Available online at www.sciencedirect.com

SCIENCE DIRECT®

Catena 65 (2006) 30 – 40

CATENA

www.elsevier.com/locate/catena

Preferential erosion of black carbon on steep slopes with slash and burn agriculture

C. Rumpel [a,*], V. Chaplot [b], O. Planchon [a,b], J. Bernadou [a,b], C. Valentin [b], A. Mariotti [a]

[a] *CNRS, Laboratoire de Biogéochimie des Milieux Continentaux, Centre INRA Versailles-Grignon, Bâtiment EGER, 78850 Thiverval-Grignon, France*
[b] *IRD, UR049, Érosion et Changement d'Usages des terres BP 06, Vientiane, Laos*

Received 23 July 2004; received in revised form 10 June 2005; accepted 2 September 2005

Abstract

In this study we investigated the quantitative and qualitative aspects of soil organic matter (SOM) losses caused by water erosion within a small catchment in Northern Laos, under steep slopes and slash and burn agriculture. The soils in the region have a high contribution of black carbon to soil organic matter and high erosion rates. The aim of the study was to quantify the erosion of black carbon and to identify the processes involved. The conceptual approach included the measurement of contents of SOM, black carbon and mineral bound SOM in bulk soils, sediments eroded from 1 m^2 plots and in sediments at the outlet of the 0.6 ha catchment. Additionally, the enrichment factors of bulk SOM, BC and mineral bound SOM were calculated for eroded sediments.

Carbon content of eroded sediments was higher than carbon content in bulk soil. The C enrichment factors in the sediments eroded from 1 m^2 plots ranged between 1.7 and 2.7, which confirms that SOM rich material such as clay or silt along with particulate organic matter such as black carbon and plant debris is preferentially eroded. At the watershed level a mean carbon enrichment factor of 1.5 showed that part of the eroded carbon is lost from the watershed. Analysis of the carbon types showed that mineral bound SOM is representing on average 30% and black carbon 15% of the carbon of the bulk soil. Mineral bound organic matter is not eroded preferentially from the soil. Enrichment factors from 1.1 to 1.8 were recorded for black carbon in sediments eroded from 1 m^2 plots. At the watershed level, black carbon represents 30% of carbon in eroded sediments. The average enrichment factor of black carbon in sediments eroded from the watershed is 2.3 compared to bulk soil C content. These results show a continuous enrichment of sediments in black carbon during the whole process of soil erosion. The preferential erosion of black carbon compared to other SOM types can be explained by its light nature, absence of mineral interactions right after its formation and its resistance to biodegradation during transport. Carbon erosion in these tropical soils may thus lead to a decrease of their ability to act as a sink for CO_2 due to the preferential exportation of a very stable SOM form. These results further suggest that erosion of black carbon could be a crucial process determining its fate in terrestrial ecosystems. Moreover, black carbon erosion could be conceptually simpler than soil erosion and could therefore be tackled by specific models.

Keywords: Tropical sloping land; Erosion; Organic matter; Black carbon

1. Introduction

Amounts of organic matter in soils are usually related to organic matter inputs (Scholes et al., 1997) and the factors controlling its stabilisation within the mineral soil (Sollins et al., 1996). When brought under agriculture, most soils, in particular under tropical climate conditions, lose about 20–30% of organic carbon by mineralisation. For instance, a 20–50% reduction of soil organic matter (SOM) has been reported by Sombroek et al. (1993) after tropical forests were cultivated. Cultivation of tropical soils in particular on sloping land increases the potential for soil erosion (McDonald et al., 2002).

Lateral movements of carbon occur in relation to soil erosion and interfere with the biological processes mentioned above. Erosion causes yearly the loss of 75 billion tons of soil (Oldeman et al., 1990) and several studies have

---

* Corresponding author.
E-mail address: cornelia.rumpel@grignon.inra.fr (C. Rumpel).

0341-8162/$ - see front matter © 2005 Elsevier B.V. All rights reserved.
doi:10.1016/j.catena.2005.09.005

shown a strong relationship between soil loss and organic matter loss (Slater and Carleton, 1938; Webber, 1964; Ritchie and McCarty, 2003). However, research information concerning this process is extremely scarce (Lal, 2003). In particular, many knowledge gaps exist, regarding the erosion of SOM types and the behaviour of eroded SOM as source or sink of atmospheric CO_2 (Liu et al., 2003).

Many complex interactions make this aspect of the global carbon balance uneasy to understand and to quantify. The first difficulty lies in the highly scale-dependent nature of soil erosion. At local scale, raindrop impact at the soil surface can detach 10 to 20 g of soil per litre of incident rainfall (Leguedois et al., 2004). However, most of this big amount settles at short distance and is not transported by running water (Planchon et al., 2000). At catchment scale again, sediments transported by running water at local scale most likely deposit before they reach the bottom of the slope, as was shown by Polyakov and Lal (2004), who measured the transport distance of sediments during a storm by tracing soil with rare earth elements.

The second difficulty lies in the physical nature of transport by water. The travel length of a particle in suspension depends on its settling velocity in water: the denser and the bigger a particle is, the faster it settles and the shorter its travel distance is. Thus, flow transport capacity is specific to each individual soil component as characterized by its size and density. Therefore, the nature of carbon in eroded sediments is unlikely to be the same as in the native soil. For example, particulate organic matter was reported to be subject to preferential erosion (Gregorich et al., 1998; Rodriguez Rodriguez et al., 2004). Moreover, enrichment ratios may be scale-dependent and may differ with carbon type.

The third difficulty lies in the possible biological and chemical transformation of carbon in its course. Beyer et al. (1993) reported that 70–80% of SOM in eroded soils might be decomposed during transport and deposition whereas Jacinthe and Lal (2001) consider that only 20% of the removed SOM is mineralised. Whether eroded organic matter is subject to decomposition or transported offsite depends on the type and chemical forms of organic carbon present as well as its type of protection.

Stabilised organic matter compounds may be used as a model substance to tackle the complex relationships between soil loss and carbon loss described above. The main interest of these compounds lies in their chemical stability, which allows us to focus on detachment and transport, disregarding chemical and biological processes along sediments course. In tropical regions mineral interactions are most important for organic matter stabilisation (Feller and Beare, 1997). Under slash and burn agriculture, black carbon resulting from burning is another important chemically stable carbon form (Schmidt and Noack, 2000).

Besides its interest in process-oriented studies concerning soil erosion, the fate of black carbon in terrestrial ecosystems is a relevant issue in itself. Mechanisms responsible for black carbon loss from soils are biodegradation and erosion. Studies have mainly been carried out to address the question of microbial decomposition of black carbon (e.g., Shneour, 1966; Shindo, 1992). No studies have been carried out to determine the potential of black carbon to be eroded. In the case of slash and burn agriculture, black carbon produced during fire represents most probably a significant part of SOM (Seiler and Crutzen, 1980), while this technique increases soil erodibility (Soto et al., 1995; Giovanni et al., 2001). Despite the fact that black carbon may be eroded from terrestrial sites (Bassini and Becker, 1990), transported in rivers (Mitra et al., 2002) and finally buried in marine sediments (Holtvoeth et al., 2003), the erosion of black carbon has never been addressed in detail because the vast majority of studies on the impacts of slash and burn agriculture on soil properties have taken place in lowlands, where sites are flat or gently sloping (McDonald et al., 2002). This study was carried out in a small watershed with steep slopes under slash and burn agriculture, where the main types of erosion are gully erosion (Chaplot et al., 2005a) and tillage erosion (Dupin et al., 2003).

To account for the complexity of the erosion process, this study was carried out at two different scales: the local scale and the catchment scale. The study at local scale consisted of sampling eroded sediments at the outlet of 1 m^2 microplots under natural rainfalls. The study at catchment scale consisted of sampling floods at the outlet of the catchment containing the microplots during the rainy season. 1 m^2 plots are too small to allow for long-distance transport processes. Therefore, erosion measured at this scale will be referred to as detachment, despite there is no doubt that some short-distance transport process must be involved to carry the sediments towards the outlet of the plots. The contribution of black carbon to the total SOM of soil and eroded sediments was analysed as oxidation resistant elemental carbon (Bird and Göcke, 1997) and the contribution of mineral bound SOM as carbon solubilised upon treatment with hydrofluoric acid (Eusterhues et al., 2003). The aims of the study were the following:

- to study the mechanisms of transport of soil carbon with eroded sediments,
- to determine the role of black carbon and mineral bound carbon in this enrichment process, and
- to compare results at watershed scale and at microplot scale in order to understand the physical processes involved in erosion of the two SOM types.

2. Material and methods

2.1. Study site

The study site is located in the mountainous areas of northern Laos (Fig. 1), a region which is highly impacted by

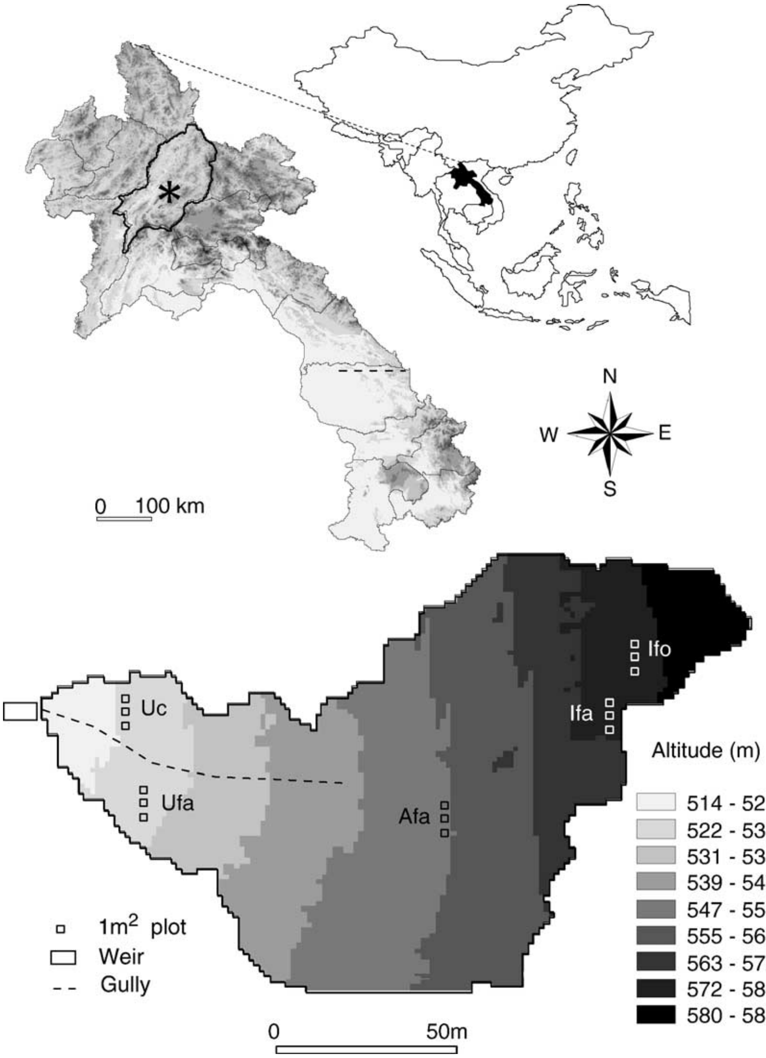

Fig. 1. Location of the experimental plots at the study site. (Uc=Ultisol cultivated; Ufa=Ultisol fallow; Afa=Alfisol fallow; Ifa=Inceptisol fallow; Ifo=Inceptisol forest).

slash and burn agriculture due to the increasing human pressure. We chose for our study the Houay Pano watershed situated 10 km south of Luang Prabang City, where steep slopes, on average 54% are prevailing. Erosion on such slopes is exceedingly high, especially on marginal land which has been converted to agricultural use (Pimentel and Kounang, 1998). The catchment covers an area of about 62 ha. Its outled is situated at 102°10′17″ longitude and 19°51′23″ latitude. This catchment is representative of slash and burn systems without inputs and submitted to a reduction of the fallow period from 5 years to 2 years. Land use within the catchment is predominantly composed of rotating land (80%), whereas less than 15% remained under secondary forest. This forest is very degraded with poor under-storey submitted nearly every year to bushfire. Upland rice and Job's tear (*Coix lacryma-jobi L.*) are the most common crop productions (de Rouw et al., 2003). On hillslopes, crops are generally located from backslope to upslope positions whereas the summits are under secondary and degraded forest and the bottomlands under bananas and tree plantations. Burning of slash and planting is usually carried out just before the rainy season in March–April.

At Luang Prabang, the average annual rainfall over the last 30 years was 1403 mm. Mean annual temperature was 25 °C. Two distinct seasons characterize the study site: a wet season from April to October and a dry season from November to March. Field observations performed over the catchment indicated that the geological substrate is mainly constituted by argilites, siltstones and fine grained sandstone from Permian to Upper Carbonifer. A cliff of limestones characterizes the northern boundary of the catchment. Altitudes estimated from a 10 m Digital Elevation Model generated from theodolithes data points ranged from 405 m at the catchment outlet to 718 m. Soils developed from these bedrocks showed a systematic distribution mainly controlled by the topography. From downslope to upslope positions, soils varied from Dystrochrepts, Alfisols and Inceptisols (Soil Survey Staff, 1999).

This study was carried out on a slope with a mean hillslope gradient of 46%. Alfisols are the most representative soils on the hillslope. They are deep (from 2.5 to 4.5 m), clayey soils marked by a typical argillic B sub-surface horizon with clay films of 1 mm or more. This argillic horizon formed below the soil surface, and is most frequently exposed at the surface by erosion. With increasing slope gradient, the argillic horizon disappears and Inceptisols occur. At downslope position, where the Dystrochrepts with redoximorphic features prevailed, average slope gradient was 30% with values from 5% to 33%. Values first increased in the upslope direction to reach a mean of 54% at backslope position and afterwards decreased to 45% midslope and 25% at the hillslope summit, where Inceptisols are present.

Carbon erosion in this watershed was studied at the detachment level (1 m^2 microplots) as well as the watershed level after natural rainfall events.

2.2. Experimental plots

The study was carried out on five experimental situations established in 2002 (Table 1, Fig. 1). At the 1 m^2 plot level, runoff and inter-rill erosion (erosion other than erosion in tillage rills) were monitored (Fig. 2). The enclosed plots of 1 m^2 size were established in triplicate in each situation. Experimental situations included Ultisols in two situations with 3 years of culture with upland rice and 3 years of natural fallow. Previous to this, these two situations showed similar land management (Uc and Ufa). The six plots were located along the same contour line with a distance interval of 2 m for replications and of 10 m for the closest plots of the two situations (Table 1). Additionally, three replicated plots were established on Alfisols under fallow (Afa) and Inceptisols under fallow (Ifa) and secondary forest (Ifo, Table 1). The plots were bounded by metal borders inserted to a depth of 0.1 m. Sowing occurred for all plots on May 15. The weeding was simultaneously operated on June 19, August 1 and 27. For weeding, plots were shallowly tilled by the farmers themselves. The intensity of such a shallow tillage was however not strong enough to allow for the establishment of rills for rill erosion to occur during the rainy season. After each rainfall event, the total runoff of each plot replication was measured and an aliquot sample was collected to estimate, after oven drying at 40 °C, the sediment concentration and afterwards the black carbon content. Between May 15, 2002 and December 15, 2002, the total runoff of each plot replication was measured and an aliquot sample was collected for measuring sediment concentration. A total of 210 samples were collected for 35 rainstorms. To obtain sufficient material for further analysis the three replicate samples collected for each situation were pooled.

Table 1
Experimental situations

Name	Sampling location	Slope angle	Slope position	Land use
Uc	Ultisol	45°	Midslope	Cultivated
Ufa	Ultisol	45°	Midslope	Fallow
Afa	Alfisol	45°	Midslope	Fallow
Ifa	Inceptisol	45°	Upslope	Fallow
Ifo	Inceptisol	45°	Upslope	Secondary forest

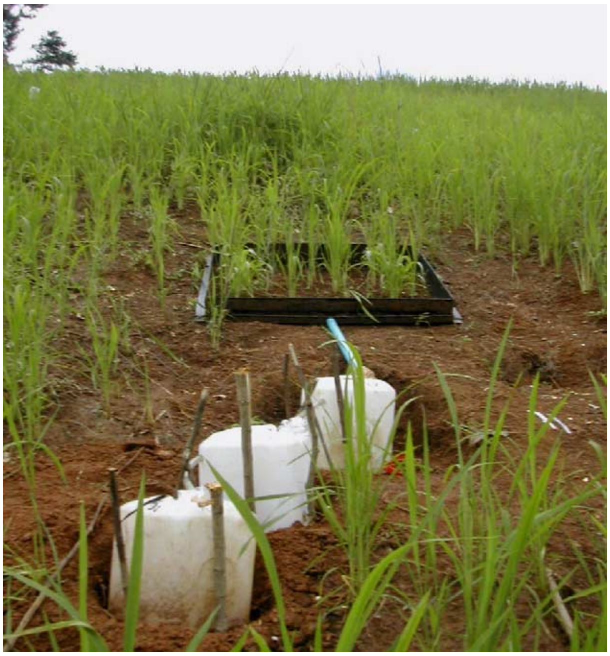

Fig. 2. 1 m^2 plots for erosion measurements.

2.3. Sampling of bulk soil

Approximately 300 g of the upper 0–5 cm soil were sampled at the end of the rainy season in 2002 from each of the 15 experimental plots. To homogenize the samples, the soil was sieved to obtain aggregates that passed a 5 mm sieve and were retained by a 3 mm sieve. The aggregate size range of 3–5 mm reflects the whole soil fraction in terms of the soil matrix because >95% of the soil was in this range. Due to the clay texture of the soil the aggregates are strong and very few fine material was lost. Usually the <2 mm fraction is used for pedological studies. We separated aggregates for structural stability analysis. The aggregates were also used for this study because the soils do not contain particles >2 mm. The samples were oven dried at 40 °C and an aliquot was ground for chemical analysis.

2.4. Sediment sampling to account for carbon erosion on the watershed level

The sediment sampling to evaluate black carbon erosion was performed during the 2002 rainy season. At the catchment level, the runoff and sediment loads were estimated using water level recorders and automatic water samplers installed within a weir. The sediment present in the

Table 2
Particle size distribution of bulk soil samples at the five experimental plots

	Coarse sand (%)	Fine sand (%)	Silt (%)	Clay (%)
Ultisol				
Cultivated	6±2	6±1	36±3	52±6
Fallow	6±1	4±0	32±2	55±2
Alfisol				
Fallow	7±2	6±1	36±4	52±8
Inceptisol				
Fallow	12±2	6±2	39±2	43±4
Secondary forest	19±3	7±1	35±9	39±3

water was recovered by filtration and oven dried at 40 °C. The content of black carbon was further estimated using the dry aliquot material of the three major rainfall events of year 2002.

2.5. Carbon, nitrogen content and $\delta^{13}C$

The carbon and nitrogen contents were determined with a Carlo Erba elemental analyser which was coupled to a Sira 10 isotope mass spectrometer yielding the ratio of stable isotopes (δ^{13}C). Soil pH ranged between 5 and 6, a range, where contribution from $CaCO_3$ cannot be expected. Therefore the total carbon content equals the organic carbon content.

2.6. Particle size distribution

Particle-size distribution was determined by the common pipette method after dispersion by Na-pyrophosphate and destruction of organic matter by H_2O_2. The sand fractions were quantified after wet sieving and the silt and clay content determined after sedimentation (Gee and Or, 1986).

2.7. Mineral bound organic matter, black carbon (oxidation resistant elemental carbon)

Removal of soil minerals was carried out by treatment of 5 g soil with 50 ml 10% hydrofluoric acid (HF, Schmidt et al., 1997). This solution was shaken for at least 2 hours. After centrifugation at 3000 RPM, HF was renewed and the soil solution shaken again. This procedure was repeated 5 times. Thereafter, the samples were washed 5 times with distilled water and freeze dried before being subjected to carbon and nitrogen measurements. The organic carbon lost upon the treatment accounts for mineral bound organic matter (Eusterhues et al., 2003).

Black carbon contribution to total carbon content was estimated as the oxidation resistant elemental carbon (OREC, Bird and Gröcke, 1997) and analysed on mineral-free samples. For determination of the black carbon content, 150 mg of an HF treated sample were submerged in an acid potassium dichromate solution and oxidised at 60 °C in an ultrasonic bath. A preliminary study showed that 24 h were an appropriate oxidation time, allowing for the oxidation of labile plant material and preserving most of the organic carbon produced during fire. Charcoal sampled from the site lost up to 40% of its carbon (Rumpel et al., in press). The solution was changed three to five times during the treatment depending on the colour change. Thereafter, the samples were washed 4 times with distilled water, before being freeze dried and weighed. The carbon content was determined on the oxidised sample and the oxidation resistant elemental carbon was expressed as percent of initial carbon after mass balance calculation.

2.8. Enrichment factors

For studying preferential erosion of organic matter, carbon enrichment factors are frequently used. In our study, we calculated the enrichment factor of carbon (E_c) as:

$$E_c = \text{mg C g}^{-1} \text{ in eroded sediments} / \text{mg C g}^{-1} \text{ in bulk soil (or sediment).} \quad (1)$$

Enrichment factors of C types (mineral bound C and black carbon) were calculated as:

$$E_{\text{C type}} = \text{mg}_{\text{C type}} \text{ g}^{-1} \text{C in eroded sediments} / \text{mg}_{\text{C type}} \text{ g}^{-1} \text{C in bulk soil or sediment} \quad (2)$$

2.9. Statistical analysis

Analyses of variance were carried out with the computer software Genstat (VSN International Ltd., UK).

3. Results

3.1. Physical and chemical properties of bulk soil

All plots have a high clay content, ranging between 39% and 55% (Table 2). Ultisols and Alfisols show a similar particle size distribution while Inceptisol plots have the lowest clay content and the highest content of coarse sand.

The carbon contents of the soil range between 18 and 25 g kg^{-1} and the C/N ratios between 11 and 14 (Table 3). The stable carbon isotope ratio (δ^{13}C) lies between −26.0‰ and −27.3‰, which is an expected value for a C3 type vegetation (Balesdent and Mariotti, 1996).

Table 3
Carbon and nitrogen contents and δ^{13}C in bulk soil at the 5 experimental plots 1 (0–52 cm)

	Carbon (mg g^{-1})	Nitrogen (mg g^{-1})	C/N	δ^{13}C (‰)
Ultisol				
Cultivated	21.9±6.9	1.9±0.3	11.6±0.9	−26.0±1.0
Fallow	22.2±1.0	1.7±0.1	12.9±0.6	−27.0±0.2
Alfisol				
Fallow	24.8±3.2	1.7±0.1	14.3±1.1	−26.9±0.4
Inceptisol				
Fallow	18.5±1.5	1.5±0.1	12.4±0.3	−26.7±0.2
Secondary forest	22.3±1.4	1.6±0.1	14.2±1.1	−27.3±0.8

Table 4
Carbon and nitrogen content, total amounts of carbon (g m^{-2}), C, N enrichment and δ^{13}C of sediments detached from soil (1 m^2 plots)

	Carbon (mg g^{-1})	Nitrogen (mg g^{-1})	C/N	Detached C (g m^{-2})	Enrichment[a]		δ^{13}C (‰)
					C	N	
Ultisol							
Cultivated	38.3	2.8	13.7	23.8	1.7	1.5	−26.9
Fallow	37.1	2.7	13.8	11.6	1.7	1.6	−27.5
Alfisol							
Fallow	64.2	3.6	17.6	30.2	2.6	2.1	−28.0
Inceptisol							
Fallow	51.3	3.3	15.4	26.3	2.8	2.2	−28.0
Secondary forest	60.1	3.4	17.5	100.9	2.7	2.1	−27.6

[a] Enriched with regards to carbon content in bulk soil (Table 3).

3.2. Content and forms of organic carbon in detached sediments

Carbon content of detached sediments ranges from 37 to 64 g kg^{-1} (Table 4). Soil type seems to have an influence on carbon and nitrogen content of detached sediments, the way that sediments detached from Ultisols regardless of the land use show a slightly lower carbon content. It is interesting to note that despite the higher carbon content of sediments detached from Alfisols and Inceptisols, only the Inceptisol under secondary forest shows a 3 fold higher C loss (g m^2) compared to the other situations, which had similar C losses. Detached sediments are enriched in carbon and nitrogen with regards to the bulk soil at all situations. Highest carbon and nitrogen enrichment was recorded for Alfisols and Inceptisols.

The C/N ratio is increased and the stable carbon isotope ratio (δ^{13}C) slightly depleted in detached sediments compared to the bulk soil (Tables 3 and 4). Therefore, we might suppose a different organic matter quality in detached particles, probably indicating a preferential erosion of certain organic matter types. Mineral bound carbon represents between 29% and 35% of the carbon of bulk soil and 17% to 34% of the organic carbon of detached sediments (Fig. 3). The enrichment values are close to unity or less than unity, suggesting that mineral bound material is not eroded preferentially. This is different for black carbon, which represents 10% to 17% of initial carbon in bulk soil and 16% to 24% of initial carbon in detached sediments (Fig. 4). The enrichment ratio is higher than unity regardless of soil type and land use.

3.3. Content and forms of carbon in sediments eroded from the watershed

The sediments eroded from the watershed collected at three different dates have a mean carbon concentration of 33 g kg^{-1} (Table 5). This is higher than the initial carbon content of the bulk soil (Table 3) but lower than the carbon content of detached sediments from the microplots (Table 4). Enrichment factors for total C and C types of sediments eroded from the watershed are shown in Table 6. Enrichment factors for total C are lower than unity when compared to eroded sediments sampled from 1 m^2 plots, but higher than unity compared to the C content of the bulk soil. C/N ratios are somewhat lower than those recorded for detached sediments. On average 25% of the eroded carbon is mineral bound versus 29% in form of black carbon (Table 5). The average enrichment factor is less than unity for mineral bound carbon and higher than unity for black carbon (Table 6). The enrichment factor for black carbon was 1.4, based on average black carbon concentrations of eroded sediments sampled at the detachment scale, and 2.3 based on black carbon concentrations of the bulk soil (Table 6).

4. Discussion

Extensive SOM loss occurs during soil erosion. Soil erodibility generally decreases with increasing SOM content and aggregate stability (Feller and Beare, 1997, Valentin et

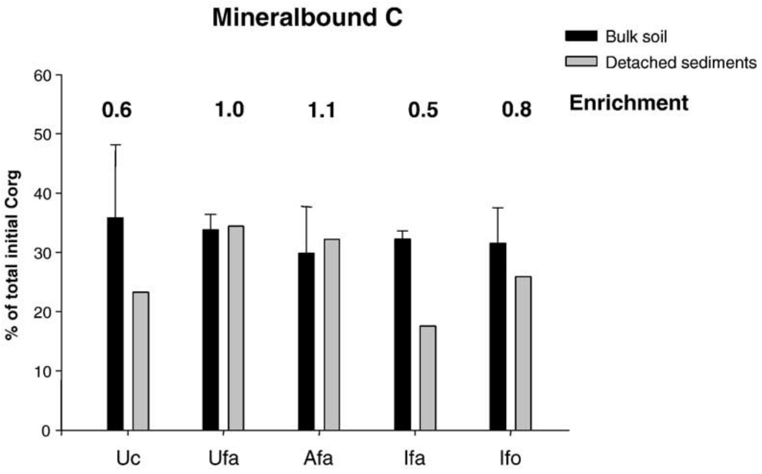

Fig. 3. Mineral bound carbon contribution (% of initial C) to the bulk soil and detached sediments. The differences between the five situations were not significant ($p = 0.875$; LSD = 13.12).

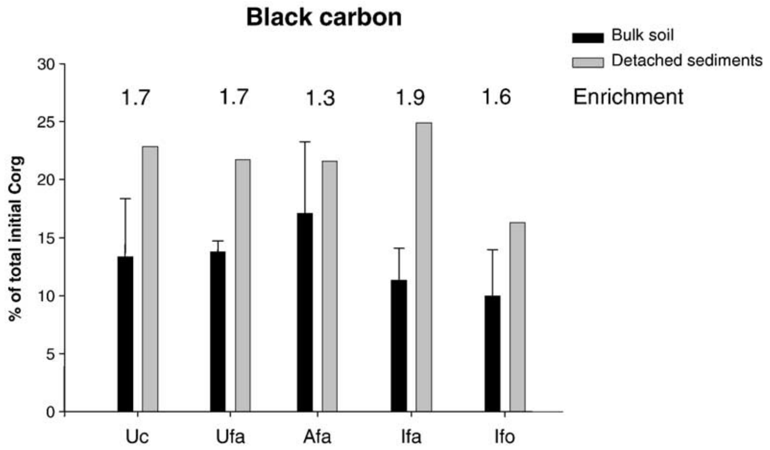

Fig. 4. Black carbon contribution as oxidation resistant elemental carbon (% of initial C) to bulk soil and detached sediments. The BC content was not significantly different at the five situations ($p = 0.354$; LSD = 7.62).

al., 2004). Albrecht et al. (1992) suggested a threshold of 20 mg C g^{-1} soil (0–5 cm) as an index of erosion risk for a Vertisol in Martinique. Regarding the high clay content and the organic carbon content of the soils, the intrinsic soil erodibility may be low (Tables 1 and 2). This corresponds to the high stability of soil aggregates shown by Chaplot et al. (2005b). The erosion process operating on sloping land affected by water erosion may be splash erosion, which relates to the effect of raindrops detaching and translocating soil particles. Moreover, tillage erosion, during soil cultivation may lead to transport of soil towards the bottom of the slope (e.g., Turkelboom et al., 1997; Dupin et al., 2003). For the evaluation of the erosion effect on SOM dynamics and carbon storage in soil, knowledge of the SOM types eroded by the different processes from the site is crucial. Therefore, we assessed the contribution of mineral bound carbon and black carbon to detached sediments translocated within the study site as well as to sediment eroded from the catchment.

4.1. Plot scale

Carbon and nitrogen are enriched in sediments eroded from 1 m^2 plots, indicating that SOM is detached preferentially from soil, with regards to mineral particles. Our results corroborate data from most studies in natural conditions showing that enrichment of SOM in sediments eroded and distributed within watershed is higher than unity (Ghadiri and Rose, 1991).

Few erosion processes are active at the square meter scale because 1 m^2 plots are too small to allow runoff to form eroding rills (unless the soil was previously tilled, which was not the case in our experiment). Thus, runoff can be efficient only at transporting sediments already detached by raindrops. Therefore, erosion at the square meter scale measures the combination of raindrop erosion and transport by discontinuous runoff (e.g., Janeau et al., 2003; Chaplot and Le Bissonnais, 2003).

Raindrops impacting the soil surface detach sediments and project them in every direction. Leguedois et al. (2004) measured detachment rates between 10 and 20 g m^{-2} mm^{-1} (that is, mass per unit source area per millimeter of rainfall) with various soil types. These values mean that the overall mass of sediment detached by raindrops represents 1% to 2% of the mass of rain that impacted the soil surface. Detached sediments are projected in every direction at various distances, which length follows a decreasing exponential (Van Dijk et al., 2003) with an average value of 0.1 to 0.2 m (Leguedois et al., 2004). This means that a significant part of the detached material may be thrown away at a distance of 0.5 m or more. For ballistic reasons analysed for long (De Ploey and Savat, 1968), splashed particles can travel farther in the downslope direction before they hit the soil surface. The consequence of this differential transport is that low points receive more sediments than they produce while high points are eroded, leading to a diffusive process that smoothes the soil surface (Planchon et al., 2002). When applied to a square-meter plot, these basics explain how the part of the surface

Table 5
Carbon and nitrogen content, C/N, $\delta^{13}C$, mineral bound and oxidation resistant carbon in sediments sampled at the collecting basin at the outlet of the watershed

Sample	Carbon (mg g^{-1})	Nitrogen (mg g^{-1})	C/N	$\delta^{13}C$ (‰)	Mineral bound (% of total C)	OREC (% of total C)
27/05/02	38.6	2.7	14.4	−27.6	19.3	20.7
17/06/02	34.8	2.4	14.4	−27.8	27.1	47.5
19/08/02	27.3	2.4	11.5	−27.4	31.4	20.4
Mean	33.5	2.5	13	−27.6	25.9	29.5

Table 6
Enrichment factors of total SOM, black carbon and mineral bound SOM of sediments sampled at the outlet of the watershed based on the carbon content of the bulk soil and the carbon content of eroded sediments

	Total SOM		Black carbon	Mineral bound SOM	Black carbon	Mineral bound SOM
	Based on C content of detached sediments	Based on C content of bulk soil	Based on content of detached sediments		Based on content of bulk soil	
Ultisol						
Cultivated	0.87	1.50	1.29	0.98	2.21	1.09
Fallow	0.89	1.52	1.35	0.75	2.13	0.74
Alfisol						
Fallow	0.52	1.35	1.36	0.85	1.66	0.79
Inceptisol						
Fallow	0.65	1.81	1.18	0.79	2.60	1.44
Secondary forest	0.55	1.50	1.81	0.80	2.97	0.98
Mean	0.70	1.54	1.40	0.83	2.31	1.00

For the calculation of the enrichment factor average SOM, black carbon and mineral bound SOM concentrations of sediments sampled at the outlet of the watershed (Table 5) have been used.

occupied by running water is continuously fed by sediments splashed from the emerged parts. If detachment rate is typically 10 g m^{-2} mm^{-1} and 10% of the plot is submerged, this process represents 1 g of sediment arriving in the running water per litre of incident rainfall. If runoff coefficient is 30%, this potentially makes a 3 g l^{-1} sediment concentration at the plot outlet, which is consistent with data obtained from erosion measurements during the 2002 rainy season, i.e. 4 to 6 g l^{-1} (Chaplot et al., 2005b).

The interesting point in the present issue is precisely that erosion rates at the square-meter scale are usually much less than 3 g l^{-1}. This is because most particles splashed in the running water settle before they cover a whole meter and are thus redistributed within the plot. However, some soil components, which have a very low settling velocity in water, may not follow this rule and reach easily the plot outlet. This is the case of black carbon for two distinct reasons: the first one is its light nature (Glaser et al., 2000), which allows the biggest particles to float; the second reason is the colloidal nature of the finest elements, which allow them to stay in suspension for long. Contrarily, mineral bound organic matter is unlikely to be detached by splash or transported by discontinuous runoff independently to the mineral fraction, and this is consistent with results in Fig. 3.

We conclude that splash by raindrop impacting the surface of bare soils well explains the enrichment of detached sediments in black carbon and the depletion of detached sediments in mineral bound carbon. Organic carbon detached from soil may be translocated within the site and not necessarily be subject to erosion from the watershed (Polyakov and Lal, 2004).

4.2. Catchment scale

Light, particulate organic matter seems to be eroded preferentially in many situations (Gregorich et al., 1998; Lal, 2003). Therefore, there may be substantial loss of SOM by oxidation in the erosion process due to aggregate breakdown during detachment and transport. Rodriguez (2004) concluded from the observation of preferential erosion of non-mineral bound OC that eroded carbon may be a source of atmospheric CO_2. In the case of black carbon, biological oxidation processes are extremely slow under natural conditions (Shneour, 1966; Shindo, 1992). The recalcitrant nature of black carbon was confirmed by radiocarbon ages of several thousand years for black carbon isolated from soil (Schmidt et al., 2002) as well as marine sediments (Masiello and Duffel, 1998). Therefore, black carbon as a chemically recalcitrant carbon form most likely survives transport within the watershed, whereas other particulate organic matter is easily decomposed. Moreover, due to its light nature, it may be preferentially transported by water (Mulleneers et al., 1999). At the watershed scale, the average enrichment factor is 2.3 with regard to the bulk soil and 1.4 with regard to the sediments already detached at the square-meter scale. The origin of black carbon observed at the outlet of the watershed remains unknown. It may come from either the drainage network (hypothesis 1) or the slopes (hypothesis 2). According to hypothesis 1, the black carbon would originate from the channels. It would be detached by rill and channel erosion. The enrichment ratio specific to the channel transport processes would therefore be 2.3 since it relates to the bulk soil. According to hypothesis 2, black carbon would be detached from the slopes by splash erosion before it enters the drainage network. In the latter case, the enrichment ratio attributed to the drainage network would be 1.4 since it would refer to the sediment already detached by splash. The real process is likely a blend of hypotheses 1 and 2. We thus can conclude that the transport processes in the drainage network lead to an enrichment ratio of 1.4 to 2.3. In the meantime, mineral bound carbon is not enriched at all by erosion and transport processes in the channel network. However, whatever the origin of the carbon at the catchment scale, for black carbon, the enrichment process is going on. For total SOM, this is not the case. As a result the enrichment of black carbon in eroded sediments is much higher than recorded for detached sediments with regards to the bulk soil and may indicate that there are (1) other erosion processes involved in black carbon erosion (e.g., rill erosion and tillage erosion) and/or (2) mechanisms leading to preferential transport of black carbon or selective loss of other carbon compounds.

4.3. Implications for erosion modelling

This experiment has shown that black carbon is able to concentrate in sediment along their transport by water. This process is probably not specific to our experimental conditions and could be general. The scope of the black carbon enrichment ratios at the 1 m^2 and the catchment scale (Fig. 3 and Table 6) is worth a more detailed discussion. Soil erosion is transport-limited in most cases. Erosion models, like WEPP among others, consider the balance between the transport capacity of a stream, and the actual sediment load. Deposition or erosion occurs to balance the stream's transport capacity. Therefore, the longitudinal variation of the transport capacity of the flow is a key parameter to localise areas where erosion and deposition occur. Our results suggest that transport of black carbon by water is easy and not limiting. Therefore, the key factor controlling erosion of black carbon would be the detachment process. This specific behaviour is drawn by the ability of black carbon to travel as individual particle of low density. The consequences of this difference in processes are that (1) black carbon erosion would be dramatically efficient since all the black carbon detached from soil is potentially able to be eroded and (2) black carbon erosion could be simpler to model than soil or SOM erosion which is finally a good news since erosion models have not yet proved their reliability (e.g., Jetten et al., 1999; Nearing, 1998, among others). Our study further demonstrates that erosion modelling needs to take into consideration the whole erosion chain including on- and offsite effects.

4.4. Implications for the tropical soils' functions on a local scale

In terms of onsite effects, the preferential erosion of black carbon has important implications in terms of carbon sequestration and soil fertility. Loss of black carbon may be detrimental to soil fertility because charcoal addition to soils was shown to be beneficial in terms of nutrient retention and availability (Glaser et al., 2002). Moreover, loss of black carbon from the site may reduce the tropical soil function to act as a sink for CO_2.

4.5. Implications for the fate of black carbon on a global scale

The nature of black carbon as a permanent sink of atmospheric CO_2 was questioned by many authors based on the assumption that at its actual production rate, the earth's surface would be converted into charcoal within a period of time of $<$100,000 years (Kuhlbusch and Crutzen, 1995). Usually, degradation of black carbon is assumed (e.g., Glaser et al., 2002). Evidence that such a degradation occurs is however scarce (Shneour, 1966). Given our results, we believe that the impact of erosion on black carbon is much more important than has been considered up to now.

5. Conclusion

Soil erosion on tropical sloping land under slash and burn agriculture is associated with organic matter loss, the carbon erosion at the 1 m^2 scale being more important than carbon erosion from the watershed. The decrease in the enrichment ratio from 1 m^2 to the catchment scale indicates that some of the carbon eroded from the microplots is deposited during transport or otherwise removed from the sediments. However, the erosion processes do not affect all organic matter types in the same manner. Black carbon was found to be preferentially eroded at the 1 m^2 scale compared to bulk SOM or mineral bound carbon. Comparison of the results obtained at the watershed scale and the microplot scale shows that in contrast to total carbon and mineral bound carbon, the enrichment of black carbon is going on until the outlet of the watershed, showing that it is more easily being transported. Black carbon is thus preferentially eroded from the watershed. Therefore, it could be used to constrain soil erosion models because all the black carbon eroded from soil is potentially able to be exported from the catchment. As erosion is the link between continents and the ocean, carbon eroded from the soils may ultimately be buried in marine sediments. Considering that this process possibly leads to loss of carbon from the active cycle, more studies are needed to address the preferential erosion of black carbon, a highly refractory organic matter type, as a real carbon sink in the global carbon cycle.

Acknowledgements

The study was financed by the department 'Institut National des Science d'Univers' of the 'Centre National de Recherche Scientifique' (France) under the framework of the project 'Dynamique et stabilisation des matières organiques dans des sols tropicaux: influence des brûlis, de l'érosion et de la mise en culture'. The authors wish to thank the National Agricultural and Forestry Research Institute (Laos) for access to the site and collaboration. Without the help of the colleagues of the Unité de Recherche 'Erosion et Changement Usage' of the Institut de Recherche pour le Developpement and the International Water Management Institute based in Laos, this study would have not been possible. We thank Naoise Nunan for the statistical analysis.

References

Albrecht, A., Rangon, L., Barret, P., 1992. Effet de la matière organique sur la stabilité structurale et la détachabilité d'un Vertisol et d'un Ferrisol (Martinique). Cahiers - ORSTOM. Pédologie 27, 121–131.

Balesdent, J., Mariotti, A., 1996. Measurement of soil organic matter turnover using ^{13}C natural abundance. In: Boutton, T.W., Yamasaki, S. (Eds.), Mass spectrometry of soils. Marcel Dekker, pp. 83–111.

Bassini, F., Becker, P., 1990. Charcoal's occurrence in soil depends on topography in Terra Firme forest near Manaus, Brazil. Biotropica 22, 420–422.

Beyer, L., Frund, R., Schleuss, U., Wachendorf, C., 1993. Colluvisols under cultivation in Schleswig-Holstein: 2. Carbon distribution and soil organic matter composition. Zeitschrift für Pflanzenernährung und Bodenkunde 156, 213–217.

Bird, M.I., Gröcke, D.R., 1997. Determination of the abundance and carbon isotope composition of elemental carbon in sediments. Geochimica et Cosmochimica Acta 61, 3413–3423.

Chaplot, V., Le Bissonnais, Y., 2003. Runoff features for interrill erosion at different rainfall intensities, slope lengths and gradients in an agricultural loessial hillslope. Soil Science Society of America Journal 67 (3), 844–851.

Chaplot, V., Coadou le Brozec, E., Silvera, N., Valentin, C., 2005a. Spatial and temporal assessment of linear erosion in catchments under sloping lands of Northern Laos. Catena 63, 167–184.

Chaplot, V., Le Bissonnais, Y., Bernadou, J., 2005b. Runoff, sediment and carbon losses within a small sloping-land catchment of the tropics showing different fallow durations. In: Roose, E., Lal, R., Feller, C., Barthès, B., Stewart, B.A. (Eds.), Soil Erosion and Carbon Dynamics, Advances in Soil Science. CRC Publisher, USA, ISBN: 1566706882, pp. 167–180.

Dupin, B., Panthahvong, K.B., Chanthavongsa, A., Valentin, C., 2003. Tillage erosion on very steep slopes in northern Laos. In: Maglinao, A.R., Valentin, C., Penning de Vries, F. (Eds.), From soil research to land and water management: Harmonizin people and nature. Proceedings of the IWMI-ADB Project Annual Meeting and 7th Management of Soil Erosion Consortium (MSEC) Assembly. IWMI, Bangkok, pp. 105–112.

Eusterhues, K., Rumpel, C., Kleber, M., Kögel-Knabner, I., 2003. Stabilisation of soil organic matter by interactions with minerals as revealed by mineral dissolution and oxidative degradation. Organic Geochemistry 34, 1591–1600.

Feller, C., Beare, M.H., 1997. Physical control of soil organic matter dynamics in the tropics. Geoderma 79, 69–116.

Gee, G.W., Or, D., 1986. Particle size analysis. In: Klute, A. (Ed.), Methods of Soil Analysis: Part 1—Physical and Mineralogical Methods, Soil Science Society of America Book Series.

Ghadiri, H., Rose, C.W., 1991. Sorbed chemical-transport in overland-flow: 2. Enrichment ratio variation with erosion processes. Journal of Environmental Quality 20, 628–633.

Giovanni, G., Vallejo, R., Lucchesi, S., Bautista, S., Ciompi, S., Llovet, J., 2001. Effects of land use and eventual fire on soil erodibility in dry Mediterranean conditions. Forest Ecology and Management 147, 15–23.

Glaser, B., Balashov, E., Haumaier, L., Guggenberger, G., Zech, W., 2000. Black carbon in density fractions of anthropogenic soils of the Brazil Amazon region. Organic Geochemistry 31, 669–678.

Glaser, B., Lehmann, J., Zech, W., 2002. Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal—a review. Biology and Fertility of Soils 35, 219–230.

Gregorich, E.G., Geer, K.J., Anderson, D.W., Liang, B.C., 1998. Carbon distribution and losses: erosion and deposition effects. Soil and Tillage Research 47, 291–302.

Holtvoeth, J., Wagner, T., Schubert, C.J., 2003. Organic matter in river-influenced continental margin sediments: The land–ocean and climate linkage at the Late Quaternary Congo fan (ODP site 1075). Geochemistry, Geophysics, Geosystems 4 (Art. No. 1109).

Jacinthe, P.A., Lal, R., 2001. A mass balance approach to assess carbon dioxide evolution during erosional events. Land Degradation and Devastation 12, 329–339.

Janeau, J.L., Bricquet, J.P., Planchon, O., Valentin, C., 2003. Soil crusting and infiltration on steep slopes in northern Thailand. European Journal of Soil Science 54 (3), 543–554.

Jetten, V., deRoo, A., Favis-Mortlock, D., 1999. Evaluation of field-scale and catchment scale erosion models. Catena 37, 521–541.

Kuhlbusch, T.A.J., Crutzen, P.J., 1995. Towards a global estimate of black carbon on residues of vegetation fires representing a sink of atmospheric CO_2 and a source of O_2. Global Biogeochemical Cycles 9, 491–501.

Lal, R., 2003. Soil erosion and the global carbon budget. Environment International 29, 437–450.

Leguedois, S., Planchon, O., Legout, C., 2004. Splash projection distance in aggregated soils: consequences on the theory of rainsplash. Soil Science Society of America Journal.

Liu, S., Bliss, N., Sunquist, E., Hungington, T.G., 2003. Modelling carbon dynamics in vegetation and soil under the impact of soil erosion and deposition. Global Biogeochemical Cycles 17, 1074–1083.

Masiello, C.A., Duffel, E.R., 1998. Black carbon in deep sea sediments. Science 280, 1911–1913.

McDonald, M.A., Healey, J.R., Stevens, P.A., 2002. The effects of secondary forest clearance and subsequent land-use on erosion losses and soil properties in the Blue Mountains of Jamaica. Agriculture, Ecosystems and Environment 92, 1–19.

Mitra, S., Bianchi, T.S., McKee, B.Q., Stutula, M., 2002. Black carbon from the Mississippi River: quantities, sources and potential implications for the global C cycle. Environmental Science and Technology 36, 2296–2302.

Mulleneers, H.A.E., Koopal, L.K., Swinkels, G.C.C., Bruning, H., Rulkens, W.H., 1999. Flotation of soot particles from a sandy soil sludge. Colloids and Surfaces. A, Physicochemical and Engineering Aspects 151, 93–301.

Nearing, M.A., 1998. Why soil erosion models over-predict small soil losses and under predict large soil losses. Catena 32, 15–22.

Oldeman, L.R., Hakkeling, R.T.A., Sombroek, W. G., 1990. World Map of the Status of Human-induced Soil Degradation: An Explanatory Note. 27 pp.

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

Planchon, O., Esteves, M., Silvera, N., Lapetite, J.M., 2000. Raindrop erosion of tillage induced microrelief: possible use of the diffusion equation. Soil and Tillage Research 56, 131–144.

Planchon, O., Esteves, M., Silvera, N., Lapetite, J.M., 2002. Microrelief induced by tillage: measurement and modelling of surface storage capacity. Catena 46, 141–157.

de Ploey, J., Savat, J., 1968. Contribution à l'étude de l'érosion par le splash. Zeitschrift für Goemorphologie 12, 174–193.

Polyakov, V., Lal, R., 2004. Modeling soil organic matter dynamics as affected by soil water erosion. Environment International 30, 547–556.

Rodriguez Rodriguez, A., Guerra, A., Arbelo, C., Mora, J.L., Gorrin, S.P., Armas, C., 2004. Forms of eroded soil organic carbon in andosols of the Canary Islands (Spain). Geoderma 121, 205–219.

Ritchie, J.C., McCarty, G.W., 2003. ^{137}Cesium and soil carbon in a small agricultural watershed. Soil and Tillage Research 69, 45–51.

de Rouw, A., Baranger, P., Soulilad, B., 2003. Upland rice and Jobs' tear cultivation in slash and burn systems under very short fallow periods in Luang Prabang Province. The Lao Journal of Agriculture and Forestry 5, 1–10.

Rumpel, C., Alexis, M., Chabbi, A., Chaplot, V., Rasse, D.P., Valentin, C. Mariotti, A., in press. Black carbon contribution to soil organic matter composition in tropical sloping land under slash and burn agriculture, Geoderma.

Schmidt, M.W.I., Noack, A.G., 2000. Black carbon in soils and sediments: analysis, distribution, implications, and current challenges. Global Biogeochemical Cycles 14, 777–793.

Schmidt, M.W.I., Knicker, H., Hatcher, P.G., Kögel-Knabner, I., 1997. Improvement of ^{13}C and ^{15}CPMAS NMR spectra of bulk soils, particle size fractions and organic material by treatment with hydrofluoric acid (10%). European Journal of Soil Science 48, 319–328.

Schmidt, M.W.I., Skjemstad, J.O., Jäger, C., 2002. Carbon isotope geochemistry and nanomorphology of soil black carbon: black charnozemic soils in central Europe originate from ancient biomass burning. Global Biogeochemical Cycles 16, 1123–1131.

Scholes, M.C., Powlson, D., Tian, G., 1997. Input control of organic matter dynamics. Geoderma 79, 25–47.

Seiler, W., Crutzen, P.J., 1980. Estimates of gross and net fluxes of carbon between the biosphere and the atmosphere from biomass burning. Climate Change 2, 207–247.

Shindo, H., 1992. Elementary composition, humus composition, and decomposition in soil of charred grassland plants. Soil Science and Plant Nutrition 37, 651–657.

Shneour, E., 1966. Oxidation of graphite carbon in certain soils. Science 155, 991–992.

Slater, C.S., Carleton, E.A., 1938. The effect of soil erosion on losses of soil organic matter. Soil Science Society of America Proceedings 3, 123–128.

Soil Survey Staff, 1999. Soil taxonomy. A Basic System of Soil Classification for Making and Interpreting Soil Surveys. (2nd ed.)USDA, Washington D.C.

Sollins, P., Homann, P., Caldwell, B.A., 1996. Stabilization and destabilization of soil organic matter: mechanisms and controls. Geoderma 74, 65–105.

Sombroek, W.G., Nachtergaele, F.O., Hebel, A., 1993. Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. Ambio 22, 417–426.

Soto, B., Basanta, R., Perez, R., Diaz-Fierros, F., 1995. An experimental study of the influence of traditional slash-and-burn practices on soil erosion. Catena 24, 13–23.

Turkelboom, F., Poesen, J., Ohler, I., van Keer, K., Ongprasert, S., Vlassak, K., 1997. Assessment of tillage erosion rates on steep slopes in northern Thailand. Catena 29, 29–44.

Valentin, C., Rajot, J.L., Mitja, D., 2004. Responses of soil crusting, runoff and erosion to fallowing in the sub-humid and semi-arid regions of West-Africa. Agriculture, Ecosystems and Environment 104, 287–302.

Van Dijk, A.I.J.M., Bruijnzeel, L.A., Wiegman, S.E., 2003. Measurements of rain splash on bench terraces in a humid tropical steepland environment. Hydrological Processes 17, 513–535.

Webber, R.L., 1964. Soil physical properties and erosion control. Journal of Soil and Water Conservation 19, 28–30.