Catena 120 (2014) 64–72

Contents lists available at ScienceDirect

Catena

journal homepage: www.elsevier.com/locate/catena

Land use and topographic position control soil organic C and N accumulation in eroded hilly watershed of the Loess Plateau

Hanhua Zhu [a,b,c], Jinshui Wu [c], Shengli Guo [a,b,*], Daoyou Huang [c], Qihong Zhu [c], Tida Ge [c], Tingwu Lei [a,b]

[a] State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau, Institute of Soil and Water Conservation, Chinese Academy of Sciences, Yangling 712100, Shaanxi, PR China
[b] State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau, Institute of Soil and Water Conservation, Northwest University of Agriculture and Forestry, Yangling 712100, Shaanxi, PR China
[c] Key Laboratory of Agro-ecological Processes in Subtropical Region, Institute of Subtropical Agriculture, Chinese Academy of Sciences, Changsha 410125, Hunan, PR China

ARTICLE INFO

Article history:
Received 29 August 2013
Received in revised form 14 March 2014
Accepted 3 April 2014
Available online 4 May 2014

Keywords:
Soil organic carbon
Total nitrogen
Land use
Topographic position
Eroded hilly watershed
Loess Plateau

ABSTRACT

Land use and topography strongly influence soil organic C (SOC) and N accumulation in eroded hilly regions. However, their combined effects and the underlying mechanism remain unclear. In this study, five land uses and three topographic positions across an eroded hilly watershed of the Loess Plateau were selected to investigate their effects on SOC and N accumulation. The restored grassland, shrubland and woodland (25 to 30 years) increased SOC and total N by 32% to 119% in the slope and 17% to 81% in the gully, respectively compared with the cropland. These restored vegetation increased soil dissolved organic C (DOC) and microbial biomass C and N (MBC and MBN) by 1.1- to 3.0-fold in the slope and 30% to 108% in the gully, respectively. Similar increases were observed in soil aggregates, MBN/total N, soil C:N and microbial C:N ratios. These improvements were higher in the shrubland than in the other land uses. The SOC significantly decreased from the gully to the slope, and the magnitudes decreased in a sequence of cropland, grassland, shrubland and woodland. Soil DOC, total N, and MBN also decreased from the gully to the slope in the cropland, but remained unchanged in the grassland, shrubland, and woodland. Land use ($P < 0.05$), topographic position ($P < 0.05$), and their interaction ($P < 0.1$) influenced SOC, total N, DOC, MBC, MBN, soil C:N and microbial C:N ratios. There were close relationships among soil aggregates, SOC and total N, and DOC, MBC and MBN. Therefore, land use, topographic position (erosion and deposition), and their interactions regulate SOC and N accumulation and their labile fractions in the eroded hilly region at a watershed scale. Our results suggested that converting cropland to shrubland is an initial strategy to restore degraded ecosystems and increase soil C sequestration in eroded hilly region of the Loess Plateau.

1. Introduction

Soil organic carbon (SOC) plays roles in improving soil quality and productivity in natural and agricultural ecosystems and in mitigating global warming through soil C sequestration (Arnalds, 2004; Lal, 2004, 2010). Land use and topography are two of the primary factors for SOC accumulation in eroded hilly regions (Eckert and Engesser, 2013; Schwanghart and Jarmer, 2011). Vegetation restoration, i.e. converting cropland to grassland, shrubland, or woodland, has been widely conducted to improve fragile ecosystems and increase SOC (Arnalds, 2004; Zhu et al., 2012). However, the efficacy of this process differs among land uses and topographic positions (Funakawa et al., 2009; Wang et al., 2001). Therefore, the combined effects of land use and topographic position on SOC accumulation should be understood to accelerate ecosystem restoration and maximize soil C sequestration in hilly regions.

The net primary productivity of plants and the associated input of litter to soils increase through vegetation restoration, i.e. converting cropland to grassland, shrubland, or woodland, thereby increasing SOC (Eckert and Engesser, 2013). However, the efficiency of SOC accumulation depends on the quantity and the quality of organic litter input in soils (An et al., 2009; Solomon et al., 2000). The gross biomass and roots of plants and the associated organic litters vary with land uses, such as cropland, grassland, shrubland, and woodland (Rasse et al., 2006; Wang et al., 2011). Organic residues from plants in shrubland and woodland contain more lignin and tannins, which have longer residence time and greater humification coefficients in the soil than those in the grassland and cropland (Solomon et al., 2000). For instance, higher stable organic C fractions and SOC accumulation are observed in shrubland soils than in grassland and cropland soils (Solomon et al.,

* Corresponding author at: Institute of Soil and Water Conservation, Chinese Academy of Sciences, Yangling, Shaanxi 712100, PR China. Tel.: +86 29 87012411; fax: +86 29 87012210.
E-mail address: slguo@ms.iswc.ac.cn (S. Guo).

http://dx.doi.org/10.1016/j.catena.2014.04.007
0341-8162/

2000). In addition, restoration with various vegetation types results in significantly different improvements in soil nutrients and soil physical conditions (Zhu et al., 2012). The improved local soil conditions may subsequently alter plant growth and organic residue input to the soil. Furthermore, such altered conditions could regulate the cycling of SOC and nutrients, through the interactions among organic residues, soil conditions, and microorganisms (Nilsson et al., 2012; Zhu et al., 2012).

Topography strongly influences the spatial distribution of surface SOC by altering hydrologic processes as well as plant litter production and decomposition (Wang et al., 2011). Topographic position, an important local parameter, influences the processes and intensity of erosion and deposition of soil, organic fractions and nutrients on slopes (Ritchie et al., 2007; Schwanghart and Jarmer, 2011). For example, surface SOC and nutrients migrate from the upper slopes and are deposited and buried in the depressional areas, particularly in severely eroded areas (Scowcroft et al., 2008; Seibert et al., 2007). The processes of soil erosion and deposition on slopes are mediated by restored vegetation (Ritchie et al., 2007). For instance, reforestation in a severely eroded clayey Plinthudult in southeastern China has significantly enhanced soil aggregation, and mitigated soil erosion and the losses of SOC and N (Zhang et al., 2004). These altered local soil conditions may regulate the gross biomass of plants, litter input in soils, and the decomposition of litter and native SOC in different topographic positions (Scowcroft et al., 2008; Wang et al., 2011). Thus, complex interactions of land use and topographic position are observed in SOC at a slope scale (Tang et al., 2010). However, the interactive effects of land use and topographic position on SOC and the underlying mechanisms remain poorly understood, particularly in eroded hilly regions at watershed scales.

The Loess Plateau, which covers an area of 640,000 km^2, is a semi-arid region characterized by a long agricultural history, ragged topography, severe soil erosion, and a fragile environmental status (Wang et al., 2011). Vegetation restoration, i.e. converting cropland to grassland, shrubland, or woodland, has been implemented by the Chinese government in the Loess Plateau since the 1980s to restore the eroded and degraded ecosystems and increase soil C sequestration (Chen et al., 2007a). Previous studies indicated that vegetation restoration significantly reduces soil erosion and increases SOC and N accumulation (Fu et al., 2011; Zhu et al., 2010). However, the effects of land use and topographic position (erosion and deposition), particularly their interactions, on SOC and N, and their labile fractions, i.e. soil dissolved organic C (DOC), microbial biomass C (MBC) and N (MBN), were rarely studied at watershed scales. Such issues deserve more investigation because they may provide relevant information to optimize ecosystem restoration and soil C sequestration on the Loess Plateau.

Small watersheds are the elementary units of natural landscapes and artificial vegetation restoration. Therefore, surface soils from five dominant land uses and three typical topographic positions were investigated across an eroded hilly watershed of the Loess Plateau. This study aimed to 1) assess differences in soil aggregates, total N and MBN, SOC, DOC and MBC among different land uses and topographic positions; and 2) study the interactive effects of land use and topographic position on these soil parameters in an eroded hilly region of the Loess Plateau at a watershed scale.

2. Material and methods

2.1. Field surveys and sampling

Five dominant land uses (cropland, orchard, grassland, shrubland, and woodland) and three topographic positions (summit, slope, and gully) were identified and selected in June (summer) 2009 across the Yangou watershed (36° 28′ N to 36° 32′ N, 109° 20′ E to 109° 35′ E), located in the hilly region of the Loess Plateau, China (Fig. 1). The watershed, covering a total area of 48.0 km^2, exhibits complex topographic variations, with a gully density of 2.74 km km^{-2} and an elevation ranging from 990 m to 1410 m. The watershed has a temperate semi-arid climate, with an annual mean temperature of 9.8 °C and precipitation of 558.4 mm. The soil is mainly Loess-Orthic Primosols (USDA Soil Taxonomy) with a loam texture, which is erodible because of weak cohesion, high infiltrability, and low water retention (Fu et al., 2010). The soil erosion modulus of the watershed is 2860 t km^{-2} a^{-1} (Xu et al., 2009).

In this watershed, the summit area covers 2.3% of the total area, whereas the slope (gradient > 5°) and the gully span 82.8% and 13.9% of the total area, respectively. Since the 1980s, vegetation restoration, that is, converting cropland to grassland, shrubland, or woodland, has been implemented at the watershed through the 'Grain for Green' Project (Chen et al., 2007a). The cropland with slopes <25° was converted to grassland, shrubland, or woodland, whereas the remaining cropland was terraced. A few terraced croplands were also cultivated into an orchard to increase the income of farmers.

The cropland in the watershed has been cultivated for mono-winter wheat or corn production for centuries, and currently acquires a maximum of 180 kg N ha^{-1} and 40 kg P ha^{-1} from chemical fertilizers to produce an average of 2500 kg grain ha^{-1} per year. Crop residues are removed for cooking or feeding cattle, and tillage is conducted twice a year to increase rainfall infiltration after the harvest (in July) and to prepare seed beds before the next sowing (in September). For the orchard, apple trees (*Malus pumila* Mill) have been planted (2 m × 3 m) for 20 years to 30 years. The orchard is tilled to control weeds and similarly fertilized as the cropland. The grassland, shrubland, and woodland have been redeveloped from cropland or sparse grassland for 25 years to 30 years, and received no fertilizer. Basic information and the dominant plant species in this watershed are described in Table 1.

A total of 314 typical sampling quadrats (approximately 10 m × 10 m) were established for the five land uses and the three topographic positions across the whole watershed according to their distributions. For each quadrat, five to eight soil cores were bulked from surface (0 cm to 20 cm) soil, and mixed as single soil sample. Each of the soil samples was sieved through a 2 mm mesh and divided into two parts. One part was air-dried for analyzing soil physico-chemical properties, and the other one was immediately stored a 4 °C for measuring soil microbial properties. The numbers of soil samples for each land use and topographic position are shown in Table 1.

2.2. Soil analysis

Soil water-stable aggregates were determined using a laser granulometer (Malvern Masterizer 2000, UK) as described by Westerhof et al. (1999). Mean weight diameter (MWD, mm) of soil aggregates were calculated as follows:

$$MWD = \sum_{i=1}^{n} (X_i \cdot W_i)$$

where X_i is the mean diameter (mm) of soil aggregate fraction *i*, and W_i is the mass proportion (%, w/w) of soil aggregate fraction *i* (Conaway and Strickling, 1962).

Soil bulk density was measured by the clod method (Culley, 1993). SOC and total N contents were determined using the K_2CrO_7–H_2SO_4 oxidation method and the Kjeldahl method, respectively (Sparks et al., 1996). The SOC and total N on mass basis (g kg^{-1}) were converted to the area basis (t ha^{-1}) by multiplying soil bulk density and a depth of 20 cm, respectively. Soil microbial biomass C (MBC) and N (MBN) were determined using the fumigation–extraction method (Brookes et al., 1985; Wu et al., 1990). Dissolved organic C (DOC) content was derived from values determined in non-fumigated soil (Brookes et al., 1985).

2.3. Statistical analysis

The means and standard errors of the data were calculated using Excel 2010 (Microsoft Corporation, USA). Multiple comparisons were

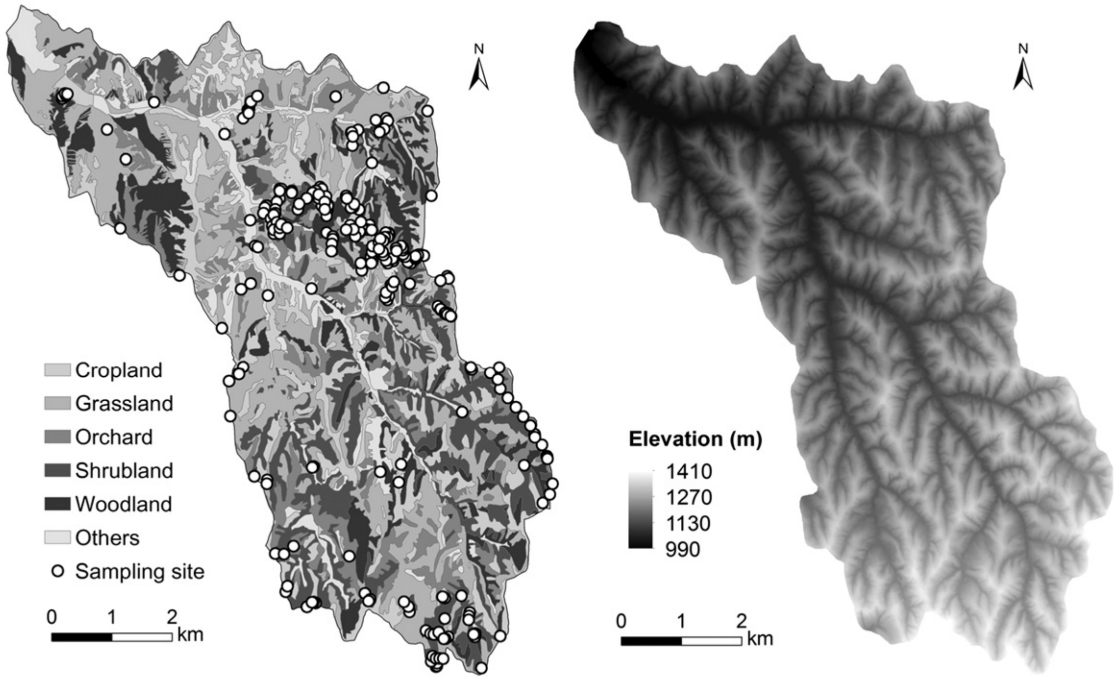

Fig. 1. Soil sampling sites, land use and topographic map of the study watershed.

conducted using a least significant difference (LSD) test at 95% confidence interval. Two-way ANOVA was also conducted, in which land use and topographic position were considered as the main factors to determine the significance of mean differences and their interaction at 95% confidence interval. Pearson correlation method was used to evaluate all the correlations among soil parameters presented in this study. Statistical analyses were performed using SPSS 11.5 (SPSS Inc. Chicago, USA).

3. Results

3.1. Soil aggregates

Soil aggregates of <0.05 mm accounted for 62% to 82% (w/w) of bulk soil, whereas aggregates of 0.05–0.25 mm and 0.25–2 mm accounted for 18% to 35%, and 1% to 3%, respectively (Fig. 2a, b and c). In the summit, soil aggregates of 0.05–0.25 mm and 0.25–2 mm were significantly higher in grassland than in cropland (Fig. 2b and c). Thus, the MWD was 13% higher in the grassland than that in cropland (Fig. 2d). However, the distributions and the MWD of soil aggregates were similar between orchard and cropland.

In the slope and gully, soil aggregates of 0.05–0.25 mm and 0.25–2 mm in the grassland, shrubland, and woodland were significantly higher than those in the cropland (Fig. 2b and c). By contrast, soil aggregates of <0.05 mm in the three land uses were significantly lower than that in the cropland (Fig. 2a). As a consequence, the MWD of soil aggregates in the three land uses were remarkably larger than that in the cropland (Fig. 2d). The magnitudes of these changes in the shrubland and woodland were notably greater than those in the grassland.

Soil aggregates of <0.05 mm in the shrubland and woodland increased by 8% and 15% from the slope to the gully, whereas soil aggregates of 0.05–0.25 mm and 0.25–2 mm decreased by 17% to 30%, respectively ($P < 0.05$; Fig. 2a).

Overall, land use significantly influenced the distributions and MWD of soil aggregates ($P < 0.01$), whereas topographic position had a significant effect on aggregates of <0.05 mm ($P < 0.05$), rather than soil aggregates of 0.05–0.25 mm and 0.25–2 mm (Table 2). In addition, interaction of land use and topographic position had no significant effect on soil aggregates and their MWD (Table 2).

Table 1
Description of the land uses, topographic positions, and sampling number in the study watershed.

Vegetation	Position	Area (ha)	Area (%)	Number of soil sample	Dominant plant species
Cropland	Summit	51.6	1.08	20	*Triticum aestivum*, *Zea mays*, *Solanum tuberosum*, and *Setaria italica* Beauv.
	Slope	517.4	10.78	38	
	Gully	83.5	1.74	10	
Orchard	Summit	39.6	0.83	15	*Malus pumila* Mill.
	Slope	512.6	10.68	41	
Grassland	Summit	61.6	1.28	5	*Artemisia gmelinii*, *Stipa bungeana*, *Bothriochloa ischaemum*, and *Setaria viridis*
	Slope	1199.9	25.00	56	
	Gully	221.0	4.60	10	
Shrubland	Slope	548.0	11.42	60	*Rosa xanthina*, *Ostryopsis davidiana* Decne., *Cotoneaster acutifolius* Turcz., and *Syringa pekinensis* Rupr.
	Gully	126.4	2.63	8	
Woodland	Slope	501.6	10.45	46	*Quercus liaotungensis*, *Betula platyphylla*, *Rhamnus davurica*, and *Robinia pseudoacacia*
	Gully	122.3	2.55	5	

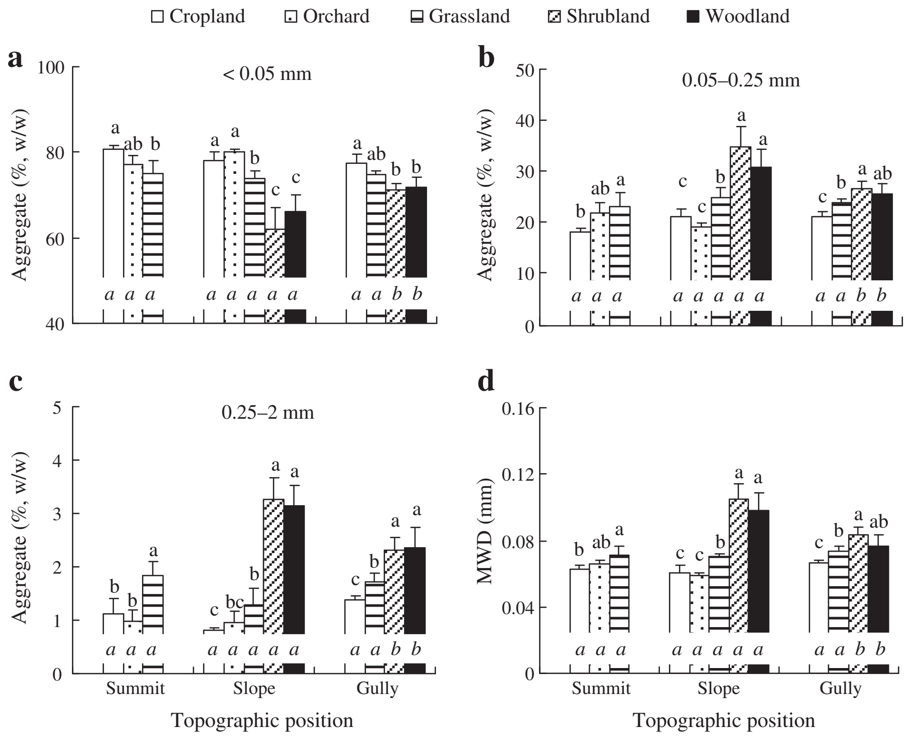

Fig. 2. Soil aggregates and their mean weight diameter (MWD) under different land uses and topographic positions. Different letters above the column indicate significant differences among land uses in the same topographic position, while different italic letters within the column indicate significant differences among topographic positions with the same land use ($P < 0.05$).

3.2. Soil total N and microbial biomass N

In the summit, soil total N was 19% higher in the orchard and 27% higher in the grassland ($P < 0.05$) compared with the cropland (1.09 t ha^{-1}; Fig. 3a). Soil MBN was 72% higher in the orchard and 75% higher in the grassland than in the cropland ($P < 0.01$; Fig. 3b). As a consequence, the MBN/total N was 53% and 46% higher in the orchard and grassland, respectively than in the cropland ($P < 0.01$; Fig. 3c).

In the slope, the total N contents in soils with grassland, shrubland, and woodland were respectively 32%, 90%, and 55% higher than those in the cropland (1.18 t ha^{-1}), ($P < 0.01$; Fig. 3a). Similar increases ranging from 1.3-fold to 3.0-fold ($P < 0.01$) were obtained for MBN in the slope (Fig. 3b). Both soil total N and MBN were similar between the cropland and the orchard. The MBN/total N in grassland, shrubland, and woodland, ranging from 4.36% to 4.56%, was 35% to 41% larger than that (3.23%) in the cropland (Fig. 3c).

In the gully, soil total N contents were 17%, 39%, and 23% higher in the grassland, shrubland, and woodland, respectively than in the cropland (1.80 t ha^{-1}), ($P < 0.05$; Fig. 3a). Similar increases, ranging from 53% to 108% ($P < 0.01$), were obtained for MBN (Fig. 3b). The MBN/total N ratios in the grassland, shrubland, and woodland with a narrow range of 4.06% to 4.43%, were 16% to 26% larger than that (3.51%) in cropland (Fig. 3c).

The soil total N of cropland increased from 1.09 t ha^{-1} in the summit to 1.80 t ha^{-1} in the gully. For grassland, soil total N increased from 1.38 t ha^{-1} in the slope to 2.10 t ha^{-1} in the gully (Fig. 3a). Similar increases were found for soil MBN (Fig. 3b). However, both soil total N and MBN of the orchard in the summit were similar to those in the slope. Soil total N and MBN of the shrubland and the woodland were similar in the slope and in the gully. In addition, the MBN/total N ratios for the five land uses were also similar among the summit, slope and gully (Fig. 3c).

Generally, soil total N and MBN in the watershed were influenced strongly by land use ($P < 0.05$), topographic position ($P < 0.05$), and proportionally by their interaction ($P < 0.1$; Table 3). The MBN/total N, however, was only affected by land use ($P < 0.05$; Table 3).

Table 2
Two-way ANOVA for soil aggregates and their mean weight diameter (MWD).

Source	df	P			
		<0.05 mm	0.05–0.25 mm	0.25–2 mm	MWD
Land use	4	0.001	0.001	0.004	0.000
Position	2	0.045	0.159	0.248	0.124
Land use × Position	7	0.125	0.253	0.298	0.209

3.3. Soil organic carbon and its labile fractions

In the summit, SOC was 18% higher in the orchard and 37% higher in the grassland than in the cropland (9.78 t ha^{-1}; $P < 0.05$; Fig. 4a). By contrast, soil DOC contents, ranging from 29.59 mg kg^{-1} to 32.79 mg kg^{-1}, were similar among the three land uses (Fig. 4b). Soil MBC contents were 63.0% and 82.5% higher in the orchard and grassland, respectively than in the cropland ($P < 0.01$; Fig. 4c). The MBC/SOC was 35% to 44% larger in the orchard and grassland than in the cropland ($P < 0.05$; Fig. 4d). Soil C:N had a narrow range (8.92 to 9.12) and was similar among the three land uses (Fig. 4e). However, microbial C:N in the grassland (5.33) was 17% larger than that in the cropland ($P < 0.05$; Fig. 4f).

In the slope, SOC contents were 41%, 119%, and 60% higher in the grassland, shrubland, and woodland respectively, than in the cropland (11.68 t ha^{-1}), ($P < 0.01$; Fig. 4a). Soil DOC contents were respectively 1.1-, 1.8-, and 1.2-fold higher in grassland, shrubland, and woodland than in the cropland, ($P < 0.01$; Fig. 4b). Soil MBC contents were 1.5-, 2.8-, and 1.9-fold higher in grassland, shrubland, and woodland, respectively than in cropland ($P < 0.01$; Fig. 4c). Soil MBC was 58% higher in the orchard than in the cropland (69.39 mg kg^{-1}), although SOC and DOC of the two land uses were similar in the slope. The MBC/SOC ratios in the grassland, shrubland, and woodland were respectively 21%, 24%

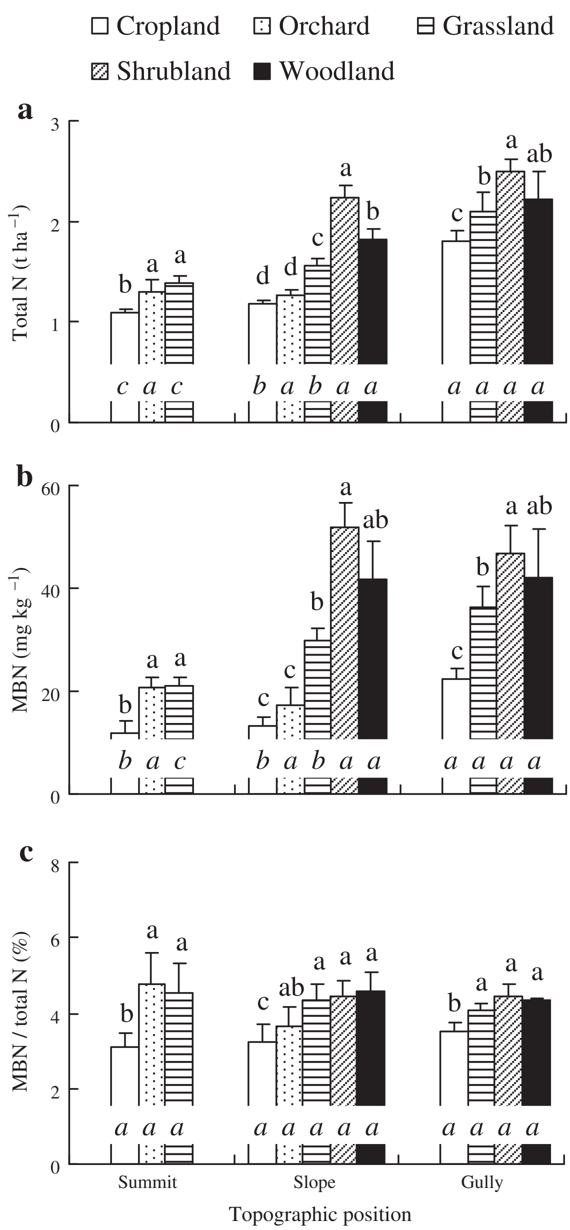

Fig. 3. Soil total N, microbial biomass N (MBN) and MBN/total N under different land uses and topographic positions. Different letters above the column indicate significant differences among land uses in the same topographic position, while different italic letters within the column indicate significant differences among topographic positions with the same land use ($P < 0.05$).

and 25% larger than that in cropland ($P < 0.05$; Fig. 4d). Both soil C:N and microbial C:N were significantly larger in the shrubland and woodland than in the cropland (Fig. 4e and f).

In the gully, SOC, DOC, and MBC in the grassland, shrubland, and woodland were significantly higher than those in the cropland (Fig. 4a, b, and c). Soil MBC contents were 38%, 100%, and 90% higher in the grassland, shrubland, and woodland, respectively compared with the cropland (Fig. 4c). Magnitudes of these increases were significantly higher ($P < 0.05$) than those of SOC (31% to 81%; Fig. 4a) and soil DOC (30% to 73%; Fig. 4b). By contrast, the MBC/SOC, with a narrow range of 1.47% to 1.57%, was similar among cropland, grassland, shrubland, and woodland (Fig. 4d). Both soil C:N and microbial C:N in the shrubland and woodland were significantly higher than those in the cropland (Fig. 4e and f).

For cropland and grassland, SOC significantly increased ($P < 0.05$) from 9.78−13.43 t ha^{-1} in the summit to 11.68–16.43 t ha^{-1} in the slope, and then to 17.76–23.34 t ha^{-1} in the gully (Fig. 4a). Soil DOC and MBC exhibited similar increases in the cropland (Fig. 4b and c). However, the MBC/SOC in the cropland and grassland decreased from 1.69%–2.47% in the summit and slope to 1.47%–1.55% in the gully (Fig. 4d). Soil C:N ratios for cropland and orchard remained within the narrow range of 8.44 to 9.34 in the watershed (Fig. 4e). But, microbial C:N in the cropland and grassland decreased from 1.69–2.28 in the summit and slope to 1.47–1.55 in the gully (Fig. 4f).

For shrubland and woodland, SOC increased by 25% and 46% from the slope to the gully, respectively (Fig. 4a). However, soil DOC and MBC for shrubland and woodland in the slope were similar to those in the gully (Fig. 4b and c). Thus, the MBC/SOC in the shrubland and woodland decreased from 2.51%–2.53% in the slope to 1.56%–1.57% in the gully (Fig. 4d). Soil C:N ratios for shrubland and woodland increased from 10.31–10.48 in the slope to 12.31–12.52 in the gully (Fig. 4e), whereas microbial C:N decreased from 6.21–6.56 in the slope to 4.69–4.83 in the gully (Fig. 4f).

In general, SOC, DOC, MBC, soil C:N and microbial C:N ratios in the watershed were influenced strongly by land use ($P < 0.05$), topographic position ($P < 0.05$), and fairly by their interactions ($P < 0.1$; Table 4). Variation in the MBC/SOC mainly depended upon topographic position and land use (Table 4).

4. Discussion

4.1. Effects of land use and topographic position on soil aggregates

Soil aggregate is an important characteristic of soil structure and indicator of soil anti-erodibility, which is closely linked to soil nutrient erosion, and SOC and N accumulation in hilly areas (Mikha and Rice, 2004; Six et al., 2004). In this study, a high proportion (62% to 82%) of <0.05 mm aggregates in bulk soil (Fig. 2a) corresponds to the weak cohesion and aggregation of the soils (Fu et al., 2010). This indicates high potential risks in soil erosion, as well as SOC and nutrient losses in the hilly watershed of the Loess Plateau. However, converting cropland to grassland, shrubland, or woodland significantly increased portions of 0.05–0.25 mm and 0.25–2 mm aggregates and MWD of soil aggregate compared with the croplands (Fig. 2b, c and d). This indicates that the restored vegetation enhances soil aggregation, which is consistent with the results in previous studies (Zhu et al., 2010). Soil aggregation is susceptible to the disturbance from erosion and tillage, and the binding agent, i.e. organic fractions (Six et al., 2004). The mitigated disturbance and the increased organic matter of soils under the restored vegetation consequently resulted in the significant improvement in soil aggregation.

Table 3
Two-way ANOVA for soil total N, microbial biomass N (MBN), MBN/total N, organic C (SOC), dissolved organic C (DOC), microbial biomass C (MBC), MBC/SOC, soil C:N, and microbial C:N.

Source	df	*P*								
		Total N	MBN	MBN/total N	SOC	DOC	MBC	MBC/SOC	Soil C:N	Microbial C:N
Land use	4	0.040	0.001	0.029	0.005	0.039	0.001	0.050	0.009	0.029
Position	2	0.050	0.047	0.182	0.026	0.050	0.047	0.000	0.031	0.000
Land use × Position	7	0.068	0.056	0.187	0.076	0.082	0.063	0.155	0.058	0.077

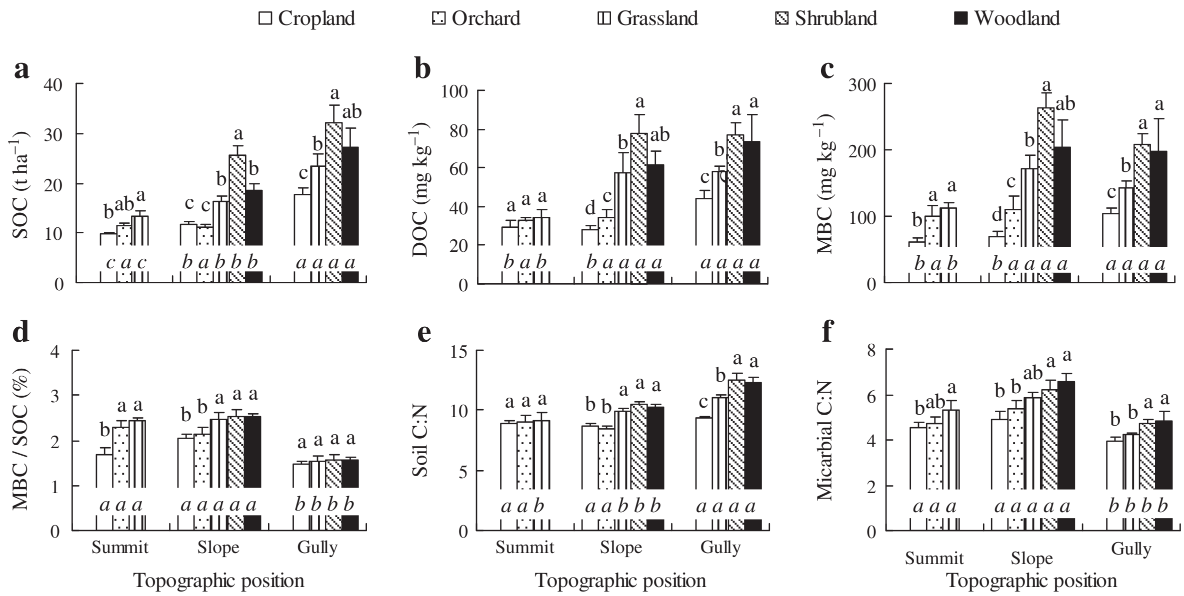

Fig. 4. Soil organic C (SOC), dissolved organic C (DOC), microbial biomass C (MBC), MBC/SOC, soil C:N and microbial C:N under different land uses and topographic positions. Different letters above the column indicate significant differences among land uses in the same topographic position, while different italic letters within the column indicate significant differences among topographic positions with the same land use ($P < 0.05$).

Usually, soil fractions in cropland are easily eroded from the slope and deposited to the gully, because of the intensive disturbance from tillage (Seibert et al., 2007; Zhu et al., 2010). In this study, the distributions of soil aggregates and the MWD for cropland remained similar in the slope and the gully. This probably attributes to the relative homogeneous and small size of soil aggregates in the cropland. However, the <0.05 mm aggregates in the shrubland and woodland increased from the slope to the gully (Fig. 2a). This reflects the erosion and deposition of large amounts of smaller soil aggregates because they are more easily eroded than the larger aggregates (Tang et al., 2010; Urich, 2002). To a certain extent, the larger amount of the relocated <0.05 mm aggregates from the summit and slope also lowered the proportion of 0.05–0.25 mm and 0.25–2 mm aggregates in the gully. Thus, enhancing aggregation of small soil particles is important to mitigate soil erosion in hilly areas.

4.2. Effects of land use and topographic position on soil total N

Nitrogen is one of the most important limiting factors of gross biomass production in natural and artificial ecosystems (Guo et al., 2012). Total N of surface soil in the grassland, shrubland, and woodland increased by 32% to 90% in the slope, and by 17% to 39% in the gully, respectively ($P < 0.05$) compared with the cropland (Fig. 3a). This clearly indicates that the N stock of surface soil was increased drastically by vegetation restoration in the past 25 to 30 years, particularly the shrubland. Similar results were found in the succession of native vegetation recovery (Zhu et al., 2010). For the cropland, the prolonged soil erosion and the intensive soil disturbance from conventional tillage lowered soil N-holding capacity, accelerated soil N loss (Chen et al., 2012; Xu et al., 2009), and thus led to the low N accumulation in the surface soil (1.09–1.80 t ha^{-1}). These processes may be further accelerated in the summit and slope (the erosional areas) because of the fair relocation of soil N to the depositional areas (Guo et al., 2010), as shown by the significantly lower soil total N of cropland in the summit and slope than in the gully (Fig. 3a). For the restored vegetation, however, soil disturbance, i.e. soil erosion and tillage, was mitigated (Carrick and Krüger, 2007; Dolan et al., 2006; Ritchie et al., 2007), and thus resulted in the low loss of soil N. The improved soil aggregation under the restored vegetation (Fig. 2) further enhanced soil N-holding capacity and reduced the loss and relocation of soil N. This is supported by the close relationships of soil total N with >0.05 mm aggregates and MWD of aggregates ($r = 0.74$–0.82, $P < 0.01$, Table 4). Similar results were obtained in previous studies (Mikha and Rice, 2004). These effects of the restored vegetation were relatively prominent in the summit and slope, as shown by the relatively greater increase of soil total N with the restored vegetation in the slope (32% to 90%) than in the gully (17% to 39%; Fig. 3a). Therefore, converting cropland to shrubland or woodland has a high potential of N accumulation in the surface soils at the watershed scale, particularly in the erosional sites.

Total N of surface soil was fairly influenced by the interaction of land use and topographic position (Table 3). This is evident from distinct

Table 4
Correlations between soil physical, chemical, and microbial parameters.[a]

	SOC	Total N	Soil C:N	DOC	MBC	MBN	MBC/SOC	MBN/total N	Microbial C:N	>0.05 mm aggregates
Total N	0.98**									
Soil C:N	0.66**	0.49*								
DOC	0.91**	0.90**	0.64*							
MBC	0.91**	0.90**	0.60*	0.84**						
MBN	0.87**	0.87**	0.55*	0.82**	0.94**					
MBC/SOC	−0.01	0.02	0.00	0.03	0.37	0.28				
MBN/total N	0.15	0.14	0.20	0.19	0.39	0.55	0.63			
Microbial C:N	0.11	0.08	0.17	0.07	0.21	−0.07	0.35	−0.37		
>0.05 mm aggregates	0.77**	0.74**	0.45*	0.84**	0.79**	0.79**	0.11	0.29	0.06	
MWD	0.85**	0.82**	0.47*	0.92**	0.83**	0.82**	0.00	0.18	0.06	0.92**

*$P < 0.05$, **$P < 0.01$.
[a] SOC, soil organic carbon; DOC, dissolved organic carbon; MBC, microbial biomass carbon; MBN, microbial biomass nitrogen; MWD, mean weight diameter of soil aggregates.

changes in total N between different land uses within the same topographic position, as well as different topographic positions for the same land use (Fig. 3a). On one hand, distinct efficacy of various land uses on soil erosion and aggregation strongly influenced surface soil N stock by altering soil N-holding capacity and the loss of soil N (as discussed above). On the other hand, topographic position also relocated soil water and aggregates, and the associated soil N, which in turn accelerated or retarded vegetation restoration and thus soil N accumulation (Fu et al., 2004; Luizão et al., 2004). These interlaced processes resulted in the interaction of land use and topographic position on the N stock of surface soil in the eroded hilly watershed. However, this interaction was only statistically significant at $P < 0.1$ in this watershed (Table 3). This is probably caused by the complex local topography and the spatial variation in vegetation restoration (Wang et al., 2001) across the eroded hilly watershed covering 48.0 km^2.

4.3. Effects of land use and topographic position on soil organic C

The SOC of grassland, shrubland, and woodland increased by 41% to 119% in the slope and by 31% to 81% in the gully compared with the croplands, respectively (Fig. 4a). This clearly reflects the restored vegetation, particularly the shrubland, remarkably increased SOC accumulation of the eroded hilly watershed in the past 25 to 30 years. For the cropland and orchard, the soils received low organic input because of conventional residue removal from the fields (Guo et al., 2012). These soils also had high loss of SOC because of the prolonged soil erosion (Xu et al., 2009) and the enhanced decomposition of SOC with intensive soil disturbance (Dolan et al., 2006). These facts result in the low SOC of the cropland and orchard, particularly in the summit and slope (9.78 t ha^{-1} to 11.68 t ha^{-1}; Fig. 4a). By contrast, the restored vegetation has relatively higher amounts of aboveground biomass and roots, resulting in relatively greater organic litter input and less soil erosion than in the cropland (Wang et al., 2001). Such effects are also enhanced by the improved soil aggregation and the increased soil total N and MBN, as shown by the close relationships of SOC with >0.05 mm aggregates, MWD, total N, and MBN ($r = 0.77$ to 0.98, $P < 0.01$, Table 4). This view is supported by previous results, in which it was reported that SOC accumulation may be accelerated by N fertilization (Gamboa et al., 2010) and the enhanced soil aggregation (Barreto et al., 2009) in secondary and native forest ecosystems. In addition, more complex organic substances from the restored vegetation, indicated by the higher soil C:N ratios (Fig. 3e), have longer residence times and greater humification coefficients in soil (Solomon et al., 2000). All of these advantageous changes may increase organic litter input and decrease SOC loss, and hence accelerate SOC accumulation. Therefore, the restored vegetation, particularly the shrubland, has a high potential of C sequestration in the surface soils in the eroded hilly watershed.

For all land uses, SOC significantly increased from the summit and slope to the gully, indicating the prolonged soil erosion and partial deposition toward the gully in the eroded hilly watershed (Fig. 4a). Similar results have been reported by Lal (2005) and Fu et al. (2011). The magnitude of these increases in the SOC followed the decreasing sequence of cropland (56%), grassland (40%) and woodland (46%), and shrubland (25%). This indicates that restored vegetation, particularly the shrubland, mitigates the loss of SOC in the slope. For cropland, intensive soil disturbance from tillage and weeding usually results in heavy soil erosion, high decomposition of SOC, and the relocation of SOC and N from the summit and slope to the gully (Lal, 2005). Meanwhile, the deposition of soil total N (Fig. 3a) and soil water (Chen et al., 2007b) may promote the growth of crops and increase their gross biomass, roots and associated organic litter input into the soils, and thus enhance SOC accumulation in the gully. For the lands with vegetation restoration, however, soil erosion and the relocation of SOC and N were mitigated by the reduced soil disturbance and the enhanced anti-erodibility from the improved surface coverage of plants and net-work of roots (Carrick and Krüger, 2007). Meanwhile, the improved soil aggregation by the restored vegetation provides microenvironments for absorbing labile organic matter and physical protection of SOC from erosion and decomposition. This is evident from the close relationship of SOC and DOC with >0.05 mm aggregates and MWD of soil aggregates (Table 4), which is in agreement with results from Six et al. (2004) and Lal (2005). Therefore, topographic position strongly influenced the efficacy of land use on soil aggregation, the erosion and deposition of soil N, and the input and relocation of soil organic fractions, and therefore resulted in their fair interaction on SOC accumulation of surface soil (Table 3) in the eroded hilly watershed.

4.4. Effects of land use and topographic position on soil microbial biomass C and N

Soil MBN and MBN/total N, potential indicators of the bioavailability of soil N, are widely used to evaluate soil quality of ecosystems and the efficacy of vegetation restoration (An et al., 2009; Zhu et al., 2012). Changes in soil MBN among various land uses and topographic positions were similar to those of soil total N (Fig. 3a and b), as shown by their close relationship ($r = 0.87$, $P < 0.01$, Table 4). This indicates changes in soil MBN depends upon soil total N at the watershed scale. Similar results were also observed in the karst ecosystems (Zhu et al., 2012). Furthermore, increases in MBN under the restored vegetation were superior to those of soil total N (Fig. 3a and b), in accordance with the significant increases in MBN/total N compared with the cropland (Fig. 3c). These results clearly indicate the restored vegetation, particularly the shrubland, enhances the potential availability of soil N in the eroded hilly watershed.

Soil MBC is the most important component of SOC, as soil microbes participate in the humification and decomposition of organic matter as well as soil C cycling and ecosystem restoration (Maharning et al., 2009; Zhu et al., 2012). In this study, soil MBC in the grassland, shrubland, and woodland increased by 1.5-fold to 2.8-fold in the slope, and by 38% to 100% in the gully, respectively, compared with the croplands (Fig. 4c). This indicates interactive influences of land use and topographic position on soil MBC and the associated soil C cycling. Usually, soil microbial biomass is regulated by the microhabitats, including primarily availability of substrates, nutrients and soil physical conditions (Maharning et al., 2009; Nilsson et al., 2012; Zhu et al., 2012). The restored vegetation remarkably increased SOC and total N, improved soil aggregation, and consequently resulted in tremendous increase in biomass of soil microbial community. This is evident from the positive correlation of soil MBC with SOC, total N, >0.05 mm aggregates and the MWD (Table 4). Similarly, synchronous increases in SOC, DOC and total N of cropland from the summit and slope to the gully led to the concomitant increase (50% to 70%) in MBC (Fig. 4c). However, soil MBC in the grassland, shrubland, and woodland remained unchanged among topographic positions, in concordance with changes in soil DOC, total N and MBN, although the SOC in the restored vegetation increased significantly from the slope to the gully (Figs. 3a, 4a, b and c). These results further reflect that availability of substrate and N determines soil MBC and the associated soil C cycling in the eroded hilly watershed. Therefore, the strong influences of land use and topographic position on labile substrates (i.e. DOC) and N supply of soils resulted in their interaction on soil MBC and the associated soil C cycling in the eroded hilly watershed.

Significantly greater changes were observed in soil MBC than in SOC among different land uses (Fig. 4a and c). This suggests that soil MBC under vegetation restoration is more sensitive than SOC, and can be used as a potential indicator to assess the efficacy of vegetation restoration on soil C sequestration in the Loess Plateau. A similar suggestion was proposed for the changes in SOC in croplands (Powlson et al., 1987). However, SOC in the restored vegetation increased by 25% to 46% from the slope to the gully (Fig. 4a), whereas soil MBC remained unchanged (Fig. 4c). Also, the MBC/SOC among different land uses changed significantly in the summit and slope, whereas remained similar in the gully (Fig. 4d). These discrepancies clearly indicate that distinct mechanisms of land use and topographic position may be involved in influencing SOC and MBC. Further studies should be conducted at spatial scales to assess the relationship of their changes in the eroded hilly areas.

Soil microbial C:N in the slope and the gully increased in a sequence of cropland, grassland, shrubland and woodland, in accordance with the change in soil C:N (Fig. 4e and f). It has been proposed that, the increase in microbial C:N indicates that the dominant soil microbes change from bacteria to the more fungi groups (Chu and Grogan, 2010). Analyzing the forest stands of 18 to 85 years old, it was found that soil fungal biomass positively correlated with soil C:N (Nilsson et al., 2012). Our results probably suggest that converting cropland to shrubland or woodland results in a higher fungal dominance within the soil microbial community through increasing soil C:N ratio. For land uses investigated in this study, however, microbial C:N decreased by 5% to 19% from the slope to the gully, which were reverse to changes in soil C:N (Fig. 4e and f). This reflects that topographic position, besides land use, also strongly influences the relative dominance of soil bacterial and fungal communities. The underpinning mechanism on these changes and their impacts on cycling of soil C and N deserved more investigation at various spatial scales.

The interactions of land use and topographic position on SOC, DOC, MBC, soil C:N and microbial C:N were only statistically significant at $P < 0.1$ (Table 3) in the eroded hilly watershed. This is probably attributed to the complex local topography and the spatial variation in vegetation restoration across the hilly watershed (Fu et al., 2004; Wang et al., 2001). The variations in spatial environment, including topography and vegetation coverage, may synchronously alter multiple soil processes, such as the erosion and deposition of soil, organic matters, and nutrients, the aggregation of soil, and the accumulation and decomposition of organic matter in soil (Lal, 2005; Tang et al., 2010). Thus, interactions of vegetation and topographic positions on soil C cycling are quite complicated and warrant further studies at various spatial scales, as well as temporal scales.

4.5. Potential implications for optimum ecosystem restoration and soil C sequestration

The restored vegetation mitigated the losses of surface soil C and N, particularly in the summit and the slope, but also increased the stocks of surface soil C and N, and concomitantly enhanced the cycling of soil C and N (indicated by the increased MBC and MBN; Figs. 3b and 4c). These improvements may, in turn, enhance the self-sustainability of ecosystems (Bell, 2001), and further accelerate the restoration of ecosystems and soil C sequestration (Zhu et al., 2010). Among the land uses investigated in this study, shrubland showed greater efficacy on these improvements. Therefore, converting cropland to shrubland is an initial strategy to restore degraded ecosystems and increase soil C sequestration in the eroded hilly region of the Loess Plateau. This is in agreement with the proposal of Chen et al. (2007a).

Topographic position, besides vegetation type, strongly influences soil C and N stocks, and the potential of soil C and N accumulation. Usually, the soil with low levels of soil C and N has high potential of soil C and N accumulation when vegetation restoration is implemented (Post and Kwon, 2000; Wang et al., 2011). In the eroded hilly watershed, the SOC and total N of cropland in the summit and the slope were 34% to 45% less than those in the gully (Figs. 3a and 4a). This implies there would be relatively higher potential of soil C sequestration if the cropland in the summit and the slope was converted to shrubland. Estimated from our current observed data, converting cropland to shrubland could increase potentially C sequestration in surface soil by 15.82, 13.92 and 14.31 t ha^{-1} in the summit, the slope, and the gully, respectively. To sequester more C in the soil, the cropland in the summit should be preferentially converted to shrubland.

5. Conclusion

Converting cropland to grassland, shrubland or woodland significantly improved soil aggregation, and increased surface SOC and N stocks, DOC, MBC, MBN, MBN/total N, soil C:N and microbial C:N in the eroded hilly watershed. These improvements were larger with shrubland than the other land uses. The SOC significantly decreased from the gully to the slope, and the magnitudes decreased in a sequence of cropland, grassland, shrubland and woodland. Soil DOC, total N, and MBN also decreased from the gully to the slope for cropland, but remained unchanged for grassland, shrubland, and woodland. Land use ($P < 0.05$), topographic position ($P < 0.05$), and their interaction ($P < 0.1$) strongly influenced SOC, total N, DOC, MBC, MBN, soil C:N and microbial C:N ratios. There were close relationships among soil aggregates, SOC and total N, and DOC, MBC and MBN. Therefore, land use, topographic position, and their interactions regulate SOC and N accumulation and their labile fractions in the eroded hilly region of the Loess Plateau at a watershed scale. Converting cropland to shrubland is an initial strategy to restore degraded ecosystems and increase soil C sequestration in the eroded hilly region of the Loess Plateau.

Acknowledgments

This study was jointly supported by the Open Fund from State Key Laboratory of Soil Erosion and Dryland Farming on the Loess Plateau (10502-Z11 and 10501-1213), and the National Natural Science Foundation of China (41071338).

References

An, S., Huang, Y., Zheng, F., 2009. Evaluation of soil microbial indices along a revegetation chronosequence in grassland soils on the Loess Plateau, Northwest China. Appl. Soil Ecol. 41, 286–292.
Arnalds, A., 2004. Carbon sequestration and the restoration of land health. Clim. Chang. 65, 333–346.
Barreto, R.C., Madari, B.E., Maddock, J.E.L., Machado, P.L.O.A., Torres, E., Franchini, J., Costa, A.R., 2009. The impact of soil management on aggregation, carbon stabilization and carbon loss as CO_2 in the surface layer of a Rhodic Ferralsol in Southern Brazil. Agric. Ecosyst. Environ. 132, 243–251.
Bell, L.C., 2001. Establishment of native ecosystems after mining — Australian experience across diverse biogeographic zones. Ecol. Eng. 17, 179–186.
Brookes, P.C., Landman, A., Pruden, G., Jenkinson, D.S., 1985. Chloroform fumigation and the release of soil nitrogen: a rapid direct extraction method for measuring microbial biomass nitrogen in soil. Soil Biol. Biochem. 17, 837–842.
Carrick, P.J., Krüger, R., 2007. Restoring degraded landscapes in lowland Namaqualand: lessons from the mining experience and from regional ecological dynamics. J. Arid Environ. 70, 767–781.
Chen, L., Gong, J., Fu, B., Huang, Z., Huang, Y., Gui, L., 2007a. Effect of land use conversion on soil organic carbon sequestration in the loess hilly area, loess plateau of China. Ecol. Res. 22, 641–648.
Chen, L., Huang, Z., Gong, J., Fu, B., Huang, Y., 2007b. The effect of land cover/vegetation on soil water dynamic in the hilly area of the loess plateau, China. Catena 2, 200–208.
Chen, H., Zhang, W., Wang, K., Hou, Y., 2012. Soil organic carbon and total nitrogen as affected by land use types in karst and non-karst areas of northwest Guangxi, China. J. Sci. Food Agric. 92, 1086–1093.
Chu, H., Grogan, P., 2010. Soil microbial biomass, nutrient availability and nitrogen mineralization potential among vegetation-types in a row arctic tundra landscape. Plant Soil 329, 411–420.
Conaway, A.W., Strickling, W., 1962. A comparison of selected methods for expressing soil aggregate stability. Soil Sci. Soc. Am. J. 26, 426–430.
Culley, J., 1993. Soil Sampling and Methods of analysis. Lewis Publishers, Boca Raton, FL.
Dolan, M.S., Clapp, C.E., Allmaras, R.R., Baker, J.M., Molina, J.A.E., 2006. Soil organic carbon and nitrogen in a Minnesota soil as related to tillage, residue and nitrogen management. Soil Tillage Res. 89, 221–231.
Eckert, S., Engesser, M., 2013. Assessing vegetation cover and biomass in restored erosion areas in Iceland using SPOT satellite data. Appl. Geogr. 40, 179–190.
Fu, B.J., Liu, S.L., Chen, L.D., Lü, Y.H., Qiu, J., 2004. Soil quality regime in relation to land cover and slope position across a highly modified slope landscape. Ecol. Res. 19, 111–118.
Fu, X., Shao, M., Wei, X., Horton, R., 2010. Soil organic carbon and total nitrogen as affected by vegetation types in Northern Loess Plateau of China. Geoderma 155, 31–35.
Fu, B., Liu, Y., He, C., Zeng, Y., Wu, B., 2011. Assessing the soil erosion control service of ecosystems change in the Loess Plateau of China. Ecol. Complex. 8, 284–293.
Funakawa, S., Makhrawie, M., Pulunggono, H.B., 2009. Soil fertility status under shifting cultivation in East Kalimantan with special reference to mineralization patterns of labile organic matter. Plant Soil 319, 57–66.

Gamboa, A.M., Hidalgo, C., De León, F., Etchevers, J.D., Gallardo, J.F., Campo, J., 2010. Nutrient addition differentially affects Soil carbon sequestration in secondary tropical dry forests: early-versus late-succession stages. Restor. Ecol. 18, 252–260.
Guo, S., Wu, J., Dang, T., Liu, W., Li, Y., Wei, W., Syers, J.K., 2010. Impacts of fertilizer practices on environmental risk of nitrate in semiarid farmlands in the Loess Plateau of China. Plant Soil 330, 1–13.
Guo, S., Zhu, H., Dang, T., Wu, J., Liu, W., Hao, M., 2012. Winter wheat grain yield associated with precipitation distribution under long-term nitrogen fertilization in the semiarid Loess Plateau in China. Geoderma 189, 442–450.
Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304, 1623–1627.
Lal, R., 2005. Soil erosion and carbon dynamics. Soil Tillage Res. 81, 137–142.
Lal, R., 2010. Managing soils and ecosystems for mitigating anthropogenic carbon emissions and advancing global food security. Bioscience 60, 708–721.
Luizão, R.C.C., Luizão, F.J., Paiva, R.Q., Monteiro, T.F., Sousa, L.S., Kruijt, B., 2004. Variation of carbon and nitrogen cycling processes along a topographic gradient in a central Amazonian forest. Glob. Chang. Biol. 10, 592–600.
Maharning, A.R., Mills, A.A.S., Adl, S.M., 2009. Soil community changes during secondary succession to naturalized grasslands. Appl. Soil Ecol. 41 (2009), 137–147.
Mikha, M.M., Rice, C.W., 2004. Tillage and manure effects on soil and aggregate-associated carbon and nitrogen. Soil Sci. Soc. Am. J. 68, 809–816.
Nilsson, L.O., Wallander, H., Gundersen, P., 2012. Changes in microbial activities and biomasses over a forest floor gradient in C-to-N ratio. Plant Soil 355, 75–86.
Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. Glob. Chang. Biol. 6, 317–327.
Powlson, D.S., Brookes, P.C., Chrstensen, B.T., 1987. Measurement of soil microbial biomass provides an early indication of changes in total organic matter duo to straw incorporation. Soil Biol. Biochem. 19, 159–164.
Rasse, D.P., Mulder, J., Moni, C., Chenu, C., 2006. Carbon turnover kinetics with depth in a French loamy soil. Soil Sci. Soc. Am. J. 70, 2097–2105.
Ritchie, J.C., McCarty, G.W., Venteris, E.R., Kaspar, T.C., 2007. Soil and soil organic carbon redistribution on the landscape. Geomorphology 89, 163–171.
Schwanghart, W., Jarmer, T., 2011. Linking spatial patterns of soil organic carbon to topography — a case study from south-eastern Spain. Geomorphology 126, 252–263.
Scowcroft, P., Turner, D.R., Vitousek, P.M., 2008. Decomposition of *Metrosideros polymorpha* leaf litter along elevational gradients in Hawaii. Glob. Chang. Biol. 6, 73–85.
Seibert, J., Stendahl, J., Sørensen, R., 2007. Topographical influences on soil properties in boreal forests. Geoderma 141, 139–148.
Six, J., Bossuyt, H., Dedryze, S., Denef, K., 2004. A history of research on the link between (micro) aggregates, soil biota, and soil organic matter dynamics. Soil Tillage Res. 79, 7–31.
Solomon, D., Lehmann, J., Zech, W., 2000. Land use effects on soil organic matter properties of chromic luvisols in semi-arid northern Tanzania: carbon, nitrogen, lignin and carbohydrates. Agric. Ecosyst. Environ. 78, 203–213.
Sparks, D.L., Page, A.L., Helmke, P.A., Loeppert, R.H., Soltanpour, P.N., Johnston, C.T., Sumner, M.E., 1996. Methods of Soil Analysis. Part 3: Chemical MethodsSoil Science Society of America Inc, Madison, WI.
Tang, X., Liu, S., Liu, J., Zhou, G., 2010. Effects of vegetation restoration and slope positions on soil aggregation and soil carbon accumulation on heavily eroded tropical land of southern China. J. Soils Sediments 10, 505–513.
Urich, P.B., 2002. Land use in karst terrain: review of impacts of primary activities on temperate karst ecosystems. Sci. Conserv. 198, 1–60.
Wang, J., Fu, B., Qiu, Y., Chen, L., 2001. Soil nutrients in relation to land use and landscape position in the semi-arid small catchment on the loess plateau in China. J. Arid Environ. 48, 537–550.
Wang, Y., Fu, B., Lü, Y., Chen, L., 2011. Effects of vegetation restoration on soil organic carbon sequestration at multiple scales in semi-arid Loess Plateau, China. Catena 85, 58–66.
Westerhof, R., Buurman, P., van Griethuysen, C., Ayarza, M., Vilela, L., Zech, W., 1999. Aggregation studied by laser diffraction in relation to plowing and liming in the Cerrado region in Brazil. Geoderma 90, 277–290.
Wu, J., Joergensen, R.G., Pommerening, B., Chaussod, R., Brookes, P.C., 1990. Measurement of soil microbial biomass by fumigation–extraction — an automated procedure. Soil Biol. Biochem. 20, 1167–1169.
Xu, Y., Yang, B., Liu, G., Liu, P., 2009. Topographic differentiation simulation of crop yield and soil and water loss on the Loess Plateau. J. Geogr. Sci. 19, 331–339.
Zhang, B., Yang, Y., Zepp, H., 2004. Effect of vegetation restoration on soil and water erosion and nutrient losses of a severely eroded clayey Plinthudult in southeastern China. Catena 57, 77–90.
Zhu, B., Li, Z., Li, P., Liu, G., Xue, S., 2010. Soil erodibility, microbial biomass, and physical–chemical property changes during long-term natural vegetation restoration: a case study in the Loess Plateau, China. Ecol. Res. 25, 531–541.
Zhu, H., He, X., Wang, K., Su, Y., Wu, J., 2012. Interactions of vegetation succession, soil biochemical properties and microbial communities in a Karst ecosystem. Eur. J. Soil Biol. 51, 1–7.