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Impact of soil movement on carbon sequestration in agricultural ecosystems

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"Capsule": *Increased sedimentation rates in a wetland ecosystem are associated with higher carbon sequestration rates.*

Abstract

Recent modeling studies indicate that soil erosion and terrestrial sedimentation may establish ecosystem disequilibria that promote carbon (C) sequestration within the biosphere. Movement of upland eroded soil into wetland systems with high net primary productivity may represent the greatest increase in storage capacity potential for C sequestration. The capacity of wetland systems to capture sediments and build up areas of deposition has been documented as well as the ability of these ecosystems to store substantial amounts of C. The purpose of our work was to assess rates of sediment deposition and C storage in a wetland site adjacent to a small first-order stream that drains an agricultural area. The soils of the wetland site consist of a histosol buried by sediments from the agricultural area. Samples of deposited sediments in the riparian zone were collected in 5 cm increments and the concentration of ^{137}Cs was used to determine the 1964 and 1954 deposition layers. Agricultural activity in the watershed has caused increased sediment deposition to the wetland. The recent upland sediment is highly enriched in organic matter indicating that large amounts of organic C have been sequestered within this zone of sediment deposition. Rates of sequestration are much higher than rates that have occurred over the pre-modern history of the wetland. These data indicate the increased sedimentation rates in the wetland ecosystem are associated with increased C sequestration rates. Published by Elsevier Science Ltd.

Keywords: Soil movement; Soil carbon; Terrestrial ecosystems; Wetlands; Sedimentation

1. Introduction

1.1. Global carbon budget and the terrestrial sink

From the period 1850 to 1998, it is estimated that 270 Gt C was emitted from industrial activity, with 176 Gt C accumulating in the atmosphere as CO_2 and 120 Gt C being assimilated by the ocean. Closure of this C budget requires a net terrestrial emission component of 26 Gt C (IPCC, 2000). In this same time period, it has been estimated that land use change by human activity has resulted in an emission of 136 Gt C. The gap between the estimated net terrestrial emission (26 Gt C) and land use change emissions (136 Gt C) has been termed the "missing C sink" (110 Gt C). In large part, the substantial size of this missing terrestrial C sink reflects our current lack of understanding of the terrestrial component. In this respect, it is noteworthy that the terrestrial component in most global C budgets is estimated primarily by the residuals after the other more easily better-characterized components have been evaluated.

Based on data from the previous decade (1990s), the Intergovernmental Panel on Climate Change (IPCC) complied a current annual global C budget as depicted in Fig. 1 (IPCC, 2000). Average annual emissions of CO_2 during this period were 6.3 Gt C year^{-1} from fossil fuel and cement production, accumulation in the atmosphere was 3.3 Gt C year^{-1} and net ocean uptake as 2.3 Gt C year^{-1}. To close this budget requires net terrestrial assimilation of 0.7 Gt C year^{-1}. It is noteworthy that the current global C budget estimates a net terrestrial uptake term for C, whereas the long term global C budget (1850–1998) estimate a net terrestrial emission for the period. This difference may be consistent with the output of two global C budget models that used different combinations of atmospheric, oceanic, and ice

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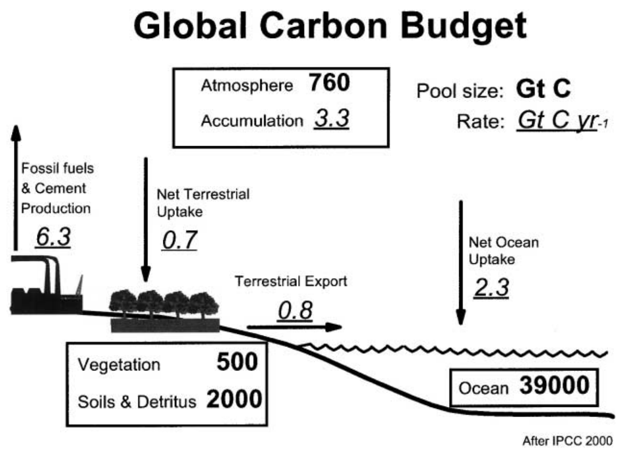

Fig. 1. Global C budget based on the 1990 decade (after IPCC, 2000).

core CO_2 data and have indicated that the dynamics of the sink/source capacity of the terrestrial ecosystem have substantially changed within the last century with a transition from a C source to sink occurring in the period 1930–1940 (Bruno and Joos, 1997; Joos and Bruno, 1998). One of the models indicates that the global terrestrial biosphere turned from being a net source of 0.5 Gt C year^{-1} into a net sink of 1.0 Gt C year^{-1} during this period and estimates that between 1950 and 1990 the terrestrial biosphere had a net assimilation of 43 Gt C (Joos and Bruno, 1998). Interestingly, a recent report (Fan et al., 1998) indicated that the North American continent south of 51° N is an important, if not the major, terrestrial sink in the global C budget.

The finding of a major transition from source to sink in the recent global terrestrial ecosystem C budget and the possibility of a major terrestrial C sink residing within the North American continent indicates the need for research on the possible dynamics within the continental ecosystem that can account for the recent development of major C sinks. Potential causes for development of the source/sink transition include CO_2 fertilization increasing the biomass accumulation of the biosphere due to increased atmospheric content, but the magnitude of the transition is considered too large for this mechanism (Joos and Bruno, 1998). Other possible causes include increased usage of fertilizer N in agriculture that may stimulate C sequestration in soil because of the need for N to stabilize soil C.

One noteworthy occurrence, however, during the period of apparent source to sink transition (1930s & 1940s) was massive soil erosion associated with agriculture, which resulted in major redistribution of surface soil resources within the continental ecosystem. It is noteworthy that the current IPCC global C budget (Fig. 1) contains a term for export of terrestrial C to the ocean which is approximately equal to the net terrestrial uptake indicating that the terrestrial ecosystem is at steady state relative to total C storage. This implicates soil erosion as an important process influencing terrestrial C cycling via terrestrial C movement and ultimate export to oceans.

1.2. Impacts of soil erosion on terrestrial carbon storage

Stallard (1998) estimated annual soil erosion to be 9.6 t ha^{-1} year^{-1} (3.4 t ha^{-1} year^{-1} water and 6.2 t ha^{-1} year^{-1} wind) with total soil erosion of about 7.5 Gt year^{-1} within the United States. Lal et al. (1998) estimated that erosion from cultivated cropland in the US has decreased from 1.9 Gt year^{-1} in 1982 to 0.6 Gt year^{-1} in 1995. This decrease in erosion was mostly the result of increases in crop land area under conservation tillage, conservation reserve programs, and afforestation programs. But even with these recent reductions, soil erosion remains the dominant force in redistributing soil C within agricultural landscapes.

A recent high profile debate over accuracy of the commonly used erosion models has brought attention to the concern about validity of current estimates of soil erosion in the United States (Trimble, 1999; Pimental et al., 1999; Nearing et al., 2000; Trimble and Crosson, 2000). This concern is based on an argument that current erosion models measure soil detachment and not the amount exported from the watershed. With this argument much of the eroded soil predicted by these models is redeposited within the watershed and as a result the estimated "soil loss" is largely accounted for by redistribution on landscapes (Trimble and Crosson, 2000).

The influence of soil erosion and redistribution on terrestrial C storage is poorly understood. Some assessments have indicated that soil erosion has an overall negative influence on C storage in the terrestrial ecosystem (Lal et al., 1998). The expectations for increased greenhouse gas emissions due to erosion are based on (1) the decrease in the capacity of eroded soils to produce biomass; (2) the breakdown of soil structure that exposes C locked in aggregates to oxidation; and (3) the mineralization of exposed C by microbial decomposition and other oxidative processes. Some have estimated that 20% of organic C associated with erosion is mineralized to CO_2 because of increased biodegradation with loss of soil structure (Lal, 1995). The magnitude of C loss from a land surface due to erosion can be substantial and the dominant loss mechanism. For example, Harden et al. (1999) found at a Mississippi site cropped for more than 100 years that nearly 100% of the original organic C in the agricultural soil had been lost. They attributed as much as 80% of the reduction in organic C to soil erosion.

Soil erosion results in redistribution of soil C across the landscape and into riparian zones, waterways, lakes, and oceans where it may be sequestered over geologic time. Ritchie (1988, 1989) provided an assessment of C storage in reservoirs and navigation pools of the Upper

Mississippi River within the United States and concluded that C sequestration in reservoirs alone may constitute a global C sink of 0.2–0.3 Gt year^{-1}. Others have estimated that globally 20 Gt year^{-1} of sediment is deposited in world oceans which translate into large exports of terrestrial C into ocean sediment with likely geologic storage. Sediment deposition in lakes, reservoirs, and ocean represents passive storage of C and such deposition is not expected to stimulate rates of net C sequestration per se.

Typically only 10% of eroded soil is exported from a watershed (Lal et al., 1998), and therefore, in large part the eroding soil represents redistribution of soil resources on the landscape. This redistribution of soil resources can lead to deep burial of terrestrial C (Lal et al., 1998) that is a mechanism for removing organic C from more reactive components of the biosphere and may constitute a largely passive form of C storage. However, consideration should also be given to influence of soil resource redistribution on active mechanisms of C sequestration within landscapes. An emerging concept is that soil resource redistribution within the terrestrial biosphere may establish states of ecosystem disequilibria that may promote C sequestration.

1.3. Potentiating terrestrial carbon sequestration by soil resource redistribution

Stallard (1998) developed a model framework for linking terrestrial sedimentation to the C cycle. The model was built on two core hypotheses: (1) significant amounts of C are buried during the deposition of sediments in terrestrial environments because of human acceleration of erosion and modification of hydrological and nutrient cycles; and (2) pedogenic C that is buried is replaced by newly fixed pedogenic C at sites of erosion or deposition. Elements of the model include (1) the consideration of enhanced mobilization of sediments and nutrients and the processes that limit their delivery to the ocean; (2) the consideration of whether newly fixed C is being added at the site of erosion or deposition and whether organic C is preserved with burial; and (3) the consideration of organic sedimentation in wetlands. With this model framework, Stallard developed a wide-ranging set of scenarios to evaluate the impacts of terrestrial sedimentation on the global C sequestration. The results ranged from 0 to 2.3 Gt C year^{-1} of C sequestration on a global basis, averaging 0.8 Gt C year^{-1} and with estimates for human-induced burial of 0.6–1.5 Gt C year^{-1} by terrestrial sedimentation using conservative assumptions. Stallard (1998) noted substantial uncertainty because of large information gaps concerning rates and locations of terrestrial sedimentation. This led to an inability to more rigorously test the hypothesis linking terrestrial sedimentation to the global C cycle and to assess magnitude of this sink. The information gap was largely attributed to the lack of good methods for tracking movement of soil redistribution, sedimentation, and C in terrestrial ecosystems.

Harden et al. (1999) tested elements of this hypothesis by measuring the potential for dynamic replacement of soil C in a highly erosive Mississippi loess soil. Using soil C, N, "bomb" ^{14}C, and measurements of CO_2 flux, in conjunction with the Century model and spreadsheet models, permitted characterization of C storage and dynamics for different erosion and tillage histories. This study found that about 100% of prehistory soil C had been lost from sites of erosion with more than 127 years of land use and subsequently that about 30% of the C at sites of erosion had been replaced since 1950. This provided strong evidence for the principle of dynamic replacement of soil C and indicates that soil erosion can induce formation of a local C sink. Harden et al. (1999) conclude, however, that the ultimate source/sink influence of soil erosion is highly dependant on fate of sediment C that includes degree and time length of burial and relative rates of organic C decomposition in the receiving ecosystem. This conclusion further emphasizes the need for improved methods to track the movement of eroded soils in terrestrial ecosystems and assess C dynamics within the ecosystem receiving eroded soils.

Based on work by Stallard (1998) and Harden et al. (1999), a conceptual relationship can be developed for the interaction between soil erosion, C sequestration and net primary productivity (NPP) at the site of erosion (Fig. 2). Within the concept, increasing soil erosion induces C sequestration by pedogenic disequilibria caused by soil C loss and this influence may at least partly be counteracted by loss of NPP. Low to moderate

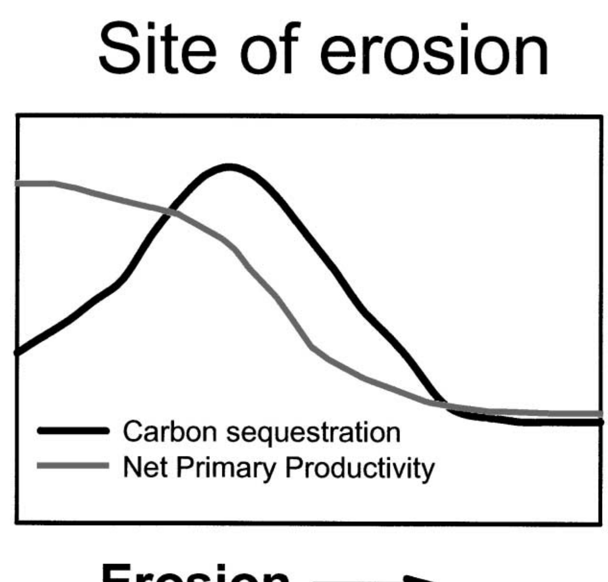

Fig. 2. Conceptual relationship between soil erosion, C sequestration and loss of net primary productivity at the site of erosion.

rates of soil erosion may stimulate C sequestration whereas severe erosion would be detrimental to C sequestration in the ecosystem because of the loss of NPP. This model addresses sites of erosion but does not account for C dynamics at zones of deposition within terrestrial ecosystems.

Watershed-scale studies are needed to more fully assess impact of soil resource movement on terrestrial C storage. We test the hypothesis that agricultural activities within a watershed can markedly increase rates of C storage within the riparian buffer by increasing sediment deposition and contributing nutrients to wetland ecosystems, which may increase primary productivity.

1.4. Characterization of study site and measurements

Research on a small gauged headwater watershed at BARC (Beltsville Agricultural Research Center) in Maryland provided a means for assessing agricultural influence at a watershed scale (Figs. 3 and 4). Characterization of the site includes an intensive topographic survey of the agricultural field and riparian ecosystem (Fig. 3). The agricultural field was sampled on a 25 m grid for assessing spatial distribution of soil C within the field (Fig. 5). Soil cores (15 cm dia.×35 cm) were taken in a series of transects across the agricultural field and the riparian ecosystem for measurement of the relationship between soil C and ^{137}Cs within the catchment (Ritchie and McHenry, 1990). Deeper profile samples (0–180 cm) were also obtained within the wetland to inventory C storage within the ecosystem. For a control site, soil samples were obtained from a pristine (undeveloped) first-order catchment located at the Patuxent Wildlife Research Center within 2.5 km of the agricultural study site. Soil samples were collected along transects within this catchment and they were analyzed for C and ^{137}Cs. Soil samples were analyzed for ^{137}Cs using a Canberra Genie-2000 spectroscopy system and analyzed for soil C by a Leco CNS 2000 dry combustion instrument. Radiocarbon (^{14}C) measurements were performed by Beta Analytic Inc. (Miami, FL).

2. Results and discussion

To explore the potential impact of terrestrial sedimentation on C sequestration consideration can be given to influence of soil resource redistribution in a typical agricultural watershed as may be represented by our agricultural study site, where soil C contents vary from roughly 1% on hilltops to 20% in the histosol bordering the stream (areal extent of histosol shown in Fig. 4). Within the agricultural field, C contents vary from roughly 1 to 2.5% (Fig. 5). Spatial patterns of soil C are highly related to topography and are usually functions of soil water relations. Typical erosion patterns will result in movement of soil down slope with deposition in concave and toe slope positions or further into the adjoining riparian wetland. This type of redistribution will generally result in depositions with C content below the pedogenic equilibrium for the zone of deposition. For example, sediment containing 1% C may become deposited on a down slope zone which supports 2.5% soil C. The pedogenic processes also promote formation of new organic C in zones of soil loss as observed by Harden et al. (1999) and likely will

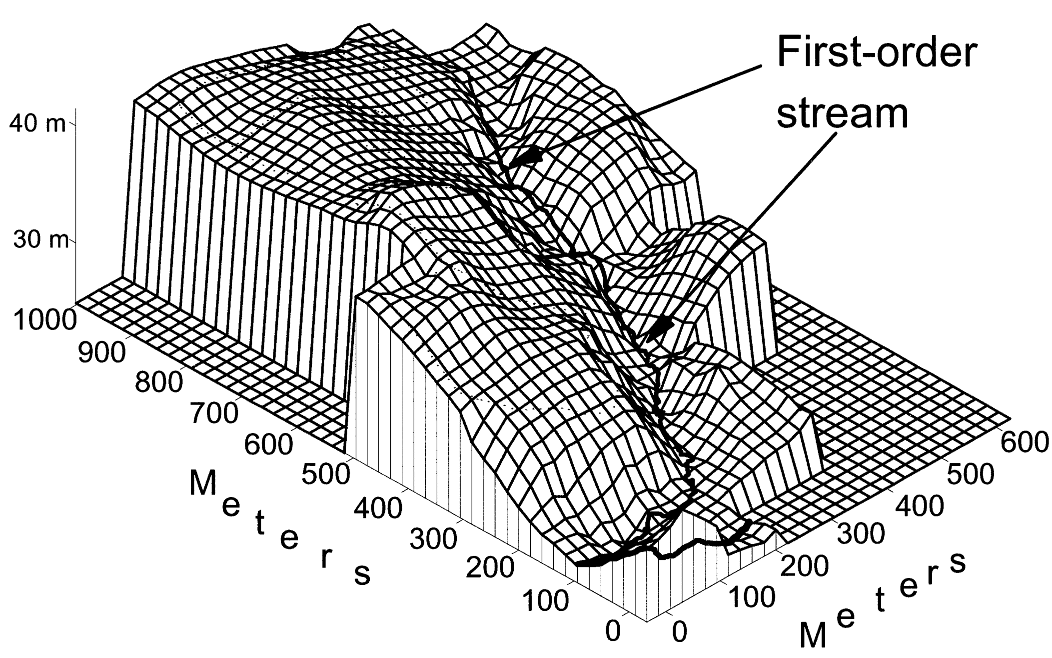

Fig. 3. Topographic surface of study area comprising an agricultural catchment containing the riparian system and associated first-order stream.

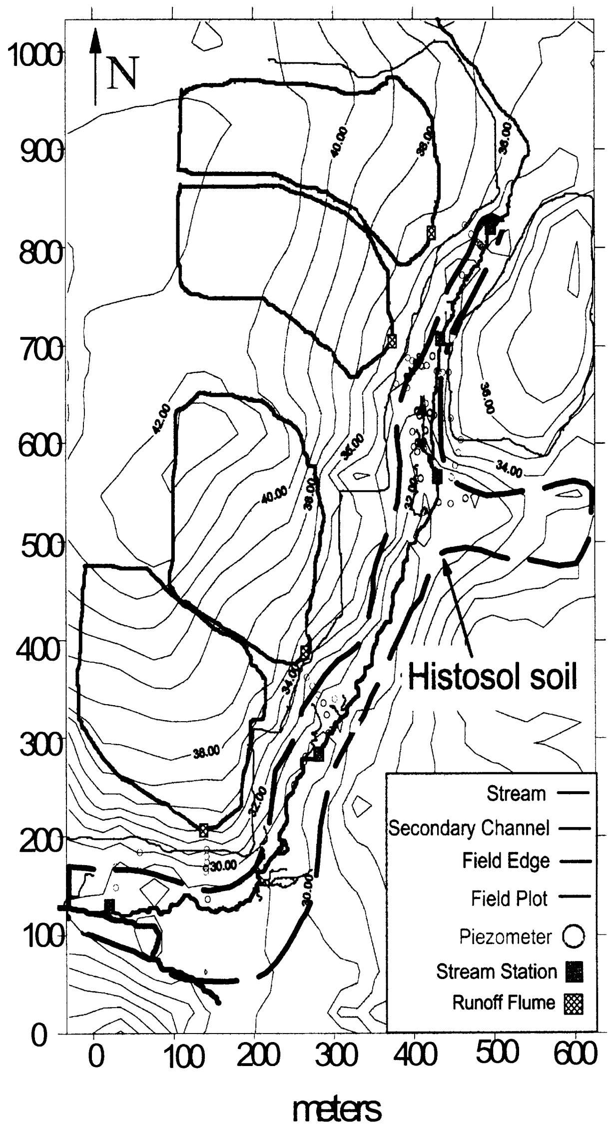

Fig. 4. Site plan of the agricultural catchment with the area containing histosol soil outlined with a dashed line.

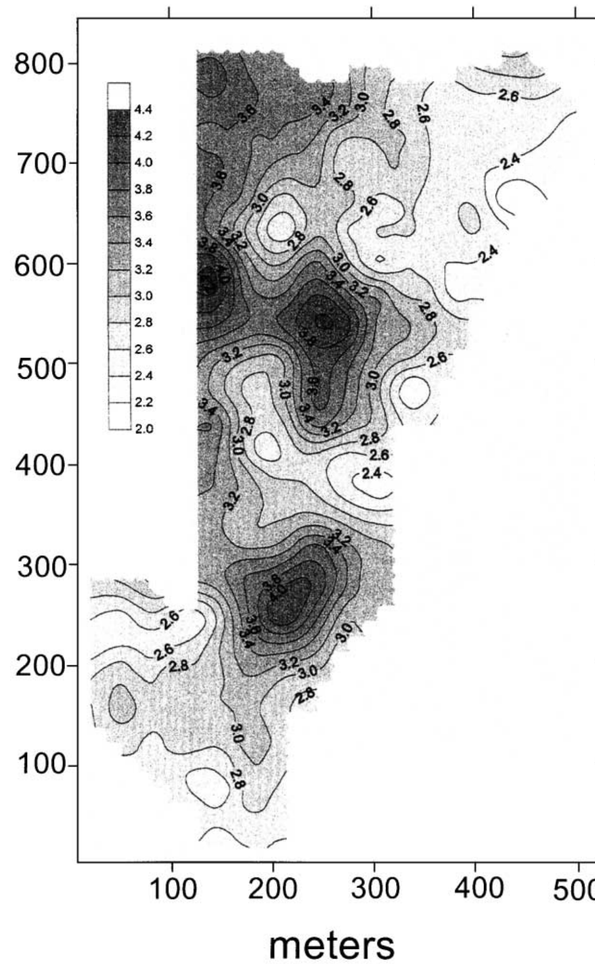

Fig. 5. Distribution of soil organic matter (%) on the agricultural field for the catchment under study.

also promote active sequestration within the zone of deposition. With the earlier scenario, the pedogenic disequilibria caused by soil movement results in much of the landscape becoming potentiated for C sequestration with greatest potential for influence in the riparian wetland. Characterizing the zone of deposition becomes important in assessing influence of terrestrial sedimentation on C storage. The influence of deposition zones on potentiating sequestration becomes greatest with movement of upland eroded soils into the wetland ecosystem that can have 20% soil C. With this influence, accounting for terrestrial sedimentation as simple C burial (i.e. passive storage) would greatly underestimate the influence of soil redistribution on terrestrial C storage.

Characterizing the zones of deposition become important in assessing influence of terrestrial sedimentation on C storage. For example, deposition of sediment in a reservoir may represent largely passive storage of terrestrial C because of low net primary production of the ecosystem, whereas deposition in wetlands with high net primary productivity may maximize active C sequestration. Other zones of deposition may also have high productivity but a lower pedogenic equilibrium for C storage. Assessing the influence of soil resource redistribution on C sequestration in an ecosystem requires the ability to track both the movement of sediment in the ecosystem and the dynamics of soil C within the zones of erosion and deposition.

An assessment of the soil C inventory for the catchment indicates that riparian buffer systems can be an important component of the overall watershed C budget. For example, our estimate of C storage in the riparian wetland of the watershed is 800 metric tons (t) C ha^{-1} that is 16 times the approximate 50 t C ha^{-1} content for agricultural soils in the watershed. In terms of the total amount of organic C stored within soil resources of this watershed, the wetland may constitute a major portion of the soil C budget. Because of these large differences in C pool size, the dynamics of C, in a relatively small wetland ecosystem, can be an important factor in determining the overall capacity of an agricultural watershed to act as a net sink for organic C. Changes in net primary production of riparian vegetation may also be an important factor. Biomass yields in natural freshwater wetland ecosystems are usually P limited. Movement of agricultural nutrients into wetlands is likely to stimulate C fixation by ecosystem vegetation (Aerts et al., 1999; Bedford et al., 1999).

Comparison of the ^{137}Cs distribution within the profiles of soil from the undeveloped (pristine) and agricultural first-order basins (Fig. 6) provides an indication of substantial differences in soil movement within the two basins. The substantial redistribution of the ^{137}Cs tracer in the agricultural basin relative to the largely unaltered distribution in the natural control basin is indicative of the strong impact of agriculture on redistribution of soil resources within watersheds. Clearly within the past 50 years, agricultural management has had a marked influence on soil movement and distribution within the watershed.

Extensive characterization of the landscape features within the agricultural watershed has led us to hypothesize that agricultural activities within a watershed markedly increase rates of C storage within the riparian buffer by increasing sediment deposition and contributing nutrients to wetland ecosystems, which may increase primary productivity. The following observations from the watershed support this hypothesis: (1) The soil of the wetland is a histosol (organic soil) which has been buried by a distinct 20–30 cm layer of recent mineral sediments; (2) Measurements of C storage in the histosol profile (i.e. storage over the history of the wetland) indicates that the typical limit for C content of this histosol is approximately 200 mg C g^{-1} soil indicating that this is an upper limit for "natural" storage of C in soils of the ecosystem (Fig. 7); (3) Dating of sediment accretion in the wetlands using fallout ^{137}Cs indicates that rapid deposition of sediment (0.2–2.0 cm year^{-1}) has occurred within the last 35 to 45 years; (4) The mineral sediment is highly enriched with organic matter and calculation of the organic C in these recent mineral sediments indicate that it constitutes a significant portion (ca. 10%) of total soil C pool for the wetland (Fig. 8 shows relationship between soil C and ^{137}Cs); and (5) Wetland accretion of sediments from agricultural lands has increased P content of the ecosystem and infiltration of agricultural N in groundwater has increased N content which raises the likelihood that net primary production of vegetation in the ecosystem has been stimulated.

2.1. Increased C sink capacity of the wetland

Our observations indicate that the recent deposition of sediment associated with agricultural activity has greatly stimulated the rate of C storage in this wetland ecosystem. The upper limit for C content of the original histosol suggests that rates of mineral sediment deposition to the wetland naturally limit C storage in the ecosystem. We have found that with deposition of agricultural sediments, recent annual rates of C storage in the wetland increased to the range of 1.6–2.2 t C ha^{-1}

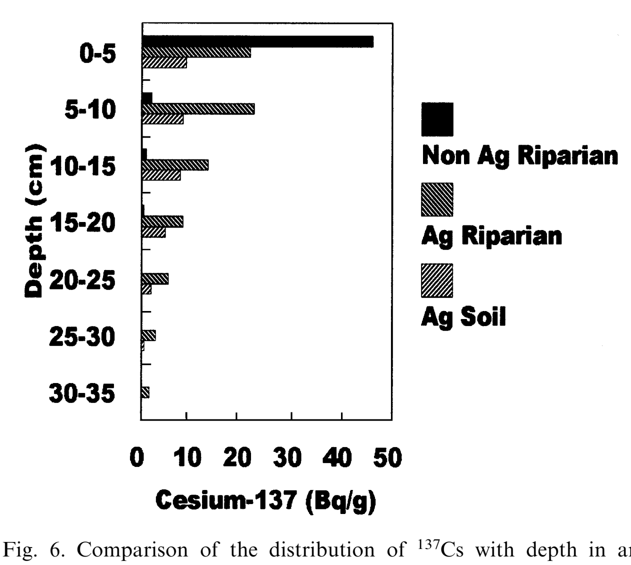

Fig. 6. Comparison of the distribution of ^{137}Cs with depth in an undeveloped (non agricultural) riparian wetland soil and soils from an agricultural riparian wetland and adjacent field under tillage.

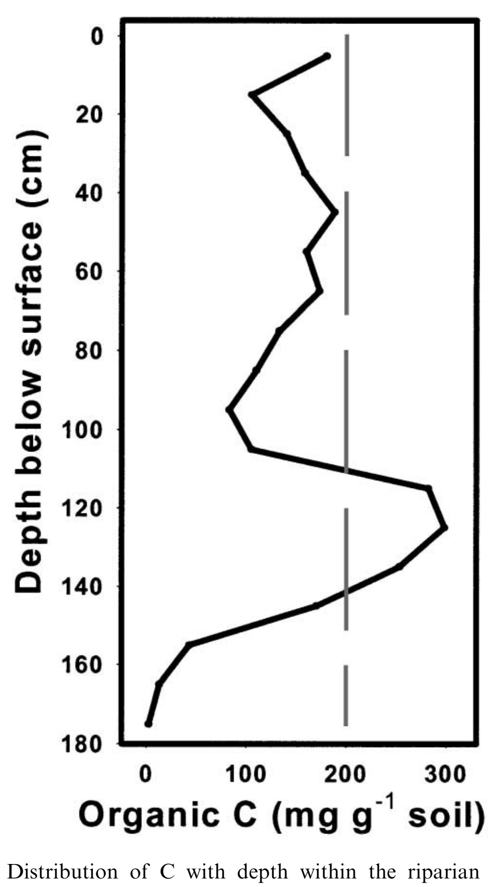

Fig. 7. Distribution of C with depth within the riparian wetland. The dashed line indicates the approximate limit for C storage in the ecosystem.

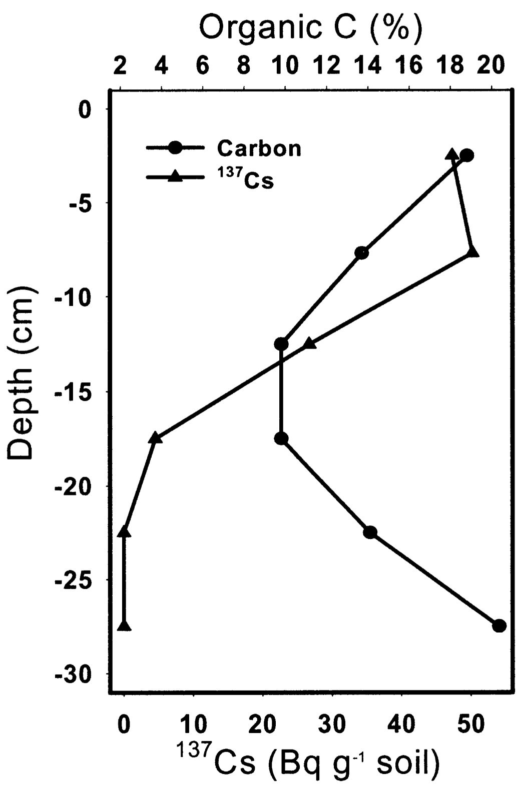

Fig. 8. Comparison of the distribution of organic C and ^{137}Cs within the post-settlement deposition portion of the agricultural wetland profile.

year^{-1}. These values are 4–7-fold higher than Lal et al. (1998) estimated rates of C storage in wetlands (0.29–0.43 t C ha^{-1} year^{-1}) that were established by methods not well suited to detect recent changes in C storage rates. Based on the mineral sediment deposited in the wetland and the apparent limit on soil C content, it is estimated that the overall capacity of this ecosystem to store C has been increased by 200–300 t C ha^{-1}. This represents a substantial increase in storage capacity of the watershed as a whole.

If rates of C storage in the wetland have not increased with introduction of agriculture to the watershed (i.e. remained constant), then extrapolation of current rates back in time would indicate that the wetland age is less than 400 years. Our knowledge on origins of land forms in the Mid Atlantic region, however, indicates that the age of the wetland is on the order of thousands of years. This is also supported by ^{14}C data for the site that indicates that the mean residence time for C in the histosol immediately below the post settlement deposition layer is 600 years BP.

2.2. Implications of increased sink capacity associated with wetlands

The 1992 USDA Natural Resources Conservation Service (NRCS) national inventory estimates 45 M ha of wetlands in the United States. Deposition of sediments and nutrients in these wetlands from anthropogenic sources is common. Large increases in rates of C storage in wetlands due to these influences will significantly influence the C storage capacity of watersheds. On a national basis, this influence could substantially increase estimates of C storage resulting from agricultural activities because of the frequent occurrence of wetlands in agricultural watersheds (7.8 M ha total agricultural wetlands: 4.6 M ha near cropland, 3.2 M ha near pastureland). Moreover, riparian wetlands are clearly long-term sinks for C, whereas residence time of C storage in soils under agricultural production is uncertain and highly dependent on future land management decisions. The increased net primary production of vegetation in wetlands due to influx of agricultural nutrients has additional significance because of the potential for highly efficient storage of plant residue C in these anoxic soil ecosystems relative to the much lower storage efficiency of the aerobic soils under agricultural production. For example, relatively modest rates of C storage (ca. 0.2 t C ha^{-1} year^{-1}) can be expected on agricultural lands with implementation of best management practices for C sequestration which is in sharp contrast to rates (1.6–2.2 t C ha^{-1} year^{-1}) we have measured in the wetland under study.

3. Conclusions

Our understanding of C cycling in the biosphere is limited as demonstrated by the magnitude of the "missing terrestrial C sink". The impact of ecosystem disturbance on C sink strength and capacity of terrestrial ecosystems has not been well characterized. New concepts about the impact of soil erosion and soil resource redistribution within terrestrial ecosystems may go a long way towards accounting for missing terms within the global C budget. Our studies on a small agricultural watershed in Maryland indicate that erosion and soil redistribution within the watershed have a major influence on the C budget for the watershed.

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