

INFLUENCE OF SOIL EROSION ON CARBON DYNAMICS IN THE WORLD

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Abstract

Natural or geologic soil erosion is a slow but constructive process. It formed some of the most fertile soils of alluvial and loess origins. Many ancient civilizations (e.g., Nile, Indus, Euphrates, Yangtze), the so-called "hydraulic societies" thrived on alluvial soils. In contrast, the accelerated erosion by anthropogenic activities, is a destructive process. It has caused the demise of once thriving civilizations. In addition to its decline in productivity, it also causes non-point source pollution, decline in air quality by dust, and emission of greenhouse gases. Erosion by water and wind preferentially removes soil organic matter (SOM), a light fraction concentrated in the vicinity of the soil surface. As a key determinant of soil quality, depletion of SOM has numerous ecological, economical and environmental consequences. Yet, the fate of eroded soil carbon (C) is not known, and is a debatable issue. Sedimentologists argue that eroded C transported into the aquatic ecosystems is buried and sequestered, and soil scientists believe that 20 to 30% of the C transported over the landscape is mineralized and released into the atmosphere. Resolving this issue is necessary to developing strategies for sustainable management of natural resources for erosion control and decreasing the rate of enrichment of CO₂. The global sediment transport to the oceans is estimated at 15 to 20 billion Mg/yr from 1.09 billion ha of land affected by water erosion of which 0.83 billion ha is at moderate + level of soil erosion. Uncertainties in the data are reflected in estimates of sediment transport, which range from 8 to 51 billion Mg/yr. The total amount of soil organic carbon (SOC) displaced by erosion annually is estimated at 4 to 6 Pg, compared to 6.3 Pg C emitted by fossil fuel combustion. The debate about the fate of eroded C is accentuated by numerous uncertainties about the lack of understanding of the complex and interacting processes involved: 1) breakdown of aggregates by erosion leads to exposure of SOC to microbial processes and enhances mineralization, 2) change in soil moisture and temperate regimes of eroded soil enhances oxidation of SOM, 3) redistribution of SOC-enriched sediments over the landscape may also increase oxidation, 4) some of the SOC buried in depressional and protected sites may be re-aggregated and sequestered, and 5) SOC transported into rivers may be mineralized depending on the climatic conditions. Assuming a delivery ratio of 13 to 20%, SOC concentration in sediment of 2% and oxidation rate of 20% of the C displaced, erosion-induced C emission is estimated at 0.8 to 1.2 Pg C/yr. Of this, 3.9% is from Oceania, 8.3% from Europe, 11.5% from North America, 6.0% from Africa, 14.2% from South America and 56.1% from Asia. As much as 0.4 to 0.6 Pg C may be transported into the ocean, and 2.8 to 4.3 Pg C is redistributed over the landscape. Yet, erosion-displaced C is not accounted for in the global C budget. Nonetheless, adoption of conservation-effective measures and erosion management techniques must be a high priority.

Key words: Eroded soil carbon, carbon burial, sedimentation of carbon, greenhouse effect, fate of eroded carbon, global warming

INTRODUCTION

Increase in atmospheric concentration of carbon dioxide (CO₂) and other greenhouse gases (GHGs) during the 20th century (IPCC, 2001), linked to the observed and projected climate change, has raised concerns regarding sources and sinks of these gases. Land use change and fossil fuel combustion are linked to the climate change, defined as a “change of climate attributed to human activity that alters the composition of the global atmosphere.” The global average surface temperature has increased by 0.6 ± 0.2 C since 1860 (IPCC, 1995; 2001), which is attributed to an anthropogenic increase in atmospheric concentration of several trace gases. The concentration of CO₂ has increased by 31% from 280 ppmv in 1750 to 367 ppmv in 1999 at an average rate of 1.5 ppmv or 0.4%/yr (Etheridge et al., 1996; IPCC, 2001), that of CH₄ from 700 ppbv to 1760 ppbv (increase of 151%) (Etheridge et al., 1998; IPCC, 2001) and that of N₂O from 270 ppbv to 316 ppbv (increase of 17%) over the same period (IPCC, 2001). Land use change, soil cultivation and erosional processes have a strong impact on the carbon (C) cycle at pedon, soilscape, landscape and watershed scales. The effects of land use change, conversion of natural to managed ecosystems and soil cultivation, on the emissions of GHGs involve complex interacting processes leading to: 1) decomposition of the biomass; 2) mineralization of soil organic matter or humus exacerbated by increase in soil temperature and decrease in soil moisture; 3) increase in susceptibility to soil erosion; 4) displacement and redistribution of soil organic carbon (SOC) over the landscape including burial in depressional sites; and 5) possible increase in emission of CO₂ and other GHGs into the atmosphere due to increase in oxidation and mineralization.

The impacts of projected climate change on soil properties, SOC pool and dynamics, and susceptibility to erosion are not known. A principal unknown is the effect of increase in global temperature on SOC pool as affected by change in net primary productivity (NPP), possible increase in the rate of mineralization of SOC and in susceptibility to soil erosion. Arnell and Liu (2001) reported that rise in GHG concentrations may lead to reduced soil moisture storage, increase in surface and shallow runoff, and increase in sediment loads in rivers. The scientific understanding of the processes set-in-motion by climate change and the attendant alterations in soil erodibility, rainfall/climate erosivity, redistribution of sediments and the associated SOC over the landscape and susceptibility of displaced SOC to mineralization under aerobic or anaerobic conditions is rather sketchy and incomplete. Mineralization of SOC depends on temperature and moisture regimes (Leiros et al., 1999), which are in turn influenced by soil erosion.

Gaseous emissions from terrestrial ecosystems are exacerbated by soil degradation. Soil erosion is by far the most widespread form of soil degradation. Total land area affected by soil erosion is 1094 million hectares (Mha) by water erosion of which 751 Mha is at moderate + level of severity, and 548 Mha by wind erosion of which 280 Mha is at moderate + level of severity (Oldeman, 1994). Because of low density (1.2 to 1.5 Mg/m³) and being concentrated in vicinity of the soil surface, the SOC fraction is strongly influenced by erosional processes. It is preferentially removed along with sediments by both runoff water and wind.

While the on-site effects of soil erosion lead to adverse impacts on soil quality and productivity, offsite impacts are even more drastic yet difficult to precisely quantify and

comprehend. Over and above the problem of eutrophication and contamination of surface waters by non-point source pollution, emission of erosion-induced GHGs is a major concern that needs to be addressed.

The objective of this manuscript is to discuss the impact of erosional processes on soil C dynamics, describe the fate of eroded soil C, and identify soil/site conditions and soil processes which lead to emission of GHGs or sequestration of C displaced by erosion.

SOIL ORGANIC MATTER: A NATIONAL RESOURCE

Soil organic matter comprises the sum of all organic substances in soil. It consists of a mixture of plant and animal residues at various stages of decomposition, of substances synthesized microbiologically and/or chemically from the breakdown products, and of the bodies of live microorganisms and small animals and their decomposition by-products (Schnitzer, 1991). Soil organic matter, comprising about 58% of SOC, is a key determinant of soil quality – biomass productivity and environment moderating capacity. The beneficial impacts of SOC on soil quality are attributed to: (i) stabilization of soil structure through formation of organo-mineral complexes, and development of microaggregates, (ii) improvement in available water holding capacity of the soil through increase in soil moisture retention at field capacity (0.3 bar suction), (iii) improvement in soil biodiversity especially activity of soil fauna (e.g., earthworms), (iv) biodegradation of contaminants, (v) buffering soil against sudden changes in pH and elemental concentrations, (vi) minimizing leaching losses of fertilizer through chelation and absorption, (vii) filtering and purification of water by sorption and degradation of pollutants, (viii) strengthening mechanisms of elemental cycling, (ix) improving soil quality and productivity, and (x) sequestering C and mitigating the climate change.

It is because of these and other multifarious benefits that led Albrecht (1938) to state that “soil organic matter is one of our most important national resources; its unwise exploitation has been devastating; and it must be given proper rank in any conservation policy as one of the major factors affecting the level of crop production in the future.” The importance of Albrecht’s statement cannot be over-emphasized in the context of the issue of the projected climate change in the 21st century. Outlining options for mitigating climate change necessitates identification of sources and sinks of atmospheric C. Soil humus has historically been a source of CO₂ and CH₄, especially in case of low-input and subsistence agriculture that had been practiced throughout the world until the advent and widespread use of chemical fertilizers after World War II. Jenny (1980) stated that “among the causes held responsible for the CO₂ enrichment, highest ranks are accorded to the continuous burning of fossil fuels and the cutting of forests. The contribution of soil organic matter appear underestimated.” Indeed, IPCC (2001) lists “land use change” as a source of atmospheric CO₂ estimated at 1.7 ± 0.8 Pg (1 Pg petagram = 1 billion metric tons = 10^{15} g) during the 1980s and 1.6 ± 0.8 Pg during the 1990s. In this context, the term “land use change” refers to tropical deforestation and the attendant release of CO₂ and CH₄ by decomposition/burning of biomass. The emission of C from soil due to anthropogenic perturbations is not specifically accounted for, partly because of the difficulty in evaluating small changes in SOC in a large global reservoir of 1550 Pg (Batjes, 1996; Eswaran, 1998), and partly because of lack of knowledge about the complex processes involved. Nonetheless, contribution

of SOC pool as a source or sink for atmospheric CO₂ at the global scale cannot be ignored (Lal, 2000).

Soil humus and its dynamics are important components of the global C pools and fluxes. Because most agricultural, degraded and drastically disturbed soils now contain lower SOC pool than that under undisturbed conditions, there exists a potential sink to absorb more C from the biomass through the process of C sequestration. Soil C sequestration implies capturing atmospheric C and securely storing it in biota, soil and other terrestrial and aquatic systems. Two principal processes of C sequestration are biotic and abiotic. Biotic processes include C storage including those in soil and biota. The latter comprises plants in terrestrial, aquatic and oceanic ecosystems, and biofuel offset is a principal component of biotic C sequestration mechanisms. Formation of secondary carbonates, dissolution of atmospheric CO₂ to form carbonic acid and its reaction with cations to form pedogenic carbonates, is also a chemical reaction facilitated by biotic processes. In contrast, abiotic processes of C sequestration involve compression of CO₂ from industrial sources and its injection in the geologic strata, saline aquifer and deep ocean. Leaching of carbonates from upper layers of soil into the ground water is another abiotic process of C sequestration.

SOIL DEGRADATION AND THE CLIMATE CHANGE

Soil degradation is defined as “decline in soil quality by several degradative processes including erosion, salinization, soil structure, depletion of SOC pool and essential nutrients.” Soil erodibility, susceptibility of soil to erosivity of rainfall and/or wind, depends on SOC concentration, soil structure, texture and water infiltration characteristics. However, the effects of climate change on soil erodibility and erosion hazard are not known. Soil erodibility increases with decrease in SOC concentration, reduction in structural stability and decline in water infiltration capacity. The SOC concentration, being in a dynamic equilibrium with input and output of biosolids and other climatic variables, may be sensitive to the projected increase in global temperature. Increase in global temperatures may lead to higher SOC turnover and increase in mineralization rate (Newton et al., 1966; Ross et al., 1996; Hungate et al., 1997; Leiros et al., 1999; Gitay et al., 2001). The attendant depletion of SOC pool may adversely affect soil structure, decrease infiltration rate and reduce available water capacity (Reilly, 1996), leading to a possible increase in erodibility, increase in runoff and high risks of soil erosion.

Soil degradation affects SOC pool directly and indirectly. Directly, it reduces the biomass or C input into the system because of reduction in NPP and decrease in water and nutrient availability with attendant reduction in biodiversity. Indirectly, it leads to disruption in biogeochemical cycles and decline in soil resilience. Soil degradation also accentuates losses of SOC pool by exacerbating the rate of mineralization, leaching and soil erosion. Soil erosion hazard may increase due to increase in erodibility and erosivity factors. Rains may become more intense and of high erosivity (Carter and La Rovere, 2001). Consequently, susceptibility to both water and rill erosion may increase (Gregory et al., 1999), with attendant effects on depletion of SOC pool (Smith et al., 1997). Depletion of SOC pool may be exacerbated by aridization of the climate, such as in the Mediterranean region (Lavee et al., 1998). The fragile ecosystems of arid and semi-arid climates may be highly sensitive to desertification even with minor changes in rainfall distribution and temperature regime (Puigdefabregas, 1998; Villers-Ruiz and Trejo-

Vázquez, 1998). Desertification and CO₂-induced climate change are intricately linked. Desertification leads to reduction in biological productivity of the ecosystem and long-term loss of natural vegetation, which reduces the biomass input into the soil. The land area prone to desertification is 7.0 million km² in Africa (assuming 25% reduction in productivity (Desanker and Magadza, 2001), and increasing.

Increase in soil degradation and desertification may increase the efflux of GHGs from soil to the atmosphere. Erosion, both by water and wind, may enhance gaseous emissions on-site and off-site. On-site erosion alters soil moisture regime and may accentuate mineralization. Off-site, erosion and deposition may alter soil moisture regime and increase susceptibility of SOC to mineralization. The eroded landscape continues to redistribute SOC and enhance its susceptibility to emission (Page et al., 2000). The effects of erosion on soil are closely linked to soil physical degradation, especially to aggregates leading to anaerobiosis either by compaction, water imbalance or both. Soil chemical degradation, especially by salinization and nutrient imbalance, can also cause depletion of SOC pool. The integrated effect of soil physical and chemical degradation on soil biological quality is to decrease the SOC pool and increase in efflux of GHGs from soil to the atmosphere.

SOIL EROSION AND CARBON DYNAMICS

Soil erosion is a complex but global problem, and its environmental impact is now being debated as to whether the process is a source or sink for atmospheric C. Resolving this issue objectively requires a thorough understanding of the mechanisms involved. As a physical process, soil erosion involves “work” in detachment and transport of soil particles. The energy for the work is supplied by agents of erosion including raindrop, runoff, wind, gravity etc. The work involves: (i) detaching soil particles from clods, (ii) breakdown of macro- into micro-aggregates and dispersion into soil separates or primary particles, (iii) transporting particles called “sediments” over the landscape by water runoff, ice, wind or gravity, and (iv) depositing sediments into depressional sites or against barriers that retard the velocity/carrying capacity of runoff water or wind. All these stages of erosion have a strong impact on SOC and soil inorganic carbon (SIC) components. Severe soil erosion, by water or wind, can lead to truncation of the soil profile by removal of the surface layer and exposure the carbonate-rich sub-soil. Carbonates thus exposed may react with acidiferous material (e.g., acid rain, fertilizers, etc.) and release C to the atmosphere. In contrast, burial of a carbonaceous soil layer by sediments may reduce the magnitude of CO₂ emission (Fig. 1).

Soil particles detached are transported over the landscape and a proportion of them eventually redeposited in depressional sites and aquatic/loessial ecosystems. The distance to which particles are transported depends on the density/weight of the particles and velocity or carrying capacity of the fluid (wind or water). The lighter particles (e.g., SOC and clay fractions) are carried longer distances than heavier fractions (e.g., silt, sand and gravel).

The fact that SOC is concentrated in the surface layer (0 to 20 cm depth), has numerous implications with regards to the erosion-induced emissions. One, SOC is easily transported by surface runoff or blowing wind, especially because it is also a light fraction. Consequently, the enrichment ratio of sediments for C is more than 1 and often as high as 5. Two, there is also an

inverse relationship between the quantity of sediments generated by different erosional processes and the amount of C displaced (Trustrum et al., 2002). Three, SOC pool is depleted on-site and enhanced at the depositional site, leading to different processes at these locations.

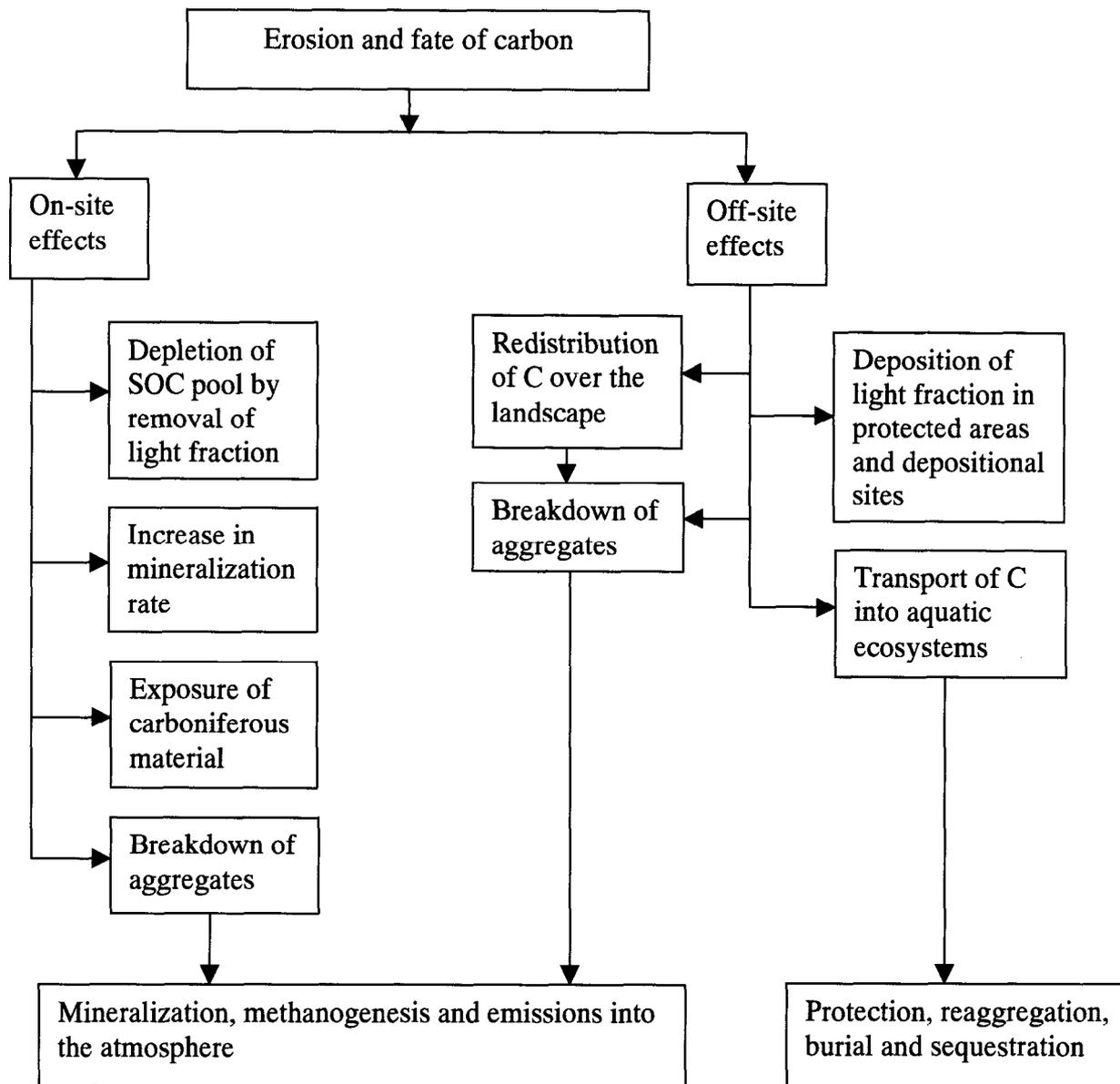
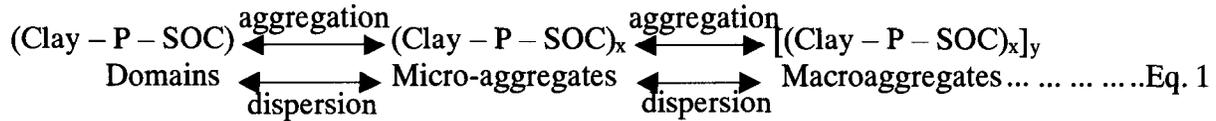


Fig. 1 Soil carbon dynamics in relation to the accelerated soil erosion by water and wind.

Similar complex processes are involved in erosion-induced SOC dynamics whereas aggregation involves formation of stable micro-aggregates or the organo-mineral complexes, slaking and breakdown of aggregates lead to dispersion of soil and emission of C (as CO₂ or CH₄) through microbial action on organic matter hitherto encapsulated within aggregates. While

the process of aggregation sequesters SOC (Edwards and Bremner, 1967; Tisdall and Oades, 1962; Chaney and Swift, 1986; Oades and Waters, 1991; Tisdall, 1996), that of dispersion releases C as shown in Eq. 1.



Therefore, breakdown of aggregates releases C, and its transport and redistribution over the landscape and deposition in depressional sites further exposes it to numerous interacting processes. Some of the SOC displaced is merely redistributed over the landscape and may never reach the stream and reservoirs or other aquatic systems (Gregorich and Anderson, 1985). A fraction of the SOC displaced, however, may reach stream and reservoirs. The time involved in the transport of SOC from up-slope to aquatic ecosystems may range from days to centuries, and SOC is subjected to numerous interacting processes en route. It is the fate of C thus being transported over the landscape that has become a topic of debate among soil scientists and sedimentologists.

FATE OF ERODED SOIL CARBON

The fate of C translocated (e.g., detached, redistributed and redeposited) from the original site is difficult to predict because of the confounding effect of numerous interacting factors such as moisture and temperature regimes, quality of the sediments, oxidation/reducing conditions, dispersion or re-aggregation. Consequently, a mass-balance approach is needed to assess the fate of displaced C at the watershed scale (Jacinthe and Lal, 2001). A schematic of the processes involved in SOC dynamics in a landscape influenced by accelerated soil erosion are outlined in Fig. 2. The erosion-induced perturbation alters SOC dynamics (Eq. 2).

$$\text{SOC} = (\text{SOC}_a + A) - (E + L + M) \dots \dots \dots \text{Eq. 2}$$

Where SOC_a is the antecedent pool, A is accretion or input, E is erosion, L is leaching and M is mineralization. A part of the SOC redistributed over the landscape may be emitted into the atmosphere either a CO_2 or CH_4 depending on the degree of aeration. Some of the dissolved organic carbon (DOC) may be leached and precipitated in the sub-soil or transported into the aquatic ecosystems and reprecipitated. Erosion may lead to C sink if $(\text{SOC}_a + A) > (E + L + M)$ or net source if $(E + L + M) > (\text{SOC}_a + A)$. In most cases, however, $(E + L + M) > (\text{SOC}_a + A)$ making erosion a net source. Consequently, there are numerous ramifications of the erosion-induced translocation of SOC:

1. On-site, SOC pool is depleted, often severely, because of the preferential removal of organic matter. Consequently, the enrichment ratio of organic matter in sediments is often as much as 2 to 5 because SOC is concentrated in the surface layer where it is easily carried by surface or shallow flow and wind currents.
2. The depletion of SOC pool leads to a decline in soil quality because of a reduction in available water capacity, decrease in effective rooting depth and depletion of the reserves of essential plant nutrients. Decline in soil quality has strong adverse impacts upon biomass production and the quantity of crop residue (both above and below ground) returned to the soil.
3. Restoration of eroded soil, through conversion to an ameliorative land use, and replacement of nutrients, can lead to SOC sequestration.

4. The magnitude of SOC sink capacity thus created depends on the extent of SOC depletion, soil profile characteristics, landscape position, soil moisture and temperature regime, prevalent climate, and the intended land use and management.

There are two schools of thought, one proposed by sedimentologists and the other by soil scientists.

1. **Sedimentologist’s View of the Deposition of Eroded Soil Carbon:** Sedimentologists argue that erosional processes transport C to the burial/depositional sites (Van Noordwijk et al., 1997; Stallard, 1998 Smith et al., 2001), and account for most of the so-called “missing sink” for CO₂ estimated at 0.5 to 2.0 Pg C (Tans et al.,1990). It is argued that erosion leads to C sequestration in at least two ways: (i) on-site erosion depletes SOC pool and creates a C sink that is filled by vegetation regrowth and residue return, and (ii) off-site SOC is transported to depressional sites and is buried, sequestered and taken out of circulation. The annual transport of particulate organic carbon to the ocean is estimated at 0.09 to 0.57 Pg C/yr (Table 1). With this assumption, Smith et al. (2001) computed the sediment and SOC budget (Table 2). Assuming no gaseous loss due to mineralization of SOC en route, SOC transported to rivers and oceans is estimated at 0.01 Pg C for the U.S. and 0.4 Pg C for the world. Sedimentologists extrapolate these projections further and surmise that the so-called “missing sink” or “fugitive carbon” is buried annually into aquatic ecosystems and depressional sites, and that accelerated soil erosion is a good thing to have.

Table 1. Estimates of particulate organic carbon transported to the ocean.

Source	Annual flux to the ocean (Pg C/yr)
Berner (1992)	0.09-0.19
Chen et al. (2001)	0.24
Ittekkot (1998)	0.231
Lal (1995)	0.57
Ludwig et al. (1996)	0.17
Schlung and Schneider (2000)	0.40-0.53
Meybeck (1993)	0.17
Meybeck and Vörösmarty (1999)	0.195
Smith et al. (2001)	0.4

Table 2. Global and the U.S. budget of sediments and soil organic carbon displaced (adapted from Smith et al., 2001).

Process	U.S.A.		Global	
	Sediment	SOC	Sediment	SOC
	-----Pg/yr-----			
Erosion	7.4	0.05	200	1.4
River+wind flux to ocean	0.7	0.01	20	0.4
Land deposition	6.7	0.04	180	1.0

Assuming no losses of SOC due to mineralization en route.

2. **Soil Scientist’s View of the Fate of Carbon Displaced by Erosion:** Soil scientists argue that accelerated soil erosion is a principal cause of the emission of greenhouse gases from soil to the atmosphere. Arguments in support of this hypothesis include the following: (i) On-site,

accelerated soil erosion depletes SOC pool and degrades soil quality. Significant reductions in crop yield on eroded soils can occur (Monreal et al., 1997) especially in low-input and resource-based agriculture (Lal, 1998). The reduction in productivity may be due to decrease in effective rooting depth, reduction in available water capacity, and exposure of edaphologically inferior sub-soil of poor structure and unfavorable elemental balance. With progressive decline in productivity of the above and below-ground biomass, both the quality and quantity of residue returned to the soil are less, which further depletes the SOC pool. Indeed, SOC pool of eroded sites is drastically lower than those of uneroded sites (Rhoton and Tyler, 1990). (ii) Off-site, erosional processes transport SOC over the landscape and redistribute eroded sediment selectively depending on the density. During the process, aggregates are subjected to disruptive forces of runoff or wind. Some of the sediments are deposited down slope in depressional sites or protected areas (Fig. 2). The SOC budget of an eroded landscape can be computed by Eq. 3.

$$(SOC)_L = (SOC)_A - (SOC)_D - (SOC)_R + (SOC)_M \dots \dots \dots \text{Eq. 3}$$

Where L represents mean SOC pool over the landscape after erosional event, A is the antecedent pool prior to the event, D is SOC deposited in depressional sites, R is SOC transported in rivers and aquatic ecosystems, and M is the fraction mineralized emitted into the atmosphere. Mineralization of SOC may occur over the landscape, in depressional sites and in aquatic ecosystems.

The principal discrepancy between sedimentologists and soil scientists lies in the assumption with regards to the magnitude of oxidation of SOC. Some sedimentologists assume that the oxidation flux is insignificant (Smith et al., 2001) and others believe that SOC lost during erosion is largely oxidized (Schlesinger, 1995). The magnitude of oxidation of eroded material may depend on the composition of particulate organic material. While humins are preserved (Hatcher et al., 1985; Hatcher and Spiker, 1988), some of the material may be re-aggregated and protected against mineralization (Gregorich et al., 1998), but easily decomposable labile fraction is mineralized (Beyer et al., 1993). The proportion of eroded SOC that is mineralized en route to and in the depositional sites may be 20 to 30% (Table 3). Therefore, about 10% of the eroded SOC is transported to the ocean, 20 to 30% is emitted into the atmosphere, and 60 to 70% is redistributed over the landscape. The magnitude of the emission may also depend on soil moisture and temperature regime (Bajracharya et al., 2000) and soil reflectance properties as influenced by erosion (Wagner-Ridle et al., 1996).

Table 3. Estimates of soil organic loss by oxidation during the erosional process.

Source	Fraction lost by oxidation (%)
Lal (1995)	20
Jacinthe and Lal (2001)	25-30
Beyer et al. (1993)	70
Schlesinger (1995)	100
Smith et al. (2001)	0

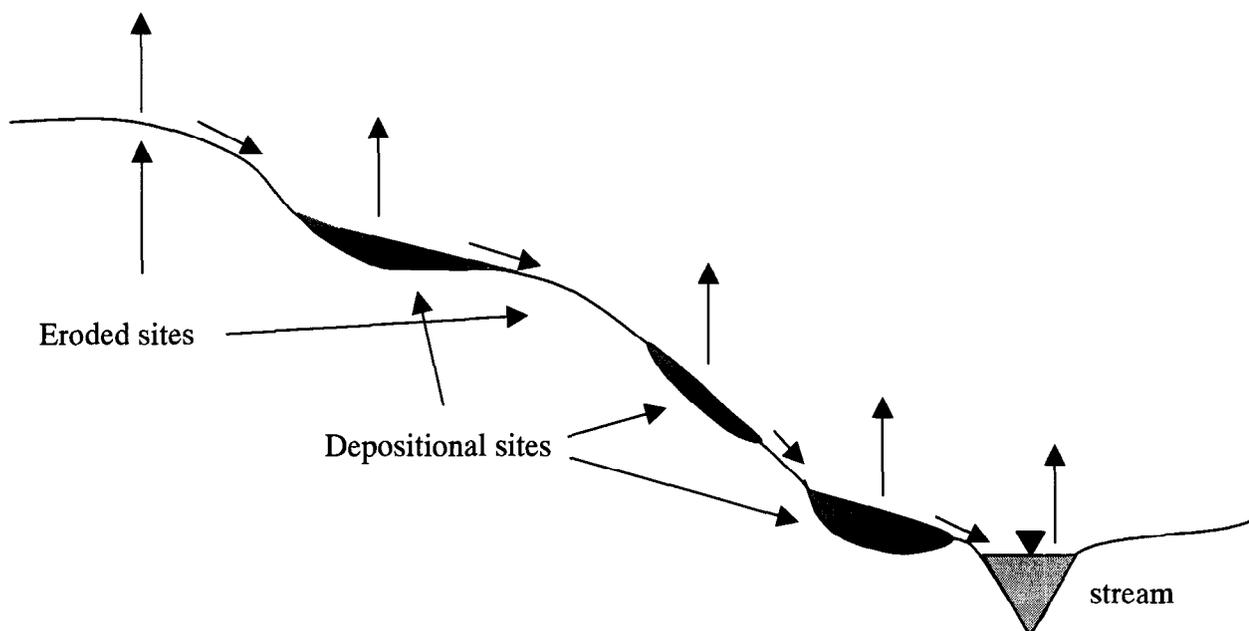


Fig. 2 Transport, redistribution and deposition of soil organic carbon on an eroded landscape.

EROSION-INDUCED GLOBAL EMISSION OF CARBON

There are two approaches to estimating erosion-induced C emission. One, sediment by world's rivers is estimated at 15 to 20 billion Mg (Walling and Web, 1996). Assuming delivery ratio of 13 to 20% (Walling and Web, 1996) and SOC content of 2%, total SOC displaced by erosion is 4.0 to 6.1 Pg. Assuming 20% is emitted into the atmosphere (Lal, 1995), erosion-induced C emission is 0.7 to 1.2 Pg C/yr (Fig. 3). Based on these assumptions, the global sediment and C budget is shown in Table 4.

There are numerous uncertainties regarding the fate of the eroded SOC, and there is a little quantitative data to assess the erosion-induced emissions. The data on continental basis is shown in Table 5. Estimates of erosion-induced emissions are shown in Table 5. Total erosion-induced emissions are estimated from 0.8 to 1.2 Pg C/yr (with a mean of 1.0 Pg C/yr) are comparable to those reported earlier at 1.14 Pg C/yr (Lal, 1995). Such large emissions must be accounted for in the global C cycle.

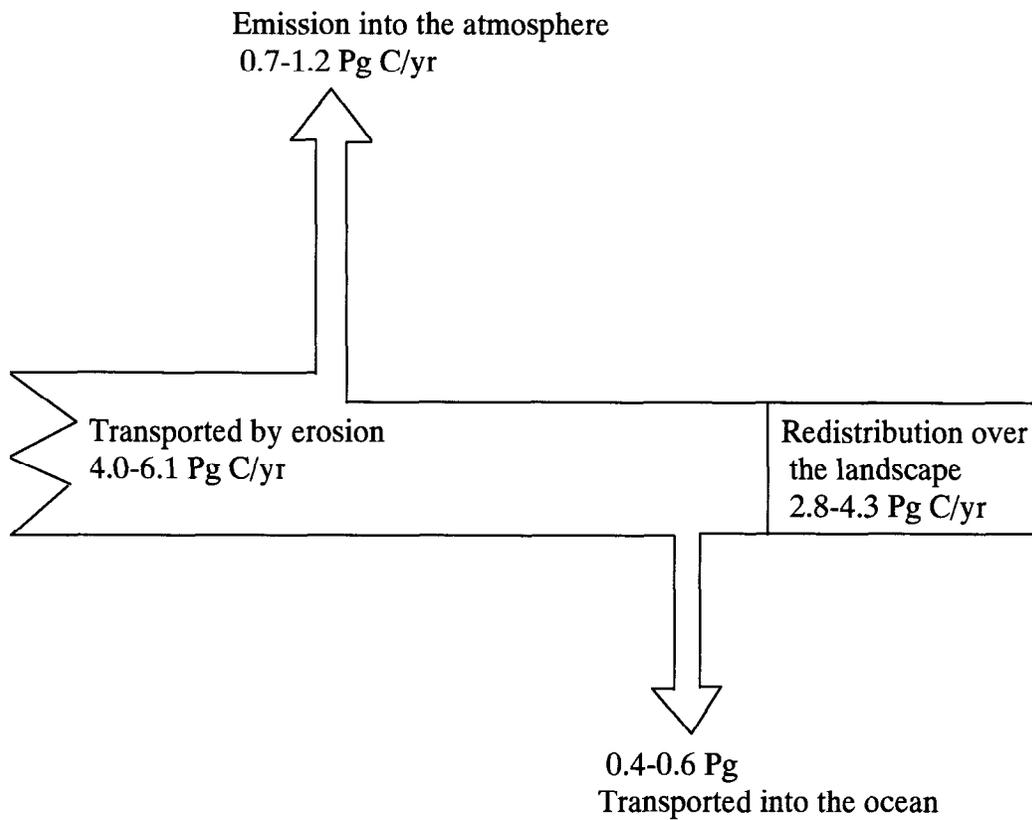


Fig. 3 Global carbon budget as influenced by soil erosion.

Table 4. Global budget of sediment and soil organic carbon.

Process	Erosion	SOC dynamic
	-----Pg/yr-----	
Erosion over the landscape	88-135	4.0-6.1
Transport to oceans	15-20	0.4-0.6
Redistributed over the landscape	73-115	2.8-4.3
Emission to the atmosphere	0	0.8-1.2

Table 5. Continental distribution of eroded soil organic carbon and emissions by decomposition.

Continent	Estimates of SOC displaced by erosion -----Tg C/yr-----	C emission by erosion (20%)
North America	456-700	91-140
South America	563-866	113-173
Africa	235-362	47-72
Asia	2220-3415	444-683
Europe	330-509	66-102
Oceania	153-236	31.47
Total	3957-6083	792-1217

Assuming sediment load estimates by Walling and Web (1996), delivery ratio of 13 to 20% and SOC concentration of 2% and emission of 20%.

SUMMARY AND CONCLUSIONS

Geologic/natural soil erosion is a constructive process. It is an important soil-forming factor that created the world's most fertile soils in the flood plains and deltas of major rivers. Annual renewal of soil fertility by sediments deposited in flood plains sustained agricultural production and supported dense populations on alluvial soils. Accelerated soil erosion, however, due to anthropogenic activities involving land misuse and soil mismanagement, is a destructive process. Offsite, it causes non-point source pollution and emission of CO₂ and other GHGs into the atmosphere. For transport of C to the oceans estimated at 0.4 to 0.6 Pg/yr, emissions as CO₂ by oxidation of eroded SOC is estimated at 0.8 to 1.2 Pg C/yr.

Sustainable management of soil and water resources is needed not only for food/biomass production but also for maintaining environment quality including mitigation of climate change. In fact, SOC sequestration is also needed for desertification control (Squire et al., 1995) and restoration of degraded ecosystems (Arnald, 2002).

There are numerous hot spots of accelerated soil erosion around the world. Principal among these are South Asia, especially the Himalayan-Tibetan ecosystem, Central Asia, the Loess Plateau of China, sub-Saharan Africa and the Maghreb region of northwest Africa, the Andean region of South America, Dominican Republic and the Caribbean, and the highlands of Central America. The problems of soil erosion in these regions are accentuated by resource-poor farmers who practice low-input and subsistence farming. A coordinated effort is needed to facilitate widespread adoption of science-based and conservation-effective agriculture.

There are numerous options of mitigating the projected climate change. Sustainable management of soil and water resources is an important option. It is a win-win strategy. Adopting effective measures of soil and water conservation can enhance productivity, improve water quality, reduce erosion-induced emissions of GHGs and sequester C in soil and biomass to reduce the rate of enrichment atmospheric CO₂ and other GHGs.

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