Evaluating Soil Quality–Soil Redistribution Relationship on Terraces and Steep Hillslope

Y. Li and M. J. Lindstrom*

ABSTRACT

Soil redistribution from tillage and water erosion have the potential to modify the spatial patterns of soil quality on terraced and steep cultivated hillslopes. However, few studies have investigated this relationship. Our objectives were to quantify soil quality parameters along terraced and steep hillslopes and determine the relationship between soil redistribution from tillage erosion and water erosion on soil quality parameters in the Chinese Loess Plateau. Soil quality indicators, i.e., soil organic matter (OM), available P, N, bulk density (D_b), and clay and silt contents were measured at 5-m intervals on a terraced field and at 10-m intervals on a steep cultivated hillslope in a down slope transect. Soil redistribution rates from tillage and overland flow were obtained by ^{137}Cs technique integrated with a tillage erosion prediction model (TEP). Water erosion was the primary cause for the overall decline in soil quality on the steep cultivated hillslope while tillage erosion had a comparable contribution to overall level in soil quality on the terraced hillslope. Soil movement by tillage controlled the spatial patterns in OM, N, and P on both terraced and steep cultivated hillslopes. Selective removal of finer particles by water erosion caused a linear decrease in clay content of 0.02% m^{-1} and corresponding increase in silt content of 0.04% m^{-1} downslope on the steep cultivated hillslope. The impact of tillage erosion on OM, N, and P on the steep cultivated hillslope can be assessed using the change in adjacent slope gradients (X) through a soil quality-topography regression model, $Y = aX + b$.

Terracing of steep hillslopes in the Loess Plateau, northern China, has been used extensively for control of water erosion. This area perhaps experiences the most severe water erosion problems in the world. The significant role of downslope soil translocation by tillage on total soil redistribution is becoming clear and has been well documented over the last decade (Lindstrom et al., 1990, 1992; Govers et al., 1994,1996; Quine et al., 1993, 1994; Lobb et al., 1995; Poesen et al., 1997). There is increasing evidence that soil translocation by tillage within terrace boundaries can be the dominant process in soil redistribution (Quine et al., 1993, 1999). The use of ^{137}Cs technique with tillage erosion models provide a new perspective to assess the contribution of water and tillage to total soil redistribution within a landscape (Govers et al., 1996; Quine et al., 1993, 1994, 1999; Zhang et al., 1998). To what extent tillage erosion affects soil variability and soil quality remains to a large extent unknown (Van Muysen et al., 1999); therefore, a quantitative evaluation of on-site impacts of soil redistribution because of tillage and water erosion is important for establishing a cause–effect relationship (Pennock, 1998; Lal, 1999, p. 329).

Schumacher et al. (1999) modeled the spatial variation in productivity due to tillage and water erosion for a 50-yr period through an empirical model and the WEPP (Water Erosion Prediction Project) hillslope model. This study demonstrated that soil redistribution from the combined effects of tillage and water erosion results in a net increase in spatial variability of crop productivity and a likely decline in overall soil productivity. Poesen et al. (1997) found that tillage erosion was responsible for the patterns of rock fragment cover that controls the spatial variability of the hydrological response in southeast Spain. Pennock (1998) suggested that tillage erosion should be an important process affecting soil quality and crop productivity in agricultural landscapes.

Despite intensive studies of soil erosion over the last 40 yr on China's Loess Plateau, effects of erosion on soil quality and crop productivity have been mostly neglected because of two reasons. First is the inherent fertility of the loess and homogeneity in distribution of particle-size composition (Liu, 1985). The second reason is the lack of quantitative information on the soil erosion–soil quality relationship across landscape positions. Therefore, in China it has been concluded that soil erosion is unlikely to have serious impact on productivity for the homogenous loess soil (Walling and Quine 1993; Zhang et al., 1997). However, measurements of suspended sediment in the Yellow River of China, indicates that water erosion causes considerable losses of OM, N, P, and other soil nutrients (Zhu, 1984). Although water erosion has exerted a strong influence on soil quality parameters, we are suggesting that soil redistribution by tillage also plays an important role in soil quality parameters across the landscape.

There is a need for evaluating the soil redistribution–soil quality relationship on terraced and steep cultivated hillslopes in the Chinese Loess Plateau. Narrow summit positions and steep linear backslopes (slope gradients up to 40°) characterize the Chinese Loess Plateau. Crop production levels depend primarily on inherent fertility, which emphasizes the importance of tillage erosion in spatial variability in soil quality.

Against this background, studies were conducted on terraced and cultivated hillslopes on China's Loess Plateau. The objectives were (i) to examine spatial patterns of soil quality on terraced and steep cultivated hillslopes; (ii) to determine the contribution of tillage and water erosion to total soil redistribution on these two contrasting landscapes; (iii) to examine the correlation of tillage and water erosion with soil quality over the landscape; and (iv) to develop a possible landscape model for assessing variations in soil quality.

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Abbreviations: D_b, bulk density; k, tillage transport coefficient; OM, organic matter; TEP, tillage erosion prediction model.

MATERIALS AND METHODS

Study Area

The field sampling and investigation were conducted in the Yangjuangou Reservoir catchment (Li et al., 1997, p. 15). The catchment has an area of 2.02 km^2, 1025 to 1250 m above mean sea level, located near Yan'an city, northern Shaanxi province in China (36°42′N, 109°31′E). It is a secondary tributary of the Yanhe River.

These soils were developed from Malan loess with uniform soil texture (16% clay, 50% silt, and 34% sand), classified as Calciustepts in the U.S. taxonomic classification system (Soil Survey Staff, 1999) and represents the Chinese Loess Plateau where many erosion studies have been conducted in the past 40 yr. The distinctive characteristic of these landscapes are the narrow summits (averaging 30 m) and long linear backslopes (150–300 m). The study area has had a long history of cultivation dating back more than 1000 yr. Water erosion problems are the result of deforestation on steep slopes up to 40° and the extremely high erodibility of the loess soils (Li, 1995, p. 133).

Field Sampling

A detailed topographic survey was conducted on two hillslopes without terraces at 5-m intervals and on a terraced hillslope at 2- to 3-m intervals during April 1997. The two hillslopes without terraces (210-m horizontal length) had similar topographic features; one hillslope was cultivated and the other had a mixed land use (Fig. 1a and Table 1). A field boundary existed at the break between the summit position of the steep cultivated hillslope and the upper backslope position, which effectively acted as a field terrace. The upper portion of the mixed land-use hillslope, 114-m horizontal length, under permanent vegetation of grass or forest was used as the reference slope to characterize the on-site impacts of topography on soil quality. The lower hillslope portion of the mixed land-

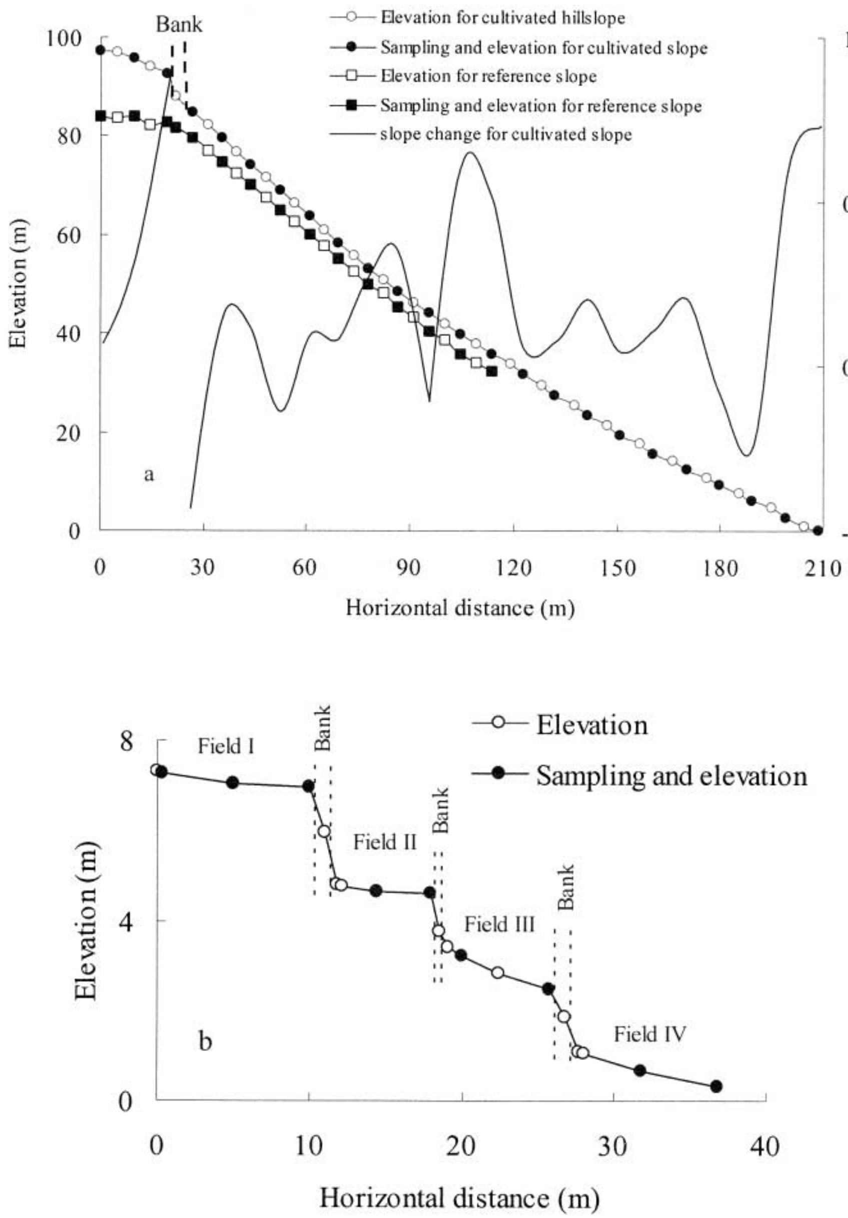

Fig. 1. Profile of the study of steep hillslopes showing (a)changes in slope gradients for the steep cultivated hillslope and (b)terraced slope on the Loess Plateau, near Yan'an, Shaanxi Province, China.

Table 1. Landscape characteristics for cultivated hillslope and reference hillslope.

		Backslope			
	Summit	Upper	Mid	Lower	Foot
Slope aspect	Southwest-facing				
Slope length, m	30	60	50	50	50
Horizontal distance, m	0–22	22–70	70–114	114–160	160–210
Soil type	Typical loess soil				
Cultivated slope					
Samples, n	8	14	14	14	14
Slope gradient					
Range, degrees	3.0–15.8	29.3–32.8	17.7–30.5	17.3–20.7	8.0–17.8
Average, degrees	10.4	31.3	25.5	19.1	14.4
Average cover, %	20	<5	10	30	30
Reference slope					
Samples, n	8	14	14		
Slope gradient					
Range, degrees	0.5–19.7	28.0–30.0	12.3–28.7		
Average, degrees	8.7	29.4	25		
Land use	Grassland	Forestland	Forestland		
Average cover, %	80	90	80		

use hillslope (114–210 m) was managed as cultivated fields. The terraced slope, 37-m horizontal length, contained four terraced fields, constructed in 1958 (Fig. 1b). The hillslopes and terraces investigated were located on southwest facing slope within the catchment. All of the ^{137}Cs survey points coincided with the elevation survey points. Samples for determination of spatial patterns in ^{137}Cs were collected using a 6.74-cm–diam. hand-operated core sampler at 10-m intervals along each hillslope transect and 5-m intervals along the downslope transect on terraces. Two cores were collected at each sampling point to a depth of 40 to 60 cm and were then bulked to make a composite sample. Sampling to this depth ensured that all ^{137}Cs inventory of the soil profile was measured.

Reference sites for determining the ^{137}Cs fallout in the study area were established at undisturbed, noneroded, level terraced fields constructed in 1954 and uncultivated grassland within the catchment. A mean value of 2390 Bq m^{-2} was determined as the actual fallout of ^{137}Cs in the study area. This value was in the range of 2365 to 2741 Bq m^{-2} as reported by Zhang et al. (1998) within 40 km of the study area. For the soil quality parameters, soil D_b was determined over the 40-cm sampling depth, while soil particle-size distribution, available N, P, and OM were for the surface 10-cm.

Sample Analysis

All samples were air dried and passed through a 2-mm sieve and weighed. All soil particles from this loess soil passed through the 2-mm sieve. Measurements of ^{137}Cs concentration were conducted on a subsample of 1000 g of each bulked core sample using a hyperpure coaxial Ge detector coupled to a multichannel analyzer. The ^{137}Cs content of samples was detected at 662 keV and using counting times of 80 000 to 86 400 s, which resulted in analytical precision of ±6% for ^{137}Cs. The results of ^{137}Cs were originally calculated on a unit mass basis (Bq kg^{-1}) and were then converted to an inventory value (Bq m^{-2}) using the total weight of bulked core soil sample and the sampling area. Soil bulk densities (Mg m^{-3}) calculations were based on volume of bulked soil cores and oven dried mass determinations (Pennock et al., 1994). Available soil N (mg kg^{-1}) was determined by using microdiffusion (Bremner, 1965), and available P (mg kg^{-1}) was determined using the method described by Olsen and Sommers (1982). Organic matter (% by weight) was measured by wet combustion (Nelson and Sommers, 1982). Particle-size distribution (%) was analyzed using the hydrometer method (Gee and Bauder, 1986).

Soil Redistribution Rate

Calculations of total soil redistribution rate at each sampling location of the terraced and steep hillslopes were derived from ^{137}Cs measurements based on a ^{137}Cs mass balance model developed by Walling and He (1997, p. 29). The validity and value of the fallout ^{137}Cs approach has been demonstrated in numerous studies in a number of environments (Ritchie and McHenry, 1990; Walling and Quine, 1990, 1993; Loughran et al., 1987; Sutherland, 1992; Pennock et al., 1995). The basis of ^{137}Cs technique involves comparing the measured inventories (total activity in the soil profile per unit area, horizontal distance) at study sites with an estimate of the total atmospheric input obtained from a reference site (Walling and Quine, 1990; Walling & He, 1997, p. 29). By comparing ^{137}Cs measurements of the study site with the reference site, one can determine whether erosion (less ^{137}Cs present than at the reference site) or deposition (more ^{137}Cs than at the reference site) has occurred. In our studies, the following ^{137}Cs mass balance model (Walling and He, 1997, p. 29) was used for estimating total soil redistribution rates at individual sampling points:

$$\frac{dA(t)}{dt} = (1 - \Gamma)I(t) - \left(\lambda + P\frac{R}{d}\right)A(t) \quad [1]$$

where $A(t)$ represents the cumulative ^{137}Cs activity per unit area (Bq m^{-2}); R represents the erosion rate (kg m^{-2} yr^{-1}); d represents cumulative mass depth representing the average plough depth (kg m^{-2}); λ represents the decay constant for ^{137}Cs (yr^{-1}); $I(t)$ represents the annual ^{137}Cs deposition flux (Bq m^{-2} yr^{-1}); Γ represents the percentage of the freshly deposited ^{137}Cs fallout removed by erosion before being mixed into the plough layer; P represents the particle-size correction factor.

Rates of soil translocation because of tillage at each ^{137}Cs sampling point were obtained using the TEP from the topographic data collected in the field (Lindstrom et al., 2000):

$$R_T = Q_S/L_C \quad [2]$$

where R_T equals the soil translocation because of tillage (kg m^{-2} yr^{-1}); Q_S equals the downslope flux of soil (kg) per unit width of slope (m) per year; L_C equals the slope length (m) over which soil is lost (convex) or accumulated (concave).

Annual downslope soil transport (kg m^{-1} yr^{-1}), assuming tillage operations occur equally often in opposing directions, at any point in the landscape can be determined using:

$$Q_S = -kq \quad [3]$$

where k represents the tillage transport coefficient (kg m^{-1} yr^{-1}) per unit slope gradient, and q represents the tangent of slope gradient.

Only limited data for the appropriate k value for animal-powered tillage has been reported. Thapa et al. (1999) determined a mean annual soil k of 423 kg m^{-1} yr^{-1} for moldboard plowing in the Philippines based on two cropping cycles per year. No significant difference in *k* values were determined between up and downslope tillage versus contour tillage. Quine et al. (1999) determined *k* values of 108 243 and 113 kg m^{-1} yr^{-1} in China, Lesotho, and Zimbabwe for animal-powered tillage when the direction of tillage was always downslope. However, in the case for continuous downslope tillage an addition constant must be added to account for the unidirectional tillage. Quine et al. (1999) concluded that net downslope translocation by animal-powered tillage always in the downslope direction may exceed those associated with mechanized agriculture. Therefore, we assigned a *k* value of 250 kg m^{-1} yr^{-1} per unit slope gradient as a reasonable approximation to simulate soil translocation because of animal-traction tillage on both the terraced field and cultivated hillslope in the Loess Plateau, China. Average water erosion rates for each sampled location were estimated by the differences between total soil redistribution and tillage erosion rates.

Statistics and Landscape Analysis

Correlation coefficient and several statistical parameters (Stein et al., 1997) were calculated to compare the relationship between the variability patterns of soil quality and soil redistribution rates at individual positions within the hillslope landscape and at selected landscape positions. Regression modeling techniques were conducted to develop a simple soil quality–topography model for evaluating on-site impacts of erosion at the hillslope scale.

RESULTS AND DISCUSSION

Spatial Patterns of Soil Quality

On the terraced hillslope, the most distinctive patterns observed in soil quality were a decrease in OM and N levels and an increase in D_b in the upper portions of the terraces ($n = 4$, duplicate samples from Fields I and III) through the mid portion ($n = 6$, duplicates samples from Fields I, II, and IV) compared with the lower end of each terrace ($n = 8$, duplicate samples from all terraced fields) (Fig. 2). Organic matter averaged 0.68% in the upper portions of the terraces as compared with 0.93% at the lower portions. Nitrogen content increased from 27 to 34 mg kg^{-1} from the upper to lower portions of the terraces while D_b showed a substantial decrease (1.29 to 1.18 Mg m^{-3}) from the upper to lower portions of the terrace. These patterns are in agreement with redistribution patterns in ^{137}Cs inventory. Over the last 38 yr, total ^{137}Cs concentration had decreased 15% within the terrace system; showing a 76% decrease at the upper portions of the terraces, a 25% decrease in the mid sections, but showing a gain of 20% at the lower portion of the terraces. Changes in available P were variable; an increase was observed in the lower portion of the upper terrace but little or no change on the three lower terraces.

On the steep cultivated hillslope, the most noticeable changes were a decrease in soil clay content (Fig. 3) and soil D_b (Table 2), and corresponding increase in silt content from the upper to lower portions in the backslope. Clay content decreased linearly with backslope length ($r^2 = 0.85$, $P < 0.01$), while silt content increased ($r^2 = 0.96$, $P < 0.01$), suggesting a sorting of soil separates by overland flow along the backslope gradient. The clay and silt contents in the upper 30- to

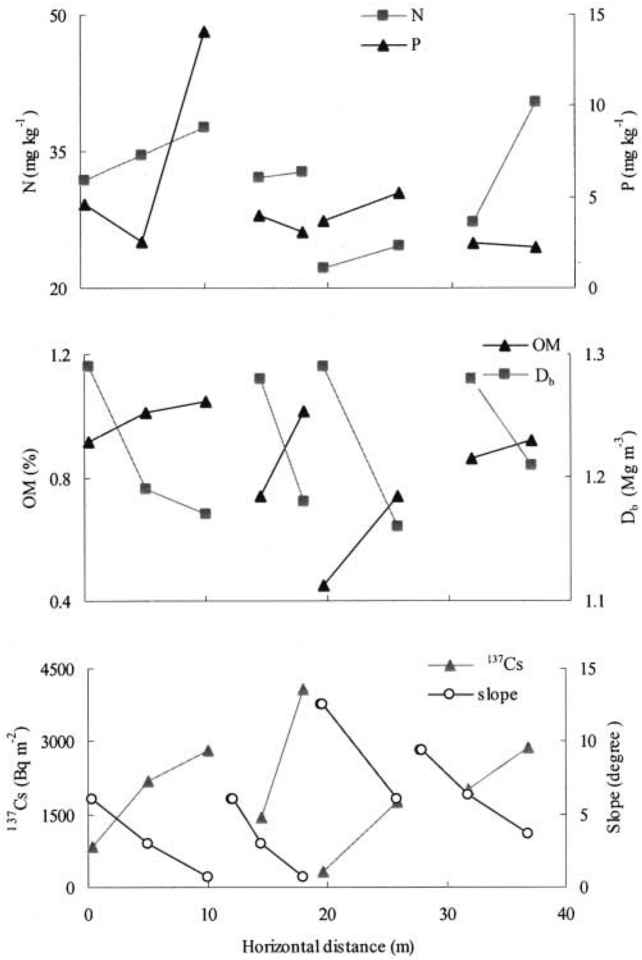

Fig. 2. Spatial distribution in soil quality parameters and ^{137}Cs inventory along the terraced slope transects.

90-m portion of the backslope were approximately equal to the values of the original loess parent materials indicating that surface soil materials have been completely removed by tillage and water erosion as bulk soil loss. The decline in clay content through the mid to lower portions of backslope (90–210 m) suggests a selective transport mechanism of fine soil materials by overland flow from the hillslope to the waterways.

The magnitudes of OM, N, P, and ^{137}Cs were signifi-

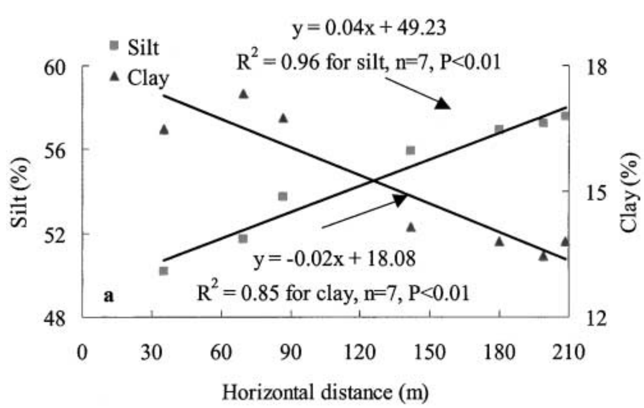

Fig. 3. Spatial distribution in clay and silt contents through the upper to the lower portions of the backslope of a steep cultivated hillslope.

cantly lower on the steep cultivated hillslope than on the terraced and reference hillslope. This is seen even more clearly when the individual data are combined according to their respective landscape locations (Tables 2 and 3). The lowest contents of soil OM and soil nutrients occurred in the upper portion of the steep cultivated backslope. Soil D_b decreased in the lower portion of cultivated hillslope transect. The steep cultivated hillslope (Table 2) had a 53% decrease in OM, a 47% reduction in N and a 63% decrease in P content when compared with the terraced hillslope (Table 3). Corresponding loss in ^{137}Cs was 61% (based on a 2390 ^{137}Cs-reference value basis) for the steep cultivated hillslope compared with 15% for the terraced hillslope. Concentrations of OM, N, and P in the upper 140- m slope position of the steep cultivated hillslope (summit, upper, and backslope) were 31, 37, and 88% of the reference hillslope. An 8% increase in D_b (Table 2) was measured on the steep cultivated hillslope. However, the spatial variability patterns in OM and P on our steep cultivated hillslope could not be explained by ^{137}Cs data. For example, higher OM ($P < 0.01$) and P ($P < 0.05$) contents were measured in the mid section of the backslope (horizontal distance 70–114 m) than in the upper and lower sections on the backslope although the ^{137}Cs inventories were similar in the three sections.

Table 3. Summary statistics for measured ^{137}Cs inventories and soil quality indicators on terraced hillslope.

Variables		Upper	Mid	Lower
	Samples, n	4	6	8
^{137}Cs	Range, Bq m^{-2}	314–828	1425–2183	1761–4059
	Average, Bq m^{-2}	571	1868	2874
	SD†	363	395	940
OM‡	Range, %	0.45–0.92	0.74–1.01	0.74–1.05
	Average, %	0.68	0.87	0.93
	SD	0.33	0.13	0.14
P	Range, mg kg^{-1}	3.68–4.59	2.41–3.96	2.20–14.06
	Average, mg kg^{-1}	4.13	2.98	6.14
	SD	0.64	0.85	5.43
N	Range, mg kg^{-1}	22.13–31.80	27.21–34.61	24.59–40.39
	Average, mg kg^{-1}	26.97	31.29	33.84
	SD	6.84	3.75	6.92
D_b§	Range, Mg m^{-3}	1.29	1.19–1.28	1.16–1.21
	Average, Mg m^{-3}	1.29	1.25	1.18
	SD	0	0.05	0.02

† Standard deviation.
‡ Organic matter.
§ Bulk density.

Impacts of Topography on Soil Quality

The soil quality parameters measured appeared to be more affected by slope gradients on the terraced hillslope than on the steep cultivated hillslope (Table 4). However, terrace borders form zones of soil transport discontinuities. Soil tillage on landscapes divided by ter-

Table 2. Summary statistics for measured ^{137}Cs inventories and soil quality indicators on the hillslopes.

		Cultivated hillslope				
Variables		Summit	Upper	Mid	Lower	Foot
	Samples, n	8	14	14	14	14
^{137}Cs	Range, Bq m^{-2}	520–4231	150–1031	348–937	452–1143	639–4070
	Average, Bq m^{-2}	1837	521	527	755	1508
	SD†	2077	290	255	258	1467
OM‡	Range, %	0.34–0.51	0.25–0.38	0.39–0.49	0.33–0.40	0.31–0.75
	Average, %	0.44	0.33	0.43	0.36	0.47
	SD	0.09	0.05	0.04	0.03	0.18
P	Range, mg kg^{-1}	1.56–2.43	1.01–1.99	1.62–2.20	1.50–1.92	1.53–1.85
	Average, mg kg^{-1}	1.91	1.39	1.96	1.70	1.75
	SD	0.46	0.33	0.24	0.15	0.13
N	Range, mg kg^{-1}	17.00–23.63	9.56–18.22	14.44–17.76	14.22–17.37	14.09–26.15
	Average, mg kg^{-1}	20.43	14.07	15.67	15.69	19.55
	SD	3.32	3.51	1.35	1.41	4.52
D_b§	Range, Mg m^{-3}	1.30–1.31	1.20–1.32	1.25–1.31	1.21–1.27	1.10–1.19
	Average, Mg m^{-3}	1.31	1.28	1.27	1.23	1.16
	SD	0.01	0.05	0.03	0.02	0.04
		Reference slope covered by forest and grass				
	Samples, n	8	14	14		
^{137}Cs	Range, Bq m^{-2}	1328–1756	1713–3011	745–2315		
	Average, Bq m^{-2}	1476	2288	1428		
	SD	242	564	640		
OM	Range, %	1.05–1.55	0.89–1.98	0.46–1.22		
	Average, %	1.36	1.52	0.99		
	SD	0.07	0.41	0.32		
P	Range, mg kg^{-1}	1.77–2.61	1.32–2.27	1.31–2.64		
	Average, mg kg^{-1}	2.15	1.79	2.01		
	SD	0.43	0.43	0.50		
N	Range, mg kg^{-1}	37.93–52.28	31.99–74.49	23.25–55.38		
	Average, mg kg^{-1}	45.44	48.22	41.62		
	SD	7.20	15.59	12.40		
D_b	Range, Mg m^{-3}	1.10–1.24	1.01–1.23	1.17–1.28		
	Average, Mg m^{-3}	1.16	1.16	1.22		
	SD	0.07	0.08	0.04		

† SD represents standard deviation.
‡ Organic matter.
§ Bulk density.

Table 4. Relations between slope gradients (S, in degree) and soil quality indicators [organic matter (OM) in %; N in mg kg^{-1}; P in mg kg^{-1}; bulk density (D_b) in Mg m^{-3}].

	Linear regression	R^2	n	P
Terraces	**OM = 1.15 − 0.06 S**	**0.92**	**9**	**<0.01**
	N = 39.48 − 1.61 S	**0.69**	**9**	**<0.01**
	P = 7.34 − 0.53 S	**0.19**	**9**	**n.s.†**
	D_b = 1.16 + 0.01 S	**0.49**	**9**	**<0.05**
Hillslope	**OM = 0.517 − 0.006 S**	**0.20**	**24**	**<0.05**
	N = 22.563 − 0.275 S	**0.37**	**24**	**<0.01**
	P = 1.989 − 0.013 S	**0.11**	**24**	**n.s.**
	D_b = 1.193 + 0.002 S	**0.10**	**24**	**n.s.**

† Not significant.

races result in soil transport away from the upper boundary primarily by tillage, while at the lower boundary soil accumulates because of a combination of soil erosion and deposition by water and tillage translocation processes (lynchet formation). As the distance between the terrace boundary decreases, mass soil transport by tillage becomes the more dominant process of soil redistribution (Turkelboom et al., 1997). The distribution of measured soil quality parameters and ^{137}Cs inventory were random and showed no association with slope gradients and changes in adjacent slope gradients at the vegetated portion of 0 to 114 m of the referenced hillslope (Tables 1 and 2). On the terraced hillslope, soil OM, and N decreased with increased slope gradient ($P < 0.01$), with a corresponding increase in D_b ($P < 0.05$). Upper portions of the terraces had lower OM and available N content in combination with a higher D_b. Organic matter and available N increase from the mid to the lower portion of each terrace because of a sharp decrease in slope gradients at the end of terraces (Fig. 2 and Table 3).

On the steep cultivated hillslope, spatial patterns in soil quality largely depended on changes in adjacent slope gradients, as indicated by their significant correlation coefficients shown in Fig. 4 for OM, N, and P. For this steep hillslope, the change in adjacent slope gradients was determined by:

$$C_{SL} = (SL_{(i)} - SL_{(i-1)})/HD \qquad [4]$$

Where C_{SL} equals the change in adjacent slope gradients (degree m^{-1}); $SL_{(i)}$ equals the upslope gradient (degrees); $SL_{(i-1)}$ equals downslope gradient (degrees), and HD equals the horizontal distance between adjacent slope segments (m).

Individual slope gradients on the steep cultivated hillslope were calculated from the elevation measurements shown in Fig. 1a. Changes in slope gradients at sampling points were determined as the difference (Eq. [4]) between the slope segment immediately above the representative sampling point and the slope segment immediately below the sampling point. Slope gradient determinations in the terraced fields (Fig. 2) were from elevation measurements within the terraced fields excluding elevation measurements in the grassed portion of the terrace.

Positive slope changes, shown in Fig. 1a, represent a concave slope configuration. The higher values in OM, N, and P found in the summit and foot portions of the steep cultivated hillslope can be attributed to concave slopes in these hillslope positions and are directly related to lower net levels of soil loss as indicated by ^{137}Cs

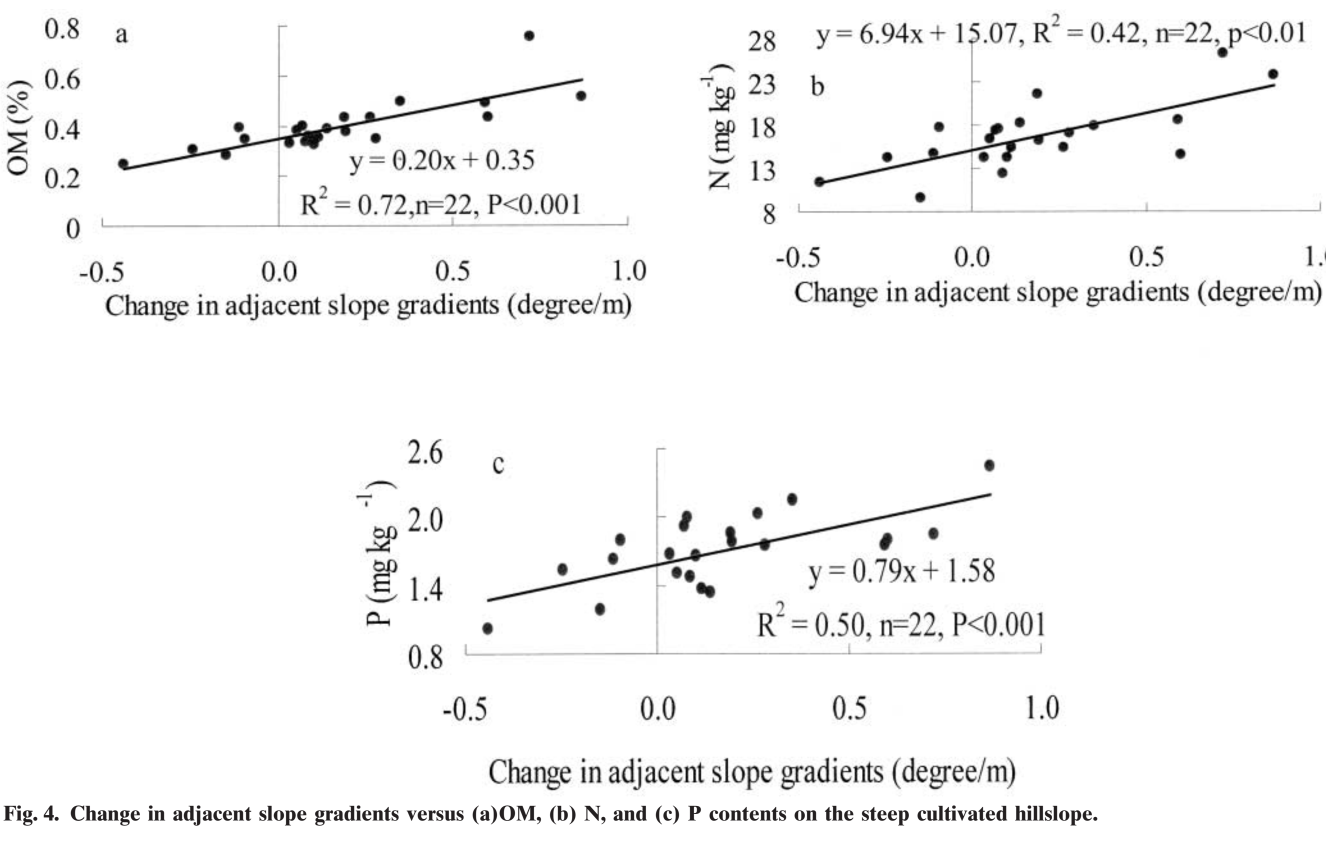

Fig. 4. Change in adjacent slope gradients versus (a)OM, (b) N, and (c) P contents on the steep cultivated hillslope.

Table 5. Gross erosion rates derived from ^{137}Cs data and tillage erosion prediction model (TEP) for the terraced slope.

Field	Slope length	Slope range	Erosion	
			Tillage	Water
	m	degrees	— Mg ha^{-1} yr^{-1} —	
I	15	0.7–6.0	13.2	13.0
II	7	0.7–6.0	16.5	16.5
III	7	6.0–12.5	28.9	35.5
IV	10	3.7–9.4	14.5	14.0

Table 6. Gross erosion rates derived from ^{137}Cs data and tillage erosion prediction model (TEP) for the cultivated hillslope.

Location	Slope length	Erosion		Deposition tillage
		Tillage	Water	
	m	— Mg ha^{-1} yr^{-1} —		
Summit	30	22.4	17.1	22.4
Backslope				
Upper	60	22.4	48.7	1.2
Mid	50	0.8	70.5	6.1
Lower	50	0.3	50.6	4.1
Foot	50	0.8	38.5	17.1

measurements. Soil translocation by tillage is directly related to slope gradients while tillage erosion (soil loss or gain) is directly related to changes in slope gradients and soil translocation rates as controlled by the tillage system (Govers et al., 1996). Soil is lost from convexities and deposited in concavities. The positive changes in slope gradients over the summit and foot position of the steep cultivated hillslope signifies concave slopes that result in soil deposition from tillage translocation. A similar situation exists in the mid backslope position (horizontal distance 70–114 m), i.e., a concave slope and higher OM, N, and P values (Fig. 1a and Table 2).

Changes in adjacent slope gradients may be used to develop a soil quality–topography relationship. A soil quality–topography landscape relationship can be described by regression analyses of the values in soil quality versus changes in adjacent slope gradients at the same locations on the steep hillslope (Fig. 4). The magnitude (Y) in soil OM, N, and P can be described using a simple linear regression model through changes in adjacent slope gradients (X): $Y = aX + b$, where a and b are constants. Constant a depends on magnitude of changes in adjacent slope gradients and is an effective constant of topography on soil quality. Constant b depends on the initial status of soil quality at upper portions of the cultivated hillslope. The mean relative error between predicted and measured values on the steep cultivated hillslope is 0.6% for OM, 1.2% for N, and 2.7% for P. If the estimated values in soil quality are accepted, it is possible to assess the magnitudes and patterns of soil quality at different landscape locations based solely on changes in adjacent slope gradients.

Contribution of Tillage and Water Erosion to Total Soil Redistribution

To compare the relationship between tillage and water erosion on soil redistribution over the terraced and steep cultivated hillslopes, total soil redistribution was determined by the ^{137}Cs mass balance model developed by Walling and He (1997, p. 29). Soil redistribution by tillage was determined by using the TEP model developed by Lindstrom et al. (2000). Soil redistribution by water erosion was determined by difference. The relationships between tillage and water erosion using this procedure are shown in Table 5 for the terraced hillslope and in Table 6 for the steep cultivated hillslope. Soil deposition shown in Table 6 were determined on the basis of changes in adjacent slope gradients (C_{SL}, Eq. [4]) within each slope segment of the steep cultivated hillslope. Soil deposition levels equal to tillage erosion rates shown in Table 5 would also take place on each of the terraced fields. Soil is not moved past field boundaries by tillage erosion.

The terraced hillslope and the summit and backslope of the steep cultivated hillslope showed the same spatial pattern in tillage erosion, i.e., a maximum soil loss at the upper portion and corresponding maximum accumulation at the lower portion because of tillage. The apparent loss and accumulation in the summit position is the result of a field boundary that acted as a terrace (Fig. 1a). This pattern is in agreement with the results reported by other authors (Quine et al., 1999; Lindstrom et al., 1990; Govers et al., 1996). The contribution of tillage and water erosion to soil redistribution was different between the terraced and steep backslope portions. Tillage erosion were found to be comparable with water erosion on 7- to 15-m width terraces and 30-m summit of the cultivated hillslope (Tables 5 and 6). High downslope soil translocation rates, but low tillage erosion values, in combination with severe water erosion characterize the erosion processes on steep cultivated hillslopes in Chinese Loess Plateau. Quine et al. (1999) and Zhang et al. (1998) reported similar results. Overland flow resulted in increased soil loss from the mid to lower portions in the landscape where the most noticeable changes were a decrease in soil clay content and soil D_b and a corresponding increase in silt content from the upper to lower portions on the backslope. Higher rates of soil accumulation from soil redistribution by tillage occurred on the summit, mid, and foot portions of the backslope where the most noticeable changes in soil quality were greater levels of soil OM and available nutrients (Table 2).

Relating Soil Redistribution to Soil Quality

To link soil redistribution rates to soil quality parameters, correlation coefficients for ^{137}Cs level and erosion processes were determined (Table 7). Soil redistribution rates because of tillage erosion gave a better correlation with soil quality parameters than overland flow in terraced and steep cultivated hillslope landscapes, particularly on the steep cultivated hillslope. Clearly tillage erosion has resulted in higher levels of soil OM and available nutrient contents at the lower boundaries of the terraced fields and the foot portions of the cultivated hillslope (Fig. 2 and Tables 2 and 3). Similar increases in soil OM and available nutrients occurred in the mid section of the cultivated backslope where a variation in adjacent slope gradient (C_{SL}) and a pronounced concave area was observed. Thus, tillage erosion plays a major

Table 7. Correlation coefficients (r) of soil redistribution rates ($Mg\ ha^{-1}\ yr^{-1}$) with ^{137}Cs ($Bq\ m^{-2}$) and soil quality parameters.

Soil quality parameters	^{137}Cs	Total erosion	Tillage erosion	Water erosion
		Terraced fields (n = 9)		
P	0.11	0.11	0.11	−0.04
OM†	0.75*	0.83**	0.59	0.15
N	0.61	0.65	0.40	0.27
D_b‡	−0.71*	−0.69*	−0.74*	0.37
		Cultivated hillslope (n = 24)		
P	0.41*	0.43*	0.63**	−0.20
OM	0.7**	0.66**	0.64**	0.14
N	0.67**	0.63**	0.53**	0.24
D_b	−0.26	−0.34	−0.23	−0.20

* Significant at the 0.05 probability level.
** Significant at the 0.01 probability level.
† Organic matter.
‡ Bulk density.

role in determining spatial variability on soil quality parameters in the Chinese Loess Plateau.

CONCLUSIONS

Soil redistribution by tillage strongly influences the spatial patterns of some soil quality parameter on steep cultivated hillslopes in the Chinese Loess Plateau, while distance between terraces in combination with slope gradients exerts a major influence on soil quality on terraces. Degradation in soil physical properties increased from the foot to upper portion of the steep backslope of the steep cultivated hillslope. This will have important theoretical implication and wide application for soil erosion–soil quality research, particularly on China's Loess Plateau. Despite the world's highest accelerated erosion rates, soil erosion effects on soil quality are not considered a problem because of the homogenous distribution of soil particle size and the loess' inherent fertility. Water erosion has been considered the dominant land degradation process in the study area. However, soil redistribution by both tillage and water erosion do affect soil quality and must be understood for establishment of cause and effect relationships on the Chinese Loess Plateau.

Based on our study results, the following conclusions were made:

1. Water erosion is the direct force driving the overall decline in soil quality on steep cultivated hillslopes whereas tillage erosion has a comparable contribution to overall level in soil quality on terraces.
2. Spatial variability patterns in soil quality parameters on terraced and cultivated hillslope are strongly controlled by tillage erosion. Soil OM and available nutrients positively increased with the increase in soil accumulation because of tillage erosion rates in the landscape.
3. The increased variability in soil quality parameters because of tillage erosion on steep cultivated hillslopes not dissected by terraces can be quantified on the basis of a simple linear regression model through changes in adjacent slope gradients at different landscape locations. This model has a high precision for prediction of OM and N and P contents (mean relative error $<3\%$).

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Potential use of Rare Earth Oxides as Tracers for Soil Erosion and Aggregation Studies

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ABSTRACT

Most existing soil loss data are spatially-averaged, though various tracing techniques have been used for obtaining spatially-distributed data. Spatially-distributed soil erosion data are needed for validating physically-based erosion prediction models and for better understanding soil erosion dynamics. The objectives of this study were to evaluate the feasibility of using rare earth element (REE) oxides directly as tracers for soil erosion studies by examining their binding ability with soil materials, and also to test a quick acid-extraction procedure. Five REE oxide powders were directly mixed with a Miami silt loam soil (fine-loamy, mixed, mesic Typic Hapludalfs) and then leached with deionized water in a soil box to evaluate the mobility of REEs. Following leaching, soil samples were sectioned in 25-mm increments and analyzed for REEs. The REE-tagged soil was wet sieved to obtain REE concentrations in each aggregate size group. A simple acid-leaching method was used to extract REEs from all soil samples. The extracts were analyzed by Inductively Coupled Plasma-Mass Spectrometry (ICP-MS) techniques. The data indicated that the maximum coefficient of variation of the proposed procedure was <10% for all REEs. The REE oxides were uniformly incorporated into soil aggregates of different sizes (>53 μm) and were bound with silt-size particles. This finding shows that the direct use of REE oxides is feasible, which should be superior to other REE-tagged particulate tracers because it eliminates the need of tagging exotic particles with REEs. Also, direct mixing of a trace amount of REEs does not substantially alter physicochemical properties of soil particles and aggregates. This work has shown that REE oxides have a great potential for tracing soil erosion and aggregation.

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Sediment delivery from small watersheds or erosion plots has been well monitored. These data are extremely useful in developing erosion prediction models, understanding erosion principles, assessing the on-site and off-site impacts, and deriving best management practices to combat soil erosion. However, these data do not contain information on soil redistribution patterns within a watershed or an erosion plot. Spatially-distributed erosion data are needed for validating physically-based erosion models and for understanding soil erosion dynamics at a process level. Since erosion processes change in time and space, physically-based models such as the Water Erosion Prediction Project model have been developed to predict soil erosion at any given time and location. Process-based models can only be thoroughly validated with spatially-distributed erosion data.

To obtain spatially-distributed data, various types of tracers have been developed and used. Most of them involve atomic bomb fallout radionuclide ^{137}Cs (Brown et al., 1981; Ritchie and McHenry, 1990; Martz and De Jong, 1991; Walling and He, 1999), naturally occurring radionuclides such as ^{210}Pb, ^{7}Be, and ^{234}Th (Wallbrink and Murray, 1993), deliberately introduced radionuclides such as ^{56}Fe and ^{60}Co (Wooldridge, 1965; Toth and Alderfer, 1960), noble metals-labeled natural particles (Olmez et al, 1994), exotic particles including glass beads, fluorescent dye coated particles (Young and

Abbreviations: ARE, average relative error; D_{50}, particle size at which 50% of particles by weight are finer than that value; DW, deionized water; ICP-MS, inductively coupled plasma-mass spectrometry; REE, rare earth element.