.79(0.04)	1.81(0.24)	0.88(0.02)	1.78(0.66)
	Secondary forest	19.83(3.79)	0.78(0.09)	2.29(0.32)	0.78(0.07)	4.09(0.82)
	Pine plantation	15.25(2.02)	0.87(0.02)	2.33(0.15)	0.85(0.02)	3.09(0.44)
Tree layer	Secondary forest	13.50(1.41)	0.87(0.01)	2.29(0.11)	0.89(0.02)	2.71(0.31)
	Pine plantation	4.75(1.31)	0.39(0.14)	0.82(0.28)	0.46(0.16)	0.81(0.29)

Table 4
Pearson correlation coefficients between habitat variables and diversity indices of herb layer (*H*, Shannon–Wiener heterogeneity index; *D*, Simpson heterogeneity index; *E*, Pielou evenness index; *Ma*, Margalef richness index; *S*, Patrick richness index).

	S	*D*	*H*	*E*	*Ma*
Time (years)	−0.515*	−0.578**	−0.611**	0.13	−0.521*
Cover	−0.218	−0.439*	−0.394	−0.152	−0.226
Soil water content	−0.098	−0.309	−0.261	−0.026	−0.105
Bulk density	0.268	0.464*	0.431	−0.037	0.278
pH	−0.442*	−0.26	−0.358	0.277	−0.433
Depth of litter	−0.382	−0.332	−0.389	0.104	−0.391
Depth of humus	−0.524*	−0.447*	−0.525*	0.135	−0.533*
Total carbon	−0.193	−0.531*	−0.434*	0.129	−0.197
Soil organic matter	−0.157	−0.542*	−0.425	0.112	−0.16
Total nitrogen	−0.092	−0.483*	−0.362	0.035	−0.094
NO_3-N	0.274	−0.122	0.047	−0.031	0.299
NH_4-N	−0.004	−0.215	−0.151	−0.059	−0.007
AN (NO_3-N + NH_4-N)	0.17	−0.197	−0.055	−0.052	0.184
C:N	−0.358	−0.285	−0.339	0.322	−0.371
Total phosphorus	0.004	0.119	0.074	−0.147	0.016
Available phosphorus	0.175	0.049	0.12	−0.15	0.197
Total potassium	−0.186	0.1	−0.005	0.27	−0.181
Available potassium	−0.237	−0.336	−0.321	0.123	−0.238

* Correlation is significant at the 0.05 level (2-tailed).
** Correlation is significant at the 0.01 level (2-tailed).

Table 5
Pearson correlation coefficients between habitat variables and diversity indices of shrub layer (*H*, Shannon–Wiener heterogeneity index; *D*, Simpson heterogeneity index; *E*, Pielou evenness index; *Ma*, Margalef richness index; *S*, Patrick richness index).

	S	*D*	*H*	*E*	*Ma*
Time (years)	0.218	−0.462	−0.096	−0.702**	0.218
Cover	0.428	−0.128	0.191	−0.427	0.428
Soil water content	0.381	−0.271	0.072	−0.473	0.381
Bulk density	−0.301	0.293	−0.028	0.553*	−0.301
pH	−0.268	−0.536*	−0.497	−0.384	−0.268
Depth of litter	0.595*	0.022	0.379	−0.438	0.595*
Depth of humus	0.37	−0.166	0.149	−0.549*	0.37
Total carbon	0.321	−0.395	−0.005	−0.656**	0.321
Soil organic matter	0.301	−0.366	0.013	−0.631*	0.301
Total nitrogen	0.321	−0.353	0.019	−0.595*	0.321
NO_3-N	0.287	−0.03	0.176	−0.137	0.287
NH_4-N	0.643**	−0.05	0.323	−0.243	0.643**
AN (NO_3-N + NH_4-N)	0.534*	−0.046	0.287	−0.218	0.534*
C:N	0.264	−0.149	0.084	−0.476	0.264
Total phosphorus	−0.166	0.019	−0.088	0.204	−0.166
Available phosphorus	0.016	0.085	0.089	0.129	0.016
Total potassium	−0.16	0.074	−0.076	0.069	−0.16
Available potassium	0.128	−0.399	−0.132	−0.418	0.128

* Correlation is significant at the 0.05 level (2-tailed).
** Correlation is significant at the 0.01 level (2-tailed).

abandonment time ($p < 0.01$), soil TC and DH ($p < 0.05$). No environmental factors were significantly correlated with the herb evenness ($p > 0.05$) (Table 4).

For the shrub layer, the differences in richness occurred only between secondary forests ($S = 19.83$) and shrublands ($S = 9.20$). Shrub species richness was significantly positively correlated with the depth of litter ($p < 0.05$) and soil NH_4-N ($p < 0.01$). Shrub species evenness was significantly in negative correlation with abandonment time, TC, DH, SOM and TN, whereas in positive correlation with BD ($p < 0.05$) (Table 5).

For the tree layer, tree species richness of the secondary forests ranged from 8 to 18 per 400 m^2 with an average of 13.5 per 400 m^2 (Table 3). Tree species richness in pine plantations ranged from 1 to 7 per 400 m^2, with an average of 4.75 per 400 m^2 (Table 3).

3.3. Changes of environmental variables after abandonment

SOM, TC, TN, and AK increased significantly with the abandonment time (Fig. 3). The AN (NH_4-N + NO_3-N) tended to increase during the first 50 years after abandonment and then decreased. TK, TP and AP did not seem to change significantly with abandonment time (Fig. 3). Community coverage, litter depth, humus depth and water content increased significantly with the increase of abandonment time, while the soil bulk density decreased significantly. There was no significant relationship between soil pH and abandonment time (Fig. 3).

3.4. Relationship between the vegetation and soil variables

The first axis in CCA, explaining 30% of the total variance of species-environmental, separated the more recent abandoned sites (grassland and shrubland) from the secondary forests and plantations (Fig. 4). The forest sites associated with high DL, DH, C:N. The secondary forests were separated from plantations and appeared to be associated with higher TC, TN, and AK, while plantations associated with higher TK and pH. Axis I was significant (*F*-ratio = 2.043, $p = 0.002$), and it seemed to represent the variations undergone in the succession process. Most of the herbaceous species were situated in the positive part of axis I, while most of the tree species were situated in the negative part, indicating the species change during the secondary succession.

The secondary forest groups were characterized by the later-successional species (Fig. 4), including *Quercus serrata* var. *brevipetiolata*, *Betula platyphylla*, *C. mollissima*, *Carpinus turczaninowii*, *B. fargesii*, *Parathelypteris nipponica*, *Arthraxon hispidus*, while the plantations were *Q. variabilis*, *P. tabulaeformis*, *T. fortunei*, *Quercus aliena var. acutiserrata*, *Platycarya strobilacea*, *Lespedeza floribunda*, *L. stenosepala var. stenosepala*, *P. aquilinum var. latiusculum*, and *Q. aliena var. acutiserrata*, *Q. variabilis*, and *L. stenosepala var. stenosepala*.

3.5. Comparisons between plantations and secondary forests

Tree species richness, evenness and heterogeneity were significantly higher in the secondary forests than those in the plantations (ANOVA, $p < 0.05$), while no significant differences were found in the shrub layer for all indices between the plantations and secondary forests. The herb layer of the secondary forests had higher species richness and heterogeneity (ANOVA, $p < 0.05$), while the differences in herb evenness were not significant.

The OM, AN, AP, TC, TN and water contents of the soils were significantly higher in the secondary forests than those in the plantations, while the plantations had the higher BD, TP, TK and depth of humus (ANOVA, $p < 0.05$) (Fig. 5). The differences in community cover, depth of litter and soil pH were not significant (ANOVA, $p > 0.05$) (Fig. 5).

4. Discussion

4.1. Soil properties and site conditions response to abandonment

The increases in soil nutrients (OM, TC, TN, and AK), improvement in site conditions (community cover, depth of litter, depth of humus, soil water content) and decrease in soil bulk density indicated the natural habitat restoration following abandonment (Fig. 3). The changes were mainly attributable to the development of vegetation, interaction between soil and vegetation, biotic fixation and atmospheric deposition (Knops and Tilman, 2000; Guariguata and Ostertag, 2001; Davidson et al., 2007; Du et al., 2007). The TP and AP did not change significantly with abandonment time (Fig. 3), implying the different biogeochemical process for the P during secondary succession in abandoned croplands (Davidson et al., 2007; Du et al., 2007).

Our study together with others in different areas of the world indicated that changes in soil properties showed diverse patterns during succession after abandonment (Lee et al., 2002; Du et al.,

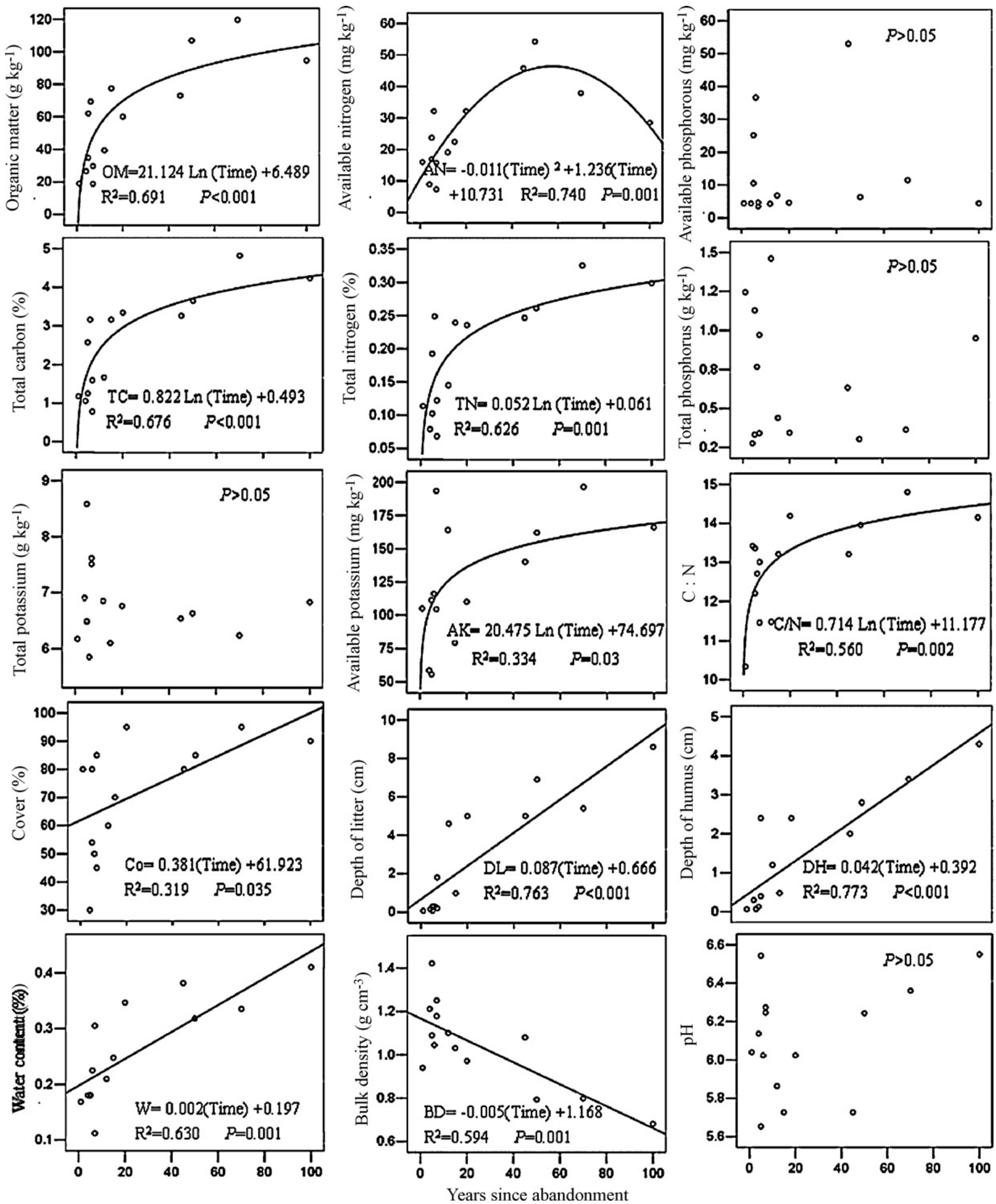

Fig. 3. Relationship between years since abandonment and the analyzed habitat variables (croplands and plantations were excluded).

2007). For example, Lee et al. (2002) found that the soil organic matter, total N, exchangeable K and available P increased with the age of abandoned field in central Korea, while the soil moisture content decreased steadily with field age after an initial rise immediately after abandonment. Du et al. (2007) demonstrated that organic matter, total nitrogen, total phosphorus, total potassium, nitrate nitrogen, ammonium nitrogen and available potassium decreased in earlier abandonment stage of succession, then increased subsequently while the available phosphorus decreased throughout succession. However, in the long run, total nitrogen and total carbon have generally increased after agricultural abandonment (Knops and Tilman, 2000).

4.2. Change of species diversity during succession

Herb species richness peaks at early stages of the secondary succession (Table 3), and the shrub species also tend to show an increase and then a decline pattern, which might be attributed to the bamboo (*B. fargesii*) establishment and colonization in the understory. The bamboo dominates the understory with higher density ($\approx$14 stem per m^2) at later-successional stages, resulting in changes in resource (*e.g.*, light) and environment condition and only a few shade-tolerant species could exist. The high richness could be the result of the coexistence of different life forms at the early or intermediate successional stages (Bonet and Pausas, 2004).

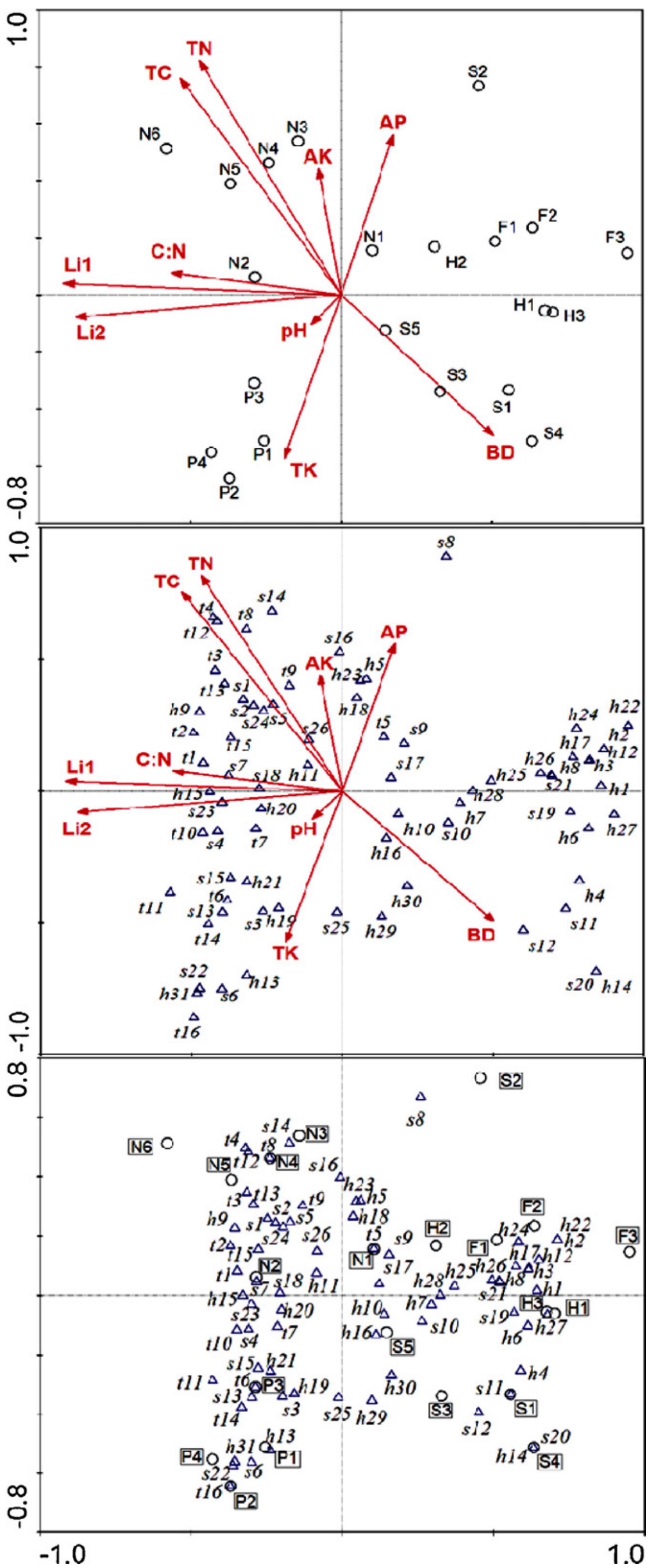

Fig. 4. Location of study sites, soil variables and species in the plane defined by the first two axes in the canonical correspondence analysis. F, cropland; H, grassland; S, Shrubland; P, Pine plantation; N, Secondary forest; TN, Total nitrogen; TC, total carbon; TK, total potassium; AP, available phosphorus; AK, available potassium; BD, soil bulk density; Li1, depth of litter; Li2, depth of humus; C:N, carbon:nitrogen; s1, *Bashania fargesii*; s2, *Smilax scobinicaulis*; s3, *Castanea mollissima*; s4, *Quercus serrata* var. *brevipetiolata*; s5, *Abelia engleriana*; s6, *Lespedeza floribunda*; s7, *Carpinus turczaninowii*; s8, *Rhus punjabensis* var. *sinica*; s9, *Rosa brunonii*; s10, *Rubus flosculosus*; s11, *Broussonetia papyrifera*; s12, *Salix sinopurpurea*; s13, *Platycarya strobilacea*; s14, *Vinurnum betulifoljkium*; s15, *Litsea veitchiana*; s16, *Rubus amabilis*; s17, *Callicarpa bodinieri* var. *rosthornii*; s18, *Quercus aliena* var. *acutiserrata*; s19, *Morus alba*; s20, *Salix permollis*; s21, *Cornus officinalis*; s22, *Quercus variabilis*; s23, *Dendrobenthamia japonica* var. *chinensis*; s24, *Schisandra sphenanthera*; s25, *Rhus chinensis*; s26, *Actinidia chinensis* var. *hispida*; h1, *Plantago asiatica*; h2, *Cirsium setosum*; h3, *Sedum lineare*; h4, *Setaria viridis*; h5, *Arundinella hirta*; h6, *Artemisia annua*; h7, *Agrimonia pilosa*; h8, *Equisetum ramosissimum*; h9, *Parathelypteris nipponica*; h10, *Viola grypoceras*; h11, *Arthraxon hispidus*; h12, *Cerastium arvense*; h13, *Pteridium aquilinum* var. *latiusculum*; h14, *Sinosenecio oldhamianus*; h15, *Rubia cordifolia*; h16, *Aster ageratoides*; h17, *Oxalis acetosella*; h18, *Duchesnea indica*; h19, *Dioscorea oppesita*; h20, *Carex lanceolata*; h21, *Thalictrum brevisericeum*; h22, *Mazus gracilis* Hemsl; h23, *Glechoma longituba*; h24, *Polygonum capitatum*; h25, *Kyllinga brevifolia*; h26, *Elsholtzia ciliata*; h27, *Conyza canadensis*; h28, *Artemisia lavandulaefolia*; h29, *Pueraria lobata*; h30, *Anemone tomentosa*; h31, *Lysimachia stenosepala* var. *stenosepala*; t1, *Castanea mollissima*; t2, *Quercus serrata* var. *brevipetiolata*; t3, *Carpinus turczaninowii*; t4, *Rhus punjabensis* var. *sinica*; t5, *Broussonetia papyrifera*; t6, *Quercus aliena*; t7, *Platycarya strobilacea*; t8, *Betula platyphylla*; t9, *Toxicodendron verniciflunm*; t10, *Quercus aliena* var. *acutiserrata*; t11, *Quercus variabilis*; t12, *Vinurnum betulifoljkium*; t13, *Rhus chinensis*; t14, *Pinus tabulaeformis*; t15, *Corylus heterophylla*; t16, *Trachycarpus fortunei*.

This unimodal pattern is consistent with studies in Mediterranean area (Bonet, 2004; Bonet and Pausas, 2004), but inconsistent with the other patterns in some studies which showed an increase, or decrease throughout succession, or fluctuating, or a decline followed by an increase (Houssard et al., 1980; Schoonmaker and McKee, 1988; Huston, 1994; Debussche et al., 1996). The discrepancy may be attributed to the different environmental conditions.

Nutrient availability at small scales has generally shown to be negatively correlated with plant diversity (Rosenzweig, 1995; Brosofske et al., 2001). In our study, herbaceous species richness was significantly negatively correlated with the depth of humus and soil pH, and the heterogeneity (*D*, *H* indices) was significantly negatively correlated with the soil TC and depth of humus, suggesting a similar pattern (Table 4). However, shrub species richness was significantly positively correlated with the depth of litter and soil NH_4-N, suggesting a different pattern (Table 5).

4.3. Natural recovery in abandoned cropland

In the Qinling Mountains, previous studies had demonstrated that the climax or later-successional communities in the region would be deciduous broad-leaves forests dominated by *Q. aliena* var. *acutiserrata*, *Q. serrata* var. *brevipetiolata*, *Q. variabilis*, *C. mollissima*, *B. platyphylla*, *C. turczaninowii*, etc., while the understory was dominated by *B. fargesii*, *C. lanceolata*, *A. hispidus*, etc. (Ren et al., 1998; Yue et al., 1999). The floristic composition and vegetation structure of some secondary forests in our study (Fig. 4, Table 2), especially in the two 'relative old' forests, were similar to the later-successional or climax communities described by other authors (Ren et al., 1998; Yue et al., 1999), indicating the successful recovery. Natural forests with shrub layer dominated by *B. fargesii* bamboo (main food of giant pandas in winter) were favorite habitats for giant panda in the Qinling Mountains (Liu et al., 2005). Our results indicated that the restoration goal could be achieved via spontaneous succession.

In another studies of the region (Ren et al., 1998; Yue et al., 1999), the soil organic matter (SOM) of 22 natural deciduous broad-leaves forests sites dominated by *Q. aliena* var. *acutiserrata*, *Q. variabilis*, *Q. serrata* var. *brevipetiolata*, and *C. mollissima* ranged from 26.4 to 140.8 g/kg, with a mean value of 70.6 g/kg. The tree species richness in these forests ranged from 6.3 to 10.7 per 400 m^2, with a mean value of 8 per 400 m^2. Compared to these forests, the SOM (ranged from 60.07 to 119.58 g/kg with a mean value of 88.6 g/kg, Appendix A) and tree species of the secondary forests in our study were a little higher or similar. In addition, the tree richness of the secondary forests in our study were higher than other two *Q. aliena* var. *acutiserrata* forests in Taibai National Nature Reserve (nearby our study area, Fig. 1), whose tree richness were 6.0 and 2.5 per 400 m^2 (Gao et al., 1997).

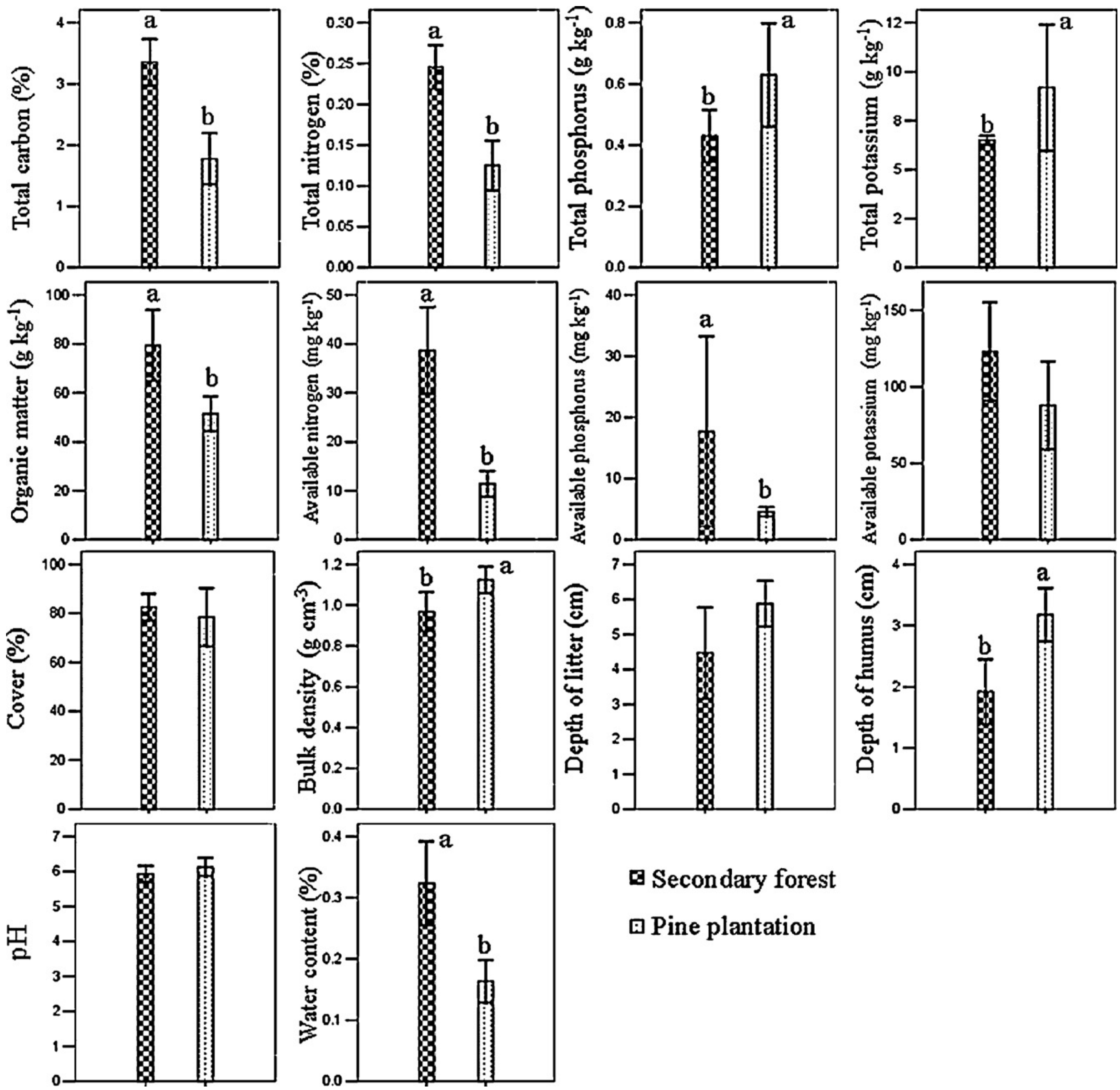

Fig. 5. Mean value and standard error of analyzed habitat variables in secondary forest and pine plantation (two relative old forests were excluded, different letters indicate significant differences by the ANOVA).

The recovery of species-rich vegetation on abandoned cropland is often constrained by depleted seed bank, poor propagule dispersal of the late-succession species and competitive weed species, even when natural abiotic conditions have been restored (Hansson and Hagelfors, 1998; Bakker and Berendse, 1999; Van der Putten et al., 2000). However, the species-rich secondary forests, recovering from abandoned cropland in our study, indicated that all the mentioned constraints might play little role in the study area. The main reasons were likely the remnant vegetation nearby the abandoned cropland in the valley and the diverse animal for seed (or propagule) dispersal in the nature reserve.

4.4. Effect of different restoration approaches on ecosystem recovery

CCA Ordination clearly separated pine plantations from secondary forests and indicated their different species composition and habitat characteristic (Fig. 4). The secondary forests were expected to have higher diversity both in tree layer and herb layer because the pine plantations were monoculture (Lugo, 1997; Ito et al., 2004). Interestingly, in the understory, the shrub (including sapling) diversity in the two forest types was similar (ANOVA, $p > 0.05$), and no significant differences were found in the understory (*i.e.*, herb layer and shrub layer) between the two vegetation types by MRPP either. Abundant late-successional tree species were found in the understory of pine plantations, some of them even became the indicator species (Table 2), *e.g.*, *Q. variabilis* and *C. lanceolata*.

Consequently, our results support the contention that 'using plantation as a nurse for establishing late-successional tree species' (Ashton et al., 1997; Lugo, 1997; Brockerhoff et al., 2003). The success of pine plantations in our study may be attributed to the indigenous *P. tabulaeformis*. Previous investigations have found that the *P. tabulaeformis* could establish natural community in the harsh conditions (*e.g.*, mountain ridge, rock cliff) in our study region (Ren et al., 1998). Similar findings were reported in pine plantations which provided conditions for the development of species-rich indigenous understory in *P. radiata* plantations in New Zealand (Brockerhoff et al., 2003; Langer et al., 2008), *P. kesiya* plantations in Thailand (Oberhauser, 1997), *P. caribaea* plantations in Sri Lankan (Ashton et al., 1997), and *P. sylvestris* plantations in Poland (Czerepko, 2004). They concluded that thinning (or self-thinnig) of the original forests might increase species diversity and relatively small amounts of litter accumulation might not inhibit seed germination (Igarashi and Kiyono, 2008).

The pine plantations had lower OM, AN, AP, TC, TN and water contents in soil and higher BD, indicating the lower soil fertility compared to secondary forests (Fig. 5). The discrepancy may be attributable to the different pattern of biomass accumulation and allocation, uptake and use efficiency of nutrients, litter return and decomposition, resulting from different species composition (Lugo, 1997; Bautista-Cruz and del Castillo, 2005). Substantial acidification, depletion of nutrients, and disruption of biogeochemical cycles are common soil changes associated with pine plantation (Scholes and Nowicki, 1998; Bautista-Cruz and del Castillo, 2005). However, in our study, the soil pH of the pine plantations did not differ from the secondary forests (Fig. 5). The depth of humus was higher in pine plantations, while depth of litter was similar. The results did not seem to suggest the habitat-degradation.

The Chinese government has invested more than 430 billion RMB (US$ 63 billion) for implementing the “Grain-for-Green” project, including compensation to farmers, seedling purchase and stand management. Abandonment and protecting natural regeneration is a low-cost but high natural value restoration practice, especially for the mountainous regions where resources and labour available for restoration are limited (*e.g.*, in the Qinling Mountains, where there is a typical impoverished area with a lack of labor in China). Nevertheless, the potential of natural restoration should be assessed at first, such as climate and soil condition, severity of degradation, availability of seeds or propagule (*e.g.*, seed bank, remnant vegetation, dispersal agents) (Lugo, 1997; Guariguata and Ostertag, 2001; Lamb et al., 2005). When site conditions are severely degraded or seed sources are absent, planting indigenous plant (especially pine with strong adaptive capacity) may be an effective approach for arresting site degradation and facilitating natural regeneration (Lugo, 1997; Ashton et al., 2001).

5. Conclusion

Our results suggested that facilitating natural succession for ecological restoration should be emphasized especially for the regions where there is lack of labor and financial support. Even at biodiversity hotspot, natural vegetation and habitat could be restored via natural succession. These spontaneous restoration forests were characterized by high diversity, high soil fertility and rich unique species. The SOM, TC, TN, AK, community cover, litter depth, humus depth and water content increased significantly with the increase of abandonment time. The AN (NH_4-N + NO_3-N) tended to increase during the first 50 years after abandonment and then decreased. The TK, TP, AP, soil pH did not seem to change significantly with time after abandonment, while the soil bulk density decreased significantly.

P. tabulaeformis plantation plays an important role in restoration and conservation in the region. The pine plantations tend to show a low level of biodiversity in tree and herb layer, but the shrub layer (including sapling) composition and diversity were similar with secondary forests. Although the pine plantation shows lower soil fertility, they do not seem to result in the habitat-degradation (*e.g.*, acidification).

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Appendix A. Summary data of characteristics at each of study sites (mean value of three replicates).

Sites	Soil water (%)	BD ($g\,cm^{-3}$)	pH (H_2O)	DL (cm)	DH (cm)	TC (%)	SOM ($g\,kg^{-1}$)	TN (%)	NO_3 ($mg\,kg^{-1}$)	NH_4 ($mg\,kg^{-1}$)	AN ($mg\,kg^{-1}$)	C:N	TP ($g\,kg^{-1}$)	AP ($mg\,kg^{-1}$)	TK ($g\,kg^{-1}$)	AK ($mg\,kg^{-1}$)
F1	0.17	1.25	6.29	0	0	1.74	45.12	0.16	29.38	11.27	40.65	11.02	1.06	41.15	7.95	106.27
F2	0.11	1.19	5.61	0	0	2.29	36.19	0.15	3.65	4.78	8.43	15.63	0.3	17.46	6.02	72.23
F3	0.15	1.22	6.22	0.01	0.01	1.41	27.33	0.11	7.34	9.98	17.32	12.72	0.85	14.55	7.33	201.49
H1	0.17	0.94	6.04	0.07	0.07	1.18	18.97	0.11	3.4	12.61	16	10.28	1.24	4.44	6.18	104.99
H2	0.18	1.09	5.65	0.3	0.07	2.57	62.11	0.19	11.86	11.86	23.72	13.4	0.33	25.18	6.49	55.75
H3	0.18	1.21	6.14	0.14	0.3	1.06	26.54	0.08	3.82	5.09	8.92	13.42	0.28	4.43	6.91	58.69
S1	0.18	1.42	6.54	0.07	0.07	1.25	34.81	0.1	1.78	15.09	16.88	12.22	1.13	10.55	8.58	111.32
S2	0.22	1.04	6.02	0.3	0.13	3.16	69.35	0.25	20.66	11.49	32.15	12.62	0.77	36.63	5.85	116.11
S3	0.31	1.18	6.25	1.8	2.4	1.59	29.63	0.12	2.42	13.31	15.73	13	0.34	3.52	7.61	193.52
S4	0.11	0.99	6.27	0.2	0.4	0.78	18.81	0.07	3.23	4.12	7.35	10.22	0.97	4.66	7.51	104.51
S5	0.21	1.1	5.86	4.6	1.2	1.67	39.33	0.15	4.61	14.49	19.1	11.48	1.46	4.3	6.85	164.09
P1	0.16	1.27	5.8	4.5	2.5	1.69	57.5	0.13	3.99	5.09	9.08	12.52	0.36	3.55	5.99	80.79
P2	0.12	1.01	6.48	5.2	2.8	2.17	51.47	0.13	4.86	6.68	11.55	16.51	0.51	3.98	13.9	56.19
P3	0.16	1.15	6.47	7.2	4.4	1.84	43.52	0.13	0.27	16.55	16.82	14.57	1.07	4.33	8.94	162.11
P4	0.21	1.07	5.76	6.6	3	1.41	53.47	0.11	1.13	7.02	8.15	12.84	0.57	6.34	7.94	52.33
N1	0.25	1.03	5.73	0.98	0.48	3.16	77.49	0.24	14.3	8.16	22.46	13.52	0.44	6.82	6.1	79.51
N2	0.35	0.97	6.02	5	2.4	3.34	60.07	0.24	7.26	24.9	32.16	14.2	0.34	4.62	6.76	109.99
N3	0.38	1.08	5.73	5	2	3.26	73	0.25	18.62	27.15	45.77	13.2	0.63	53.01	6.53	140.19
N4	0.32	0.79	6.24	6.9	2.8	3.64	106.92	0.26	25.14	29.08	54.22	13.82	0.3	6.35	6.63	162.03
N5	0.34	0.8	6.36	5.4	3.4	4.82	119.58	0.33	17.74	20.16	37.9	14.78	0.36	11.51	6.24	196.53
N6	0.41	0.68	6.55	8.6	4.3	4.23	94.64	0.3	6.42	22.07	28.5	14.13	0.95	4.44	6.83	166.16

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