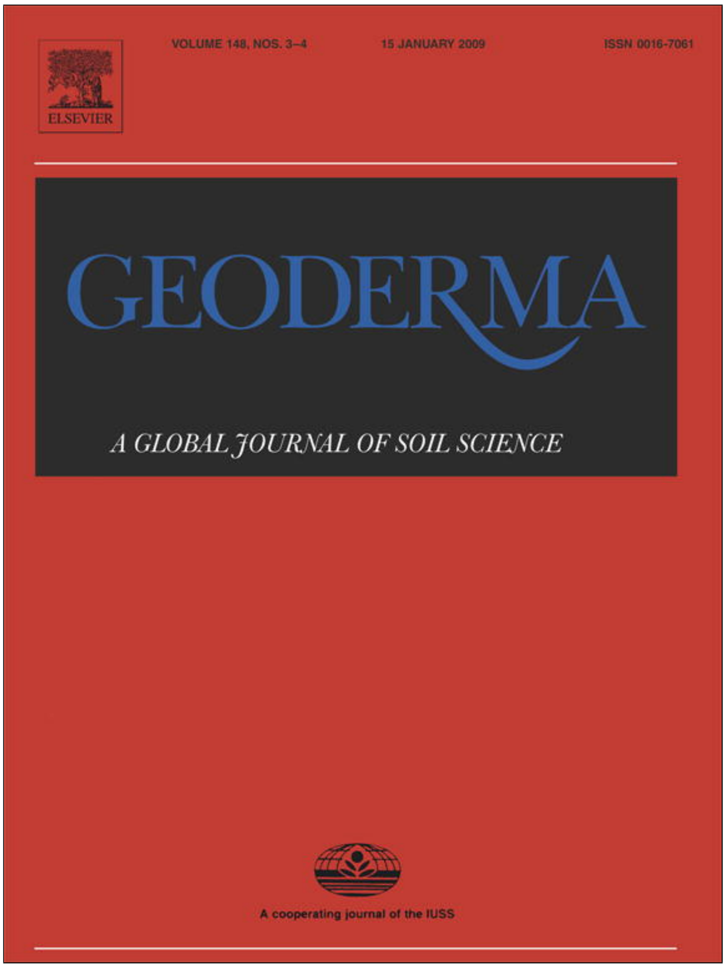

VOLUME 148, NOS. 3–4 15 JANUARY 2009 ISSN 0016-7061

ELSEVIER

GEODERMA

A GLOBAL JOURNAL OF SOIL SCIENCE

A cooperating journal of the IUSS

Geoderma 148 (2009) 405–412

Contents lists available at ScienceDirect

Geoderma

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

Application of stable isotope analysis to quantify the retention of eroded carbon in grass filters at the North Appalachian experimental watersheds

P.A. Jacinthe [a,]*, R. Lal [b], L.B. Owens [c]

[a] *Department of Earth Sciences, Indiana University Purdue University, Indianapolis, IN, USA*
[b] *Carbon Management and Sequestration Center, The Ohio State University, Columbus, OH, USA*
[c] *USDA-ARS, North Appalachian Experimental Watersheds, Coshocton, OH, USA*

ARTICLE INFO

Article history:
Received 4 July 2008
Received in revised form 18 October 2008
Accepted 11 November 2008
Available online 9 December 2008

Keywords:
Soil erosion
13C isotope
Grass filters
Corn-derived C
Terrestrial deposits

ABSTRACT

The entrapment of eroded soil organic carbon (SOC) in grass filters could affect watershed C export, but the magnitude of the process is rarely quantified. In order to assess the retention of eroded C in these settings, SOC stock was measured in grass buffers receiving runoff from cropped watersheds under long-term (>20 y): chisel-till (CT) corn (*Zea mays*, L.)-soybean [*Glycine max*, (L.) Merr.] rotation, moldboard plowing (MP) continuous corn (CC), and no-till (NT) CC. Adjacent reference grasslands not affected by erosion were also sampled. In the CC watersheds, the $\delta^{13}C$ of bulk soil and soil separates was determined to identify the source of SOC in the grass filters. After accounting for differences in the number of corn crops, SOC stock in the MP watershed showed a corn-C deficit of 9.3 Mg C ha^{-1} compared to NT. Corn-C accounted for 2 to 16% (mean: 5.2 Mg C ha^{-1}) of the total SOC pool in the grass filter and, assuming water erosion as the main determinant of C distribution, this corn-C gain translates into the retention of 55% of eroded C in the grass strip. Relative to the reference grasslands, SOC stock in the grass filters was up to 30 Mg C ha^{-1} higher, an amount deemed too large to be attributed solely to retention of eroded C (export rate: 0.05-0.08 Mg C ha^{-1} y^{-1}). Periodic delivery of nutrients may have enhanced biomass production and indirectly contributed to the observed SOC accrual in grass filters. Higher extractable P and higher C:N ratios at these locations support that hypothesis. These results demonstrate the applicability of ^{13}C isotope to trace SOC sources in buffers receiving runoff from areas supporting C_4 vegetation. They also underscore the need to incorporate *in-situ* biomass production and burial processes in assessing the temporal evolution of SOC stocks in terrestrial deposits and the contribution of these landscape segments to watershed C budget.

1. Introduction

Water erosion affects the redistribution and dynamics of soil organic carbon (SOC) in terrestrial landscapes. Worldwide, an estimated 1.11×10^9 ha of land are affected to varying degree by water erosion (Oldeman, 1994) and, in anticipation of more frequent erosive storms in a warmer climate (Phillips et al., 1996; Favis-Mortlock and Guerra, 1999), the impact of water erosion on the global C cycle would likely increase. Highlighting the significance of soil erosion on C budget, Izaurralde et al. (2007) noted that C exported in runoff events compares in magnitude to C sequestration achievable with adoption of no-till (NT) practices. Several fates of eroded C in the environment have been identified including mineralization during transport and entrapment in depositional environments (Jacinthe and Lal, 2001). Although attempts have been made to quantify these various pathways (Stallard, 1998; Jacinthe and Lal, 2001; Van Oost et al., 2007), published estimates are not well constrained and major uncertainties remain with regard to the net impact of water erosion on the C budget of terrestrial ecosystems.

Water erosion involves the breakdown of soil aggregates, the transport of detached materials and deposition of suspended soil particles. During a storm event, eroded materials can be transported off-site in runoff, but could also be subject to several deposition-resuspension cycles (Hairsine et al., 1999) whereby mobilized materials are deposited near the point of detachment. The significance of each of these mechanisms varies with rainfall characteristics, surface roughness and geomorphic settings. The amount of eroded materials transported out of a drainage basin is the balance between soil detachment and its retention in upland storage zones. It has been estimated that most (70-85%) eroded material remains near the point of detachment (Walling, 1983). Ritchie et al. (2007) combined ^{137}Cs concentration data and krigging techniques to delineate erosion and deposition zones in farmlands, and observed strong correlation between landforms and SOC distribution. In western Ohio, Jacinthe et al. (2001) identified depressional areas downslope of intensively-managed croplands as entrapment zones for eroded C. In addition to being more recalcitrant, SOC pool (0–45 cm depth) in these depositional zones was 1.7 and 2.7 fold higher than in adjacent forest

---

* Corresponding author. Department of Earth Sciences, Indiana University Purdue University Indianapolis (IUPUI), 723 W. Michigan Street, SL 122, Indianapolis, IN 46202, USA. Tel.: +1 317 274 7969; fax: +1 317 274 7966.
E-mail address: pjacinth@iupui.edu (P.A. Jacinthe).

0016-7061/$ – see front matter © 2008 Elsevier B.V. All rights reserved.
doi:10.1016/j.geoderma.2008.11.013

and cropland, respectively. This had led to the conclusion that terrestrial depressions are favorable for the sequestration of eroded C originating from cultivated uplands (Jacinthe et al., 2001). However, since pedogenetic and other landscape processes could also contribute to the observed C accumulation, it has been difficult to draw definitive conclusions regarding the source of organic C sequestered in these terrestrial entrapment zones.

Grass filters (GF) have demonstrated capacity to reduce runoff flow energy and off-site export of eroded materials (Le Bissonnais et al., 2004). Depending on runoff volume, surface conditions and landscape attributes, several studies have shown that as much as 80% of the suspended sediment load in runoff can be removed within the first 15 m of grass buffers (Le Bissonnais et al., 2004; McKergow et al., 2004; White et al., 2007). Thus, by facilitating the deposition of eroded materials, grass filters can greatly reduce the export of both nutrients and organic C from terrestrial to aquatic ecosystems. Data from Le Bissonnais et al. (2004) also indicated that, grass filters can be as selective as water erosion itself, and tend to predominantly retain coarse silt and sand-size materials. White et al. (2007) also reported a lower retention of silt/clay-size particles in forested filters and proposed that a filter width of at least 16 m is required for effective trapping of suspended materials in that size fraction. While the impact of grass strips on sediment trapping and retention of nutrients such as phosphorus (P) has been investigated (Le Bissonnais et al., 2004; McKergow et al., 2004; Owens et al., 2007; White et al., 2007), their impact on C retention is less well understood.

The present investigation was conducted at the North Appalachian Experimental Watershed (NAEW). The landscape settings and the well-documented land management history at the site made it possible to quantitatively assess the retention of eroded C in grass filters adjacent to cultivated fields. Available at this experimental station are several well delineated (constructed berms and natural boundaries) watersheds from which runoff is directed to instrumented outlets (in some instances) and then flows through well-defined grassy areas. One of these watersheds (included in this study) was under continuous corn (*Zea mays* L.) and conventional tillage for 20 years. Runoff from the cultivated upland portion of the watershed flows through a downslope grass strip before entering a nearby ditch. This unique aspect of the site allows application of stable C isotopic technique to quantify the retention of eroded C in the grass filter. To our knowledge, there are few studies in which ^{13}C isotope was employed as a soil erosion tracer. Fox and Papanicolaou (2007) relied on the $\delta^{13}C$ difference between upland (mean $\delta^{13}C$: $-26.34 \pm 0.09‰$) and floodplain soils (mean $\delta^{13}C$: -27.14 ± 0.07 ‰) to determine the relative contribution of these landscape elements to sediment loadings during various rainstorm events. De Gryze et al. (2008) attempted the application of ^{13}C isotopes to trace the distribution of eroded C among landform elements in fields under continuous corn, but the short duration (<4 y) of their study made it difficult to separate corn-derived C against the large background of native C.

Affected for at least two decades by runoff from a continuous corn (CC) field, soil organic C (SOC) in the grass filter was thus assumed to be a mixture of C from the current C_3 grass vegetation and corn-derived C fractions (C_4 vegetation) deposited onto the grass filter during runoff events. It is well established that, by virtue of the difference in photosynthetic pathways, tissues of C_3 ($\delta^{13}C$: −35 to − 20‰) and C_4 ($\delta^{13}C$: −19 to −9‰) vegetation have distinct ^{13}C isotopic composition (Farquhar et al., 1989). This difference in ^{13}C signature has extensively been used in assessing C storage and dynamics at sites where shifts in C_3–C_4 vegetation can be documented (Balesdent and Balabane, 1996; Bernoux et al., 1998; Puget et al., 2005). Therefore, in grass buffer strips receiving runoff from sites supporting C_4 vegetation for several years, variation in the ^{13}C signature of SOC may make it possible to quantitatively trace the origin of eroded C entrapped in these settings.

The objective of the present study was to determine the amount and source of eroded C retained in grass filters. Stable C isotope technique was used to document the source of eroded C retained in the grass filters. This application of the stable C isotope approach has not been attempted previously.

2. Material and methods

2.1. Description of the study site

The sampling sites are located at the North Appalachian Experimental Watershed near the town of Coshocton in Northeast Ohio (40° 22′ N, 81° 48′ W, elevation: 300–600 m). At this research station, established in 1938 by the US Department of Agriculture (USDA), watershed-size cultivated fields have been used in long-term studies evaluating the impact of crop rotation and tillage practices on soil erosion and water quality. The landscape is representative of the unglaciated rolling uplands of west-central Appalachia. Soils are generally well drained silt-loam developed from inter-bedded coarse-grained sandstone, shale and limestone. The dominant soil series in the sampled watersheds include Rayne (fine-loamy, mixed, mesic Typic Hapludults) and Keene (fine-silty, mixed, mesic Aquic Hapludalfs). Long-term mean annual temperature is 10.4 °C and rainfall is 950 mm. October is the driest, and June and July are the wettest months of the year (Kelley et al., 1975).

Three watersheds (WS) were included in the present study: (i) chisel till (CT, WS123) under corn-soybean [*Glycine max* (L.) Merr.]/rye (*Secale cereale* L.) rotation since 1984 (rye used as a winter cover crop following soybean), (ii) moldboard plow (MP, WS128) under continuous corn since 1984, and (iii) no-till (NT, WS188) under continuous corn since 1970. Historical land-use record (Puget et al., 2005; Izaurralde et al., 2007) showed that WS128 supported 12 corn crops during the period 1938–1983. Between 1938 and 1970, corn was also grown for 8 years in WS188. Thus, at the time of sampling for the present study in 2004, both WS128 and WS188 had 32 and 42 corn-years, respectively.

The CT watershed was plowed to a depth of 25 cm with a chisel each spring. The MP watershed was moldboard plowed each year to a depth of 25 cm, then disked and harrowed before planting. Cropped fields received fertilizer at an average rate of 150 kg N ha^{-1} y^{-1} during the corn year, and herbicides at recommended rates for weed control. Based on soil tests, P fertilizer was applied at average rates of 101.3 (range: 25.8–154.7), 72.6 (range: 47.1–101.9) and 84.1 (80.7–87.4) kg P ha^{-1} y^{-1} to WS123, WS128 and WS188, respectively. All tillage and planting operations were performed along slope contour. Additional information regarding land use history, tillage operations and general soil properties at the sites are available in previous publications (Kelley et al., 1975; Shipitalo and Edwards, 1998; Owens et al., 2002; Puget et al., 2005; Blanco-Canqui et al., 2005).

Runoff from the cultivated upland section of WS123 and WS128 was channeled through the grass filters located at the outlet of these watersheds. Dominant vegetation in the grass filters included orchard grass (*Dactylis glomerata*, L.) and bluegrass (*Proa pratensis*, L.). With implementation of NT management, runoff-generating events were rare in WS188. In addition to being not well-defined, the grassy vegetation near the bottom of this watershed was deemed not affected by water erosion and therefore was not sampled.

2.2. Soil sampling

Soil sampling was carried out in April–May 2004, and soil samples were taken both in the cultivated portion (up-, mid-, and down-slope positions) of each watershed and along triplicate transects in the grass filters downslope. Samples were collected at depths 0–2.5, 2.5–5, 5–10, 10–20 and 20–40 cm. Within each grass filter, samples were taken at 1, 3 and 6 m from the edge of the cultivated field. Adjacent to each grass filter, a grassy knoll in a summit position and apparently not affected by runoff was also sampled to serve as reference. Intact soil cores were also taken for determination of soil bulk density (ρ_b).

At the time of soil sampling, grass and corn residues were also collected. These materials were dried at room temperature, finely crushed, sieved and analyzed for C and ^{13}C. The average ^{13}C content of the grass and corn residues was −29.05 and −12.16‰, respectively.

Soil samples were air-dried, and a fraction of each sample (<2 mm) was crushed to pass through a 250 µm sieve. The finely-ground soil was used for determination of extractable inorganic P, total P and SOC. Most of the soil samples contained negligible amount of inorganic C (SIC) and, in the few samples where the presence of carbonates was detected (depth >20 cm), concentration was generally low (mean: 1.4 ± 0.2 g kg^{-1}). The SOC was calculated as the difference between total C and SIC.

Soil samples were tested for inorganic P using the Bray P method. Duplicate 1 g of soil was extracted with 10 mL of extracting solution (0.03 M NH_4F and 0.025 M HCl). The soil suspension was shaken for 5 min and centrifuged (3000 rpm) for another 5 min. The supernatant was filtered through a Whatman No. 42 filter paper and P concentration in the filtrate was determined colorimetrically by the malachite green (MG) method as described by D'Angelo et al. (2001). Absorbance was read at 630 nm using a VersaMax microplate reader (Molecular Devices, Sunnyvale, CA).

2.3. Physical fractionation of soils

Selected soil samples from WS 128 were included in the fractionation. The procedures described by Haile-Mariam et al. (2000) and Cambardella and Elliott (1992) were followed with some modifications. Air dried (5 g, 2 mm size) soil was dispersed in 20 mL of Na-polytungstate solution (specific gravity 1.85 g cm^{-3}, Geoliquids, IL). The suspension was shaken for a brief period (~5 min) and centrifuged for 15 min (18,000 *g*). The supernatant was passed through a 1.6 µm GF/A filter and the light fraction (LF) material was collected on the filter. While still on the filter, the LF was washed several times with deionized water to remove excess Na-polytungstate. Finally, the LF fraction was washed back into a beaker.

Another soil sub-sample (5 g, <250 µm) was dispersed with Na-hexametaphosphate (20 mL, 5 g L^{-1}) and the suspension shaken overnight on a reciprocal shaker. The suspension was passed through a 53 µm nylon filter (Gilson Company, OH) and thoroughly rinsed with deionized water. The material retained on the filter, representing particulate organic matter (POM) and the sand fraction (53–250 µm), was washed back into a centrifuge tube. Following centrifugation, any floating material was collected on a 1.6 µm GF/A filter and combined with LF fraction previously obtained. The soil slurry (silt+clay) that passed through the 53 µm sieve) was re-suspended in a 1 L glass cylinder and, after 3 h of decantation the supernatant (clay fraction, <2 µm) was siphoned off and transferred into a beaker. The silt fraction (2–53 µm) deposited at the bottom of the glass cylinder was washed back into another beaker.

All fractions were dried in a forced-air oven at 65 °C, finely ground and stored in glass vials for total C and ^{13}C analysis.

2.4. Assessment of total C pool and corn-derived C in grass filter

Carbon stock (Mg C ha^{-1}) in each soil layer was computed as:

$$SOC_{stock} = \frac{C}{10^3}\rho_b T \frac{10^4 m^2}{ha} \qquad (1)$$

where, C is carbon concentration (g C kg^{-1}), ρ_b is the bulk density (Mg m^{-3}) and T thickness (m) of the soil layer.

For a given soil layer, the amount of SOC attributable to corn vegetation (Cc) was determined using the isotope mixing equation (Bernoux et al., 1998)

$$C_c = C_t \frac{\delta_t - \delta_r}{\delta_c - \delta_r} \qquad (2)$$

where C_t and δ_t are SOC content and ^{13}C enrichment (‰), respectively at a given soil depth; δ_r: ^{13}C enrichment (‰) of the reference grassland soil at that depth, and δ_c: mean ^{13}C enrichment (−12.162 ‰) of corn residue. The ^{13}C at the grassland located in the grassy knoll (upper ridge) was used as the reference since this area was not affected by runoff.

2.5. Analytical methods and data analysis

Soil samples were analyzed for total C and N concentration by dry combustion (960 °C) using a Vario-Max C-N analyzer (Elementar Americas, NJ). The SIC was determined using a procedure involving acid-decomposition of carbonates (Loeppert and Suarez, 1996) in a sealed serum bottle (0.2 g of soil, 0.5 mL of 2 M HCl) and measurement of CO_2 evolved using an infra-red gas analyzer (PP Systems, Amesbury, MA). Vegetation and whole soil samples were analyzed for $^{13}C/^{12}C$ using isotope ratio mass spectrometry (IRMS) at the Stable Isotope Facility of the University of California, Davis (Model 2020 Europa mass spectrometer, PDZ Europa, Crewe, UK). Soil fractions were analyzed for $^{13}C/^{12}C$ using isotope ratio mass spectrometry (IRMS) at the Carbon Management and Sequestration Center, Ohio State University.

Because the sampling sites were not replicated, only descriptive statistics are used to present the data and compare SOC inventories at the various sampling units included in the study. Linear regression analysis was also performed to test relationships among parameters.

3. Results and discussion

3.1. Tillage practices and SOC pools in the cropped sections of the watersheds

Relative to the cultivated field, both SOC and total N concentrations in the reference grasslands were 1.6 fold higher (Tables 1 and 2). The $\delta^{13}C$ of soil organic matter (SOM) in the reference grassland ranges between −26.9‰ and −24.4‰ (Table 3), probably reflecting the ^{13}C signature of grass residues at the study site (mean $\delta^{13}C$: −29.05‰). In the reference grassland, the $\delta^{13}C$ becomes less negative with depth

Table 1
Organic carbon, C:N ratio, and extractable inorganic P (Bray P) in cropland, reference grassland and in grass filters receiving runoff from a cultivated field under chisel-till (CT) and corn-soybean rotation

Soil depth, cm	Reference grassland	Cultivated field	Distance within grass filter from field edge		
			1 m	3 m	6 m
Organic carbon, g C kg^{-1}					
0–2.5	20.3	13.7	34.2	31.4	31.4
2.5–5	19.6	15.2	33.7	30.8	32.7
5–10	11.5	12.9	27.7	29.7	25.6
10–20	9.1	9.4	19.7	18.6	13.2
20–40	8.6	7.8	12.8	11.8	8.3
Total nitrogen, g N kg^{-1}					
0–2.5	1.98	1.37	2.71	2.48	2.60
2.5–5	1.92	1.41	2.66	2.33	2.54
5–10	1.23	1.32	2.43	2.45	2.25
10–20	1.01	1.07	1.84	1.68	1.31
20–40	0.95	0.87	1.32	1.15	0.89
C:N ratio					
0–2.5	10.3	10	12.6	12.7	12.1
2.5–5	10.2	10.8	12.7	13.2	12.9
5–10	9.4	9.8	11.4	12.2	11.4
10–20	9	8.8	10.7	11.1	10.1
20 – 40	8.9	8.9	9.7	10.2	9.3
Bray P, mg P kg^{-1}					
0–2.5	6.4	16.7	23.9	24.4	24.4
2.5–5	1.6	20.1	25.1	25.5	24.2
5–10	1.3	18.4	21.9	25.2	23.4
10–20	1.6	13.7	21.3	5.9	16.3
20–40	0.2	7.4	18.8	16.1	0.7

Table 2
Organic carbon, C:N ratios and extractable inorganic P (Bray-P) in cropland, reference grassland and grass filters receiving runoff from a cultivated field under moldboard plow (MP) and continuous corn

Soil depth cm	Reference grassland	Cultivated field	Distance within grass filter from field edge		
			1 m	3 m	6 m
Organic carbon, g C kg^{-1}					
0–2.5	31.4	11.8	34.4	31.8	33.0
2.5–5	33.3	10.7	33.6	30.9	24.9
5–10	25.8	11.2	28.0	29.8	19.0
10–20	13.4	11.5	19.6	18.8	13.4
20–40	8.3	10.8	12.7	11.7	11.6
Total nitrogen, g N kg^{-1}					
0–2.5	3.09	1.01	2.73	2.53	2.62
2.5–5	2.40	0.97	2.67	2.36	2.6
5–10	1.88	1.02	2.48	2.47	2.29
10–20	1.36	1.03	1.87	1.7	1.33
20–40	1.20	1.02	1.34	1.18	0.9
C:N ratio					
0–2.5	10.7	11.7	12.6	12.6	11.9
2.5–5	10.4	10.9	12.6	13.1	12.8
5–10	10.1	11	11.3	12.1	11.3
10–20	9.8	11.1	10.5	11.1	10
20–40	9.6	10.6	9.5	10	9.2
Bray P, mg P kg^{-1}					
0–2.5	2.1	5.8	7.9	10.8	10.5
2.5–5	3.3	7.0	6.7	12.9	11.5
5–10	3	8.1	7.4	6.0	8.9
10–20	0.7	3.7	4.5	9.1	3.8
20–40	0.6	5.3	3.1	6.0	2.9

and, between surface and 40 cm depth, the δ^{13}C enrichment was in the order of 1.78‰.

Depending on tillage practice and the depth considered, SOC stock in the cultivated sections of the watersheds (Table 4) was either lower or similar to the level in the reference grasslands (Fig. 1). The average SOC stock (55.4 Mg C ha^{-1}, Table 4) recorded in the chisel-tilled cropland (WS123) was 14% lower than the 63.4 Mg C ha^{-1} (0–40 cm) measured in the reference grassland (Fig. 1). Likewise, SOC stock in the MP (WS 128) and NT (WS 188) cropland was 15 and 8% lower (Table 4) than in the reference grassland soil (72.5 Mg C ha^{-1}, Fig. 1). However, if the 0–20 cm depth is considered, SOC stocks in the NT watershed (WS188, Table 4) and the reference grassland (Fig. 1) were similar (46.1 vs 45.5 Mg C ha^{-1}). This trend confirms a widely reported observation that the effect on NT is generally limited to the topsoil layer.

When SOC stocks in the surface soil (0–20 cm depth) between the adjacent MP (WS 128) and NT (WS 188) watersheds are compared, the results also showed a difference in SOC equivalent to 11 Mg C ha^{-1} (Table 4). Past studies conducted at the study sites before (Fall 2001, Puget et al., 2005) and after (summer 2004; Blanco-Canqui et al., 2005) the present research have also shown that the NT watershed (WS188) contained between 9.1 (Puget et al., 2005) and 10.7 Mg C ha^{-1} (Blanco-Canqui et al., 2005) more C than the adjacent MP (WS128) watershed.

Table 4
Total soil organic carbon (SOC) pool and contribution of corn to SOC in croplands under different tillage practices

Depth range	SOC pool in cultivated fields as related to tillage practices		
	CT † (WS 123)	MP (WS 128)	NT (WS 188)
Total soil organic C pool, Mg C ha^{-1}			
0–20 cm	31.2 (0.5)	35.1 (0.5)	46.1 (3.1)
0–40 cm	55.4 (0.8)	61.1 (0.6)	66.6 (2.3)
Corn-derived soil organic C pool, Mg C ha^{-1}			
0–20 cm	nd‡	13.2 (0.2)	25.3 (0.9)
0–40 cm	nd	21.2 (0.2)	29.5 (0.5)

Values are means with standard deviations in parentheses.
†Abbreviations: CT = chisel till; MP = moldboard plow; NT = no-till.
‡nd = not determined.

In the NT watershed (WS188), corn residue contributed an average 69.8% (range: 54.6–78%) of the SOC stock in the top 10 cm soil layer (Table 3). The contribution of corn decreased to 20.6% in the 20–40 cm depth range. In the MP practice, the average corn contribution was 40% and 33% in the 0–10 and 10–20 cm depths, respectively. The amount of corn-derived C retained in the top 20 cm soil layer of WS188 (NT) and WS128 (MP) was 25.3 and 13.2 Mg C ha^{-1}, respectively (Table 4). Puget et al. (2005) reported slightly (1.2 times) greater corn-derived SOC pools for these treatments, most likely due to the more negative δ^{13}C signal in the forest soil used as reference in their study. Nonetheless, it is noteworthy that the difference (Table 4) in corn-derived SOC stock ($\Delta = 12.1$ Mg C ha^{-1}) between these two adjoining MP and NT watersheds was of similar magnitude as the difference in total SOC ($\Delta = 11$ Mg C ha^{-1}). This observation suggests that SOC loss mechanisms operating in the MP watershed primarily affect the recently-deposited corn-derived C pool. Water erosion is one of the mechanisms, and despite year to year variability, studies conducted at NAEW and elsewhere have shown measurable reduction in soil loss with implementation of NT. Other studies (Puget et al., 1995; Angers and Giroux, 1996) have also documented the predominant association of recently-deposited SOM fractions with soil macro-aggregates as well as the greater susceptibility of these large aggregates to the destructive action of raindrops. Since water erosion preferentially releases the more recent and more labile SOC fractions (Gregorich et al., 1996; Jacinthe et al., 2004; Zhang et al., 2006), one may thus propose that water erosion is as the likely mechanism contributing to the lower corn-derived SOC stock in WS128 (MP) than in WS188 (NT). A portion of the eroded C could remain entrapped in the grass filter located at the outlet of WS128.

3.2. Carbon retention in grass filters

Deposition of eroded C in the grass filters is clearly indicated by the much greater SOC stock recorded in these settings relative to adjacent croplands and reference grasslands (Tables 1, 2 and 4). In WS123, the average SOC stock (0–40 cm) in the grass filters (103.4 Mg C ha^{-1}, Fig. 1) was 1.6 times the level (64.3 Mg C ha^{-1}) in the reference

Table 3
Isotopic composition (^{13}C) and percent of soil organic carbon (SOC) derived from corn in moldboard plowed (MP), no-till (NT) croplands and in grass filters receiving runoff from a continuous corn field at the North Appalachian Experimental Watershed (NAEW)

Soil depth, cm	Reference grassland	Continuous corn croplands as related to tillage				Distance within grass filters from cultivated field edge					
		MP		NT		1 m		3 m		6 m	
	δ^{13}C (‰)	δ^{13}C (‰)	Corn-C (%)	δ^{13}C (‰)	Corn-C (%)	δ^{13}C (‰)	Corn-C (%)	δ^{13}C (‰)	Corn-C (%)	δ^{13}C (‰)	Corn-C (%)
0–2.5	−26.66 ± 0.27	−20.13 ± 0.26	44.7	−15.49 ± 0.18	76.9	−24.71 ± 0.31	13.9	−26.03 ± 0.26	4.4	−26.64 ± 0.28	<1
2.5–5	−26.42 ± 0.56	−20.66 ± 0.22	39.2	−15.24 ± 0.2	78	−24.72 ± 0.32	12.3	−25.40 ± 0.27	7.4	−26.30 ± 0.53	1
5–10	−25.85 ± 0.74	−20.68 ± 0.19	36.2	−18.23 ± 0.3	54.6	−24.34 ± 0.39	11.4	−25.13 ± 0.47	5.4	−25.68 ± 0.67	1.3
10–20	−25.25 ± 0.53	−20.29 ± 0.06	35.9	−21.02 ± 0.42	30.2	−24.28 ± 0.26	7.7	−24.48 ± 0.59	6.1	−24.72 ± 0.76	4.2
20–40	−24.88 ± 0.48	−20.75 ± 0.06	30.8	−22.01 ± 0.2	20.6	−23.86 ± 0.26	8.3	−23.95 ± 0.71	7.6	−24.26 ± 0.79	5

The reference grassland was located in a summit position, thus not affected by cropland runoff.

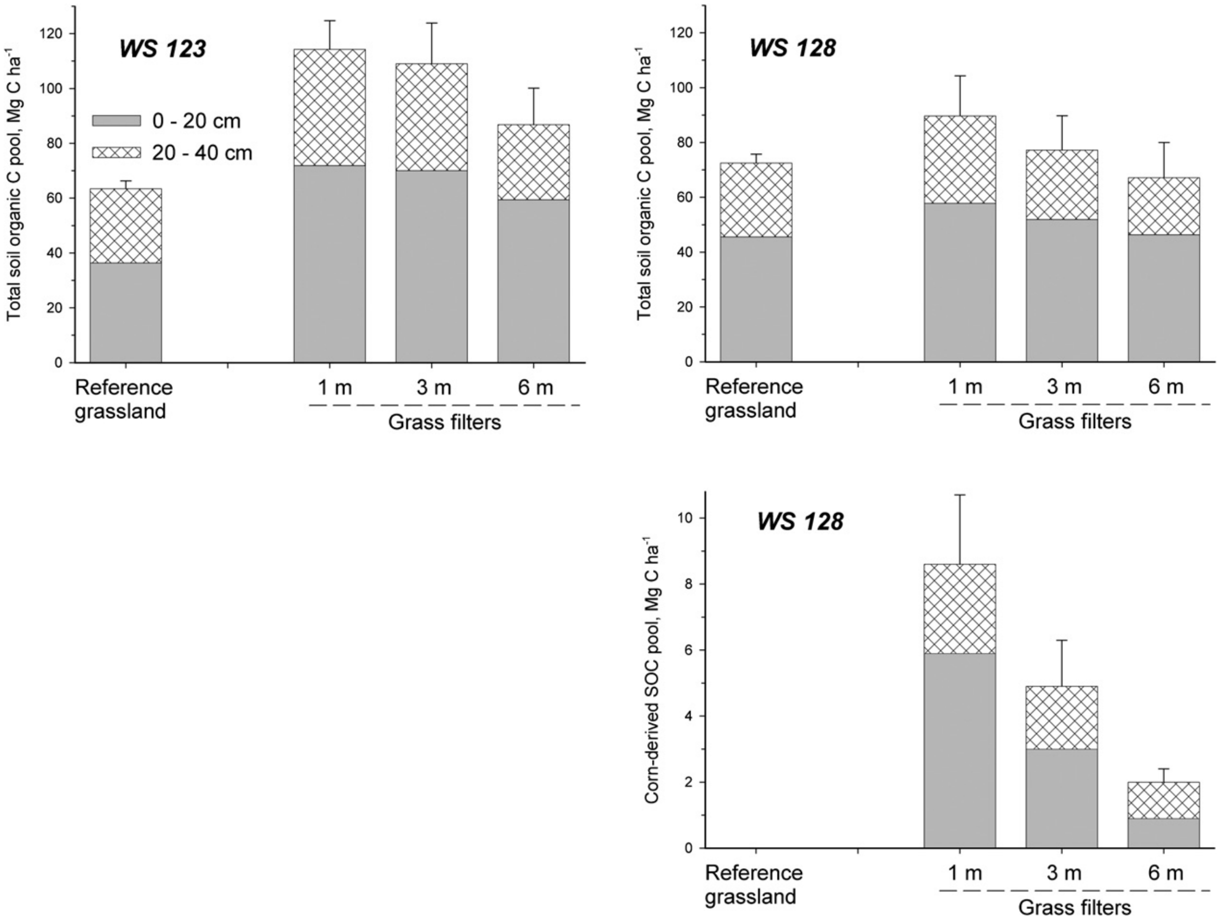

Fig. 1. Total soil organic carbon pool (upper graphs), and corn-derived SOC pool (bottom graph) in reference grassland and grass filters receiving run-off from cultivated sections of the watersheds. Tillage practices included chisel-till in WS 123 and conventional moldboard plowing in WS 128. The reference grasslands were not affected by runoff. The grass filters were sampled at distances 1, 3 and 6 m from the edge of the cultivated fields.

grassland. That translates into a gain of 39.1 Mg C ha^{-1} in the grass filter located downslope of WS123 (CT). Shipitalo and Edwards (1998) reported soil loss averaging 3.76 Mg soil ha^{-1} y^{-1} from WS123 during the period 1945–1966. In 2001–2002, annual soil loss and eroded C export from WS123 averaged 2.9 Mg soil ha^{-1} and 0.053 Mg C ha^{-1}, respectively (Jacinthe et al., 2004). Thus, under the current tillage practice, total C export during the last 20 years is likely to be in the 1.1 – 1.3 Mg C ha^{-1} range. Even if one assumes that the grass filter was 100% efficient in trapping eroded C, this total C export corresponds to only 4% of the 39.1 Mg C ha^{-1} C increment in the grass filter relative to the reference grassland. Therefore, SOC gain in the grass filter cannot be solely attributed to entrapment of eroded C.

Likewise, in WS128 (MP), SOC pool (0–40 cm) averaged 78 and 72.7 Mg C ha^{-1}, respectively in the grass filters and in the adjacent reference grassland (Fig. 1). Thus, the data indicate a net gain of approximately 5.3 Mg C ha^{-1} in the grass filter over the years. WS128 is not instrumented and therefore, C export measurements are not available. Nonetheless, using the EPIC (Erosion Productivity Impact Calculator) model and available soil C data for the site, Izaurralde et al. (2007) estimated an export of 2.96 Mg C ha^{-1} from this watershed during the last 35 years. These results showed that the C gain (5.3 Mg C ha^{-1} above the reference grassland) measured in the grass filters could represent 55% of potential C exported from the watershed. Thus, although a contributing factor, entrapment eroded C alone appears not adequate to explain SOC buildup in the grass filters.

As noted above, a smaller amount (difference of 12.1 Mg C ha^{-1}) of corn-derived C was measured in the MP than in the NT soil profile (0–20 cm depth). Puget et al. (2005) attributed this difference to less efficient incorporation of corn-C into the stable SOC pool under MP. After considering difference in C sequestration rate (0.28 Mg C ha^{-1} y^{-1}, Puget et al., 2005) between the two tillage practices, and the 10 corn-years difference between WS 128 (32 y) and WS 188 (42 y), SOC stock in NT still exceeded the level in MP by 9.3 Mg C ha^{-1} [=12.1 − (0.28 × 10)]. Thus, one may argue that export of C in runoff is an additional soil C loss pathway that needs to be taken into account. With this in mind, it is further hypothesized that if corn-derived C is retained in down-slope landscape positions, the amount of entrapped C can be quantified based on shifts in the ^{13}C isotopic signal of the soil organic matter.

Results (Tables 3 and 4) showed that corn-derived C accounted for 2 to 16% of the total SOC pool measured in the grass filters. The amount of corn-C in the grass strips declined sharply with distance from the cultivated field edge averaging 8.6, 4.9 and 2 Mg C ha^{-1} after 1, 3 and 6 m, respectively (Table 3). Thus, assuming that water erosion is the main process contributing to the 9.3 Mg corn-C ha^{-1} deficit in the MP cropland and taking the average corn-C in the grass filter (5.2 Mg corn-C ha^{-1}), present results suggest that approximately 55% of eroded C may have been retained in this 6 m wide grass filter.

Fractionation of soil samples (WS128 grassland and grass filter) was carried out in an effort to assess the contribution of POM and primary mineral particles to C retention in grass filters. Results showed that in the reference grassland, all the soil fractions were isotopically-depleted (lower δ^{13}C values) relative to the bulk soil (Fig. 2). In the grass filter, the pattern varied with the fraction considered with clay being consistently more enriched (above the 1:1 line). As reported by others (Solomon et al., 2002; Puget et al., 2005),

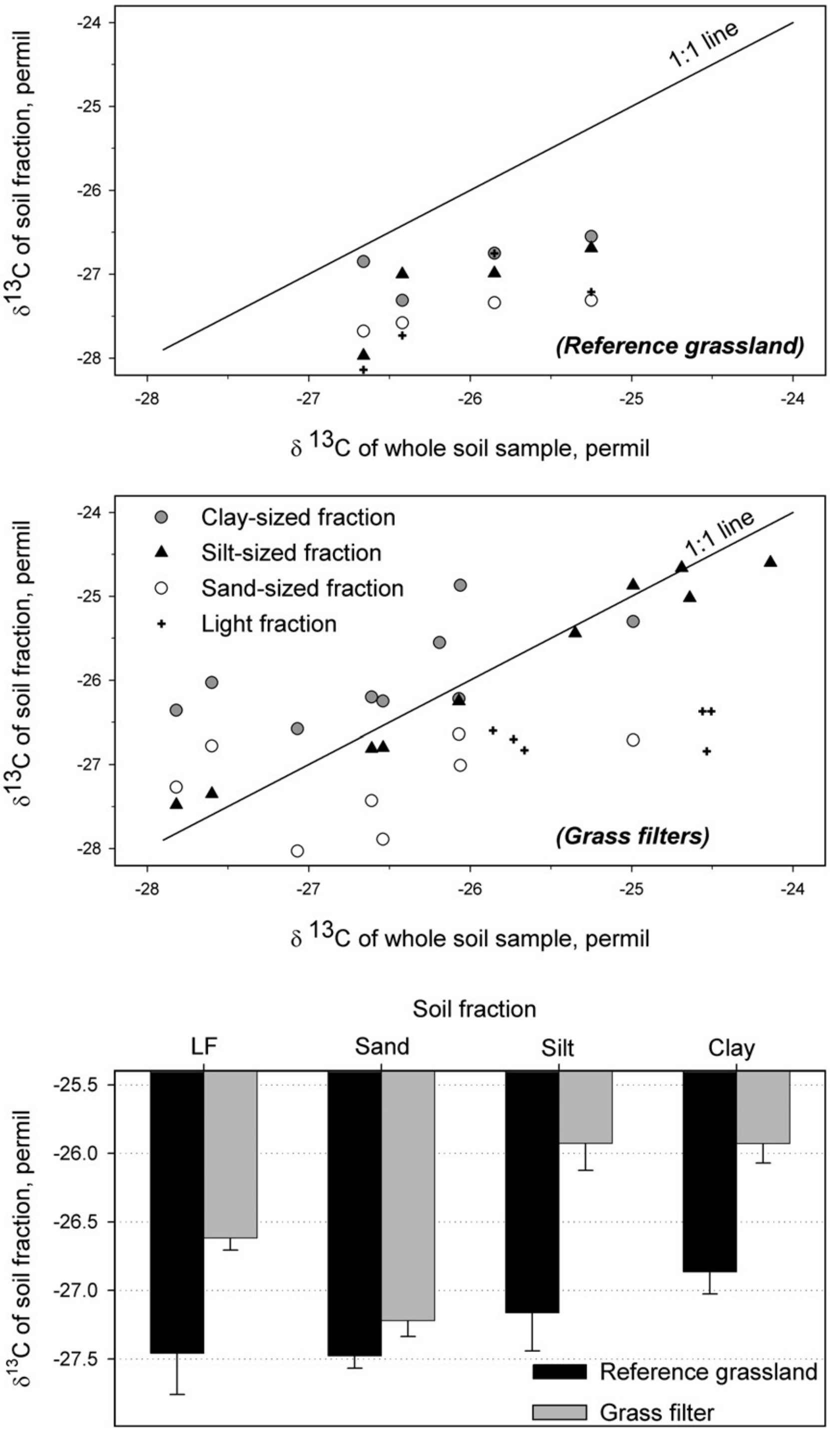

Fig. 2. Isotopic composition (δ^{13}C) of whole soil and soil separates in reference grassland (upper panel) and in grass filters (middle panel). The average δ^{13}C of organic matter in soil samples from the reference grassland and grass filters is shown in the lower panel.

there was an increase in δ^{13}C with decrease in particle size. This trend has generally been ascribed to more advanced decomposition of SOM associated with the finer soil particles. However, in the context of the present study, the trend observed with the soil fractions from the grass filter suggests a preferential association of eroded corn-C with silt- and clay-sized particles. The fact that the δ^{13}C values for these size fractions plotted over and above the 1:1 line with bulk SOC - which was not observed with the reference grassland samples - provides further support for that interpretation (Fig. 2). Based on difference in the δ^{13}C values of the fractions, assuming that the δ^{13}C signal in soil fractions from the grass filter is related to deposition of corn-derived C and comparing their δ^{13}C values with those of similar size fractions in the reference grassland, it was determined that corn-C accounted for 6, 2, 5 and 4.5% of the SOC in the light fraction, sand, silt and clay fractions, respectively. Thus, despite their light weight and greater propensity to be transported off-site, corn-derived C associated with clay and silt may have been retained in the grass filters. The retention of these colloidal-size particles in the grass filter likely involves reduction in flow velocity resulting in settling of particles and greater water infiltration. Le Bissonnais et al. (2004), White et al. (2007) and others have discussed the contribution of these physical processes to sediment trapping efficiency of grass buffers.

3.3. Eroded carbon retention versus in-situ production

The C:N ratio of SOM was consistently higher in the grass filters compared to either adjacent cultivated fields or reference grasslands (Tables 1 and 2). Other studies (Jacinthe et al., 2001; Yoo et al., 2005; Zhang et al., 2006) have reported similarly higher C/N ratios in depositional areas relative to surrounding landscape elements. In the 0–20 cm depth, Bray P concentrations were 3.8 to 8 fold higher in the grass filters than in the reference grasslands (Tables 1 and 2). Based on the vertical stratification of P, it is reasonable to ascribe this P enrichment to retention of sediment-associated P in the grass filters.

In light of past studies (Herbert and Fownes, 1995; Thiel-Egenter et al., 2007) that have documented a link between P availability and terrestrial ecosystem productivity, increased inorganic P may have enhanced soil fertility and biomass production in grass filters affected by periodic delivery of nutrients. Thus, the retention of sediment and nutrients could affect C dynamics in terrestrial deposits in two major ways. The retention of eroded C is the direct effect and most past studies (Beyer et al., 1993; Jacinthe et al., 2001) have examined the impact of terrestrial deposits on watershed C budget from that angle only. An indirect effect of sediment/nutrient entrapment in terrestrial deposits could be an increase in *in-situ* biomass production and consequently residue-C input, the main driver of C accrual in terrestrial ecosystems (Gregorich et al., 1998). A tenet of the “dynamic C replacement hypothesis” postulated in Stallard's (1998) paper is that soil erosion can induce formation of a local C sink as long as the eroded C is replaced by net primary production (Stallard, 1998; Harden et al., 1999). However, enhanced productivity at depositional sites was not explicitly considered in formulating that hypothesis. In a continental-scale C budget analysis, McCarty and Ritchie (2002) hypothesized that, during the massive soil loss affecting North America in the 1930's, the continent as a whole acted as a net C sink, and speculated that erosional transport of nutrients to riparian areas may have enhanced primary productivity and C sequestration in these zones, thereby contributing to the regional C sink. Along the same line, Yoo et al. (2005) also concluded that C sinks in eroding hillslopes are driven, not by preservation of eroded C, but by burial of autochtonously produced biomass. Our data also indicated that the indirect effect of erosion on primary production and C sequestration at depositional sites may be significant, and thus need to be better quantified and incorporated into watershed-scale C budgeting exercises.

4. Conclusions

Water erosion contributes to the distribution of SOC within terrestrial landscapes and its export to aquatic ecosystems during runoff events. Grass filter strips are erosion control measures with demonstrated effectiveness in reducing sediment and nutrient loading to surface waters. In this study, stable ^{13}C isotope technique was employed to quantify the retention of eroded C in grass filters, information that is needed to determine the impact of soil erosion on watershed C budget. ^{13}C enrichment of organic matter in a grass filter receiving runoff from a conventional tillage (MP) continuous corn field clearly showed a transfer of organic C from the upland cultivated field to the grass strip located downslope. Corn residue accounted for 2 to 16% of the total C pool in the grass filter. The amount of corn-derived C (mean: 5.2 Mg C ha^{-1}) in the grass filters corresponded to 55% of the corn-C pool determined to have been exported over the years from the MP cropland. Overall, SOC stocks in the grass filters were much larger (between 5.5 and 30 Mg C ha^{-1} above reference grassland) than in adjacent reference grasslands. Considering the high C/N ratios and the P enrichment observed in the grass filters, these elevated SOC pools could, besides eroded C entrapment, be related to enhanced productivity and burial of locally-produced biomass. In addition to demonstrating C retention and storage in grass buffers, these results highlight the complexity of C biogeochemistry in terrestrial deposits.

References

Angers, D.A., Giroux, M., 1996. Recently deposited organic matter in soil water-stable aggregates. Soil Sci. Soc. Am. J 60, 1547–1551.

Balesdent, J., Balabane, M., 1996. Major contribution of roots to soil carbon storage inferred from maize cultivated soils. Soil Biol. Biochem 28, 1261–1263.

Bernoux, M., Cerri, C.C., Neill, C., de Moraes, J.F.L., 1998. The use of stable carbon isotopes for estimating soil organic matter turnover rates. Geoderma 82, 43–58.

Beyer, L., Fründ, R., Schleuss, U., Wachendorf, C., 1993. Colluvisols under cultivation in Schleswig-Holstein. 2. Carbon distribution and soil organic matter composition. J. Plant Nutr. Soil Sci 156, 213–217.

Blanco-Canqui, H., Lal, R., Owens, L.B., Post, W.M., Izaurralde, R.C., 2005. Strength properties and organic carbon of soils in the North Appalachian region. Soil Sci. Soc. Am. J 69, 663–673.

Cambardella, C.A., Elliott, E.T., 1992. Particulate soil organic matter changes across a grassland cultivation sequence. Soil Sci. Soc. Am. J. 56, 777–783.

D'Angelo, E., Crutchfield, J., Vandiviere, M., 2001. Rapid, sensitive, microscale determination of phosphate in water and soil. J. Environ. Qual. 30, 2206–2209.

De Gryze, S., Six, J., Bossuyt, H., Van Oost, K., Merckx, R., 2008. The relationship between landform and the distribution of soil C, N and P under conventional and minimum tillage. Geoderma 144, 180–188.

Farquhar, G.D., Ehleringer, J.R., Hubick, K.T., 1989. Carbon isotope discrimination and photosynthesis. Annu. Rev. Plant Physiol. Plant Mol. Biol 40, 503–537.

Favis-Mortlock, D.T., Guerra, S.J.T., 1999. The implications of general circulation model estimates of rainfall for future erosion: a case study from Brazil. Catena 37, 329–354.

Fox, J.F., Papanicolaou, A.N., 2007. The use of carbon and nitrogen isotopes to study watershed erosion processes. J. Am. Water Resour. Assoc 43, 1047–1064.

Gregorich, E.G., Ellert, B.H., Drury, C.F., Liang, B.C., 1996. Fertilization effects on soil organic matter turnover and corn residue C storage. Soil Sci. Soc. Am. J 60, 472–476.

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

Haile-Mariam, S., Cheng, W., Johnson, D.W., Ball, J.T., Paul, E.A., 2000. Use of carbon-13 and carbon-14 to measure the effects of carbon dioxide and nitrogen fertilization on carbon dynamics in ponderosa pine. Soil Sci. Soc. Am. J 64, 1984–1993.

Hairsine, P.B., Sander, G.C., Rose, C.W., Parlange, J.Y., Hogarth, W.L., Lisle, I., Rouhipour, H., 1999. Unsteady soil erosion due to rainfall impact: a model of sediment sorting on the hillslope. J. Hydrol. 220, 115–128.

Harden, J.W., Sharpe, J.M., Parton, W.J., Ojima, D.S., Fries, T.L., Huntington, T.G., Dabney, S.M., 1999. Dynamic replacement and loss of soil carbon on eroding cropland. Global Biogeochem. Cycle 13, 885–901.

Herbert, D.A., Fownes, J.H., 1995. Phosphorus limitation of forest leaf area and net primary production on a highly weathered soil. Biogeochemistry 29, 223–235.

Izaurralde, R.C., Williams, J.R., Post, W.M., Thomson, A.M., McGill, W.B., Owens, L.B., Lal, R., 2007. Long-term modeling of soil C erosion and sequestration at the small watershed scale. Clim. Change 80, 73–90.

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

Jacinthe, P.A., Lal, R., Kimble, J., 2001. Organic carbon storage and dynamics in croplands and terrestrial deposits as influenced by subsurface tile-drainage. Soil Sci 166, 322–335.

Jacinthe, P.A., Lal, R., Owens, L.B., Hothem, D.L., 2004. Transport of labile carbon in runoff as affected by land use and rainfall characteristics. Soil Till. Res 77, 111–123.

Kelley, G.E., Edwards, W.M., Harold, L.L., McGuinness, J.L., 1975. Soils of the North Appalachian Experimental Watershed. USDA ARS Publ. 1296. United States Department of Agriculture, Washington, DC. 145 pp.

Le Bissonnais, Y., Lecomte, V., Cerdan, O., 2004. Grass strip effects on runoff and soil loss. Agronomie 24, 129–136.

Loeppert, R.H., Suarez, D.L., 1996. Carbonate and gypsum. In: Sparks, D.L. (Ed.), Methods of Soil Analysis, Part 3- Chemical Methods. Soil Science Society of America, Madison, WI., pp. 437–474.

McCarty, G.W., Ritchie, J.C., 2002. Impact of soil movement on carbon sequestration in agricultural ecosystems. Environ. Pollut 116, 423–430.

McKergow, L.A., Prosser, I.P., Grayson, R.B., Heiner, D., 2004. Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 2. Water quality. Aust. J. Soil Res. 42, 485–498.

Oldeman, L.R., 1994. The global extent of soil degradation. In: Greenland, D.J., Szabolcs, I. (Eds.), Soil Resilience and Sustainable Land Use. CAB International, Wallingford, UK., pp. 99–118.

Owens, L.B., Malone, R.W., Hothem, D.L., Starr, G.C., Lal, R., 2002. Sediment carbon concentration and transport from small watersheds under various conservation tillage practices. Soil Till. Res 67, 65–73.

Owens, P.N., Duzant, J.H., Dreks, L.K., Wood, G.A., Morgan, R.P.C., Collins, A.J., 2007. Evaluation of contrasting buffer features within an agricultural landscape for reducing sediment and sediment-associated phosphorus delivery to surface waters. Soil Use Manage 23, 165–175.

Phillips, D.L., Lee, J.J., Dodson, R.F., 1996. Sensitivity of the US corn belt to climate change and elevated CO_2: II. Soil erosion and organic carbon. Agr. Syst 52, 503–521.

Puget, P., Chenu, C., Balesdent, J., 1995. Total and young organic matter distributions in aggregates of silty cultivated soils. Eur. J. Soil Sci 46, 449–459.

Puget, P., Lal, R., Izaurralde, C., Post, M., Owens, L., 2005. Stock and distribution of total and corn-derived soil organic carbon in aggregate and primary particle fractions for different land use and soil management practices. Soil Sci. 170, 256–279.

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.

Shipitalo, M.J., Edwards, W.M., 1998. Runoff and erosion control with conservation tillage and reduced-input practices on cropped watersheds. Soil Till. Res 46, 1–12.

Solomon, D., Fritzsche, F., Lehmann, J., Tekalign, M., Zech, W., 2002. Soil organic matter dynamics in the sub-humid agroecosystems of the Ethiopian highlands: evidence from natural C-13 abundance and particle-size fractionation. Soil Sci. Soc. Am. J 66, 969–978.

Stallard, R.F., 1998. Terrestrial sedimentation and the carbon cycle: coupling weathering and erosion to carbon burial. Global Biogeochem. Cycle 12, 231–257.

Thiel-Egenter, C., Risch, A.C., Jurgensen, M.F., Page-Dumroese, D.S., Krüsi, B.O., Schütz, M., 2007. Response of a subalpine grassland to simulated grazing: above ground productivity along soil phosphorus gradients. Community Ecol 8, 111–117.

Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., Marques da Silva, J.R., Merck, R., 2007. The impact of agricultural soil erosion on the global carbon cycle. Science 318, 626–629.

Walling, D.E., 1983. The sediment delivery ratio problem. J. Hydrol 65, 209–237.

White, W.J., Morris, L.A., Pinho, A.R., Jackson, C.R., West, L.T., 2007. Sediment retention by forested filter strips in the Piedmont of Georgia. J. Soil Water Conserv. 62, 453–463.

Yoo, K., Amundson, R., Heimsath, A.M., Dietrich, W.E., 2005. Erosion of upland hillslope soil organic carbon: coupling field measurements with a sediment transport model. Global Biogeochem. Cycle 19, GB3003.

Zhang, J., Quine, T.A., Ni, S., Ge, F., 2006. Stocks and dynamics of SOC in relation to soil redistribution by water and tillage erosion. Global Change Biol 12, 1834–1841.