

Agricultural activities and the global carbon cycle

R. Lal

Carbon Management and Sequestration Center, The Ohio State University, Columbus, OH 43210, USA;
(e-mail: rlal@magnus.acs.ohio-state.edu)

Key words: Conservation tillage, Soil carbon sequestration, Soil organic matter dynamics, Soil quality, Terrestrial carbon pool

Abstract

The observed and projected increase in emission of greenhouse gases, with attendant effects on global warming and sea level rise, have raised interests in identifying mitigation options. Terrestrial C sequestration involves capture of atmospheric C through photosynthesis and storage in biota, soil and wetlands. Land use, vegetation and soil management have a strong impact on the biotic processes of C sequestration. Losses of C from the terrestrial ecosystems are exacerbated by deforestation, biomass burning, plowing, resource-based and subsistence agriculture, and practices that mine soil fertility and deplete the soil organic C (SOC) pool. Biomass burning may also produce charcoal, which is an inert carbon with long residence time. Practices that enhance C sequestration include afforestation and reforestation, conservation tillage and mulch farming, integrated nutrient management and adopting systems with high biodiversity. Net C sequestration within an ecosystem can be assessed by taking into account the hidden C costs of fertilizers, pesticides, tillage, irrigation and other input. Restoration of degraded soils and ecosystems has a vast potential of C sequestration. The Kyoto Protocol provides for C sequestration in terrestrial sinks and C trading through Clean Development Mechanisms. Terrestrial C sequestration, besides being a win–win strategy, offers a window of opportunity for the first few decades of the 21st century. It is a natural process of reducing the rates of gaseous emissions while alternatives to fossil fuel take effect.

Introduction

Global surface temperatures have increased between 0.4 and 0.8 °C (0.6 ± 0.2 °C) since the late 19th century. The rate of increase has been 0.15 °C/decade since 1976, and the greatest warming has occurred during the Northern Hemisphere winter and spring along with some warming during summer (Baede 2001). The largest rates of warming are observed in mid- and high-latitude continental regions of the Northern Hemisphere. In addition, the land surface precipitation is increasing in the Northern Hemisphere mid- and high latitudes. There has also been a widespread increase in heavy and extreme precipitation events in regions, which have experienced an increase in precipitation. In eastern Asia, increases in heavy rainfall have been observed even with slight or

no increase in precipitation. Consequently, the snow cover and ice extent have decreased (IPCC 2001).

The observed climate change is attributed to emissions of greenhouse gases (GHGs) and anthropogenic activities. Concentrations of atmospheric GHGs and their radiative forcing have continued to increase since 1850. For example, the concentration of carbon dioxide (CO₂) has increased by 31% from 280 ppmv in 1850 to 380 ppmv in 2004, and since 1980 is increasing at the rate of 1.5 ppmv/yr or 0.4%/yr. Similar to CO₂, atmospheric concentrations of CH₄ and N₂O have also progressively increased because of anthropogenic activities (Etheridge et al. 1998; IPCC 2001; Prather and Ehalt 2001).

The term ‘radiative forcing’ denotes an externally imposed perturbation in the radiative energy budget of the Earth’s climate system (Ramaswamy 2001).

Such a perturbation can be caused by changes in solar irradiance incidents upon the planet, the concentration of GHGs or other changes that affect the radiative energy absorbed by the surface. The total radiative forcing of all GHGs (CO_2 , CH_4 , N_2O and halocarbons) since 1850 is estimated at 2.43 W/m^2 (IPCC 2001). Of this, the radiative forcing of CO_2 is 1.46 W/m^2 or 60% of the total changes in all GHGs. The current direct radiative forcing of CH_4 is 0.48 W/m^2 , which is 20% of the total from all of the long-lived GHGs. The radiative forcing of N_2O is estimated at 0.15 W/m^2 or 6% of the total. The IPCC (2001) report indicates a direct link between anthropogenic activities and the observed climate change.

Sources of greenhouse gases

From 1850 to 1998, approximately $270 \pm 30 \text{ Pg}$ of CO_2 has been net emitted into the atmosphere from fossil fuel burning and cement production. In comparison, about $136 \pm 55 \text{ Pg}$ has been emitted as a result of land use change (IPCC 2001). The latter comprises an estimated emission of $78 \pm 12 \text{ Pg}$ from world soils caused by anthropogenic depletion of soil organic carbon (SOC) (Lal 1999). Most of the emissions since 1980 are due to fossil fuel burning with 10 to 30% due to land use change or tropical deforestation.

During the 1980s, the global C budget comprised $5.4 \pm 0.3 \text{ Pg C/yr}$ emissions by fossil fuel combustion and $1.7 \pm 0.8 \text{ Pg C/yr}$ emission due to land use change, with a total source of 7.1 Pg C/yr . The known sinks included $3.3 \pm 0.2 \text{ Pg C/yr}$ accumulated in the atmosphere, $2.0 \pm 0.8 \text{ Pg C/yr}$ absorbed by the ocean, and $1.9 \pm 1.3 \text{ Pg C/yr}$ absorbed by unknown terrestrial sinks (IPCC 2001; Schimel et al. 2001). The global C budget changed for the 1990s. It comprised $6.3 \pm 0.6 \text{ Pg C/yr}$ emissions by fossil fuel combustion and $1.6 \pm 0.8 \text{ Pg C/yr}$ emission by land use change, leading to a total source of 7.9 Pg C/yr . The known sinks for the 1990s included $3.3 \pm 0.2 \text{ Pg C/yr}$ absorbed by the atmosphere, $2.3 \pm 0.8 \text{ Pg C/yr}$ absorbed by the ocean and $2.3 \pm 1.3 \text{ Pg C/yr}$ absorbed by unknown terrestrial sinks (IPCC 2001). There are large uncertainties associated with estimating the CO_2 release due to land use change. Therefore, the magnitude of terrestrial sinks is also uncertain.

Soils and the global carbon cycle

There are five principal global C pools, and the C circulates among these pools. The oceanic pool is the largest, estimated at $38,000 \text{ Pg}$ of C. The geologic pool is the second largest, estimated at 5000 Pg of C and comprising three sub-pools including 4000 Pg of coal, 500 Pg of oil and 500 Pg of gas. The third largest pool is pedologic or the soil C pool, comprising two distinct components: SOC and soil inorganic carbon (SIC). The SOC pool is estimated at about 1220 to 1550 Pg to 1-m depth and 2376 to 2450 Pg to 2-m depth (Eswaran et al. 1995). The SIC pool is estimated at 695 to 748 Pg to 1-m depth (Batjes 1996). The atmospheric C pool is 760 Pg , and increasing at the rate of 3.3 Pg C/yr . The biologic pool, comprising C contained in all biota, is 560 Pg . Therefore, the soil C pool estimated at 2300 Pg to 1-m depth (SOC = 1550 Pg and SIC = 750 Pg) is 4.1 times the biotic pool and 3.0 times the atmospheric pool. The atmospheric pool is increasing at the expense of the geologic, pedologic and the biotic pools. The soil and biotic pools combined are referred to as the terrestrial C pool.

The SOC pool is primarily humus, which comprises mixtures of plant and animal residues at various stages of decomposition and of microbial by-products. Humus is a dark brown or black amorphous material, characterized by large surface area and high charge density. Because of its high reactivity, it is generally bound to clay-sized particles as organo-mineral complexes, which may involve hybrid compounds of organic molecules attached to clay through polyvalent cations (Al^{+3} , Ca^{+2}). The density of the SOC pool to 1-m depth may range from 30 Mg/ha to 150 Mg/ha . Soils in their natural state contain a large SOC pool, which is stratified in the surface horizons and decreases abruptly with depth. Some organic soils contain as much as 800 Mg C/ha . The magnitude of the accumulative SOC pool depends on temperature, moisture, and texture (Jenny 1980). The SOC pool is higher in cool than in warm climates, and in wet and poorly drained soils than in dry and well-drained soils. The SOC has numerous functions, especially as a key determinant of soil quality or "soils' ability to produce economic goods and services and moderate the environment" (Doran and Parkin 1996; Gregorich et al. 1997). The SOC pool and its dynamics have a strong impact on the global C cycle because it can be a major source or sink for atmospheric CO_2 .

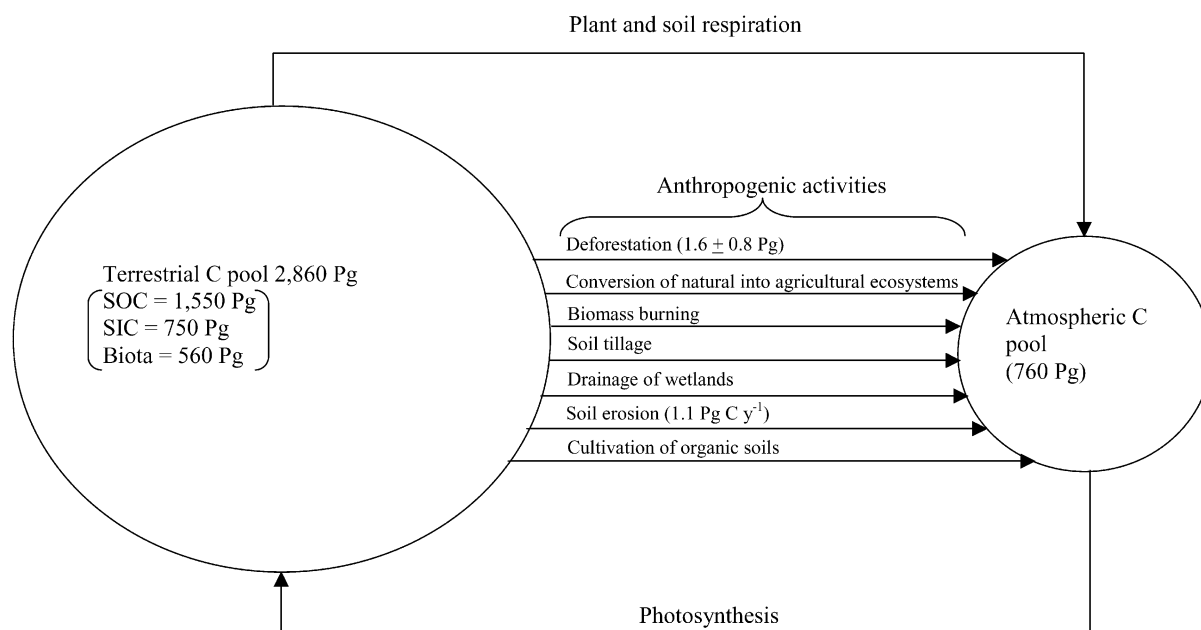


Figure 1. Anthropogenic activities affecting C emission from the terrestrial to the atmospheric pool. The direction of the arrow indicates the flux of C from one pool to another. Photosynthesis and plant/soil respiration are natural activities. All others are anthropogenic activities that cause emission of CO₂ (and other gases) from the terrestrial ecosystem to the atmosphere. The magnitude of emission caused by all anthropogenic activities is not known.

The SIC pool, although less reactive than the SOC pool, also impacts the global C cycle. It comprises primary and secondary carbonates. The primary carbonates are derived from the parent material and include primary minerals like calcite, dolomite and gypsum, and are called lithogenic carbonates (LIC). The secondary carbonates, also called pedogenic carbonates or PIC, are formed through the reaction of atmospheric CO₂ with Ca⁺² or Mg⁺² brought in (by dust, runoff, manure, ocean drift, etc.) from outside the ecosystem. Formation of PIC is a principal mechanism of soil C sequestration in arid and semi-arid climates.

There exists a strong link between the pedologic (soil) and the atmospheric pool via the biotic C pool. Every year about 60 Pg of atmospheric C is photosynthesized by the biotic pool and an equivalent amount is returned back to the atmosphere through soil and root respiration. If 10% of the photosynthate (6 Pg C) is retained in the terrestrial ecosystem, it can balance the C emissions due to fossil fuel combustion (6.3 Pg C/yr). Thus, the importance of strategic management of the terrestrial C pools cannot be over-emphasized.

Factors affecting emission of terrestrial carbon into the atmosphere

There is a strong linkage between the cycles of H₂O, C, N, P and S. In addition, the H₂O cycle is also closely linked with the energy budget. Thus, anthropogenic perturbation of any of the elemental cycles, the energy balance and the water cycle can have a drastic impact on the pedologic and biotic C pools, leading to enrichment of the atmospheric C pool. Anthropogenic activities that lead to depletion of the terrestrial pool and enrichment of the atmospheric pool include the following (Figure 1): (1) deforestation and biomass burning and other activities related to conversion of natural to agricultural ecosystems, (2) tillage and other soil disturbances, (3) drainage of wetlands, (4) cultivation of organic soils, (5) removal of biomass for fuel, fodder and other uses, and (6) accelerated soil erosion. While biomass burning is a source of emission, it also produces charcoal that is inert or recalcitrant and has a very long residence time in soil. The magnitude and rate of SOC loss due to agricultural activities is higher for soils with a high compared to a low pool, tropics compared to temperate regions, croplands compared to pastures and other

perennial vegetation (tree plantations). The loss of the SOC pool from agricultural soils may be as much as 30 to 60 Mg/ha, depending on the antecedent pool, climate, land use and management systems (Lal 2000). Most of the SOC loss occurs during the first few years upon conversion of natural to managed ecosystems (Davidson and Ackerman 1993; Gregorich et al. 1998). The loss of the SOC pool is accentuated when input of C in managed ecosystems, as crop residues and root biomass, is less than the losses by erosion, mineralization and leaching. The historic loss of SOC, difficult to precisely assess, has been estimated at 66 to 90 Pg (Lal 1999).

Soil degradation and emission of greenhouse gases

Conversion of natural to managed ecosystems and disruption in cycles of H₂O, C and other elements can lead to decline in soil quality as measured in terms of its biomass productivity and environment moderation capacity. Soil degradation may occur due to natural factors but is drastically accentuated by anthropogenic activities. Three principal forms of soil degradation comprise physical (e.g., decrease in quantity and stability of aggregates, crusting, compaction, reduction in infiltration rate, and erosion by water and wind), chemical (e.g., reduction in pH, acidification, accumulation of salts in the root zone by salinization or alkalinization, elemental imbalance leading to toxic concentration of some and deficiency of others) and biological degradation (e.g., reduction in activity and species diversity of soil fauna and flora, depletion of SOC pool). All degradative processes result in decline in the SOC pool primarily because of the decrease in biomass production, but also due to increase in losses by mineralization, erosion or leaching. Anthropogenic activities leading to soil degradation with attendant emission of CO₂ and other GHGs include deforestation, biomass burning, residue removal, drainage of wetland and cultivation of organic soils, excessive plowing, indiscriminate or inadequate use of fertilizers such as in subsistence agriculture, inappropriate use of irrigation, etc. Soil degradation is a biophysical process driven by socio-economic and political factors. It affects the SOC pool by decreasing productivity and reducing the amount of biomass C returned to the soil. It also accentuates emissions by decreasing aggregation, exposing hitherto encapsulated C to microbial processes, altering soil–water

and temperature regimes and influencing activity and species diversity of soil fauna.

Accelerated soil erosion is the principal degradative process. Together with the desertification in dry regions, vast areas of soil are affected globally by water and wind erosion (Dregne 1992; Oldeman 1994). In addition to decreasing productivity and the quantity/quality of biomass returned to the soil, soil erosion also slakes/disperses aggregates and disrupts the cycles of C, N, P, S and H₂O, and changes the energy balance of the soil. Erosion-induced alterations in these processes lead to emission of GHGs from soil to the atmosphere.

Soil erosion is a 4-step process involving (1) detachment and breakdown of aggregates, (2) transport of soil particles by runoff water or wind, (3) redistribution of eroded material over the landscape, and (4) deposition of sediments in depressional sites and protected areas. The SOC pool is concentrated in the vicinity of the soil surface and is lighter than mineral particles (density of SOC is 1.2 to 1.5 Mg/m³ compared with 2.5 to 2.7 Mg/m³ for mineral particles). Therefore, it is preferentially removed by runoff water and wind currents (Avnimelech and McHenry 1984; Lowrance and Williams 1989; Cihacek et al. 1992), although the transported material may not always be enriched in SOC. Consequently, SOC content decreases drastically on eroded compared with uneroded soils (Voroney et al. 1981; Woods 1989; Rhoton and Tyler 1990; Gregorich 1996). Depletion of the SOC pool by erosion on-site is exacerbated on plow till compared with no-till soil (Tiessen et al. 1982). In addition to depleting the total pool, erosion also affects SOC quality by removing the light fraction, which is both habitat and substrate for soil biological activity. The light fraction is deposited in depressional/protected sites where it may enhance re-aggregation of the sediments (Van Veen and Paul 1981), and protect losses against mineralization (Schimel et al. 1985; Elliott 1986). Many sedimentologists believe that SOC carried into depositional sites and aquatic ecosystems is sequestered and taken out of circulation (Stallard 1998; Smith et al. 2001). Therefore, the depositional aspect of the erosional process may lead to SOC sequestration.

While the SOC pool is depleted on-site and enriched in depositional sites, the fate of SOC en route to redistribution over the landscape and translocated to depositional sites is not known (Lal 2003a). Schlesinger (1999) argued that SOC lost during erosion is largely oxidized. In contrast, Stallard (1998)

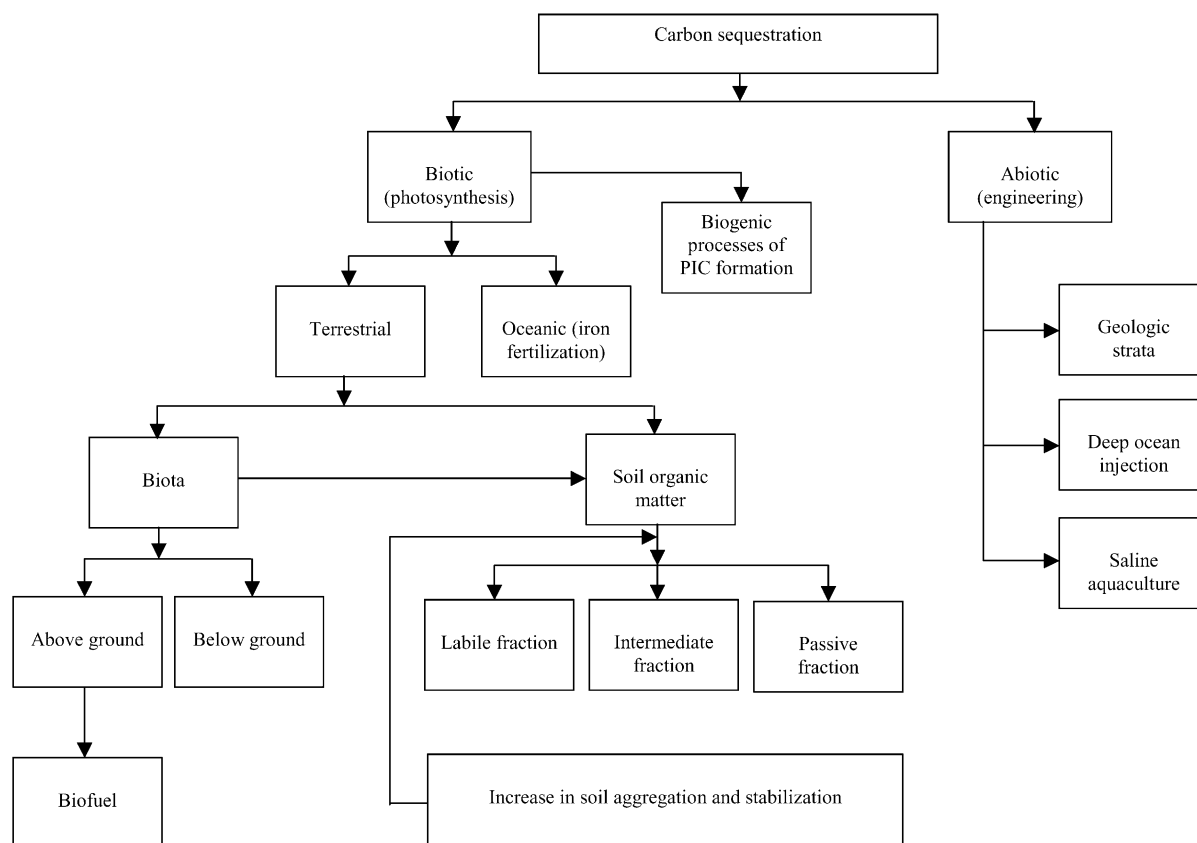


Figure 2. Mechanisms of carbon sequestration by biotic and abiotic processes.

and Smith et al. (2001) argued that SOC is largely being carried to depositional sites and aquatic ecosystems and is sequestered. Indeed, a large fraction of eroded C is being carried to the oceans (Trustrum et al. 2002). Sedimentologists argue that SOC carried into the oceans accounts for the so-called 'missing sink' for CO₂ (Tans et al. 1990). The real world situation lies somewhere in between the two extreme views. It is likely that a sizeable proportion of the SOC being transported and redistributed over the landscape is oxidized prior to its deposition in depositional/protected sites and aquatic ecosystems. Some mineralization may also happen in depositional sites. The fraction oxidized may be 20% (Lal 1995) or 20 to 30% (Jacinthe and Lal 2001). Assuming 20% mineralization of the displaced SOC, erosion-induced emission of C is estimated at 15 Tg C/yr for the USA (Lal et al. 1998) and 1.14 Pg C/yr for the world (Lal 1995).

Soil erosion is a serious problem, even in cold and humid climates such as in Iceland (Arnalds et al.

2001, 2002). Other hot spots of soil erosion include South Asia, Central Asia, the Himalayan-Tibetan ecosystem, the Caribbean, highlands of Central America, the Andean region, Australia and sub-Saharan Africa. There is a strong need to adopt conservation-effective measures in these regions, and develop a procedure to account for C emissions due to erosional processes.

Carbon sequestration in terrestrial ecosystems

Carbon sequestration implies the net removal of CO₂ from the atmosphere into long-lived pools of C, such as terrestrial and geologic. In other words, it is capturing and securely storing C by biotic (photosynthesis) and abiotic (injection into geologic strata or ocean) processes (Figure 2). Biotic processes of C sequestration are based on photosynthesis and conversion of atmospheric C into the terrestrial pool comprising biomass (above- and below-ground), soil

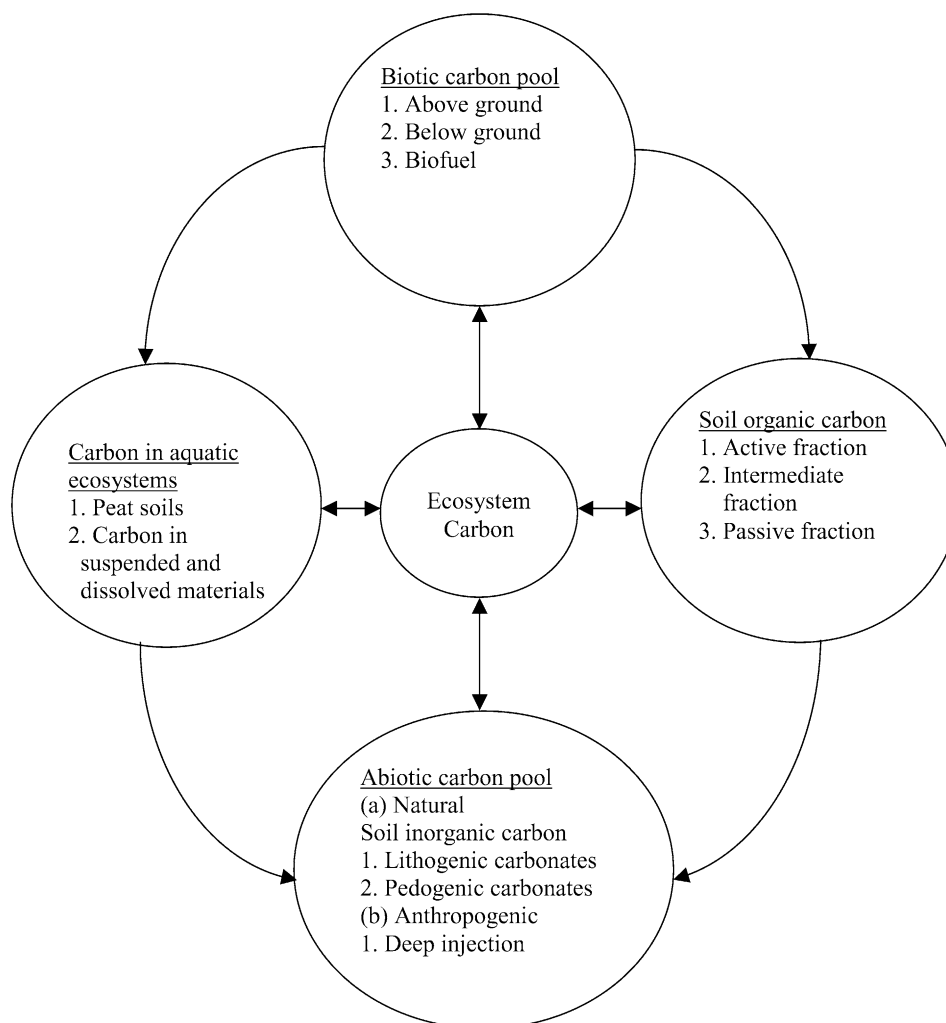


Figure 3. Components of the ecosystem carbon and interactions among them.

humus, and biogenic carbonates. Biotic sequestration also occurs in the ocean (Chisholm and Morel 1991), although its effectiveness through iron fertilization may be a myth (Chisholm et al. 2001). Enhancing the soil organic matter pool is an important strategy of biotic C sequestration. The soil organic matter has three functional fractions: labile, intermediate and passive (Parton et al. 1987). The mean residence time is 0.1 to 1.5 years for the labile fraction, 8 to 50 years for the intermediate fraction and 400 to 2200 years for the passive fraction. The passive fraction is formed through humification of the labile fraction. Formation of secondary carbonates through biogenic processes (Monger and Gallegos 2000) is also a biotic process. The abiotic sequestration involves engineering strategies of capturing and compressing CO₂

from power plants and other industrial structures, and transporting and injecting it deep into geological strata or the ocean. The abiotic process may be expensive and have unknown adverse impacts on the ecosystem. However, the sink capacity of the abiotic options is extremely large (Halman and Steinberg 1999; USDOE 1999; Battelle 2000).

In the context of biotic processes of C sequestration in terrestrial ecosystems, understanding the dynamics of ecosystem C cannot be over-emphasized. The ecosystem C is defined as 'the sum of all C in soil, biota and the wetlands'. Therefore, it comprises four distinct but inter-related components (Figure 3). Terrestrial C sequestration through biotic processes can occur in either or all of these components. Land use and management, especially agricultural and for-

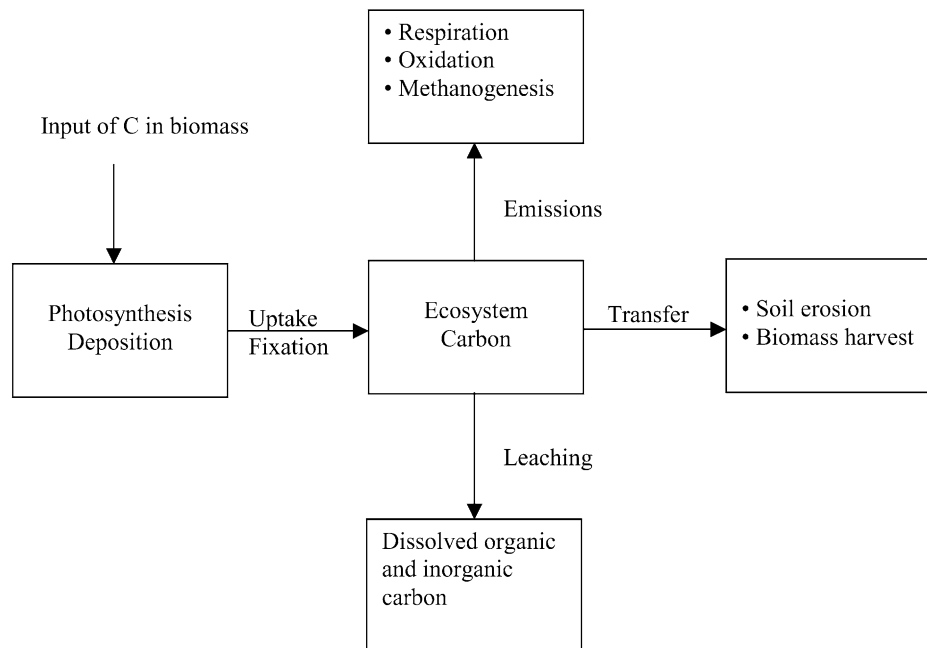


Figure 4. Dynamics of ecosystem carbon.

estry practices, play an important role in terrestrial C sequestration through biotic processes.

Agriculture (and forestry) can be defined as ‘anthropogenic manipulation of carbon through uptake, fixation, emission and transfer’:

$$CU + CF = CE + CT \quad (1)$$

where symbols U, F, E and T refer to uptake, fixation, emission and transport.

At equilibrium, C uptake and fixation equal C emission and transfer. In subsistence and resource-based farming practiced in developing countries, C emission and transfer (harvest) usually exceed C uptake and fixation. In case of slash-and-burn agriculture, large emissions occur due to deforestation and biomass burning. In developed economies where agriculture involves large off-farm input, C emissions are related to hidden C costs of fertilizers, pesticides, irrigation, tillage and traffic operations. In view of the increasing food demand globally and the threat of climate change, sustainable agricultural production must be defined in terms of the production efficiency of C:

$$I_s = \left(\frac{C_U + C_F}{C_E + C_T} \right)_t \quad (2)$$

where I_s is the index of sustainability and t is time. An I_s value of > 1 with a positive trend over time indicates a sustainable agricultural or forestry system.

Land use and management practices for terrestrial carbon sequestration

The dynamics of ecosystem C is illustrated in Figure 4. Terrestrial C sequestration can occur if the losses (emissions, erosion and leaching) are less than the gains by photosynthesis and deposition. Land use and management practices that lead to net sequestration of terrestrial C vis-à-vis those which exacerbate C emission and transfer are outlined in Table 1. The strategy is to enhance biomass production, humification of biomass returned to the soil, transfer of C deep into the sub-soil through bioturbation and deep root system development, and formation of organo-mineral complexes leading to improvement and stabilization of soil structure. Enhancing soil biodiversity (e.g., earthworms and other soil fauna) improves soil quality (Wright and Milner 1994). Earthworm density and SOC pool are highly correlated for a wide

Table 1. Land use and management options that enhance or deplete the terrestrial carbon pool and save avoidable CO₂ emission at production level of fertilizers and pesticides.

Practices which enhance fixation and uptake	Practices which accentuate emission and transfer
<p>1. Land use</p> <ul style="list-style-type: none"> • Afforestation, reforestation and establishing perennial vegetation • Retiring agriculturally marginal soils • Enhancing biodiversity 	<ul style="list-style-type: none"> • Deforestation • Conversion of natural to managed ecosystems • Arable land use
<p>2. Croplands</p> <ul style="list-style-type: none"> • Conservation tillage • Mulch farming • Integrated nutrient management involving judicious use of mineral and organic fertilizers • Precision farming • Growing cover crops • Crop rotations and forage crops • Integrated pest management that reduces hidden C costs • Erosion control with buffer strips, riparian filters, etc. 	<ul style="list-style-type: none"> • Inversion tillage • Bare soil • Low input, subsistence farming and fertility mining practices • Indiscriminate use of fertilizers with low use efficiency • Summer fallow • Continuous row crops • Indiscriminate use of pesticides with low use efficiency • No erosion management
<p>3. Grazing lands</p> <ul style="list-style-type: none"> • Controlled/rotational grazing • Improved forage species • Integrated nutrient management • Soil and water conservation • Prescribed or no burning 	<ul style="list-style-type: none"> • Excessive grazing • Traditional/native species • Little or no fertility enhancement • Little or no erosion management • Indiscriminate or excessive burning
<p>4. Forest/plantation lands</p> <ul style="list-style-type: none"> • Appropriate harvesting practices • Desired seedbed/site preparation and drainage • Growing new/improved species • Stand management with adequate weed control • Nutrient management 	<ul style="list-style-type: none"> • Indiscriminate use of heavy machinery • Lack of optimum seedbed/site preparation • Traditional/native species • Excessive or poor stand with no weed control • No soil fertility management

range of soil types and vegetation (Hendrix et al. 1992). Practices that enhance C sequestration in agricultural soils include conservation tillage, mulch farming, growing cover crops and forages in rotation with row crops, integrated nutrient management and adopting recommended agricultural practices (Lal 1999, 2000, 2001, 2002, 2003a) (Table 2).

Impact of land use and management must be assessed on the basis of net C sequestration. This implies that the gross C sequestration in terrestrial ecosystems must be adjusted for hidden C costs in all input of fertilizers, pesticides, tillage operations, etc.

$$\text{Net C sequestration} = (\text{gross terrestrial sequestration}) - (\text{hidden C costs}) \quad (3)$$

Hidden C costs are high for nitrogenous fertilizers, pesticides, and plow tillage. In addition to hidden C costs, some of these mitigation options may also lead to increased emission of N₂O and CH₄. Application of nitrogenous fertilizers may exacerbate emission of

N₂O. Similarly, application of manure, compost and other biosolids accentuates emissions of CH₄ and N₂O (Minami et al. 1994). Therefore, improving use-efficiency of these inputs is important to enhancing net C sequestration (Schlesinger 1999; Robertson et al. 2000; Smith et al. 2000).

The question of permanence must also be addressed. Once sequestered in soil through adoption of conservation tillage and other recommended management practices, how long will the C stay in the soil? On a global scale, the mean residence time (MRT) of C, as computed by dividing the pool by flux, is more in soil than in trees but less than that in ocean. At the soil/pedon or farm level, however, the permanence of C sequestered in soil depends on the continuation of the recommended management practices. Once converted from plow till, the no-till system of seedbed preparation must be continued in perpetuity. Reicosky et al. (1999) showed that even a single plowing can drastically accentuate emission of CO₂ from soil because of an increase in the rate of mineralization.

Table 2. Soil organic carbon sequestration potential in agricultural and forestry ecosystems of temperate cool climates (adapted from Lal 1999, 2000, 2004; IPCC 2001).

Activity	Rate of soil organic carbon sequestration (kg C/ha/yr)
I. Agriculture and forestry	
1. Cropland (conservation tillage, rotations, cover crops, fertility management, water management)	200–600
2. Grazing land (improved species, controlled grazing, fire management)	200–400
3. Forest land (harvesting, site preparation, improved species, nutrient management)	100–200
II. Land use conversion	
1. Restoring severely degraded soils (e.g., conservation reserve program)	200–800
2. Minesoil reclamation	200–500
3. Wetland restoration (e.g., wetland reserve program)	500–1000
III. Urban land	
1. Urban forestry	200–500
2. Recreational land	400–600

Change in soil water and temperature regimes and in the addition of quantity and quality of crop residue returned to the soil can also influence the MRT of C in soil. Any perturbation that influences soil aggregation, and activity and species diversity of soil fauna and flora can influence the MRT of C in soil.

There are several bright spots of C sequestration in terrestrial ecosystems (Figure 5). While CO₂ fertilization will enhance biomass production in general (Kimball et al. 2002), C sequestration can be enhanced by restoration of degraded ecosystems (biota, soils, wetlands) and land use change. Degraded ecosystems have lost a large proportion of their native C pool and the present pool is much below the potential capacity. Such ecosystems include soils degraded by severe water and wind erosion, salinization, nutrient depletion, compaction, contamination and pollution, and drastic disturbance by mining activities. In addition to soil, desertified ecosystems have also suffered C loss because of the degradation of vegetation. About one-half to two-thirds of the C pool of agricultural ecosystems has been lost. Most degraded and agricultural ecosystems may have lost 50 to 60 Mg C/ha, and a high proportion of the historic C loss may be sequestered through a change in land use and adoption of recommended management practices. Restoring wetlands has a large potential of C sequestration, because erosion of topsoil and organic matter from upland catchment areas is deposited in wetlands and the decomposition rate is slow (Gleason and Euliss 1998). Emission of CO₂ from managed peat soils can be decreased by raising

the water table. The strategy is to balance CH₄ emissions from peatlands with CO₂ emission from drained wetlands, so that the wetlands can be a net sink rather than the source. Similarly, desertification control through water resources development, erosion control, and revegetation (Hillel and Rosenzweig 2002) has a large potential of C sequestration (Squire et al. 1995; Lal 2001).

Soil quality and soil organic carbon pool

There is a strong link between soil resilience and SOC pool, and soil quality and SOC pool. Soil resilience refers to the soil's ability to restore its processes and properties following a major perturbation or stress. Soil resilience (S_r) is a function of numerous soil properties including SOC concentration:

$$S_r = f(\text{SOC}, S_i, R_d, e_d, N_r, B_d)_t \quad (4)$$

where SOC is the soil organic carbon concentration and quality, S_i is the structural index of soil, R_d is the rooting depth, e_d is the charge density, N_r is the improvement in nutrient reserve, B_d is soil biodiversity and t is the time (Lal 1997). There is a critical/threshold level of SOC concentration below which soil may lose its resilience and never recover.

Similar to soil resilience, soil quality is also related to the amount and composition of the SOC pool (Doran and Jones 1996; Doran et al. 1998; Karlen and Andrews 2000; Singer and Ewing 2000). Soil quality

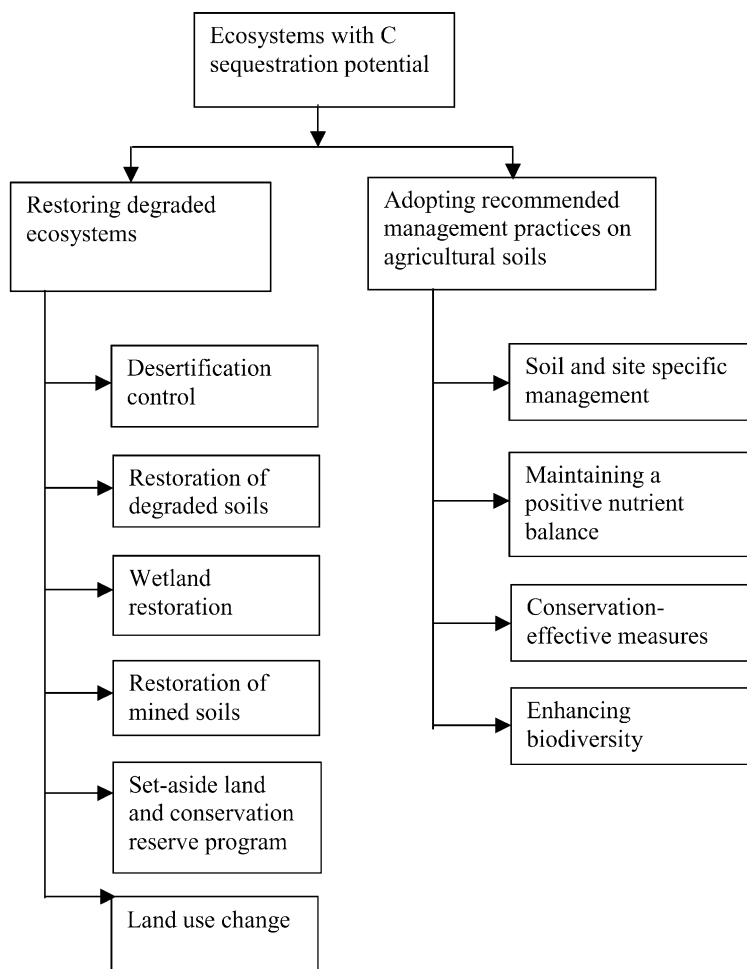


Figure 5. Bright spots of carbon sequestration in terrestrial ecosystems.

refers to biomass productivity and environmental moderation capacity. There is a close inter-relationship between soil quality and soil resilience; resilient soils have high soil quality and vice versa. Similar to soil resilience, therefore, soil quality (S_q) is also a function of numerous soil properties including SOC concentration:

$$S_q = f(\text{SOC}, R_d, K, N_r, \text{AWC}, C_1)t \quad (5)$$

where N_r is the nutrient reserve, AWC is the plant available capacity, K is soil erodibility and C_1 is the clay content and type. There is a critical limit of SOC content (and other soil properties) below which soil may be irreversibly degraded. Aune and Lal (1995) observed that the critical limit of SOC concentration for most tropical soils with regard to crop yield and

response to fertilizer input is 1.1%. The critical limit for irreversible soil degradation may be considerably less. There may, however, be subtle differences in critical limits with regard to the onset of degradative processes and threshold values for soil resilience, and these values are often not the same. Because of a strong inter-dependence on soil quality and SOC concentration, those land use and management practices that enhance the SOC pool also enhance soil quality.

There is a strong relationship between SOC concentration and crop yield for low-input agricultural systems. For example, Petchawee and Chaitep (1995) observed a strong correlation between soil organic matter content and maize grain yield in Thailand ($y = 2.91x + 0.21$, $r^2 = 0.88$, where y is in Mg/ha and x is in%). In this case, maize grain yield increased by

2.91 Mg/ha for every 1% increase in soil organic matter content. Data from a long-term Sanborn experiment also showed a close relationship between decline in soil organic matter content and agronomic productivity (Mitchell et al. 1991). In addition to improvement in availability of plant nutrients, an increase in SOC concentration also improves plant available water capacity (Salter and Hawroth 1961; Hollis et al. 1977; Hudson 1994; Emerson 1995). Increase in soil organic matter enhances soil water retention because of its hydrophilic properties and its ability to form aggregates and improve soil structure (Hamblin and Davis 1977; Tisdall and Oades 1982; Haynes and Beare 1996; Haynes and Naidu 1998; Kay 1998). In accord with improvement in water retention and soil physical properties, enhancement of SOC concentration also decreases the risks of soil erosion, decreases sedimentation and improves water quality (Lal 2003b).

Soil carbon and other elemental cycles

There exists a close link between soil C and components of the hydrologic cycle (Lucas and Vitosh 1978). Increase in SOC concentration enhances soil water retention capacity, improves soil structure and water infiltration capacity, and decreases runoff and soil erosion. Therefore, components of the hydrological cycle strongly affected by changes in the SOC pool are soil water storage, infiltration rate, runoff and evapotranspiration:

$$ET = P - (\Delta S + I + R + D) \quad (6)$$

where ET is evaporation, P is precipitation, ΔS is change in soil water storage, I is infiltration, R is surface runoff and D is deep drainage. Increase in the SOC pool through soil C sequestration increases ΔS , I and D, and decreases R components of the hydrologic cycle. Therefore, an increase in SOC would lead to an increase in ET or the proportion of the precipitation utilized by plants for their growth and biomass production.

Similar to water, there is also a close link between the SOC pool and the amount of N, P, S and other elements in the soil (Stevenson 1982, 1986). The turnover of N in soil is closely associated with that of C (Jenkinson 1990). The biogeochemical cycles of most elements (e.g., N, P, S, Cu, Mn, Zn) are medi-

ated by micro-organisms. Decomposition of biomass in soil transforms N, P, S and other micro-nutrients into a form that can be readily absorbed by plant roots. Elements thus absorbed are resynthesized into microbial tissue and plant biomass.

Carbon is only one of the building blocks of humus. Other important building blocks of humus include N, P, S, Cu, Zn, Mn and other minor elements. Thus, transformation of C contained in the biomass requires the availability of all these elements. The elemental ratio of C:N decreases from 30–70 in crop residue to 10–12 in humus, C:S from 200–400 in residue to 40–60 in humus, and C:P from 100–400 in residue to 60–80 in humus (Allison 1973; Stevenson 1982; Jenkinson 1988). Based on these ranges of elemental ratios, Himes (1998) estimated that sequestration of 10,000 kg of C contained in crop residues would require 833 kg of N, 200 kg of P and 143 kg of S. Such a transformation would increase the SOC concentration in the plow layer of 20-cm depth by 0.7%. Because of the strong linkages between C and other major (N, P, S) and minor (Cu, Zn, Mn, Fe, Mo, B) elements, the challenge of soil C sequestration is to make available C and other essential elements over and above the needs for producing good crop yields.

The Kyoto protocol and terrestrial carbon sequestration

The Kyoto Protocol (Oberthür and Ott 2000) provides for C sequestration in terrestrial ecosystems. The meeting in July 2001 in Bonn stipulated that: (1) countries are allowed to subtract from their industrial C emissions certain increases in C sequestered in sinks such as forest and soils, and (2) Annex 2 countries are allowed to trade C with Annex 1 countries under the Clean Development Mechanism. Therefore, C sequestration in terrestrial ecosystems is relevant to both developed and developing countries.

Is terrestrial C sequestration adequate to mitigate climate change? No, terrestrial C sequestration (in soil and biota) has a finite capacity. Yet, soil C sequestration is the most efficient and natural strategy during the first few decades of the 21st century (Battelle 2000).

In the long run, reducing the rate of enrichment of atmospheric concentration of GHGs necessitates finding sustainable alternatives to fossil fuels. At present, 80% of the global power production comes from fossil and nuclear fuels. Terrestrial C sequestra-

tion provides a 'window of opportunity' between now and 2050, to develop renewable sources of energy. Clean and inexhaustible renewable energy (e.g., biomass, solar, wind, ecothermal) capacity exists in all continents. While these energy sources are being developed, terrestrial C sequestration can reduce the rate of enriching the atmospheric concentration of GHGs. It is a win-win strategy.

Conclusions

Agriculture has an important role to play towards an effort to mitigate the climate change due to atmospheric enrichment of CO₂ and other greenhouse gases. Contrary to the common perception, scientific agriculture can be a solution to environmental issues but especially to reducing the rate of enrichment of CO₂ in the atmosphere. Recommended management practices include conversion from plow till to no-till, incorporation of cover crops and forages in the crop rotation, liberal use of crop residues and biosolids like mulch, integrated nutrient management including compost/manures and judicious use of fertilizers, integrated pest management, etc. There are hidden C costs of fertilizers and pesticides, and risks of increasing emissions of N₂O and CH₄. Therefore, enhancing use efficiency of these inputs is important.

Restoration of degraded soils and ecosystems is an important strategy of SOC sequestration. Most degraded soils have lost a large fraction of the original SOC pool, and it can be enhanced through restorative measures. Soil erosion by water and wind is the most widespread degradative process on a global scale, and adoption of conservation-effective measures can curtail erosion-induced emissions and replenish the depleted SOC pool. Alleviation of soil-related constraints to biomass production can also improve soil quality and increase the SOC pool. Indeed, there exists a close inter-relationship between SOC pool and soil quality and resilience.

The permanence of soil C sequestered depends on the continuity of the recommended practices adopted. Rotational plowing, as is commonly practiced in the U.S. Corn Belt, can lead to depletion of the SOC sequestered. With continuation of recommended management practices, however, the sequestered C stays for a relatively long time in the soil pool and decreases the rate of enrichment of atmospheric CO₂ concentration.

The SOC sequestration has numerous ancillary benefits. Yet, the potential of SOC sequestration is finite. Therefore, it is a short-term solution. The long-term solution to climate change lies in developing carbon-neutral fuel sources.

References

- Allison F.E. 1973. *Soil Organic Matter and Its Role in Crop Production*. Elsevier Scientific Publishing Co., New York.
- Arnalds O., Porarinsdottir E.F., Metusalemsson S., Jonsson A. and Arnason E.G.O.A. 2001. *Soil Erosion in Iceland*. Soil Conservation Service, Agricultural Research Institute, Reykjavik, Iceland, 121 pp.
- Arnalds O., Aradottir A.L. and Gudbergsson G. 2002. Organic carbon sequestration by restoration of severely degraded areas in Iceland. In: Kimble J.M., Lal R. and Follett R.F. (eds), *Agricultural Practices and Policies for Carbon Sequestration in Soil*. Lewis Publishers, Boca Raton, FL, pp. 267–280.
- Aune J.B. and Lal R. 1995. The tropical soil productivity calculator – A model for assessing effects of soil management on productivity. In: Lal R. and Stewart B.A. (eds), *Soil Management: Experimental Basis for Sustainability and Environment Quality*. Lewis Publishers, Boca Raton, FL, pp. 499–520.
- Avnimelech Y. and McHenry J.R. 1984. Enrichment of transported sediments with organic carbon, nutrients and clay. *Soil Sci. Soc. Am. J.* 48: 259–266.
- Baede A.P.M. 2001. The climate system: An overview. In: *Inter-government Panel on Climate Change, Climate Change 2001: The Scientific Basis*. Cambridge University Press, Cambridge, UK, pp. 85–198.
- Batjes N.H. 1996. Total C and N in the soils of the world. *Eur. J. Soil Sci.* 47: 151–163.
- Battelle 2000. *Global Energy Technology Strategy: Addressing Climate Change*. Battelle, Washington, DC, 60 pp.
- Chisholm M.S.W. and Morel F.M.M. 1991. What controls phytoplankton production in nutrient-rich areas of the open sea. *American Society Limnology and Oceanography Symposium* 22–24 February 1991, San Marcos, CA, *Limn. Ocean.* 36: 41507–41511.
- Chisholm M.S.W., Falkowski P.G. and Cullen J.J. 2001. Discrediting ocean fertilization. *Science* 294: 309–310.
- Cihacek L.J., Sweeney M.D. and Deibert E.J. 1992. Characterization of wind erosion sediments in the Red River Valley of North Dakota. *J. Environ. Qual.* 22: 305–310.
- Davidson E.A. and Ackerman I.L. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* 20: 161–193.
- Doran J.W. and Parkin T.B. 1996. Quantitative indications of soil quality. In: *Methods for Assessing Soil Quality*. Soil Science Society of America Special Publication #49. SSSA, Madison, WI, pp. 25–38.
- Doran J.W. and Jones A.J. 1996. *Methods for Assessing Soil Quality*. Soil Science Society of America Special Publication #49. SSSA, Madison, WI.
- Doran J.W., Jones A.J., Arshad M.A. and Gilley J.E. 1998. Determinants of soil quality and health. In: Lal R. (ed.), *Soil Quality and Soil Erosion*. CRC Press, Boca Raton, FL, pp. 17–36.

- Dregne H.E. (ed.) 1992. Degradation and Restoration of Arid Lands. Texas Technical University, Lubbock, TX.
- Elliott E.T. 1986. Aggregate structure, and carbon, nitrogen and phosphorus in native and cultivated soils. *Soil Sci. Soc. Am. J.* 50: 627–633.
- Emerson W.W. 1995. Water retention, organic carbon and soil texture. *Austr. J. Soil Res.* 33: 241–251.
- Eswaran H., Van den Berg E., Reich P. and Kimble J.M. 1995. Global soil C resources. In: Lal R., Kimble J., Levine E. and Stewart B.A. (eds), *Soils and Global Change*. Lewis Publishers, Boca Raton, FL, pp. 27–43.
- Etheridge D.M., Steele L.P., Francey R.J. and Langenfelds R.L. 1998. Atmospheric methane between 1000 AD and present: Evidence of anthropogenic emissions and climatic variability. *J. Geophys. Res.* 103: 15979–15993.
- Gleason R.A. and Euliss N.H. Jr. 1998. Sedimentation on prairie wetlands. *Great Plains Res.* 8: 97–112.
- Gregorich E.G. 1996. Soil quality: A Canadian Perspective. In: Cameron K.C., Cornforth I.S., McLaren R.G., Beare M.H., Basher L.R., Metherell A.K. and Kerr L.E. (eds), *Soil Quality Indicators for Sustainable Agriculture in New Zealand*. Proceedings of Workshop. Lincoln University, Christchurch, New Zealand, pp. 40–52.
- Gregorich E.G., Carter M.R., Doran J.W., Pankhurst C.E. and Dwyer L.M. 1997. Biological attributes of soil quality. In: Gregorich E.G. and Carter M.R. (eds), *Soil Quality for Crop Production and Ecosystem Health*. Elsevier, Amsterdam, The Netherlands, pp. 81–113.
- Gregorich E.G., Greer K.J., Anderson D.W. and Liang B.C. 1998. Carbon distribution and losses: erosion and deposition effects. *Soil Tillage Res.* 47: 291–302.
- Halmann M.M. and Steinert M. 1999. *Greenhouse Gas Carbon Dioxide Mitigation: Science and Technology*. Lewis Publishers, Boca Raton, FL, 568 pp.
- Hamblin A.P. and Davies D.B. 1977. Influence of organic matter on the physical properties of some east Anglian soils of high silt content. *J. Soil Sci.* 28: 11–22.
- Haynes R.J. and Beare M.H. 1996. Aggregation and organic matter storage in meso-thermal humid soils. In: Carter M.R. and Stewart B.A. (eds), *Structure and Organic Matter in Agricultural Soils*. CRC/Lewis Publishers, Boca Raton, FL, pp. 213–261.
- Haynes R.J. and Naidu R. 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. *Nutr. Cycl. Agroecosyst.* 51: 139–153.
- Hendrix P.F., Mueller B.R., Bruce R.R., Langdale G.W. and Parmelee R.W. 1992. Abundance and distribution of earthworms in relation to landscape factors on the Georgia Piedmont, U.S.A. *Soil Biol. Biochem.* 24: 1357–1361.
- Hillel D. and Rosenzweig C. 2002. Desertification in relation to climate variability and change. *Adv. Agron.* 77: 1–38.
- Himes F.L. 1998. Nitrogen, sulfur and phosphorus and the sequestering of carbon. In: Lal R., Kimble J.M., Follett R.F. and Stewart B.A. (eds), *Soil Processes and the Carbon Cycle*. CRC/Lewis Publishers, Boca Raton, FL, pp. 315–319.
- Hollis J.M., Jones R.J.A. and Palmer R.C. 1977. The effects of organic matter and particle size on the water retention properties of some soils in the West Midlands of England. *Geoderma* 17: 225–231.
- Hudson B.D. 1994. Soil organic matter and available water capacity. *J. Soil Water Conserv.* 49: 189–193.
- Intergovernment Panel on Climate Change 2001. *Climate Change 2001: The Scientific Basis. Summary for Policy Makers*. IPCC, Cambridge University Press, Cambridge, UK, pp. 1–21.
- Jacinthe P.A. and Lal R. 2001. A mass balance approach to address carbon dioxide evolution during erosional events. *Land Degrad. Dev.* 12: 329–339.
- Jenkinson D.S. 1988. Soil organic matter and its dynamics. In: Wild A. (ed.), *Russell's Soil Conditions and Plant Growth*. Longman, Essex, UK, pp. 505–561.
- Jenkinson D.S. 1990. The turnover of organic carbon and nitrogen in soil. *Phil. Trans. Royal Soc. London (B)* 329: 361–368.
- Jenny H. 1980. *The Soil Resource: Origin and Behavior*. Springer, New York, 377 pp.
- Karlen D.L. and Andrews S.S. 2000. The soil quality concept A tool for evaluating sustainability. In: Elmholt S., Stenberg B., Gronlund A. and Nuutinen V. (eds), *Soil Stresses, Quality and Care*. DIAS Rep. #38, Danish Inst. Agric. Su., Tjele, Denmark.
- Kay B.D. 1998. Soil structure and organic carbon: a review. In: Lal R., Kimble J.M. and Stewart B.A. (eds), *Soil Processes and the Carbon Cycle*. CRC Press, Boca Raton, FL, pp. 169–197.
- Kimball B.A., Kobayashi K. and Bindi M. 2002. Response of agricultural crops to free-air CO₂ enhancement. *Adv. Agron.* 77: 293–368.
- Lal R. 1995. Global soil erosion by water and carbon dynamics. In: Lal R., Kimble J.M., Levine E. and Stewart B.A. (eds), *Soils and Global Change*. CRC/Lewis Publishers, Boca Raton, FL, pp. 1–34.
- Lal R. 1997. Degradation and resilience of soils. *Phil. Trans. Royal Soc. London (B)* 352: 997–1010.
- Lal R. 1999. Soil management and restoration for C sequestration to mitigate the accelerated greenhouse effect. *Prog. Env. Sci.* 1: 307–326.
- Lal R. 2000. World cropland soils as a source or sink for atmospheric carbon. *Adv. Agron.* 71: 145–191.
- Lal R. 2001. Potential of desertification control to sequester carbon and mitigate the greenhouse effect. *Clim. Change* 15: 35–72.
- Lal R. 2002. The potential of soils of the tropics to sequester carbon and mitigate the greenhouse effect. *Adv. Agron.* 74: 155–192.
- Lal R. 2003a. Soil erosion and the global carbon budget. *Env. Intl.* 29: 437–450.
- Lal R. 2003b. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* 22: 151–184.
- Lal R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304: 1623–1627.
- Lal R., Kimble J.M., Follett R.F. and Cole C.V. 1998. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*. Ann Arbor Press, Chelsea, MI, 128 pp.
- Lowrance R. and Williams R.G. 1989. Carbon movement in runoff and erosion under simulated rainfall conditions. *Soil Sci. Soc. Am. J.* 52: 1445–1448.
- Lucas R.E. and Vitosh M.L. 1978. *Soil organic matter dynamics*. Research Report 358. Michigan State University, East Lansing, MI.
- Minami K., Mosier A. and Sass R. (eds) 1994. *CH₄ and N₂O. Global Emissions and Controls from Rice Fields and other Agricultural and Industrial Sources*. National Institute Agro-Environmental Sciences, Tsukuba, Japan, 234 pp.
- Mitchell C.C., Weserman R.L., Brown J.R. and Peck T.R. 1991. Overview of long-term agronomic research. *Agron. J.* 83: 24–29.

- Monger H.C. and Gallegos R.A. 2000. Biotic and abiotic processes and rates of pedogenic carbonate accumulation in the southwestern United States – Relationship to atmospheric CO₂ sequestration. In: Lal R., Kimble J.M., Eswaran H. and Stewart B.A. (eds), *Global Climate Change and Pedogenic Carbonates*. CRC/Lewis Publishers, Boca Raton, FL, pp. 273–290.
- Oberthür S. and Ott H.E. 2000. *The Kyoto Protocol: International Climate Policy for the 21st Century*. Springer, Berlin, Germany.
- Oldeman L.R. 1994. The global extent of soil degradation. In: Greenland D.J. and Szabolcs I. (eds), *Soil Resilience and Sustainable Land Use*. CAB International, Wallingford, UK, pp. 99–118.
- Parton W.J., Schimel D.S., Cole C.V. and Ojima D.S. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Sci. Soc. Am. J.* 51: 1173–1179.
- Petchawee S. and Chaitep W. 1995. Organic matter management for sustainable agriculture. In: Lefroy R.D.B., Blair G.J. and Craswell E.T. (eds), *Organic Matter Management in Upland Systems in Thailand*. ACIAR, Canberra, Australia, pp. 21–26.
- Prather M. and Ehhalt D. 2001. Atmospheric chemistry and greenhouse gases. In: *Intergovernment Panel on Climate Change, Climate Change 2001: The Scientific Basis*. Cambridge University Press, Cambridge, UK, pp. 239–287.
- Ramaswamy V. 2001. Radiative forcing of climate change. In: *Intergovernment Panel on Climate Change, Climate Change 2001: The Scientific Basis*. Cambridge University Press, Cambridge, UK, pp. 349–416.
- Reicosky D.C., Reeves D.W., Prior S.A., Runion G.B., Rogers H.H. and Raper R.L. 1999. Effects of residue management and controlled traffic on carbon dioxide and water loss. *Soil Tillage Res.* 52: 153–165.
- Rhoton F.E. and Tyler D.D. 1990. Erosion-induced changes in the properties of a Fragipan soil. *Soil Sci. Soc. Am. J.* 54: 223–228.
- Robertson G.P., Paul E.A. and Harwood R.R. 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *Science* 289: 192–194.
- Salter P.J. and Haworth F. 1961. The available water capacity of sandy loam soil. II. The effects of farmyard manure and different primary cultivations. *J. Soil Sci.* 12: 335–342.
- Schimel D.S., Coleman D.C. and Horton K.A. 1985. Soil organic matter dynamics in paired rangeland and cropland toposequences in North Dakota. *Geoderma* 36: 201–214.
- Schimel D.S., House J.I. and Hubbard K.A. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature* 414: 169–172.
- Schlesinger W.H. 1999. Carbon sequestration in soil. *Science* 284: 2095.
- Singer M.J. and Ewing S. 2000. Soil quality. In: Sumner M.E. (ed.), *Handbook of Soil Science*. CRC Press, Boca Raton, FL, pp. G271–G278.
- Smith W.N., Desjardins R.L. and Pattey E. 2000. The net flux of carbon from agricultural soil in Canada. *Global Change Biol.* 6: 557–568.
- Smith S.V., Renwick W.H., Buddenmeier R.W. and Crossland C.J. 2001. Budgets of soil erosion and deposition of sediments and sedimentary organic carbon across the conterminous United States. *Global Biogeochem. Cycles* 15: 697–707.
- Squire V.R., Glenn E.P. and Ayoub A.T. (eds) 1995. *Combating global climate change by combating land degradation*. UNEP, Nairobi, Kenya, 347 pp.
- Stallard R.F. 1998. Terrestrial sedimentation and the carbon cycle: coupling weathering and erosion to carbon burial. *Global Biogeochem. Cycles* 12: 231–257.
- Stevenson F.J. 1982. *Humus Chemistry. Genesis, Composition, Reactions*. John Wiley and Sons, New York.
- Stevenson F.J. 1986. *Cycles of Soil: Carbon, Nitrogen, Phosphorus, Sulfur and Micronutrients*. John Wiley and Sons, New York, 380 pp.
- Tans P.P., Fung I.Y. and Takahashi T. 1990. Observational constraints on the global atmospheric CO₂ budget. *Science* 247: 1431–1438.
- Tiessen H., Stewart J.W.B. and Betany J.R. 1982. Cultivation effects on the amount and concentration of carbon, nitrogen and phosphorus in grassland soils. *Agron. J.* 74: 831–834.
- Tisdall J.M. and Oades J.M. 1982. Organic matter and water stable aggregates in soils. *J. Soil Sci.* 33: 141–163.
- Trustrum N.A., Tate K.R., Page M.J., Sidorchuk A. and Baisden W.T. 2002. Towards a national assessment of erosion-related soil carbon losses in New Zealand. 12th ISCO Conference, 26–31 May, Beijing, China, Vol. 3: pp. 182–186.
- US-DOE 1999. *Carbon Sequestration Research and Development*. National Technical Information Service, Springfield, VA.
- Van Veen J.A. and Paul E.A. 1981. Organic carbon dynamics in grassland soils. 1. Background information and computer simulation. *Can. J. Soil Sci.* 61: 185–201.
- Voroney R.P., Van Veen J.A. and Paul E.A. 1981. Organic C dynamics in grassland soils. 2. Model validation and simulation of the long-term effect of cultivation and rainfall erosion. *Can. J. Soil Sci.* 61: 211–224.
- Woods L.E. 1989. Active organic matter distribution in the surface 15-cm of undisturbed and cultivated soil. *Biol. Fertil. Soils* 8: 271–278.
- Wright S.F. and Milner P.D. 1994. Earthworms and other fauna in the soil. In: Hatfield J.L. and Stewart B.A. (eds), *Soil Biology: Effects of Soil Quality*. CRC/Lewis Publishers, Boca Raton, FL, pp. 29–60.