

Review

Carbon sequestration in soil

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Abstract

Carbon (C) sequestration in soil implies transfer and secure storage of atmospheric CO₂ into the soil organic carbon (SOC) pool as recalcitrant humus/biochar and into the soil inorganic carbon (SIC) pool as secondary carbonates. Its importance lies in the urgent need to offset increases in atmospheric enrichment of CO₂ (from 280 ppm in 1750 to 385 ppm in 2008) and its benefits to agronomic yield and soil quality. The soil C sink capacity, created by historic land use and soil degradation, is estimated at 78 ± 12 Pg or 10–60 Mg/ha (Lal, 1999). Principal strategies of SOC sequestration involve: (i) restoration of degraded/desertified soils through conversion to a perennial land use, and (ii) adoption of recommended management practices including no-till farming, manuring, agroforestry and use of biochar as a soil amendment. The mean rate of C sequestration is 300–500 kg/ha/yr for SOC and 2–10 kg/ha/yr for SIC. Accelerated soil erosion is a net source of atmospheric CO₂ and must be effectively controlled. Soil C sequestration is also enhanced by adoption of nanotechnology, biotechnology, information technology and trading of C credits. Avoidance of deforestation, and afforestation of degraded/desertified soils are cost-effective and have a large potential to offset emissions, influence the global C cycle and stabilize the atmospheric CO₂. Soil C sequestration is a win–win–win strategy because it advances food security, improves the environment and mitigates global warming.

Keywords: Trading of C credits, Global warming, Biochar, Soil quality, Soil restoration, Global C cycle, Erosion and C cycle

Introduction

The risks of global warming [1] have raised interest in identifying potential sinks for atmospheric CO₂, and strategies to sequester carbon (C) in these sinks. The characteristics of a good C sequestration system include: (i) high sink capacity or the amount of CO₂ that it can absorb, (ii) long residence time or a high sink stability, (iii) low cost, (iv) positive environmental impact or minimal negative impact and (v) numerous ancillary benefits. Indeed, the annual increase in atmospheric abundance of CO₂ depends on the balance between global C sources and sinks. Principal sources of CO₂ are fossil fuel combustion and land use change, and the sinks are the ocean,

land and atmosphere. The schematic diagram and data in Figure 1 indicate two significant points:

- (1) Of the total anthropogenic emissions, only 40–45% of CO₂ remains in the atmosphere because oceanic and terrestrial sinks combined absorb the rest even without human intervention.
- (2) The capacity of the natural sinks, and especially those of terrestrial sinks, has declined during the 2000s (Figure 1; [5]) probably because of increase in soil degradation and desertification. For example, the capacity of the natural sinks was 56.3% in the 1980s, 60.0% in the 1990s and 54.9% in the 2000s.

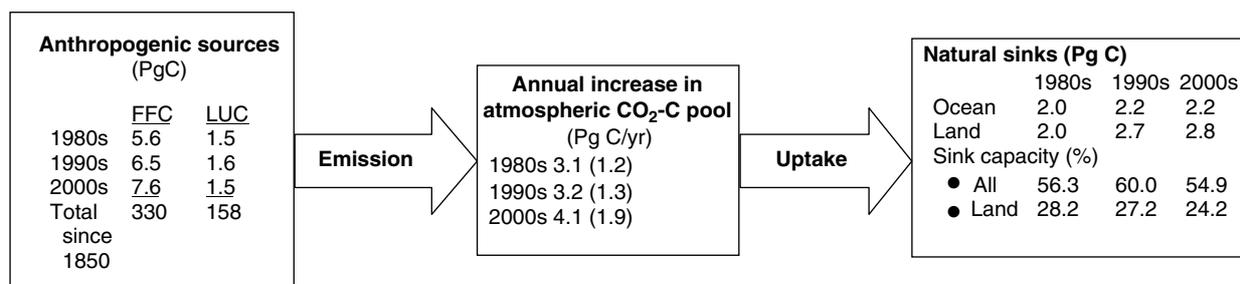


Figure 1 Estimates of anthropogenic sources and sinks, and annual increase in atmospheric C pool. The data (in parentheses) are the annual increase (ppm/yr) in atmospheric CO₂ (data are based on [1–6])

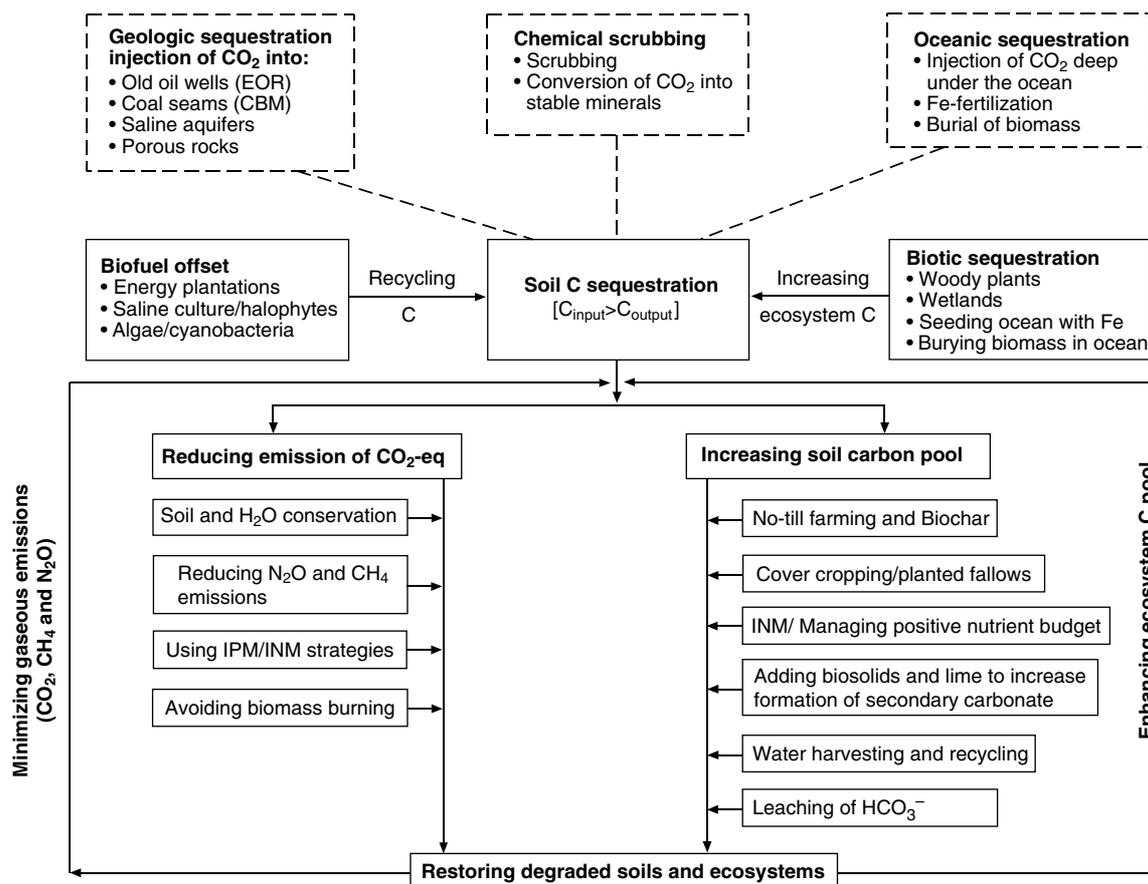


Figure 2 Processes and practices of C sequestration in soil vis-à-vis other global C pools

Of the total capacity of natural sinks, the capacity of the terrestrial biosphere (land sinks) was 28.2% in the 1980s, 27.2% in the 1990s and 24.2% in the 2000s. The process of C sequestration implies the transfer and secure storage of atmospheric CO₂ into other long-lived C pools through natural or managed processes. Because of the high risks of global warming, there is strong interest in accentuating the process through transfer of atmospheric CO₂ into oceanic, geological, biotic and pedological pools. The wide range of options, including engineering and biotic, are outlined in Figure 2 and have been discussed elsewhere (e.g. [6]). Specifically, C sequestration in soil is the transfer and secure storage of atmospheric CO₂ into the

pedological/soil pools, comprising the soil organic carbon (SOC) and soil inorganic carbon (SIC) components. Therefore, the objective of the present review is to describe the processes, factors, causes, strategies and impacts of C sequestration in soils of natural and managed ecosystems.

Soil Carbon Pool

The pedological or soil C pool is the third largest of the five global pools (Figure 3). Its magnitude of 2500 Pg to 1 m depth is about 3.3 times the atmospheric pool

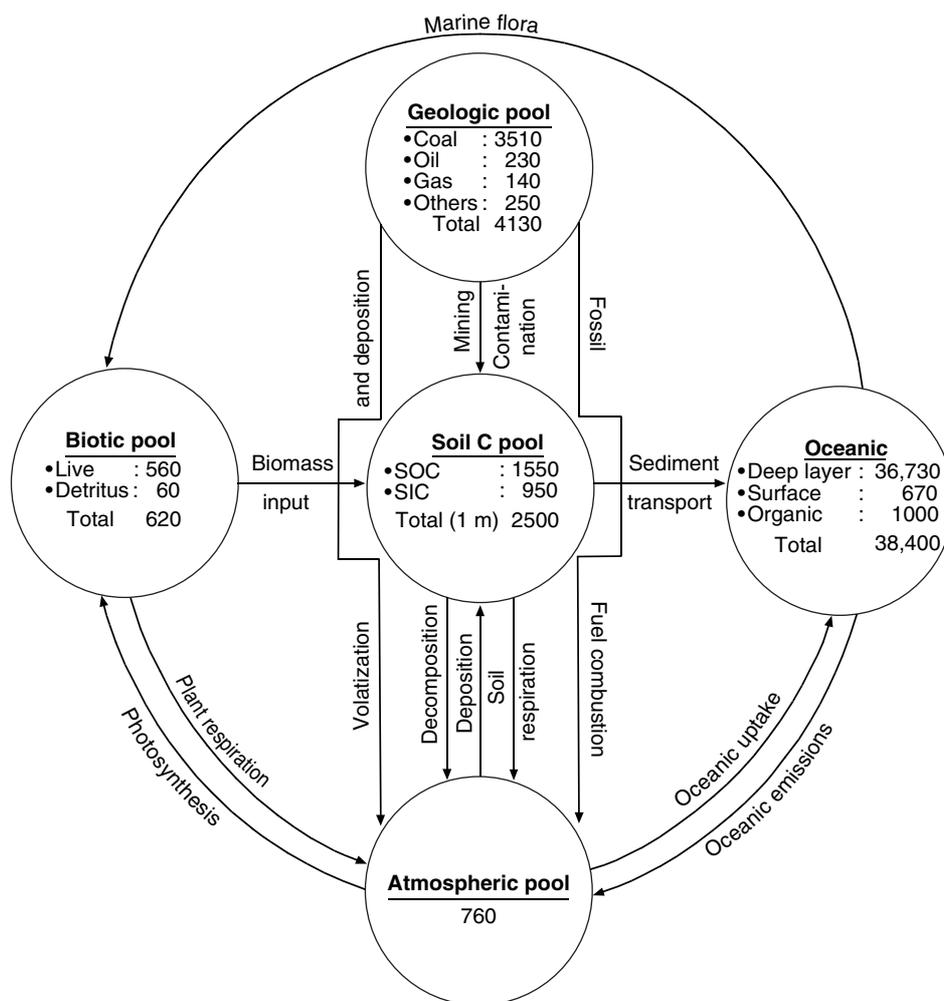


Figure 3 The soil carbon pool and its interaction with other global carbon pools. All figures are in Pg of C (data on pools are from [7–9])

of 760 Pg and 4.0 times the biotic pool of 620 Pg. The soil C pool consists of two distinct components: an SOC pool of about 1550 Pg and an SIC pool of 950 Pg, both to 1 m depth [7, 10]. The SOC pool comprises any organic C assembly, large or small, dead or alive, including microbial biomass [11], and consists of the following constituents [12, 13]: (i) small amounts of plant and animal tissues as remains of the original biomass input, (ii) the products of the biological and chemical decomposition of the biomass addition to the soil, (iii) living and dead microbial cells, (iv) degradation of soil organisms, and (v) interaction products of any or all of these substances. Many of these substances are of colloidal nature and occur in close interaction, both within and outside, of the clay lattices. The SOC pool is often referred interchangeably as soil organic matter (SOM) or humus. While the SOM consists of all five components listed above, humus is a dark brown or black amorphous material. It is a highly decomposed component of the SOM, and is characterized by a large surface area, high charge density, high reactivity and high affinity for the clay fraction. Humic

substances are dark-brown organic macromolecules rich in phenolic compounds and are derived from plant remains and microbial synthesis [14]. It is highly dynamic, being simultaneously formed from plant and animal residues, and also decomposed by microbial processes [15, 16]. It is the process of decomposition that is the source of energy for all biological processes in soil [17–22].

The SIC pool, an important component in soils of dry climates, includes elemental C, primary carbonates (e.g. calcite, dolomite and gypsum) and secondary carbonates. The primary carbonates are derived from the weathering of the parent material. In comparison, secondary carbonates are derived from the dissolution of CO_2 in the soil air and the reaction of the weak carbonic acid with Ca^{2+} , Mg^{2+} and other cations brought into the system through atmospheric deposition, runoff, irrigation water, manure and other amendments, and weathering of minerals. Thus, soil C sequestration involves conversion of atmospheric CO_2 to the soil C pool through: (i) addition of biosolids (live or dead) to the soil (the biosolids eventually go

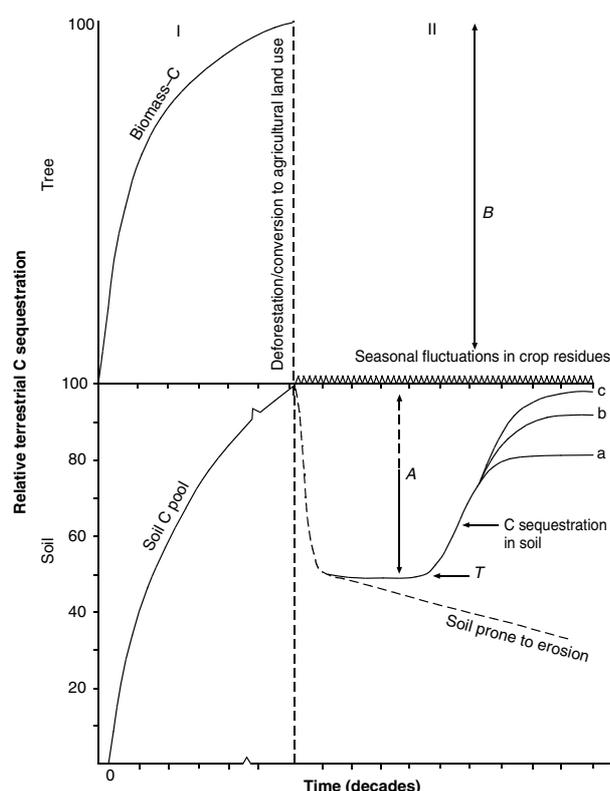


Figure 4 The ecosystem C pool (trees and soil) captured under the natural vegetation cover (I) and depleted with conversion to agricultural ecosystems (II). The sum of A (soil) and B (trees) represents the potential ecosystem sink capacity. Adoption of RMPs at time T commences C sequestration in soil

through the process of humification) and (ii) formation of secondary carbonates.

The SOC pool in wetland soils is also an important component of the global soil C pool. The total area of organic soils (histosols) is 15 Mha in the USA and 153 Mha in the world. The SOC pool in histosols is estimated at 54 Pg. Most soils of the tundra and boreal regions, and also predominately wetland soils comprising mostly peat, contain large SOC pools. Soils of tundra (permafrost) and boreal regions contain SOC pools of 393 and 382 Pg, respectively [7, 10]. Global warming and other disturbances of these soils can have a strong positive feedback with large emission of CO₂ and other greenhouse gases (GHGs). Disturbance of permafrost can alter these soils from a sink to a major source.

Conversion of natural to managed (cropland, pastureland and woody plantations) ecosystems usually leads to the depletion of the SOC pool (Figure 4). Most soils lose one-third to one-half of their original pool, and those of the tropics may lose as much as 90% of antecedent pool [23]. The magnitude of loss (10–60 Mg/ha) is accentuated when the input of C into the managed ecosystem is lower, and losses through erosion and mineralization are higher than those under prior land use or management. The rate

and magnitude of loss are more drastic in the tropics than in temperate regions, in coarse-textured soils than in fine-textured soils, in soils with a high antecedent pool than in soils with a low antecedent pool, in intensively drained soils than in undrained soils and in terrains prone to erosion than in terrains prone to deposition. Furthermore, the magnitude and rate of loss are higher in agroecosystems managed with extractive farming practices leading to negative nutrient and water budgets than in those managed with recommended management practices (RMPs), which lead to positive nutrient and water budgets. Similarly, the magnitude of the depletion is lower in ecosystems converted to an appropriate land use (as per the land capability assessment) than in ecosystems converted to an inappropriate land use (e.g. conversion to cropland on sloping lands, and drainage of peat soils). Depletion of the SOC pool leads to decline in soil quality, and emission of CO₂ and other gases into the atmosphere. Jenny [11] has stated that contributions of SOM to atmospheric CO₂ appear underestimated. He has argued that 'more CO₂ would become oxidized from debris, roots and humus for a number of years after cutting or clearing than would be released promptly by fire and immediate decay'. The magnitude of the total loss of the SOC pool has been varyingly estimated to range from 40 to 537 Pg, with a mean range of 78 ± 12 Pg [24].

It is this historic depletion of the soil C pool, especially the SOC component, that has created a C-sink capacity in soils of the managed or agro-ecosystems. The sink capacity may also be created by improvement in soil fertility through application of N and of P. The latter is especially important in acid soils of the savannah ecosystem in South America (e.g. Cerrados and Llanos). Restoration of wetlands and prohibiting cultivation of organic soils (peat) can also enhance the C-sink capacity. A goal of the soil and water management strategy is to fill this C-sink capacity through conversion to a restorative/appropriate land use and adoption of improved management practices or RMPs. Restoration of the SOC pool through C sequestration also leads to improvement in soil quality and resilience.

Rational for Carbon Sequestration in Soils

There are numerous reasons for sequestering C in the terrestrial biosphere in general, and in soils in particular. Important among these (e.g. agronomic, ecological and climatic) are the ones briefly discussed below:

Agronomic Effects of SOC Sequestration

The SOM has been considered as an elixir of plant life ever since the dawn of settled agriculture some 10–13 millennia ago. Allison [12] stated that 'man has

appreciated the fact that dark soils, commonly found in the river valleys and broad level plains, are usually productive soils. He also realized at a very early date that colour and productivity are commonly associated with organic matter derived chiefly from decaying plant materials'. Lawes and Gilbert [25] observed that a soil with more SOC is a better soil, a more productive soil. Albrecht [26] stated, in the *USDA Year Book of Agriculture* entitled 'Soils and Men', that 'Soil organic matter is one of our most important national resources; its unwise exploitation has been devastating; and it must be given the proper rank in any conservation policy as one of the major factors affecting the level of crop production in the future'. Ever since these visionary statements, the literature is replete with the importance of SOM in enhancing agronomic production and advancing food security [23, 27], especially with the threat of declining crop yield with the projected climate change [28]. Lal [29] synthesized the available literature relating crop yield to incremental increase in the SOC pool. He reported that increase in the SOC pool by 1 Mg C/ha/yr can increase crop yield by 20–70 kg/ha for wheat, 10–50 kg/ha/yr for rice, 30–300 kg/ha for maize, 20–50 kg/ha for soybeans and 30–60 kg/ha/yr for beans. This increase in crop yield is due to increase in use efficiency of input because of improvements in the available water-holding capacity, soil structure and aggregation, and cation/anion exchange capacity. Lal estimated that an increase in the SOC pool by 1 Mg C/ha/yr would increase agronomic production in developing countries by 24–40 million Mg/yr for food grains and 7–11 million Mg/yr of roots and tubers [30]. The agrarian stagnation in sub-Saharan Africa and in the dry farming regions of South Asia can only be broken if the soil quality can be improved through increase in the SOC pool. As has Jenny [14] stated, 'injecting crop stimulation fertilizers into soil does not recreate soil mass lost or restore natural soil structures and life.' Indeed, attempts at increasing production by applying chemical fertilizers to depleted and degraded soils of Africa have met only with modest, if any, success.

The question 'Should SOC be maintained or increased?' must be answered in the context of soil type, land use and management. Similar to crop seed (e.g. maize, wheat or soybean), SOC is the means of production as well as an end product. Thus, how much of it must be consumed, and how much saved for future use, needs a careful appraisal. Tisdale and Nelson [31] stated that all the edaphologically important functions of SOM in soil (e.g. storehouse of nutrients, increase in exchange capacity, source of energy for micro-organisms, improvement in soil structure and tilth, and increase in infiltration rate through protection of soil surface against crusting) depend on its decomposition. Thus, they argued that production of large quantities of biomass (residues) and their subsequent decay is necessary to good crop and soil management. Janzen [32] also debated the following dilemma: Can we both conserve organic matter and profit

from its decay? Albrecht [26] wondered about the wisdom of hoarding much SOM 'like a miser hoarding gold. Organic matter functions mainly as it is decayed and destroyed. Its value lies in its dynamic nature'. However, the answer also depends on the level of SOC pool and the critical limit. There exists a critical limit of SOC pool for agricultural soils and this limit may be different for temperate [33] and tropical soils [34]. Enhancing the SOC pool is essential to harnessing the agronomic, ecological and climatic benefits along with improving the use efficiency of input if its level is below the critical limit.

Strategies of Carbon Sequestration in Soil

Dyson [35] was among the first soil scientists who suggested building upon C reserves in the form of humus. Several reports have since been written about the management systems to build 'carbon bank' in soil. Activities and the processes of building the C bank through C sequestration in soils are schematically outlined in Figure 5. Activities that deplete the SOC pool include deforestation, ploughing, extractive farming, negative nutrient budget, uncontrolled grazing and residue removal. These activities accentuate soil degradation processes including accelerated erosion, topsoil removal or truncation, mineralization of SOM, and loss of nutrients and water out of the ecosystem. In these scenarios, $C_{input} < C_{output}$, thus ΔSOC is negative and the soil C pool is depleted (Figure 5). In contrast, activities that enhance the SOC pool include afforestation, no-till (NT) farming, mulching, cover cropping, integrated nutrient management (INM), controlled grazing, agroforestry and liberal use of manure and biosolids. These activities accentuate soil restoration processes including humification, aggregation, illuviation, deposition, strengthening of cycles of H_2O and elements, and formation of secondary carbonates. In these scenarios, $C_{input} > C_{output}$, the ΔSOC is positive and the soil C pool is enhanced.

The relevance of NT farming and conservation agriculture requires a special mention. NT farming, developed for row crops in the early 1960s, is practised on about 100 Mha (6% of the world cropland area) to control erosion on sloping lands, conserve water in the root zone, save energy and time, and stabilize agronomic yields against vagaries of climate. In addition, conversion of plough tillage (PT) to NT also enhances the SOC pool, especially in the surface layers. The mean rate of SOC sequestration under NT is about 500 kg/ha/yr, with more sequestration in cold and humid than in warm and arid climates (Table 1). The potential of C sequestration is more when NT is practised in conjunction with cover cropping, complex rotations and INM including manuring. The rate is also high for adoption of improved practices on depleted and degraded/desertified soils of the developing countries (e.g. sub-Saharan Africa, South Asia and the Caribbean and Andean regions). However, the

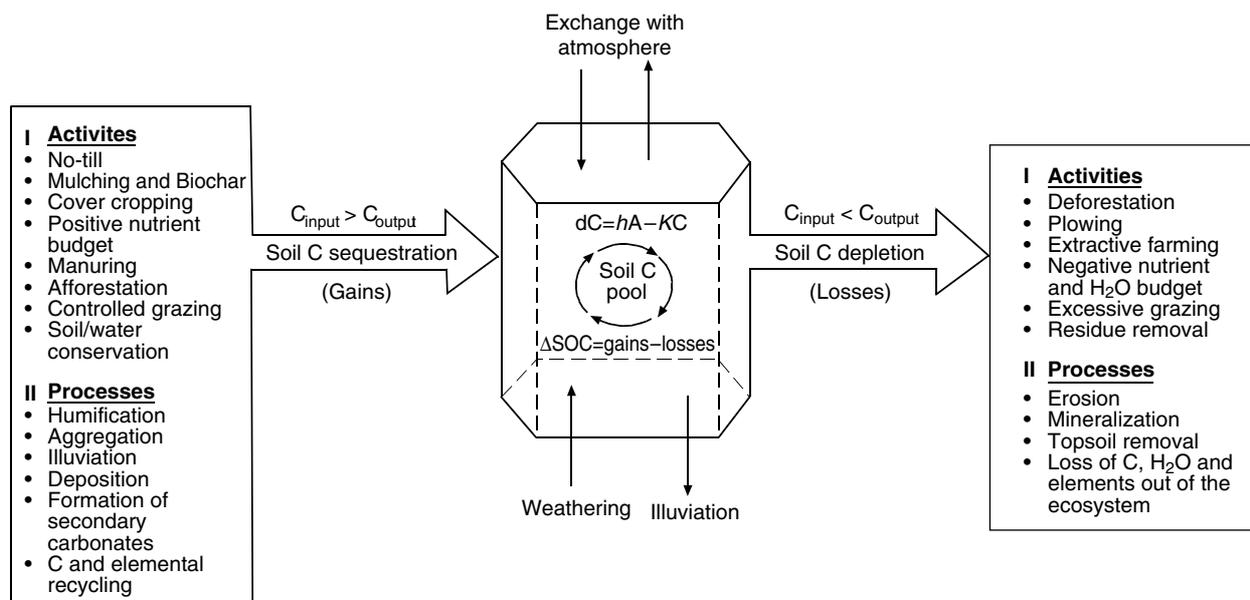


Figure 5 The soil carbon (C) pool is in a dynamic equilibrium with activities and processes that affect input into and output from the soil ecosystem (SOC is soil organic C, K is decomposition constant, h is humification constant and A is the addition of biomass to the soil. At equilibrium, when $\Delta\text{SOC}=0$, C in soil is equal to hA/K).

adoption of NT by resource-poor and small landholders is severely constrained by the lack of or non-availability of essential input such as herbicides, NT planter and sprayers, round-up ready seeds and fertilizers. Competing uses of crop residues (e.g. fodder, residential fuel, and fencing) and numerous social and economic issues are among major constraints to adopting NT farming in developing countries. Similar problems of soil degradation without residue mulching (e.g. crusting, compaction, high water runoff, accelerated erosion, none or minimal activity of earthworms, disruption of elemental cycling and depletion of the SOC pool) would occur if the crop residues from NT soils were to be harvested for cellulosic ethanol from the US cropland. Estimates of the global potential of SOC sequestration in agricultural soils through conversion of PT to NT is about 1 Pg/yr [9], and vary among ecoregions, soil types and management systems. Realizing the environmental and economic benefits of NT farming, especially with regard to SOC sequestration, requires soil-specific research information for: (i) a well-defined baseline, (ii) hidden C costs of all inputs, (iii) C storage in the sub-soil to at least 50 cm depth, (iv) the composition of SOM and its recalcitrance against decomposition, (v) ecosystem C budget at the landscape, farm or watershed scale, (vi) nutrient requirements for humification of crop residues of high C/N, C/P and C/S ratios in terms of the coupled cycling of C, N, P, S and H₂O, (vii) different rates of residue return to meet the needs for ligno-cellulosic ethanol, (viii) soil quality and agronomic production, (ix) changing climate including high temperatures and more variable rainfall/precipitation, (x) use of biochar carbon as a soil amendment, (xi) how long before a change in the SOC pool can be detected, (xii) trading of

C credits, (xiii) using modern innovations including nanoenhanced materials, zeolites and biotechnology, and (xiv) developing cost-effective methods for measuring changes in the SOC pool at regional and national scales.

Once C is sequestered in soil, it must be secured so that it is not re-emitted to the atmosphere. In addition to increasing C_{input} over C_{output} , it is also important that the residence time is increased, through several mechanisms. Important among these are: (i) biochemical alteration and (ii) physicochemical protection [36]. The biochemical alteration transforms SOM to chemical forms that are more resistant to decomposition. The physicochemical protection inhibits biochemical attack by formation of organomineral complexes (e.g. stable micro-aggregates). It is the occlusion or encapsulation of decomposable SOM within stable aggregates and its deposition in pores that render it inaccessible to micro-organisms. Improving the soil structure and favouring the activities of fungi would increase the residence time of C in soil. Translocation of C deep into the sub-soil, away from the zone of natural and anthropogenic perturbations, is another important strategy [37]. Conversion of biomass C to biochar C (discussed later) is another option for increasing the residence time.

SOC

The rate of C sequestration in soil ranges from negative or zero under arid and hot climates to about 2000 kg/ha/yr under cool, humid climates (Table 1) [23]. Most commonly observed rates under on-farm conditions are 300–500 kg/ha/yr (Table 1). Higher rates of SOC sequestration

Table 1 Rate of carbon sequestration in soils with improved management

No.	Location	Soil/region	Management	Soil depth (cm)	Duration (years)	Carbon sequestration rate (kg/ha/yr)	References
1	USA	Minesoil, Ohio	Reclamation	30	20–25	1480–3160	[38]
2	Sweden	Climate transect	Afforestation (Norway spruce)	–	–	150–400	[39]
3	Sweden	Boreal/Temperate	Forest	–	–	40–410	[39]
4	World	161 sites	NT	–	25–30	400–480	[40]
5	Switzerland	Central Plateau	Intensive management	–	3	570–1470	[41]
6	USA	North America	NT	–	–	0	[42]
7	Brazil	Cerrado	NT	20	25	300–600	[43]
8	UK	Scotland	Peat soil	–	1000	140–720	[44]
9	USA	Ohio	Mulching	50	10	1200–2200	[45]
10	Norway	Southern region	Tillage/fertilization	12–24	50	72–370	[46]
11	USA	Columbia	N fertilization, rotation	–	–	500–1500	[47]
12	Canada	Saskatchewan	NT	15	11	3200–3500	[48]
13	Canada	Saskatchewan/Chernozem	Eliminating fallow	15	17	135–441	[49]
14	Brazil	Rainforest	NT	30	–	270–4000	[50]
15	Spain	Northwest	NT	5	–	1240	[51]
16	USA	Ohio	NT	30	30	433	[52]
17	USA	Minnesota	NT	50	23	0	[33]
18	Norway	Askov region	Manuring	20	10–16	440–2600	[54]
19	USA	California	Irrigation	2 m	75	0.1–1 (Carbonates)	[55]
20	Tropics	Africa/LA	Manuring	–	50	80–170	[56]
21	Caribbean	Tropics	Rotation/tillage	30	10	200–2000	[57]
22	USA	Southeastern	NT	–	5–15	420	[58]
			Cover cropping	–	5–15	530	
			Cropping intensity	–	5–15	220	
			N fertilizer	–	5–15	240	
			Poultry manure	–	5–15	720	
			Forages	–	15–32	1030	
23	USA	Southern Piedmont	NT	–	7–10	490	[59]
24	USA	Indiana	NT	100	28	357–821	[60]
25	Belgium	Southern	Grassland	30	50	438	[61]
26	USA	Appalachian region	NT, manuring	30	60	417	[62]
27	Burkina Faso	Savannah soil	Manuring	30	13	690	[63]
28	USA	Minesoils, Ohio	Reclamation	30	20–50	700–3000	[64]
29	USA	Illinois	N fertilization	30	23	158–324	[65]
30	Canada	Prairie	Rotation	20	–	300	[66]
31	USA	California	Cropping systems	–	10	(–350)–560	[67]
32	India	Himalayan	Manuring	45	33	900	[68]
33	Ethiopia	Southwestern	Afforestation	–	–	1000–3200	[69]
34	China	Northern	NT	5	5	0	[70]
35	USA	Colorado	Forest	1	–	576–805	[71]
36	Italy	Northeastern	Manuring	–	25	400	[72]
37	Canada	Ontario	NT	20	11	0	[73]
38	Costa Rica	Rainforest	Agroforestry	–	10	600	[74]
39	USA	Midwest	NT	100	24	1790	[75]
40	World	River Basin Scale	Modelling	–	–	60–150	[76]
41	UK	Rothamsted	Afforestation	30	100	400	[77]
42	USA	Colorado	Native grass mix	20	–	1400–1520	[78]
43	Australia	Victoria	Phosphorus	10	68	174	[79]
44	USA	Iowa/Mollisol	N fertilization	–	12	Negative	[80]
45	Brazil	Parana	NT	20	20	500–800	[81]
46	USA	Great Plains	NT	5	5	294	[82]
47	Tajikistan	Sierozen	Mulching	50	15	1260–3950	[83]
48	Norway	Southern region	Crop rotation/ley farming	20	37	325	[84]
49	Belgium	Cropland	Manuring/mulching	–	20	94	[85]
50	Australia	Brisbane	Tillage/rotation	10	20–25	100–1600	[86]
51	Argentina	Pampas	NT	–	–	(2760, cumulative)	[87]
52	Italy	Southern Alps	Pastures	–	62	360	[88]

Table 1 (Cont.)

No.	Location	Soil/region	Management	Soil depth (cm)	Duration (years)	Carbon sequestration rate (kg/ha/yr)	References
53	Senegal	Peanut Basin	Agroforestry	–	25–50	120–220	[89]
54	Norway	Southern region	Residue	20	43	104–114	[90]
55	Norway	Southern	Manuring	20	37–74	68–227	[91]
56	USA	Ohio/Minesoil	Reclamation	30	50	64–2400	[92, 93]
57	World	276 paired data	NT, crop rotation	–	–	570 ± 140	[94]
58	World	276 paired data	Rotation	–	–	200 ± 120	[94]
59	USA	Texas	NT	15	20	(47–62%)	[95]
60	Brazil	Acrisol/Southern	NT	20	18	190–650	[96]

Table 2 Potential of soil carbon sequestration

No.	Location	Soil/region	Management	C sequestration potential (Tg C/yr)	Reference
1	USA	Central region	Elimination of fallow	6.2	[97]
2	Brazil	Cerrado region	Improved management	42	[98]
3	Amazon	Rainforest	Improved management	508	[50]
4	South America	Pampas	NT farming	9.4–12.5	[99]
5	Mexico and Central America	Agricultural soils	Soil/crop management	3.8–16.7	[100]
6	Caribbean	Agricultural soils	Soil management	6.5–7.5	[57]
7	USA	Grazing land	Pasture management	17.5–90.5	[101]
8	EU countries	15 countries	Improved management	16–19	[102]
9	USA	Forest land use	Forest management	48.9–185.8	[103]
10	West Asia and North Africa (WANA)	Dryland	Improved management	200–400	[104]
11	World	Tropical soils	RMPs	278–516	[105]
12	China	Agricultural soils	Improved management	112–236	[106]
13	World	Arid lands	Afforestation	900–1900	[107]
14	Central Asia	Dry lands	Improved management	10–22	[108]
15	World	Cropland soils	Improved	400–800	[23]
16	India	Agricultural soils	Improved management	39–49	[108]
17	Tropical forest	Forest ecosystem	Improved management	200–500	[109]
18	Europe	Agricultural soils	Improved land use and RMPs	70–190	[110]
19	USA	Cropland	Improved	144–432	[111]
20	World	Slash and burn	Biochar	210	[112]
21	Sub-Saharan Africa	Agricultural land use	Soil and crop management	28.5	[113]

are observed in degraded agricultural soils converted to a restorative land use such as perennial cover (e.g. afforestation, improved pasture, and *in situ* water conservation). Estimates of regional and global potential are summarized in Table 2. For example, the global potential of C sequestration in cropland soils is estimated at 0.4–1.2 Pg/yr [23]. Pacala and Socolow [9] estimated that conversion of all cropland to NT farming would sequester 1 Pg C/yr. However, only about 100 Mha or 6% of the world's cropland was managed by NT farming in 2007 [114]. Stewart *et al.* [115] concluded that world agricultural soils observe a linear relationship between soil C pool and C input, and saturation of soil C does occur. Therefore, the greatest efficiency in C sequestration is in soils farther from C saturation (e.g. the highly degraded

and desertified soils severely depleted of their original SOC pool). The saturation concept was schematically illustrated by Jenny [11] and is graphically depicted in Figure 4.

SIC

Formation of secondary carbonates is a geological process, and occurs over a geological time scale. Thus, the rate of C sequestration as secondary carbonates is lower than that of SOC sequestration, and is usually 2–10 kg C/ha/yr [6], but can be as much as 25 kg/ha/yr. In Boreal grassland and forest soils of Canada, Landi *et al.* [116] reported that the rate of pedogenic carbonate

accumulation increases from 12.5 kg C/ha/yr for semi-arid grasslands (Brown soils) to 21.5 kg C/ha/yr for forest soils (Grey soils). Secondary carbonates occur in various forms including films, threads, concretions, pendants, laminar caps, caliche and calcrete. Carbonate pendants are formed in an environment that may contain lithogenic carbonates. Wang and Anderson [117] observed minute crystals (0.5–4 μm in diameter) of secondary carbonates in three chernozemic soils in Saskatchewan, Canada. In the Gangetic Plains of India, Srivastava [118] observed pedogenic/secondary carbonates in the form of dense micrite and diffused needles.

There are numerous factors that affect the process of formation of secondary carbonates (e.g. climate, moisture regime, profile characteristics and biogenic activity). Formation of secondary or pedogenic carbonates usually occurs in soils with dry or ustic moisture regime, and in those with sudden textural breaks in the lower solum [119]. Over the geological time scale, the rate of formation of secondary carbonates was inhibited during the ice age when CO₂ production in soil was low and the water supply was ice-locked. In dry climates, the rate of formation is also limited by high aridity. Indeed, the presence versus absence of secondary carbonates is a good indicator of paleoprecipitation above or below the annual value of 760 mm [120]. Formation of secondary carbonates is accentuated by the presence of a biogenic CO₂ source [42]. In addition, some soil micro-organisms also set in motion the process of formation of secondary carbonates. In Spain, Rivadeneira *et al.* [121] observed that the presence of *Deleya halophila* enhanced the formation of secondary carbonates. The sequential process seemingly commences with a nucleus formed by the aggregation of a few calcified bacterial cells. This is followed by accumulation of more calcified cells and carbonates cement the bacteria together. Rivadeneira *et al.* [121] observed that this process leads to the formation of spherical bioliths, and termed it as 'biomineralization'. Microbial precipitation of dolomite in dilute natural waters was also confirmed in field and laboratory experiments by Roberts *et al.* [122] who observed that methanogens are the dominant metabolic guild and drive dolomite precipitation via nucleation on the cell wall, as another example of biomineralization.

There are two modern applications of the natural process of formation of secondary carbonates. First, the strategy of geological sequestration of CO₂ into the saline aquifer is based on the principle of immobilization of CO₂ through carbonate precipitation (see Figure 2). For example, Xu *et al.* [123] assessed the feasibility of CO₂ trapping by secondary carbonate minerals such as calcite, dolomite, siderite and dawsonite in the presence of high-pressure CO₂. They observed the trapping capability of 0.5–1.4 kg CO₂-C (2–5 kg CO₂)/m³ of formation. The formation of secondary carbonates into the solid matrix decreased porosity and reduced permeability. Rush *et al.* [124] also conducted tests to assess the possibility of

in situ storage of CO₂ in geological strata, particularly in deep brackish to saline non-potable aquifers, as an application of formation of secondary carbonate minerals over time. The data showed mineral dissolution with an increase of desirable ions (Ca²⁺, Fe²⁺ and Mg²⁺) in solution that can form the carbonate minerals (e.g. calcite, siderite and magnesite). However, the methodology must be refined if this technology is to be implemented safely, efficiently and predictably [125]. Furthermore, the process of CO₂ immobilization through precipitation of secondary carbonates would not occur to permit significant sequestration of CO₂ if few minerals are present in the host rock.

Secondly, the principles of the formation of secondary carbonates can be used to study geological processes on other planets. For example, Vecht and Ireland [126] hypothesized that occurrence of Vaterite (formation at ~25°C), aragonite (~70°C) and calcite (~80°C) may indicate possible reaction of CO₂ with an aqueous solution of CaCl₂ in the presence of ammonia. These conditions may have existed at the surface of Mars in the past and may imply formation of pseudo-biogenic carbonate structure under exobiology conditions. In contrast, Golden *et al.* [127] presented experimental evidence that the zoned Mg–Fe–Ca carbonates in a Martian meteorite may have been formed by simple inorganic processes. Gleason *et al.* [128] studied a Martian meteorite and concluded that complex zoning in carbonates is indicative of non-equilibrium processes in their formation. The data suggested that CO₂-rich fluids of variable composition infiltrated the rock while on Mars, indicating inorganic origin of the carbonates.

The formation of secondary carbonates also takes place in building materials when atmospheric CO₂ reacts with Ca²⁺ present in the pre-solution. Martinez-Ramirez *et al.* [129] used micro-Raman spectroscopy to establish the existence of various forms of CaCO₃ in fully carbonated lime mortar. Calcite was found to be the most thermodynamically stable form in the mortar.

Biofuel and Carbon Sequestration in Soil

The atmospheric concentration of CO₂ has increased from 280 ppm in the pre-industrial era to 381 ppm in 2006 [5]. Some consider mitigating the CO₂-induced climate change as an energy problem [130]. In addition to the CO₂-induced threat of global warming, increase in the price of oil is also the driving force behind the urgency for developing viable alternatives to fossil fuel, including the biofuel. For example, CO₂-C emission increased from <0.1 Pg/yr between 1750 and 1799 to 0.5 Pg/yr by 1899 and then increased to 6.8 Pg/yr by 1999, and is projected to be 12 Pg/yr by 2054 [131]. The price of a barrel of oil increased from about \$25/barrel in 2000 to about \$100/barrel in December 2007, and \$135/barrel in May 2008, which is provoking the new energy crisis [132]. The

strategies being proposed are a mandatory cap on emissions (Intergovernmental Panel on Climate Change (IPCC) meeting in Bali, Indonesia (December 2007)), and developing alternative (C-neutral or C-negative) fuel sources. Schlamadinger and Marland [133] suggested that forest and bioenergy strategy offers the prospect of reduced CO₂ emissions through five mechanisms: (i) use of wood products that displace other products that require more fossil fuel for their production, (ii) use of biofuels to displace fossil fuel, (iii) storage of C in forest products, (iv) C sequestration in the forest biomass and (v) C sequestration in soil. However, the use of biomass as a fuel source has several issues: (i) a low power density of photosynthesis for biofuels (0.6 W/m²), thus the need for a large land area [130] along with the competition for water and nutrients, and (ii) the low-energy production efficiency. Ethanol production in the USA has increased drastically since 2000. It was 1.7 billion gallons (BG) in 2000, 1.9 BG in 2001, 2.1 BG in 2002, 2.8 BG in 2003, 3.4 BG in 2004, 4.0 BG in 2005, 4.9 BG in 2006 and 7 BG in 2007. The 2007 bill mandates 36 BG by 2022, with 21 BG coming from the so-called cellulosic ethanol [134]. Thus, there is an urgent need for procuring 1000 billion tonnes of biomass [135], which has necessitated identifying alternative sources including crop residues [136, 137]. In addition to the adverse impact of residue removal on soil quality and non-point source pollution [14, 45, 138], there are also challenges in engineering microbes for production of cellulosic ethanol [139, 140]. While ethanol production from sugarcane has been successful in Brazil [141], the potential of that from maize grain in the USA is debatable [134, 142], and is also a threat to the Conservation Reserve Program (CRP). Thus, other sources of biomass (e.g. switchgrass or prairie grass) are being considered [143, 144]. Pimentel and Patzek [145] observed that ethanol production using maize grain required 29% more fossil energy than the ethanol fuel produced. Establishment of energy plantations (e.g. poplar, willow, miscanthus and switchgrass) may have the advantage of restoring degraded lands and sequestering C in soil while producing biomass. However, the energy efficiency seems to be low with current technology. Pimentel and Patzek [145] estimated that ethanol production required 50% more fossil energy and wood biomass, 57% more than ethanol fuel produced from these sources. Scharlemann and Laurance [146] posed the question 'How green are biofuels?' It is because of these issues with harvesting crop residues, using soybean/maize grains and establishing biomass plantations, that biofuels are considered as 'the cure worse than the disease' [147]. Some have argued whether expecting food and fuel for all from scarce land resources is foolish or realistic [148]. Others have lamented that there is no such thing as free biofuel from crop residues [149], because removal of crop residues would increase soil erosion risks, increase the need for additional fertilizers and deplete the SOC pool. Jenny [14] argued against indiscriminate conversion of biomass and

organic wastes to fuels. He emphasized that 'the humus capital, which is sustainable, deserves being maintained because good soils are national asset'. Crop residues can either be used to sequester C in soil or produce ethanol and be used for other competing uses. Furthermore, use of maize and soybeans to produce biofuels is causing food shocks in terms of rising food prices [42]. Thus, there is a strong need to identify new biofuel-producing techniques that are more economical and efficient. Identifying and harnessing ancillary benefits of biofuel industry may be an important consideration. Using residues from forestry may be a viable option, especially when forest biomass is burnt directly for heat production, without conversion to ethanol. The strategy of direct combustion (or co-combustion) may be a more effective way of using biofuels. However, other competing uses (timber for construction) must also be critically evaluated. There may also be a time lag in the way that biofuels may initially increase emissions but the long-term use may decrease emissions.

Ragauskas *et al.* [150] emphasized the importance of several by-products of biorefineries. These biomaterials are useful industrial products of economic importance. Use of residues as soil amendments to sequester C in soil is another important consideration. Johnson *et al.* [151] observed that land application of maize stover fermentation by-products, which are about 70% lignin, may reduce the environmental risks from crop residue harvest by stabilizing the structure and sequestering C in soil.

Biochar and Carbon Sequestration in Soil

There is growing interest in the application of biochar to soil to improve its quality, sequester C [152] and affect the global C budget [153]. The concept is based on archaeological evidence of a technique developed by pre-Columbian civilization in the Amazon basin [154]. These anthropic soils, developed through enrichment with biochar, are called 'terra preta' or dark earth [152]. Biochar (charcoal; a biomass-derived BC (black C)) can be a long-term sink for atmospheric CO₂ in terrestrial ecosystems. Lehmann *et al.* [112] observed that conversion of biomass C to biochar C leads to sequestration of about 50% of the initial C, compared with the low amounts retained after burning (3%) and the slow process of humification (10–20%). Biochar produces more recalcitrant C, which has a long residence time in soil, and as an amendment also leads to immediate benefits to agronomic productivity. Lehmann *et al.* hypothesized that replacement of slash-and-burn agriculture by slash-and-char agriculture could offset as much as 0.2 Pg C annually. Similar conclusions were arrived at by Rumpel *et al.* [155]. Experiments conducted in Central Amazonia by Steiner *et al.* [156] indicated that recalcitrance of charcoal C is attributed to the presence of refractory compounds, and its application enhances soil fertility and ameliorates soil quality. Warnock *et al.* [157] observed that biochar affects

microbial population and soil biochemistry and concluded that biochar influences mycorrhizal abundance and functioning through the following mechanisms: (i) altering soil physicochemical properties, (ii) affecting other soil microbial populations and activities, (iii) interfering with plant–fungus signalling and detoxifying allelochemicals on biochar and (iv) providing refugia from fungal grazers. While biochar application to soil by itself may not be feasible, its production in conjunction with those of biofuel may be an important strategy for offsetting emission by producing C-neutral fuels.

Production of biochar from residues of the forest and animal industry is being done by numerous processes including the Charcoal, Heat and Power (CHaP) process [158]; the flash carbonization process by which biomass is converted to biocarbon quickly and efficiently [159]; a low cycle time biomass char production system [160] and a slow pyrolysis system that uses urban and other wastes that cannot be economically recycled or reused [161, 162]. Indeed, a promising technique for lowering CO₂ in the atmosphere while producing energy is biochar bioenergy based on low-temperature pyrolysis [112, 163–165]. The process involves capturing the off-gases from thermal decomposition of biomass to produce energy (heat, electricity or biofuel). Biochar, a major by-product of the process, has many favourable environmental properties. When applied to soil, it has a long residence time, high cation retention capacity and high absorption capacity. Extracting BC from biomass can be permanently sequestered as a geomass [166]. Thus, the process of char production generates energy while also producing BC, which can be used as an amendment to improve soil quality and sequester C in the pedosphere.

Schmidt and Noack [167] listed numerous benefits of using BC, a continuum from partly charred material to graphite and soot particles, to various biological, geochemical and environmental processes including C sequestration in soil. Indeed, BC represents a significant sink in the global C cycle [36, 168–170], especially a sizeable component of the total C pool in mixed-grass savannah [170] and in fire-prone ecosystems [171]. It represents a major sink of atmospheric CO₂ [172] and has important implications on soil fertility [173]. The global rate of BC formation in fire-prone ecosystems is estimated at 50–270 Tg C/yr [172], which is a sink of C because it is highly recalcitrant.

Soil Erosion: A Source of Sink for Atmospheric CO₂

Globally, soil erosion affects about 1100 Mha by water and 550 Mha by wind [174], and the process strongly alters the global C cycle [107]. Erosional processes by water runoff and wind preferentially remove SOC because it has low density (0.8–1.2 Mg/m³) and is concentrated in the

vicinity of the soil surface. Thus, loss of SOC on highly erodible soils is more than the loss due to mineralization [175], and the sediments transported by water and wind are enriched by 2–30 times as much SOC as contained in original soil [107]. The SOC translocated by erosion is redistributed over the landscape; some of it is buried in depressional sites and the rest is carried into the aquatic ecosystems [176–178]. In addition to more biomass production, higher SOC concentration in soils of the foot slopes than those on summit or side slopes may also be the result of deposition [176, 179]. It is estimated that global soil erosion by water annually transports 4–6 Pg C [107] and is an important factor that affects the global C cycle [180]. Yet, the fate of the erosion-displaced SOC is a debatable issue. Sedimentologists argue that it is a C sink [181–187] by as much as 1.5 Pg/yr on a global scale. Van Oost *et al.* [188] have estimated that the sink capacity of the erosional process on a global scale is only 0.12 Pg/yr. In contrast, agronomists and soil scientists believe that soil erosion is a major source of CO₂ [189]. Lal [107] calculated that of the 4–6 Pg C/yr displaced by water erosion, 2.8–4.2 Pg C/yr is redistributed over the landscape, 0.8–1.2 Pg C/yr is emitted into the atmosphere and 0.4–0.6 Pg C/yr is transported to the aquatic ecosystems and burial sites.

The change in SOC pool over a watershed can be described as per Equation (1):

$$\Delta\text{SOC} = \text{SOC}_a - \text{SOC}_f = (D_b + B_a) - (O_i + O_l + O_d + O_r) + A_n, \quad (1)$$

where SOC_a is the antecedent pool, SOC_f is the final pool, ΔSOC is the change in SOC pool following an erosional event, D_b is deep burial in protected and depositional sites, B_a is transport and eventual burial into the aquatic ecosystems, and A_n is the new accumulation over the watershed. The term O refers to oxidation *in situ* (subscript 'i'), over the landscape (subscript 'l'), in depressional sites including methanogenesis (subscript 'd'), and in rivers and other water bodies (subscript 'r'). An important component that can make ΔSOC positive or lead to SOC sequestration is A_n. It implies that the erosional event can lead to SOC sequestration if A_n is more after than before the event, implying that eroded soils have more net primary production (NPP) than uneroded soils. This is precisely the assumption made by Smith *et al.* [187] and Stallard [186] under the term 'dynamic replacement'. However, the NPP of eroded soils is usually lower than those of uneroded soils even with the additional input of fertilizer, irrigation and tillage, all of which have high hidden C costs.

Soil erosion is a three-phase process: detachment, transport and redistribution, and deposition. While the C is protected against mineralization during the depositional phase, it is exposed to microbial processes and climatic factors during the detachment and transport/redistribution phases. With low NPP of eroded soils and a long time

lag (decades) to restoration, the net effect of erosion is an increase in emission, making erosion a net source of atmospheric CO₂.

Resolving the debate necessitates integrating field and modelling studies to minimize uncertainties in scientific understanding of the fate of C at each of the three erosional phases: detachment, transport and redistribution, and deposition. Data are needed with regard to C flows and transformations in the vertical and horizontal directions including soil C pools, erosion/deposition, labile C pool and flux of CO₂, CH₄ and N₂O. These data are also needed to build the three-dimensional net ecosystem C budget.

Modern Scientific Innovations: Nanotechnology, Biotechnology, and Information Technology

Nanotechnology is defined as utilization of structure with at least one dimension of nanometre size (10⁻⁹ m) for the construction of material, devices or systems with novel or significantly improved properties because of their nano-size [190]. Nanotechnology has numerous applications in agriculture [191], especially in nanofertilizers and the nano-delivery systems for delivery of nutrients to plant roots which enhance NPP and the amount of biomass C added to the soil. They can also be used as sorbents of environmental contaminants [192]. Lombi *et al.* [193] have used nano X-ray fluorescence and nano X-ray absorption near-edge structure (n-xanes) to study distribution and exchangeability. Using nanosensors, which can detect soil quality as an index of SOC pool and flux, is an emerging field [194, 195]. In combination with remote sensing technology, use of nanosensors can assess edaphic parameters in areas with poor accessibility. Zeolite materials, natural or synthetic, can be used to enhance nutrient and water retention and for use as a soil amendment for slow release of nutrients [196–199]. Zeolites are strong soil modifiers [200]. Weak hydrogen bonds with framework oxygen atoms implies that the NH₄⁺ molecule can be easily exchanged or desorbed. This property is attractive for agronomy and soil remediation [201, 202]. Hydrogels and zeolites can be used to enhance soil water-holding capacity [203, 204], and enhance C input into the soil. Nanofilms, used appropriately, can prevent escape of H₂O molecules from plants without inhibiting the CO₂ exchange.

Biotechnology has numerous applications in C sequestration in soil through genetically modified (GM) plants with a favourable root/shoot biomass ratio, high concentration of recalcitrant compounds with a long residence time and tolerance to biotic/abiotic stresses. Plants under stress can emit molecular-based signals [205] that can be detected using nanosensors and remote sensing technology, and targeted treatment imposed prior to a strong adverse impact on NPP. GM plants can be developed that are characterized by nanometre-scale pores

called protein channels as biosensors [206]. Forest biotechnology has numerous applications in C sequestration [207]. In combination with nanotechnology, GM trees can have recalcitrant compounds (e.g. phenolics and seuberin) included in cells to enhance the residence time of C. Advances in information technology and creating digital highways can enhance connectivity and provide information about modern innovations to farmers in remote areas. Creation of automated decision support systems can promote extension services for resource-poor farmers in developing countries. Wireless transmitters based on solar power are an innovation to promote connectivity among and within rural communities [208]. Farmers in developing countries can leapfrog to modern innovations that enhance C sequestration in soils and biota by using improved/GM crops and earning extra income through trading of C credits.

Promoting Carbon Sequestration in Soil Through Carbon Trading

The potential of C sequestration in soils is the largest and the challenge the greatest in severely eroded, degraded and desertified ecosystems where extractive farming is the norm. Resource-poor small landholders of the tropics do not have the financial capability to invest in soil restoration. The required inputs, if available, are prohibitively expensive. However, generating another income stream for farmers through trading of C credits can promote adoption of technology that enhances C sequestration in soils. Carbon trading can be done under the Clean Development Mechanism (CDM) of the Kyoto Protocol [133, 209–211] through UN Organizations [212], and industry [213] such as the Chicago Climate Exchange [214] and the European Exchange [215]. Broecker [216] proposed a global C-pie as a tool to stabilize atmospheric CO₂. The magnitude of C-pie is computed by assessing the surplus C that must be offset through emission trading. The surplus is computed on the basis of the desired level of CO₂ concentration in the atmosphere (e.g. 560 ppm) by a specific time (e.g. 2100). Assuming that 4 Pg of C absorbed by the atmosphere equals 1 ppm of CO₂ concentration, the global C-pie is 720 Pg [(560 ppm–380 ppm) × 4 Mg/ppm = 720 Pg]. The total quota of 720 Pg can be allocated to different countries on the basis of their population. Any country that has a surplus quota beyond its industrial/developmental needs can sell it to others. Developing countries can also trade C sink in soil (and trees) as viable strategies to offset emissions under CDM. However, commodification of C sequestered in soil requires the development of a methodology to assess changes in the C pool (ΔSOC; kg/ha/yr) over a short period of 1–2 years. Furthermore, the SOC pool must be measured over the landscape, watershed, farm or regional scale with reference to a baseline (e.g. conventional tillage vis-à-vis NT). Trading of

C credits may be facilitated by imposition of emission taxes and caps [217], and creation of a mechanism to aggregate the SOC pool for the participating community. In addition to the baseline [218], the issue of net C sequestration must be also resolved with reference to: (i) the hidden C costs of all input [108, 219] and (ii) emission of all GHGs (e.g. CO₂, CH₄ and N₂O) [220]. The depth of SOC measurement (to at least 1 m) is also critical.

The Cost-curve for Carbon Sequestration in Soil

Operationalizing the process of trading of C sequestered in soils would need estimate of the cost incurred per unit quantity (\$/Mg). An estimate of the cost may be done as follows:

$$\text{Cost (\$/Mg)} = (C_r - C_c) / \Delta\text{SOC}, \quad (2)$$

where C_r is the cost of all inputs required for the RMP (e.g. NT), C_c is the cost of all inputs used for the conventional or reference system also termed baseline (e.g. conventional till) and ΔSOC is the change in soil C pool (Mg) over the specified period. The ΔSOC must be the net gain in due consideration of N₂O and CH₄ fluxes, and the hidden C costs of all input. The cost of capital equipment (e.g. NT seeder and herbicide applicator) must be allocated appropriately considering the expected life of the equipment. Since RMPs are not necessarily adopted for C sequestration but to enhance agronomic yield, total agronomic production in each of the systems must also be considered. While the price of C is low in the Chicago Climate Exchange (\$2/Mg of CO₂ in December 2007), C sequestration in soil may also become economically profitable with possible emission tax and cap in the future. The resolution adopted at the IPCC meeting in Bali, Indonesia (December 2007), is a step in that direction. Enkvist *et al.* [221] and the authors of [222] estimated the relative costs of each possible method of reducing emissions. They developed an integrated fact base and the related cost curves showing the significance and cost of each available approach. They observed that a substantial share of the overall opportunities, including a large potential to reduce emission by protecting and replanting trees, lies in developing countries. Furthermore, power generation and manufacturing industry offer less than half of the potential of reducing emissions, and about 25% of a possible reduction would occur from energy conservation (e.g. insulation of the buildings).

The option of protecting and replacing trees in developing countries identified by Enkvist *et al.* [221] has two components of the terrestrial biosphere: (i) tree biomass and (ii) soil. Afforestation of degraded/desertified soils in developing countries has a potential to sequester 0.9–1.9 Pg C/yr [223]. Restoration of these degraded ecosystems is a matter of urgency for economical, ecological and environmental reasons. It is a truly win-win strategy.

While sequestering C in soil to mitigate the climate change, it increases biodiversity, improves the environment and advances the food security.

Conclusions

The importance of C sequestration to manage soil fertility has been recognized for millennia since the dawn of settled agriculture. However, its impact on the atmospheric chemistry as a source of atmospheric CO₂ and its role in stabilizing atmospheric chemistry as a natural C sink gained attention during the 1970s. As the third largest global pool, the soil/pedological C pool plays an important role in the global C cycle through natural and anthropogenic changes in both organic and inorganic components. The maximum or potential soil C sink capacity is equivalent to the historic C loss estimated at 78 ± 12 Pg. Restoration of wetlands, prohibiting cultivation of peat soils and minimizing disturbances of Cryosols (permafrost soils) are important for reducing the positive feedback through the global warming. The realizable potential, however, may be only about two-third of the sink capacity.

Harnessing the potential of C sequestration in soil necessitates addressing several issues through research and development. Important among these are the following:

- (i) identification of soils and ecoregions with high C sink capacity, and of land use and management practices that are major sources of CO₂, CH₄ and N₂O,
- (ii) restoration of wetlands, prohibiting the cultivation of peat soils and minimizing disturbance of permafrost and frozen soils of tundra and alpine regions,
- (iii) validation and adaptation of those land use and management techniques that create a positive ecosystem C budget ($C_{\text{input}} > C_{\text{output}}$),
- (iv) assessment of the net C gain in the soil in consideration of all gases (CO₂, CH₄ and N₂O), hidden C costs of input and the baseline,
- (v) evaluation of C gain intensity in relation to NPP, agronomic productivity and other ecosystem services,
- (vi) determination of permanence or residence time of C in soil,
- (vii) establishment of the relation between the SOC pool and soil quality,
- (viii) development of a routine methodology to measure the SOC pool at the landscape, farm or watershed scale non-destructively and economically,
- (ix) development of a method to trade credits of C sequestered in soil and provide incentives to farmers for investments in soil restoration,
- (x) identification of mechanisms of C sequestration (e.g. aggregation, humification, illuviation and recalcitrance) under soil-specific conditions, and managing

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soils to enhance these processes in mineral and organic soils,

- (xi) development of a holistic approach to soil C management involving INM, manuring, biofuel production, application of amendments (biochar) and use of modern innovations such as nanotechnology, biotechnology and information technology, and
- (xii) replacement of other CO₂-emitting materials by viable alternatives that do not emit GHGs.

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