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SOILS ISSUES

ACHIEVING SOIL CARBON SEQUESTRATION IN THE UNITED STATES: A CHALLENGE TO THE POLICY MAKERS

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Carbon (C) sequestration in soil implies enhancing the concentrations/pools of soil organic matter and secondary carbonates. It is achieved through adoption of recommended management practices (RMPs) on soils of agricultural, grazing, and forestry ecosystems, and conversion of degraded soils and drastically disturbed lands to restorative land use. Of the 916 million hectares (Mha) comprising the total land area in the continental United States and Alaska, 157 Mha (17.1%) are under cropland, 336 Mha (36.7%) under grazing land, 236 Mha (25.8%) under forest, 14 Mha (1.5%) under Conservation Reserve Programs (CRP), and 20 Mha (2.2%) are under urban land use. Land areas affected by different soil degradative processes include 52 Mha affected by water erosion, 48 Mha by wind erosion, 0.2 Mha by secondary salinization, and more than 4 Mha affected by mining. Adoption of RMPs can lead to sequestration of soil organic carbon (SOC) at an annual rate of 45 to 98 Tg (teragram = 1 × 10^{12} g = 1 million metric tons or MMT) in cropland, 13 to 70 Tg in grazing land, and 25 to 102 Tg in forestlands. In addition, there is an annual soil C sequestration potential of 21 to 77 Tg by land conversion, 25 to 60 Tg by land restoration, and 15 to 25 Tg by management of other land uses. Thus, the total potential of C sequestration in soils of the United States is 144 to 432 Tg/y or an average of 288 Tg C/y. With the implementation of suitable policy initiatives, this potential is realizable for up to 30 years or when the soil C sink capacity is filled. In comparison, emission by agricultural activities is estimated at 43 Tg C/y, and the current rate of SOC sequestration is reported as 17 Tg C/y. The challenge the policy makers face is to be able to develop and implement policies that are conducive to realization of this potential. (Soil Science 2003;Volume 168:827–845)

Key words: Climate change, humus, secondary carbonates, soil carbon dynamics, conservation tillage, land use, soil restoration, soil degradation.

IDENTIFYING and implementing appropriate options for reducing the rate of enrichment of atmospheric concentration of CO_2 are important national and international priorities. Rising atmospheric concentrations of carbon dioxide (CO_2) and other greenhouse gases (GHGs) caused by anthropogenic activities are a concern because of the attendant risks of global climate change. Technological options for mitigating climate change may be either adaptive or mitigative (Halman and Steinberg, 1999). Conversion to appropriate land use and adoption of sustainable options for managed terrestrial and aquatic ecosystems under changed climate constitute the adaptive response. Reducing net emissions, either by capturing CO_2

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from industrial plants or by transferring atmospheric CO_2 into long-lived pools, constitutes the mitigative response.

Transferring atmospheric CO_2 into long-lived pools and keeping it stored securely so that it is not immediately re-emitted is called carbon (C) sequestration and involves both biotic and abiotic processes. The biotic process of C sequestration involves removal of atmospheric CO_2 by chlorophyll-bearing plants through photosynthesis and the conversion of some photosynthesized C and its storage into terrestrial or aquatic pools with long turnover times. In contrast, abiotic sequestration involves using engineering techniques for separating, capturing, compressing, transporting, and injecting CO_2 from power plant flue gases or from other sources into ocean or geologic strata (DOE, 1999). Finding non-C fuel or decarbonizing energy sources is another long-term strategy for reducing emissions (Parson and Keith, 1998). Formation of secondary carbonates and/or leaching and transfer of carbonates into groundwater and rivers also constitute abiotic processes of C sequestration (Raymond and Cole, 2003; Lal and Kimble, 2000).

Terrestrial C sequestration has two distinct but interrelated components: soil and biota. Carbon sequesters into soils as humus and secondary carbonates. Biotic C sequestration occurs in forests as above- and below-ground biomass. Indeed, the terrestrial C pool is the third largest among the five global C pools. The largest, or oceanic pool, contains 38,000 Pg (Pg = petagram $= 1 \times 10^{15}$ g = 1 billion metric tons), followed by the geologic pool (comprised of fossil fuel) containing 5000 Pg (4000 Pg of coal, 500 Pg of oil, and 500 Pg of gas), the soil/pedologic pool containing 2300 Pg to 1-meter depth (comprised of 1550 Pg of soil organic carbon or SOC and 750 Pg of soil inorganic carbon or SIC), the atmospheric pool containing 760 Pg, and, finally, the biotic pool containing 600 Pg of live mass and detritus material (Eswaran et al., 1993; Houghton, 1995; Batjes, 1996; Batjes and Sombroek, 1997).

Two components of the soil C pool, SOC and SIC, are linked through biogenic transformations. The SOC pool, comprising active humus and relatively inert charcoal, plays an important role in the global C cycle and has a strong impact on biomass productivity, agricultural sustainability, and environment quality (Jenkinson et al., 1991; Stevenson, 1994). The SIC pool is an important constituent in subsurface horizons of soils of arid and semiarid regions. It includes elemental C and carbonate and bicarbonate minerals (e.g., calcite and dolomite). There are two types of carbonates, referred to as primary and secondary. The primary carbonates are derived from the weathering of the parent material, whereas the secondary carbonates are formed through the conversion of CO_2 in soil air into carbonic acid and its reaction with Ca^{+2} and Mg^{+2} (Lal and Kimble, 2000). Thus, "soil C sequestration" implies an increase in soil concentration/pool of humus and secondary carbonates, whereas "terrestrial C sequestration" implies an increase in soil and biotic C pools.

The Kyoto Protocol (Oberthür and Ott, 2001), ratified by several countries but not by the United States, is an international treaty the purpose of which is to reduce net industrial CO_2 emissions. If it comes into effect, it will mandate that industrialized nations reduce their net emissions by an agreed amount with reference to 1990 levels. While not accepting the mandatory Kyoto Protocol, the Economic Report of the President to the U.S. Congress highlights the opportunity for "sequestration of greenhouse gases in agricultural soils and forestry sinks" (White House, 2002). The President has outlined voluntary options as an alternative to the Kyoto Protocol by stating that "we look for ways to increase the amount of C stored by American farms and forests through a strong conservation title in the Farm Bill" (Kerr, 2002). The President also highlighted two initiatives as a part of his 2003 budget request to Congress: The Climate Change Technology Initiative and the Climate Change Research Initiative. The President's strategy aims for an 18% reduction in GHG emissions by 2012 and has a strong research component (Malakoff, 2003). Another alternative is the "BioCarbon Fund," which was initiated by the World Bank to encourage soil C sequestration and use soil C as a tradeable commodity (World Bank, 2003). However, an important prerequisite to implementation of these policies is an accurate and reliable assessment of the soil C sequestration potential.

The objective of this manuscript is to provide reliable estimates of the potential of C sequestration in soils of the United States in order to meet the goals of voluntary reduction of CO_2 emissions; to compare these estimates with those from other reports; and to identify policy options to help achieve the potential. This report synthesizes the results of three independent but related studies on the estimates of the potential of soil C sequestration in croplands (Lal et al., 1998b), grazing lands (Follett et al., 2001a), and forestlands (Kimble et al., 2003).

LAND USE IN THE U.S.

Land use data for the continental United States and Alaska are shown in Table 1. Of the total land area of 916 million hectares (Mha), 17.1% is under cropland, 36.7% under grazing land, 25.8% under forestland, and the remaining 20.4% is in other land uses. Judicious management of cropland is important to soil C sequestration (Lal et al., 1998b). The cropland area of 157 Mha comprises 119 Mha of rainfed, 22 Mha of irrigated, and 16 Mha of idle cropland. There were some fluctuations in the cropland area in the U.S. during the 20th century (Lal et al., 2003). The area increased from 133 Mha in 1910 to 150 Mha in 1920 and then remained stable until 1952; it decreased to 133 Mha between 1952 and 1962 and remained stable until 1972; it then increased again to 155 Mha between 1972 and 1982 and has remained stable through the first half-decade of the 21st century.

Similar to cropland, the management of U.S. grazing land is also important to soil C sequestration (Follett et al., 2001a). U.S. grazing land consists of 336 Mha (Sobecki et al., 2001; Schuman et al., 2001a; Lal et al., 2003), of which 212 Mha are privately owned, and includes 161 Mha of rangeland and 51 Mha of pasture. The area under grazing land changed little between 1992 and 1997 (Sobecki et al., 2001). Whatever change occurred was attributed to conversion to or from cropland.

There is also a large potential of forestlands for soil C sequestration (Heath et al., 2003). The total area of forestland in the conterminous U.S. has been stable over the 20th century (Birdsey and Lewis, 2003), with a net decrease of only 4 Mha. However, there have been significant regional changes in the forest area. The Northeast and North Central regions have gained forestland by 43% and 7%, respectively, while the Pacific Coast and South Central regions have lost forestland by 14% and 13%, respectively (Birdsey and Lewis, 2003).

Urban land use is another category of relevance to soil C sequestration, and the area under urban land use is rapidly changing. The urban land area is presently estimated at 20 Mha (Table 1) and includes urban forestry and recreational land uses. With intensive use of chemicals (fertilizers, pesticides) and irrigation, and the return of biomass (clippings), these lands have a high potential for soil C sequestration (Pouyat et al., 2003; Qian and Follett, 2003).

CARBON POOL IN SOILS OF THE U.S.

There have been several estimates of the C pool in soils of the United States. Kern (1994) estimated the total SOC pool in soils of the 48 contiguous states at 80.7 ± 18.6 Pg. Waltman and Bliss (1997) estimated the SOC pool at 59.4 Pg for the 48 contiguous states and 13.5 Pg for Alaska. However, the present pool is much lower than it was before the time of settled agriculture. Conversion of natural to agricultural ecosystems caused severe decline in the SOC pool of agricultural soils (Cole et al., 1990, 1993; Donigian et al., 1994; Flach et al., 1997). Most grassland and forest soils tend to lose from 20 to 50% of their orig-

TABLE 1

Land use in the U.S. (NRI, 1997)

Land use	Category	Area (Mha)	Total area (Mha)
1. Cropland			156.9
	(i) Rainfed	118.9	
	(ii) Irrigated	22.3	
	(iii) Idle	15.7	
2. Grazing land	(a) Privately owned		336.0
	(i) Pasture	51.0	
	(ii) Grassland	161.0	
	(b) Public land	124.0	
3. Forest land			236.1
4. Conservation Reserve Program (CRP)			13.8
5. Wetland Reserve Program (WRP)			0.6
6. Special use			101.4
7. Urban			20.0
8. Other land			51.5
Total			916.3

The total land area of 916.3 Mha includes 147.8 Mha of Alaska and 124.0 Mha of public grazing land.

inal SOC pool in the first 40 to 50 years of cultivation (Mann, 1985; 1986; Johnson and Kern, 1991; Rasmussen and Collins, 1991; Camberdella and Elliott, 1992; Grigal and Ohmann, 1992; Kern, 1994; Rasmussen and Parton, 1994; Gebhart et al., 1994; Houghton, 1995; Ajwa et al., 1998). Lal et al. (1998b) estimated that the conversion from natural to agricultural ecosystems between 1750 and 1950 caused depletion of 3 to 5 Pg of C from soils of U.S. croplands. Additional C loss came from soils of the forest and grazing lands. Most croplands have lost between 20 to 40 Mg C/ha (IPCC, 2001). The depletion of the SOC pool is caused by an increase in the rate of oxidation/mineralization and accelerated soil erosion. Soil degradation by erosion and other processes exacerbates depletion of the SOC pool (Parton et al., 1987; Rhoton and Tyler, 1990; Pennock et al., 1994; Lal, 2003a). A large part of this SOC loss is emitted into the atmosphere as CO_2.

Three important implications of the depletion of the SOC pool are: (i) a decline in soil quality and biomass/grain yields, (ii) an increase in risks of environmental pollution, and (iii) the creation of soil C sink capacity through adoption of recommended management practices (RMPs). Indeed, the historic SOC loss has created a soil C sink capacity and vast opportunities to increase the present SOC pool. In addition, enhancing the SOC pool would increase biomass/agronomic productivity and improve the environment—a truly win-win strategy (IPCC, 2000).

SOIL DEGRADATION IN THE U.S.

Soils of croplands and other steeplands are prone to erosion and other degradative processes. Total land area subject to accelerated (more than the tolerable or T level) erosion is estimated at 44 Mha of cropland (27.9% of all cropland) and 51 Mha of grazing land (15.2% of all grazing land) (Table 2). In addition, soils disturbed drastically by mining activities constitute more than 4 Mha. Organic soils, especially those that have been drained for growing crops, are prone to decomposition and subsidence. Such lands cover about 15 Mha. The United States has about 4 Mha of coastal wetlands, much of it along the Louisiana coast alone (Groat, 1989). Drainage of wetlands and cultivation of organic soils is a type of degradation that exacerbates the rate of decomposition of organic matter. Indeed, organic soils are endangered soils and may eventually become extinct as a result of drainage and cultivation.

Accelerated soil erosion affects the SOC pool by the slaking of aggregates, preferential removal of C in runoff water or dust storms, and an increase in mineralization on-site and in depositional sites (De Jong et al., 1983; Janzen et al., 1997; Lal, 2000). However, deposition and burial of eroded C, which is not lost to the atmosphere, is placed in long-term storage (Stallard, 1998). Losses from the SOC pool caused by erosion on sloping land may be several times greater than that caused by mineralization (Daniel and Langham, 1936; Slater and Carleton, 1938; Webber, 1964). Consequently, the SOC pool in eroded soils is much lower than on uneroded phases (Fahnestock et al., 1996). In Saskatchewan, Canada, Anderson et al. (1986) reported depletion of the SOC pool by 70% on convex eroded slopes compared with only 40% depletion on slightly eroded soils. De Jong and Kachnoski (1988) reported that about 50% of the total SOC loss occurred as a result of soil erosion. Cultivated soils not prone to erosion usually attain a stable level of SOC pool within 30 to 50 years, whereas the SOC pool of those subjected to severe erosion may continue to decline over a long period of time (Voroney et al., 1981; Tiessen et al., 1982; Gregorich et al., 1998). Because of the severe depletion of the SOC pool in eroded and degraded soils, the soil C sink capacity is higher than found for slightly or nondegraded soils. Conversion to a

TABLE 2
Land area affected by different soil degradative processes in the U.S. (adapted from Lal et al., 2003; Dregne and Chow, 1994)

	Area (Mha)
A. Water erosion	
1. Cropland	26.3
2. Grazing land	25.8
B. Wind erosion	
1. Cropland	22.1
2. Grazing land	25.6
C. Total wind and water erosion	
1. Cropland	43.7
2. Grazing land	51.4
D. Desertified lands	
1. Irrigated land	4.0
2. Rainfed cropland	3.6
3. Rangeland	276.0
D. Salinization	
1. Cropland	0.17
2. Grazing land	0.01
E. Mining	
1. Unreclaimed	4.4
F. Organic soils	15.4

restorative land use can enhance the SOC pool and resequester a large proportion of the C lost.

Soil degradation and desertification are interrelated processes. Desertification refers to degradation of soil and vegetation in dryland ecosystems. National estimates of the desertification of soil and vegetation were provided by Dregne and Chou (1994) (Table 2).

PROCESSES OF SOIL CARBON SEQUESTRATION

Much of the soil C depleted through conversion of natural to agricultural ecosystems and by soil erosion and degradation can be resequestered through adoption of Recommended Management Practices (RMPs) and restoration of degraded soils (Lal et al., 1998b; Lal, 2001; Follett et al., 2001a; Kimble et al., 2002, 2003). Important pedospheric processes that lead to SOC sequestration include the following:

Humification: This involves converting biomass into humic substances that are relatively resistant to microbial decomposition and have a long turnover time. The SOC pools and processes of C sequestration are described by Follett (2001b) and by Follett et al. (2001a). Depending on soil temperature and moisture regimes, reports of the fraction of the biomass returned to the soil that is converted to soil humus range from 10 to 20% (Duiker and Lal, 1999; Jacinthe et al., 2002) or, in the case of a long-term small grain fallow system, from 5 to 10% (Follett et al., 1997).

Aggregation: Formation of stable organo-mineral complexes is an important mechanism of soil C sequestration. Soil C thus encapsulated within stable microaggregates as organo-mineral complexes is protected against microbial processes and enzymatic reactions (Tisdall and Oades, 1982; 1979; Tisdall, 1997; Elliott, 1986).

Leaching of carbonates: In addition to the formation of secondary carbonates, leaching of carbonates into groundwater (Drees et al., 2000) and transport of alkalinity into rivers (Raymond and Cole, 2003) are important processes in the sequestration of inorganic carbon.

Translocation of carbon into subsoil horizons and burial by sedimentation: Translocation of C from surface into the subsoil is also an important mechanism of C sequestration. Translocation may occur by movement with water as dissolved C (both organic and inorganic) or by the growth and proliferation of roots in the subsoil. The eluviation and deposition of humus and stable microaggregates within the soil profile is another important mechanism of C translocation into the subsoil. Burial of C in depositional sites and aquatic ecosystems also leads to long-term storage (Stallard, 1998; Smith et al., 2001).

The strategies of soil C sequestration include: (i) adoption of RMPs on cropland, grazing land, and forestlands, (ii) land conversion, (iii) land restoration, and (iv) improved systems of urban land use and management. These strategies are based on the principle of agricultural intensification on prime agricultural soils while converting agriculturally marginal and degraded soils to restorative land uses.

Soil Carbon Sequestration in Croplands

Important among RMPs for SOC sequestration on cropland is conservation tillage and residue management (Larson et al., 1972; Rasmussen et al., 1980, 1998; Bouquet et al., 1997; Power et al., 1998; Bruce et al., 1999; Schlesinger, 1999). Tillage decreases the SOC pool by exacerbating emissions (Reicosky and Lindstrom, 1993; Reicosky, 1998) and increasing losses by erosion (Shipitalo and Edwards, 1998). Benefits of soil C sequestration by conservation tillage are linked closely with those of residue management, crop rotation and cover crops, and the judicious use of fertilizers and organic amendments (Follett, 2001a). In Pennsylvania, Drinkwater et al. (1998) reported an increase in SOC by 2.2 Mg C/ha over a 15-year period by using chemical nitrogenous fertilizers. However, the rate of SOC sequestration was 3 times greater when a legume-based rotation was adopted and 5.5 times greater when organic manure was used. Soil application of biosolids also increases the SOC pool (Logan et al., 1997; Duiker and Lal, 1999). In the Sacramento Valley, Clark et al. (1998) reported that application of animal manure over 8 years increased SOC concentration drastically. In Akron, Colorado, Halvorson et al. (1999) observed that SOC concentration increased in the 0- to 15-cm-depth from 15 Mg/ha for no N fertilizer use to 17 Mg/ha for 134 kg N/ha after 11 crops. There is a strong need to enhance the use efficiency of nitrogenous fertilizers to reduce N_2O emissions (Matson et al., 1998), especially by using site-specific farming technology (Schnug et al., 1998). Other RMPs with potential for SOC sequestration include growing winter cover crops (Hargrove, 1986; McVay et al., 1989; Kuo et al.,

1997a and b; Hu et al., 1997; Lal et al., 1998a), crop rotations (Havlin et al., 1990; Omay et al., 1997), and water recycling through subirrigation (Lal et al., 1998b). The observed rates of SOC sequestration through these practices range from 200 to 800 kg/ha/y.

After extrapolating experimental and other data to the national scale, the potential of SOC sequestration in U.S. cropland for different RMPs is shown in Table 3. The potential of SOC sequestration is 24 to 40 Tg C/y (Tg = teragram = 1×10^{12} g = 1 million metric tons) through adoption of conservation tillage on 100 Mha, including that in judicious management of crop residue; 5 to 11 Tg C/y through irrigation and water table management on 64.4 Mha; and 16 to 47 Tg C/y through adoption of improved cropping systems. Total potential of SOC sequestration in U.S. cropland is 45 to 98 Tg C/y (average 72 Tg C/y).

To compare Table 3 data, but excluding residue management, with that of other researchers, land conversion data from Table 7 (10–16 Tg C/y) is added to Table 3 data for an average total potential of 84 Tg C/y, a number very near the value of 82 Tg C/y reported by Sperow et al. (2003) and of 75 Tg C/y as reported in an earlier paper by Bruce et al. (1999) (Fig. 1). Thus when the mineral soil data from Lal et al. (1998b) and Sperow et al. (2003) are compared for a C change resulting from the use of conservation tillage, elimination of bare summer fallow, inclusion of winter cover crops in annual crop rotations, and conversion of marginal cropland to perennial grass and other set-aside land, they are very similar. A major difference noted is that the overall summary of data by Lal et al. (1998b) is much more inclusive as it also reports potential biofuel production, irrigation/water management, savings in fuel consumption, residue management, and other information to arrive at a total average potential of 142 Tg C/y.

CO_2 emissions from organic agricultural soils in the above comparisons; however, estimates are that agricultural lands in organic soils emitted 9.5 Tg CO_2-C/y in 2001, and this amount can then be subtracted from the above estimate by Sperow et al. (2003) and the 84.4 Tg C/y reported above from Tables 3 and 5.

The above discussion shows that both the Sperow et al. (2003) potential C sequestration estimate of 82 Tg C/y, and their estimate of changes in agricultural and land management occurring between 1982 and 1997 in total soil C sequestration of 17 Tg C/y for mineral soils as seen in Fig. 1, are obtained using IPCC methodology. Thus policy makers need to recognize that present estimates of sequestration rate (17 Tg C/y) that are reported for cropland based upon IPCC methodology are only about 20% of the total potential of C sequestration (82 Tg C/y) in U.S. cropland soils. Because of this, we must also assume that present estimates generated using the IPCC methodology for grazing lands, forestlands, or other types of land uses should also be expected to represent only a small part of the potential soil C that other land types can sequester.

Soil Carbon Sequestration in Grazing Lands

The area under U.S. grazing lands is more than twice that under cropland (336 Mha vs 157 Mha). Publicly owned grazing lands and most

TABLE 3

Soil carbon sequestration in U.S. croplands (adapted from Lal et al., 1998b)

Scenario	Area (Mha)	Rate (kg C/ha/y)	Tg C/y	Cumulative potential (Tg C/y)
1. Conservation Tillage and Residue Mgt.	100	240–400	24–40	24–40
2. Irrigation/Water Management				5–11
(a) Supplemental	21	50–150	2–6	
(b) Sub-irrigation on poorly drained soils	43.4	70–115	3–5	
3. Improved Cropping Systems				16–47
(a) Fertilizer Management	117.5	50–150	6–18	
(b) Organic Manures and By-Products	117.5	?	3–9	
(c) Rotation and Winter Cover Crops	51	100–300	5–15	
(d) Summer Fallow Elimination	9.4	100–300	1–3	
(e) Management of Rice Straw	1.3		0.5–1.5	
Subtotal (not including residue mgmt).				45–98

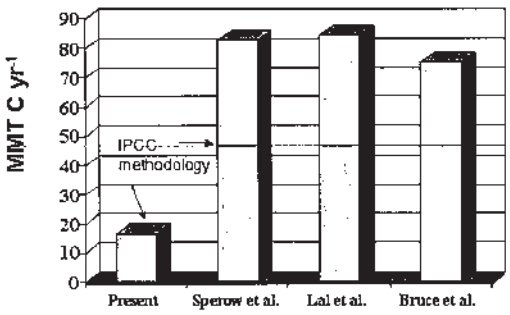

Fig. 1. Present and potential estimates of rate of soil C sequestration using IPCC methodology and estimates by other methods.

U.S. range land areas have never been cultivated. Thus, grazing land soils in the same ecozone will generally have higher SOC levels than similar soils that are cropped. Factors conducive to SOC sequestration in grazing lands include: low levels of soil disturbance, maintenance of the root biomass year round, returning residue produced even when grazing a large percent of the area, and regular application of cattle dung and manure. Furthermore, adopting RMPs on grazing lands also lead to SOC sequestration (Follett et al., 2001a). Data from several long-term experiments have shown a larger SOC pool under soils of well-managed pastures than under croplands (Garwood et al., 1977; Rice and Owensby, 2001; Schnabel et al., 2001; Schuman et al., 2001b). Important RMPs for grazing lands include grazing management, growing improved/adopted pasture species, applying fertilizers, and a judicious use of fire. Converting marginal croplands to pasture may also improve the overall quality of surface soils. Conant et al. (2001) outlined processes and practices of SOC sequestration in grasslands and observed that improving grazing lands have the potential to sequester about 500 Kg C/ha/y. Conant and colleagues concluded that grasslands can be a significant C sink with a widespread adoption of RMPs.

Extrapolating experimental and other data to national scale, Follett et al. (2001a) estimated the potential of only the 212 Mha of privately owned U.S. grazing lands for soil C sequestration. The soil C sequestration potential on these U.S. grazing lands is −4.1 to 14 Tg C/y for nonintensively managed grazing lands and 16 to 50 Tg C/y for improving/intensifying management (Table 4). The total potential of soil C sequestration on the 212 Mha of privately owned grazing lands, primarily through adopting intensive/improved management practices, is 12 to 64 (average = 38) Tg C/y. Soil C sequestration on the ~124 Mha of publicly owned grazing lands is not included in the above figure but, where policies by public agencies result in good grazing management practices, a conservative estimate of accumulation might be 10 to 50 kg C/ha/y for public lands or an additional 1–6 (average = 3.5) Tg C/y above that for privately owned lands. Thus, total potential of soil C sequestration in grazing lands is 13 to 70 Tg C/y. As in the case of cropland, IPCC estimates of the present rate of soil C sequestration for hayland and grazing land is ~17 Tg C/y (USDA 2003) and represents about 41% of the potential discussed above and by Follett et al. (2001a).

Soil Carbon Sequestration in Forestlands

Several forest soil management practices have a beneficial impact on SOC sequestration. Important among these are prescribed fire, fertilization and liming, harvesting and site preparation, and choice of appropriate species (Hoover, 2003; Heath et al., 2003). There is also the potential of agroforestry practices in specific soil and ecoregional conditions (Nair and Nair, 2003). Site preparation is important to minimize risks of compaction and accelerated erosion (Lal, 2003b). The SOC sequestration may be faster on sites not specifically developed (Silver et al., 2000), probably because of low soil compaction. Soil disturbance at harvesting can cause a 20 to 30% reduction in the SOC pool (Jackson et al., 2000; Knoepp and Swank, 1997; Black and Harden, 1995; Law et al., 2001; Wigginton et al., 2000; Giese et al., 2000). Fire and its management can affect SOC pool over the long term (Parker et al., 2001). Soils of the Boreal region can become warmer because of the decrease in canopy cover and reduction in litter fall and detritus material following burning. The resultant soil warming can increase the depth of the active thaw layer (O'Neill et al., 2002). Uncontrolled fire in forests of northern latitudes can transform these ecosystems from a net sink to a net source.

Forest management systems that maintain a continuous canopy cover and mimic natural forest ecosystems are likely to enhance the SOC pool (Thornley and Cannell, 2000). For example, maintenance of the understorey can enhance the SOC pool (Shan et al., 2001). Fertilization is important to SOC sequestration because most forests are N-limited (Magill and Aber, 2000; Resh et al., 2002). In addition to N, growth in

TABLE 4
Soil carbon sequestration in U.S. grazing lands (adapted from Follett et al., 2001a)

Scenario	Area (Mha)	Rate (kg C/ha/y)	Quantity sequestered (Tg C/y)	Cumulative potential (Tg C/yr)
1. Non-intensively Managed Grazing Lands				−4–14
1. Soil inorganic carbon	262.4	0.12–13.0	0.032–3.4	
2. Soil Organic C (rangelands)	54	(100)–100	−5.4–5.4	
3. Pasture land (25% currently unmanaged)				
(a) SOC (12.5% remains unmanaged)	6.38	50–200	0.32–1.27	
(b) SOC (12.5% improved management)	6.38	150–600	0.96–3.82	
4. Cold Region Systems		(200)–200		
II. Improve/Intensify Management				16–50
1. Improved Rangeland Management	107	50–150	5.4–16.0	
2. Improved Pastureland Management				
(a) Fertility Management = liming, and fertilizer-N; rate = 200 to 300 kg/ha				
1. Estimated current area (20%)	10.2	100–200	1.0–2.1	
2. Potential additional area (10%)	5.1	100–200	0.5–1.0	
(b) Application of manure from confined livestock with a N; rate = 250 kg/N ha				
1. Estimated current area (25%)	12.75	200–500	2.6–6.4	
2. Potential additional area (10%)	5.1	200–500	1.0–2.6	
(c) Planting improved plant species				
1. Estimated current area (5%)	2.6	100–300	0.3–0.8	
2. Potential additional area (10%)	5.1	100–300	0.5–1.5	
3. Grazing Management on Pasture				
(a) Estimated current area (20%)	10.2	300–1300	3.1–13.3	
(b) Potential additional area (10%)	5.1	300–1300	1.5–6.6	
4. N-fertilization of Mountain Meadows fertilizer-N; rate = 100 to 200 kg/ha				
(a) Estimated current area (20%)	0.32	100–200	0.03–0.07	
(b) Potential additional area (10%)	0.16	100–200	0.02–0.03	
III. Carbon Sequestration on Public Land	112	10–50	1–6	1–6
Total				13–70

many forests can also be improved by application of micronutrients such as Fe (Benemann, 1992). Forests also have the potential to produce bioenergy crops, both for biofuel and for SOC sequestration (Tolbert et al., 2000).

Heath et al. (2003) extrapolated field and experimental scale data to the national scale and estimated the potential of C sequestration in forest soils of the United States. The data in Table 5 show the potential of 25 to 102 Tg C/y by forest management.

Urban forestry is an important land use system in the United States. Transformation of landscapes from nonurban to urban land use can alter the SOC pool and fluxes drastically. Further, urban trees can play an important role in SOC sequestration (Nowak, 1993; Pouyat et al., 2002, 2003). Nowak and Crane (2002) observed that urban trees in the conterminous United States store 700 Tg of C with a gross sequestration rate of 23 Tg C/y. The observed rates of SOC sequestration may range from 0 to 1200 Kg C/ha/y, depending on species, soil type, management, and climate. There are special forestry practices that also have soil C sequestration potential. These include agroforestry techniques (Nair and Nair, 2003), urban forest (Pouyat et al., 2003; Nowak, 1993), turf management (Qian and Follett, 2003), and conservation buffers and windbreaks (Lal et al., 1998b). The potential for SOC sequestration by the use of these practices is 15 to 25 Tg C/y (Table 6).

Carbon Sequestration through Land Conversion

Change in land use from cropland to pasture and forest can enhance the SOC pool. Afforesta-

TABLE 5

Soil C sequestration potential of U.S. forestlands (adapted from Heath et al., 2003)

Activity	Area (Mha)	Rate of soil C sequestration (kg C/ha/y)	Potential (Tg C/y)
(a) Regeneration	59.7	70–419	4–25
(b) Fertilization	20.0	875–3061	18–61
(c) More partial cutting/less clear cuts	1.5	0–1195	0–2
(d) Lengthen rotations	0.7	0–1195	0–1
(e) Manage to increase soil C	125	20–100	3–13
Total			25–102

•Impact of management activities on wildfire are not clear.
•Although past afforestation is not an activity that can now be influenced, it is included here as a contribution to total potential soil carbon changes.
•Plantation forest (16 Mha including 1 Mha/y of afforestation) and wildlife protection (21 Mha) are included under this category.
•There is no data on the impact of fire management on soil C sequestration.

tion, converting nonforest land to forest, is an important land use change for SOC sequestration. The rate of afforestation at any given time depends on land characteristics, socioeconomic conditions, political factors, and technological advances. Afforestation also depends on the multiple uses of intended land use and forest products. Afforestation of marginal agricultural lands for timber production is also important to soil C sequestration. There are 16 Mha of plantation forests in the U.S. (Birdsey and Lewis, 2003), and the average rate of afforestation is about 1 Mha/y, of which only a small fraction is used for short-rotation production of biomass or energy (Tolbert et al., 2000). In addition, 21 Mha of reserved forestland is withdrawn from timber production and converted to wilderness and wildlife protection (Birdsey and Lewis, 2003).

The Conservation Reserve Program (CRP) in the U.S. has been extremely effective in reducing the sediment load in rivers and enhancing the SOC pool. It involves converting highly erodible land (HEL) and idle/marginal cropland to CRP (Follett, 2001a). The rate of SOC sequestration under CRP may be 600 to 900 Kg C/ha/y (Follett et al., 2001b). An issue neither discussed yet at the national policy level nor evaluated, but that has the potential to result in additional C sequestration under CRP, is recognizing that these lands represent a large area of land use in the U.S. They are often, in part, degraded by erosion or are marginal lands, and there are presently no requirements for soil testing, different management practices, or other evaluation that might aid in enhancing C sequestration on these lands. At this time, CRP land remains ungrazed by domestic livestock. Follett (unpublished data) has noted a substantial increase in the rate of soil C sequestration on N-deficient CRP land with the application of fertilizer N; fertilization with other nutrients (perhaps phosphorus) also needs to be evaluated. In addition to having adequate fertility on CRP lands, grazing management might also be expected to aid in the control of undesirable species and promotes domination by perennial grasses and plant community stability and diversity to promote C sequestration. Grazing management aids litter decomposition into soil hu-

TABLE 6

Soil carbon sequestration by other land uses in forestry (modified from Heath et al., 2003)

Scenario	Area (Mha)	Rate (kg C/ha/y)	Potential (Tg C/y)	
			Actual	Cumulative
1. Alley cropping	80	173–288	14–23	
2. Riparian buffers			0.4–0.6	
3. Windbreaks	85	8–15	0.7–1	
4. Urban forest			0–3	
5. Urban management	1	0–3000	0–3	
Total C sequestration				15–25

mus through the effects of grazing and animal traffic, whereas removal of excess standing dead material can allow earlier spring green-up, sunlight penetration, and soil warming (Lecain et al., 2000; Schuman et al., 1999).

Estimates of the potential of soil carbon sequestration through land conversion in the United States are shown in Table 7. The potential is 11 to 18 Tg C/y for land conversion from cropland to CRP, WRP, restoration of wetlands and organic soils, and conservation buffers; 2 to 6 Tg C/y for land conversion from cropland to pasture; 2 to 3 Tg C/y for land conversion from forest to pasture; and 7 to 52 Tg C/y by afforestation practices. Thus, the total potential of soil C sequestration through land conversion in the United States is 22 to 109 Tg C/y.

Restoration of Degraded Ecosystems

Restoration of eroded and drastically disturbed soils also provides an opportunity to resequester some of the SOC depleted by degradative processes. There are several techniques for restoring degraded/desertified soils and ecosystems: increasing vegetative cover by afforestation and other perennial vegetation; improving soil fertility by application of biosolids and fertilizers/amendments; and conserving and recycling water. Similar techniques are applicable to the restoration of minesoils (Akala and Lal, 2000). Municipal sludge and other biosolids are also effective in reclaiming minesoils (Seaker and Sopper, 1988).

Measured rates of SOC sequestration through soil restoration vary widely, depending on soil, land use, and ecoregional characteristics. Reclamation of minesoils in Ohio increased the SOC pool to 30-cm depth by 35 to 37 Mg C/ha over a 25-year period (Akala and Lal, 2000). The Wetland Reserve Program (WRP) also leads to C sequestration in restored wetlands and their watersheds (Lal et al., 1998b). The importance of coastal wetland soils is neither their area nor their rate of C sequestration, although it is can exceed 800 kg C/ha/yr (Connor et al., 2001), but that they accrete and bury organic-rich sediments continuously over centuries. If an assumption is made that an average rate C sequestration is 400 to 800 kg C/ha/yr (Connor et al., 2001; Rabenhorst, 1995), as long as these coastal wetlands can be protected, the huge stores of C buried in them accumulate continuously and unabated. The potential for SOC sequestration through the restoration of wetlands and conversion of organic soils, with a combined area of 19 Mha, is 8 to 15 Tg C/y. The potential of soil C sequestration in the U.S. is estimated at 13 to 31 Tg C/y for restoration of eroded croplands, 2 to 10 Tg C/y for restoring eroded grazing lands, 2 to 4 Tg C/y for restoring minelands, 0.01 to 0.03 Tg C/y for restoring salt-affected soils, and 8 to 15 Tg C/y for restoration of organic soils. Total potential of soil C sequestration through restoration of degraded ecosystems in the U.S. is 25 to 60 Tg C/y (Table 8). Urban land use (e.g., household and corporate lawns, golf courses, sports arenas, recreational land use) utilizes intensive management including fertilizers, pesticides, irrigation, mowing etc. Judicious use of input and return of clippings can enhance soil C sequestration (Pouyat et al., 2002, 2003; Qian and Follett, 2003).

TABLE 7

Soil carbon sequestration through land conversion (adapted from Lal et al., 1998b; Follett, 2001a; Follett et al., 2001a)

Scenario	Area (Mha)	Rate (kg C/ha/y)	Potential (Tg C/y)	
			Actual	Cumulative
1. Land conversion from cropland				10–16
Conservation Reserve Program	14.7	600–900	9–13	
Conservation buffers	3.2	300–700	1–2	
Wetland reserve program	2.0	200–300	0.4–0.6	
2. Land conversion from cropland to pasture				2–6
Estimated current area (4%)	2.4	400–1200	1–2.9	
Potential additional area (4%)	2.4	400–1200	1–2.9	
3. Land conversion from forest to pasture				
Silvopasture	70	25–40	2–3	2–3
4. Land use change in forest ecosystems				7–52
(a) Increase afforestation	10.0	0–676	0–7	
(b) Reduce deforestation	0.8	1740–3461	1–3	
(c) Past afforestation continuing to accrue C[b]	62.0	100–676	6–42	
Total				21–77

TABLE 8

Soil carbon sequestration through land restoration (adapted from Lal et al., 1998b; Follett et al., 2001a)

Scenario	Area (Mha)	Rate (kg C/ha/y)	Potential (Tg C/y)	
			Actual	Cumulative
1. Eroded cropland	43.7	300–700	13–31	
2. Eroded grazing land	51.4	50–200	2–10	
3. Mine lands	4.4	500–1000	2.2–4.4	
4. Salt affected soils	0.2	50–150	0.01–0.03	
5. Restoration of organic soils	19.0	400–800	8–15	
				25–60

HIDDEN CARBON COSTS OF RECOMMENDED MANAGEMENT PRACTICES

Adopting RMPs involves C-based inputs (e.g., fertilizers and pesticides, irrigation, tillage, crop drying). Furthermore, agricultural activities also lead to emission of several GHGs (e.g., N_2O). Thus, many researchers have suggested that hidden C costs of agricultural intensification be deducted from gross C sequestration to assess the net rate of C sequestration (Schlesinger, 1999; Robertson et al., 2000; Smith et al., 2001; West and Marland, 2002).

Estimates of the hidden C costs of adopting RPMs on croplands, grazing lands, forestlands, and urban soils are shown in Table 9. Hidden C costs are estimated on the basis of fertilizer and pesticide use, fuel combustion, and irrigation water use. The footnote in Table 9 provides the statistics used to compute these numbers. The hidden C costs are 18 to 22 Tg C/y for cropland, 3 to 4 Tg C/y for grazing lands, about 1 Tg C/y

TABLE 9

Estimates of hidden carbon costs (Tg TCE/y)

Land use	Category	Operation	N_2O emissions	Total
1. Cropland				
	(i) Rainfed	2.6–2.8		
	(ii) Irrigated	2.0–4.4		
	(iii) Idle		13.0–14.3	
	Sub-total	4.6–7.2	13.0–14.3	17.6–21.5
2. Grazing land				
	(i) Pasture	0.14–0.16		
	(ii) Grassland	0.10–0.12	3.0–3.3	
	Sub-total	0.24–0.28	3.0–3.3	3.2–3.6
3. Forest land		0.03–0.05	0.7–0.8	0.7–0.9
4. Conservation Reserve Program (CRP)				
5. Wetland Reserve Program (WRP)				
6. Special use				
7. Urban		4.0–4.4	1.9–2.1	5.9–6.5
8. Other land				
Total		8.8–11.9	18.6–20.5	27.4–32.4

•Estimates of hidden C costs are based on statistics on fertilizer, pesticide and irrigation (USEPA, 1999), and hidden C costs required for manufacture and use of chemicals, and for N_2O emissions (Robertson et al., 2000; West and Marland, 2002). All emissions are calculated as carbon equivalent using GWP of 1 for CO_2 and 310 for N_2O (IPCC, 2001). Farm chemical use in the U.S. includes 11.2 Tg/y of N, 4.2 Tg/y of P_2O_5, 4.8 Tg/y of K_2O, 2.6 Tg/y of lime, 0.166 Tg/y of herbicides, 0.023 Tg/y of fungicides and 0.027 Tg/y of insecticides. Of these chemicals, it is assumed that 70% are used on cropland, 16% on grazing lands, 4% on forestlands and 10% on urban lands.

•Hidden C costs for tillage methods include 62–72 kg C/ha/y for conventional tillage, 40–65 kg C/ha/y for minimum tillage and 20–23 kg C/ha/y for no-till.

•Hidden C costs of fertilizers are 0.86 kg C/kg of N, 0.17 kg C/kg P_2O_5, 0.12 kg C/kg K_2O, and 0.36 kg C/kg lime.

•Hidden C costs of pesticides are 4.7 kg C/kg of herbicides, 5.2 kg C/kg of fungicides and 4.9 kg C/kg of insecticides.

•Hidden C costs for irrigation include 150 kg C/ha/y (Follett, 2001a).

•Computations of hidden C costs include 10% additional fertilizer and pesticide use for adopting RMPs.

for forestland and 6 to 7 Tg C/y for urban land. Therefore, the total hidden C costs of adopting RMPs are estimated at 27 to 32 Tg C/y. It is the intensive management of urban land use that is responsible for high levels of hidden C costs. Indeed, 1 Mha of urban land use has 30% of the hidden C costs of 157 Mha croplands. Thus, a judicious use of chemicals, irrigation and mowing frequency is prudent to reduce gaseous emissions and risks of water pollution from urban lands.

Despite the arguments in favor of assessing the net C sequestration, justification to do so requires the recognition that adoption of RMPs is to provide food and fiber for both domestic and export needs, enhance agricultural productivity, improve farm income, and achieve the sustainable use of soil and water resources. Soil C sequestration is a by-product of adopting RMPs, which is needed to enhance production and achieve global food security. Farmers are not using fertilizers and irrigation water to sequester C but to improve and sustain agricultural production. Similarly, livestock are raised for producing meat and milk rather than dung for manuring agricultural fields to enhance SOC sequestration.

CARBON SEQUESTRATION POTENTIAL OF SOILS OF THE U.S.

The data in Table 10 show the total C sequestration potential for soils in the U.S. by adopting RMPs on all land uses. The potential is 144 to 432 Tg C/y or an average of 288 Tg C/y. Of the total potential, 25.0% is from croplands, 21.9% from forestland, 14.6% from grazing lands, 17.0% from land conversion, 14.6% from land restoration, and 6.9% from improving other land uses (Fig. 2). In contrast, CO_2 emission from agricultural activities is 43 Tg C/y (Lal et al., 1998b). Therefore, the potential of C sequestration is 6.7 times the emission from agricultural activities. Total U.S. emissions for 2001 are about 1892 Tg C.E. (USEPA, 2001). Thus, soil C sequestration potential of 288 Tg C/y is 15% of the total emissions. There is an additional potential of U.S. forests biomass to offset 15% of the total national emissions (USEPA, 2001). Thus, the total potential of terrestrial C sequestration in the U.S. is about 30% of the national emissions of CO_2 equivalent.

TABLE 10

Carbon sequestration potential of soils in the U.S.

Land use	Soil C sequestration potential (Tg C/y)	
	Range	Mean
1. Cropland	45–98	72
2. Grazing land	13–70	42
3. Forest lands	25–102	63
4. Land conversion	21–77	49
5. Soil restoration	25–60	42
6. Other land use	15–25	20
Total	144–432	288

Except for cultivated organic soils, agricultural soils in the U.S. are currently estimated to be accumulating C at the rate of about 16 to 17 Tg C/y (Sperow, et al., 2003; USDA, 2003). When the Century Model is used, much higher rates would be expected (USDA 1993). Much of the current sequestration is attributed to CRP, adopting conservation tillage, either eliminating or reducing the intensity of summer fallow, and applying manure and biosolids. Compared with the present rate of sequestration, the potential of soil C sequestration is 18 times greater.

CHALLENGE TO POLICY MAKERS

Managing C is one of the more critical issues facing society at the on-set of the 21st century (Kimble et al., 2002). Without technological or behavioral intervention, atmospheric concentration of CO_2 may double by 2100, with a drastic impact on the climate and natural and managed ecosystems. There is a need to identify market-based policy mechanisms to facilitate adopting RMPs to achieve the potential of soil C sequestration. Policies are needed, particularly at the national and state levels, to encourage good land stewardship, and adoption of land practices and management systems are needed to restore ecological functions and rehabilitate the land and landscape. Policies for adopting RMPs must be in place irrespective of the argument that "the jury is still out on the projected climate change." Soil C sequestration is needed to restore soil, air, and water quality even if the global climate change does not happen, and it is needed even more in the event that global climate change does happen.

Estimates presented herein are potential and cannot be realized without implementing policies to promote the adoption of existing and improved RMPs. Policy makers need to recognize that the IPCC methodology, when used to estimate the annual rate of C sequestration that occurs at the present time under current management practices and then when estimates are made, again using the same IPCC methodology but with improved RMPs, results in data that is only about 20% of the potential (Sperow et al., 2003). Potential estimates made in this paper for U.S. croplands are nearly identical to the potential estimates made by

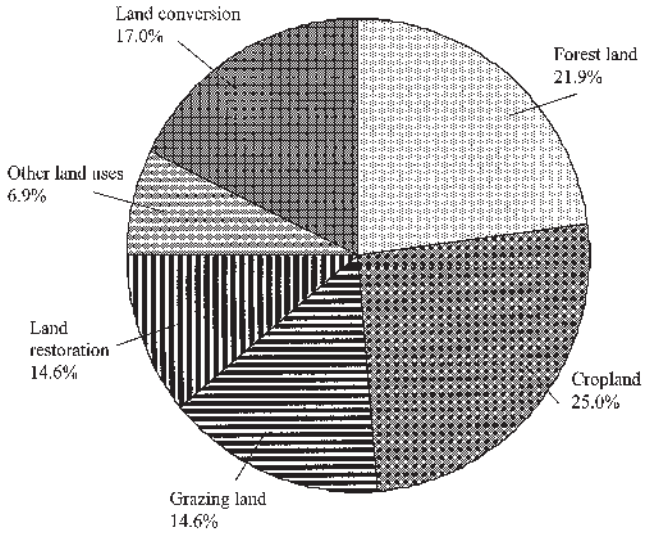

Fig. 2. The potential of U.S. soils for C sequestration (total = 288 Tg C/y).

Sperow et al. (2003) with the IPCC methodology. In addition to and because of the above, present estimates generated using the IPCC methodology for grazing lands or other types of land uses are low and represent a much reduced percentage of the potential soil C that these other land types can sequester as well.

Carbon trading policies developed at the national level to reduce GHG emissions could benefit U.S. farmers while also advancing the goals of other conservation initiatives. A properly tailored C trading program has the potential of being a cost-effective option for offsetting GHG emissions while also complementing existing air, water, and soil quality programs. Soil C sequestration would result in the sustainability of soil resources, maintenance of agronomic productivity, and improved water quality. The financial benefits that farmers may receive and attendant environmental gains can only be achieved if the policy makers adopt a pragmatic policy. The administration has announced the USDA's plans to invest $3.9 billion in agriculture and forestry conservation on private land in 2004, leading to C sequestration of 12 Tg/y by 2012 (USDA, 2003; Knight, 2003). This initiative is most welcome, and it is indicative of the acceptance of soil C sequestration as a viable option for mitigating the climate change. However, there is little incentive to producers to increase C sequestration or to traders to develop C markets when the goal is unrealistically low. For a goal of merely 12 Tg C/y, where is the incentive to do more even on cropland, or on grazing lands or forestland? Recognizing that there are emissions from organic soils equivalent to about 9.5 Tg C, these amounts become minor when compared with the estimates of the potential range from 45 to 98 Tg C/y for cropland alone (Bruce et al., 1999; Lal et al., 1998b; Sperow et al., 2003) and 136 to 417 Tg C/y for all land uses in the U.S. (Table 10). If the estimate of 12 Tg/y comprises C sequestered in soil and biomass, it is trivial, and, more realistically, needs to be 5 times larger or 60 Tg by 2010, 10 times larger or 120 Tg by 2020, and 20 times larger or 240 Tg by 2030. Setting a goal that is below the current rate of sequestration can jeopardize the continuation of even the existing programs (e.g., CRP conservation compliance) and dampen the enthusiasm of implementing new programs. Achieving these realistic goals is a challenge to the policy makers.

We are also faced with the question, often framed as a challenge, of how we can accurately, transparently, and economically measure soil C pool and flux, and whether soil C sequestration is a "pie in the sky" theory. The science of measuring soil C concentration has existed since the latter part of the 19th century (Wolff, 1864; van Bemmelen, 1890). The wet combustion method of determining soil organic matter has been taught in beginning soils classes since the 1930s (Walkley and Black, 1934) and newer, more accurate combustion methods are now used routinely.

The scaling procedures for extrapolating data from the pedon level to landscape, watershed, and regional scales have been available since the 1980s (Lal et al., 2001). Numerous advances are being made in monitoring soil C at the landscape level through remote sensing techniques (Shepherd and Walsh, 2002). Admittedly, the soil C pool is variable over time and space, but no more than other natural systems, including trees or forests. Similar to the methods of soil C assessment, the idea that policies are needed to manage soil C (albeit for soil fertility enhancement) is not new either. Albrecht (1938) pointed out "up to the present, the policy – if it can be called a policy – has been to exhaust the supply rather than to maintain it by regular additions according to the demands of the crop produced or the soil fertility removed. To continue very long with this practice will mean a further sharp decline in crop yields." Although intensive use of chemical fertilizers during the later half of the 20th century masked the adverse impacts of decline in soil organic matter on crop yields, mining of soil organic matter continues unabated at national and global scales.

Fortunately, many RMPs, when adopted, do result in increased rates of C sequestration under on-farm conditions similar to those reported in long-term experiments. Policies that we feel are needed include those that can help reduce risks and costs associated with, for example, conversion from conventional till to no-till systems, which can be done with strong incentives in conservation programs. Such incentives might include assistance for purchase of equipment, costs associated with the use of cover crops, reduction of crop insurance costs to cover the 3- to 5-year period during the restoration to improve soil health. Many farmers are not willing to make changes in management now because of associated risks, such as lower yields for the 3 to 5 years it takes for no-till to work effectively, and a lack of incentives and low commodity prices make it difficult for them to bear the costs of making such changes. Unfortunately, failure to adopt RMPs results in lower rates of C sequestration, continued higher net losses of soil C to the atmosphere as CO_2, and excess soil erosion and sediment/chemical losses that can degrade or lower water and soil quality.

There is an urgent need for intensive and interactive dialogue on a continuous basis between soil scientists and land managers on the one hand and between soil scientists and policy makers on the other. While policy makers are still debating the science and impact of climate change (Malakoff, 2003), it is important to establish a *modus operandi*, a continued interaction between soil scientists and policy makers. It is important for soil scientists to ensure that their findings are accessible and made available in a user-friendly language for policy and legislative use. It is equally important for policy makers and legislators to consult with the scientific community when formulating policies of national and international scope. Barriers to greater interactions between and among the scientific and policy-making communities lie in the partisan nature of the decision-making process and the narrow or discipline-based training of the scientists. Indeed, a bipartisan approach is needed to address this issue of global significance. There is also a need for nurturing the delicate relationship between soil scientists and policy makers. Because scientists have neither the training nor the opportunity to interact in the arena where policies are created and legislation is debated, it is imperative that policy makers create both the opportunity and the environment to obtain scientific views, especially when the problems are complex and the policies may have a long-lasting impact. The gravity of the problem of soil C sequestration to mitigate climate change and to enhance the environment is one such complex issue that warrants a continuous interactive dialogue between soil scientists and policy makers.

CONCLUSIONS

The potential of C sequestration in soils of the United States estimated at 288 Tg C/y is 6.7 times the emission of 43 Tg C/y by agricultural activity and 16.9 times the current rate of soil C sequestration estimated at 17 Tg C/y. Thus, there is a vast scope for enhancing soil C sequestration through the adoption of recommended management practices on prime lands and conversion to restorative land use for degraded/desertified and drastically disturbed lands. Methods of measuring soil C have existed since the dawn of the 20th century, and rapid advances were made during the 1980s and 1990s in measuring soil C economically and precisely and in extrapolating the data to watershed regional scales through developments in scaling procedures. However, achieving the potential of soil C sequestration remains a challenge to policy makers. There is a need to develop realistic but scientifically credible goals of soil C sequestration. Targeting soil C sequestration at 60 Tg C/y by 2010, 120 Tg C/y by 2020, and 240 Tg C/y by 2030 is feasible, and at a modest trad-

ing cost of $20/Mg it would enhance farm income directly as these targets were implemented by 1.2 billion, 2.4 billion, and 4.8 billion dollars/y, respectively. These targets are achievable provided that policies are implemented to reduce risks and costs associated with adopting recommended management practices, such as conservation tillage and use of cover crops in the rotation cycle, restoration of eroded and degraded soils, use of fertilizers on grazing lands and forest ecosystems, etc. Many farmers/land managers are not willing to adopt recommended management practices because of the risks involved. Soil C sequestration policies can be implemented only if there is a political will to act to develop the needed policies and there are supporting institutions and infrastructures for on-site verification. There is a strong need for implementing a National Soil Policy to safeguard national soil resources and to sequester C for enhancing productivity and improving the environment.

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