Mitigation of Climatic Change by Soil Carbon Sequestration: Issues of Science, Monitoring, and Degraded Lands

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I. INTRODUCTION

Farmers, gardeners, and, of course, agronomists know that adding organic matter to soils is a good thing to do. Organic matter increases soil water-holding capacity, imparts fertility with the addition of nutrients, increases soil aggregation, and improves tilth. Depending on its type—humus, manure, stubble, litter—organic matter contains between 40 and 60% carbon.

It is also known with certainty that carbon (C, hereafter), in the form of carbon dioxide (CO_2), is currently accumulating in the atmosphere at the rate of about 3.4 Pg year^{-1} (1 Pg is equal to 1 billion tonnes or 10^{15}) as the result of fossil fuel combustion and land use change (Table I). The atmospheric concentration of CO_2 has increased by about 30% since the beginning of the industrial revolution (ca. 1850) from about 280 to about 370 ppmv today. There is a strong consensus among atmospheric scientists that continued increase in the concentration of atmospheric CO_2 and other greenhouse gases such as methane (CH_4) and nitrous oxide (N_2O) will enhance the earth's natural greenhouse effect and lead to global warming (Intergovernmental Panel on Climate Change, IPCC, 1996). Some scientists argue that the warming "footprint" is already detectable.

Recognizing the prospect that emissions of greenhouse gases at increasing or even continuing rates will likely lead to global warming, the United Nations adopted a Framework Convention on Climate Change (UNFCCC) in Rio De Janeiro in

Table I
Global C Flux Budget

Carbon Flows	Pg C year^{-1}
Sources	
Fossil fuels	6.4
Land use change	1.1
Tropical deforestation	1.6
Total sources	9.1
Sinks[a]	
Atmospheric increase in CO_2	3.4
Terrestrial in temperate regions	2.0
Oceans	2.0
"Missing"	1.7
Total sinks	9.1

[a]Potential sinks in croplands 40–80 Pg in 50–100 years according to IPCC (1996).

1992 (United Nations, 1992). The convention aims at the "stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system." The parties to the UNFCCC met in Kyoto, Japan, in December 1997 and drafted a protocol to place binding limits on greenhouse gas emissions and to begin the process of stabilizing their atmospheric concentrations (United Nations, 1997). The protocol recognizes that its objectives can be met either by decreasing the rate at which greenhouse gases are emitted to the atmosphere or by increasing the rate at which they are removed from it. In the case of CO_2, removal from the atmosphere and its capture in the biosphere is possible.

Table 1 gives current estimates of global sources and sinks for C, and indicates that about 2 Pg C is "missing," indicating that the fate of this C is still unknown. Some C may be going into oceanic sinks. It is believed, however, that most is being taken up by the terrestrial biosphere primarily because of the regrowth of forests in the Northern Hemisphere (e.g., Houghton *et al.,* 1999). Stimulation of photosynthesis, primarily in plants with C-3 metabolism, and reduction of transpiration in both C-3 and C-4 plants due to the so-called CO_2-fertilization[1] effect may also contribute to increased CO_2 uptake. Where vegetation is changing in kind or in vigor, above-ground and below-ground inventories of C must change too.

There is also evidence that, globally, soils have the capacity to accommodate the addition of substantial amounts of C drawn from the atmosphere by photosynthesis and to sequester it for a long enough time to significantly reduce the accumulation rate of atmospheric CO_2. The Intergovernmental Panel on Climate Change, in its Second Assessment Report (IPCC, 1996), estimated that it may be possible over the next 50–100 years to sequester 40 to 80 Pg of C in cropland soils (Cole *et al.*, 1996; Paustian *et al.*, 1998; Rosenberg *et al.*, 1998). If so, these soils could capture enough C to offset any further increase in the atmospheric inventory for a period lasting between 12 and 24 years. There is additional C sequestration potential in managed forests and grassland soils. These calculations are crude and cannot be taken as certain, but they do suggest a potential to offset significant amounts of CO_2 emissions. It must be remembered, however, that with growing populations and rising standards of living future CO_2 emissions may be much higher than they are today (see Fig. 1).

Another way of looking at the potential role of soil C sequestration is illustrated in Figure 1, produced with the integrated assessment model MiniCAM 98.3 (Edmonds *et al.,* 1996a, 1996b; Rosenberg *et al.,* 1999). The top line in the figure represents the anticipated increase in carbon emissions to the atmosphere from the year 2000 to the end of the 21st century under the so-called "business-as-usual"

[1]Photosynthesis in C-3 species is stimulated by increases in $[CO_2]$ of the ambient air. In both C-3 and C-4 species elevated $[CO_2]$ increases stomatal resistance and hence reduces transpiration. By reducing moisture stress on plants, the latter effect also fosters greater photosynthetic production.

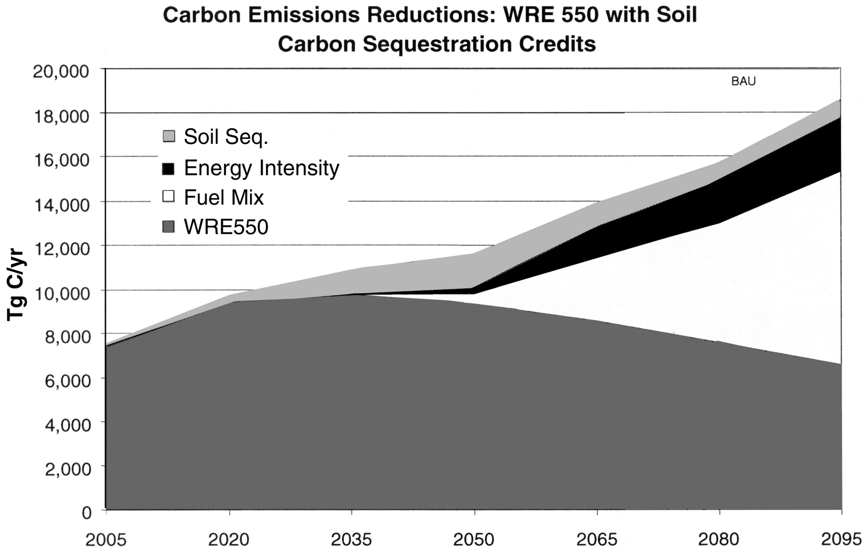

Figure 1 Global carbon emissions reductions: WRE 550 (Wigley *et al.*, 1996, 550 ppmv atmospheric CO_2 concentration). This figure shows a hypothetical path to carbon emissions reductions from MiniCAM's business as usual (BAU) emissions pathway to the WRE 550 concentration pathway, under a scenario in which credit for soil carbon sequestration is allowed. Soil sequestration of carbon alone achieves the necessary net carbon emissions reduction in the early part of the century. From the middle of the century on, further emissions reductions must come from changes in the energy system (such as fuel switching and the reduction of total energy consumption). (From Rosenberg *et al.* (1999). With permission.)

scenario of IPCC (1990). It also shows a more desirable emissions trajectory that allows atmospheric [CO_2] to reach a maximum of 550 ppmv by 2035 and lowers it somewhat thereafter. Increased efficiency in the uses of fossil fuels and changes in the fuel mix (i.e., a greater role for biomass, solar, wind, and nuclear energy) will bear the brunt of lowering CO_2 emissions. While soil C sequestration alone cannot solve the problem, Figure 1 shows that it can play a very strategic role. Sequestration alone can achieve the net C emissions reduction in the early part of the 21st century needed to keep to the desired trajectory.

The calculations shown in Figure 1 are based on the assumption that from 2000 to 2100 agricultural soils sequester C at global annual rates ranging from 0.4–0.8 Pg year^{-1}, with rates twice as great in the initial years and half as great in the later years. It is further assumed that the full potential of soil C sequestration is realized without any additional net cost to the economy. This is not an unreasonable assumption since, as stated above, the addition of organic substances to soil improves agricultural productivity.

How realistic are the estimates of potential soil C sequestration on which the economic modeling is based? The IPCC estimates for cropland assume the restitution of up to two-thirds of the soil carbon released since the mid-19th century by the conversion of grasslands, wetlands and forests to agriculture. The experimental record confirms that C *can* actually be returned to soils in such quantities. A few examples: Carbon has been accumulating at rates exceeding 1 Mg ha^{-1} year^{-1} in former U.S. crop lands planted to perennial grasses under the Conservation Reserve Program (CRP) (Gebhart *et al.,* 1994). Soil C increases ranging from 1.3 to 2.5 Mg ha^{-1} year^{-1} have been estimated in experiments on formerly cultivated land planted to switchgrass (*Panicum virgatum*), a biomass crop (preliminary data, Oak Ridge National Laboratory). Further, there have been a substantial number of experiments over the last two or three decades with minimum tillage and no-till management of farm fields demonstrating that such practices lead to increases in soil C content (Lal *et al.,* 1998a; Nyborg *et al.,* 1995; Janzen *et al.,* 1998).

Managed forests, wetlands, and rangelands provide further opportunity for significant C storage. For example, when agriculture is converted (or allowed to revert) to forest vegetation in systems with very little management to improve growth, soil C may accumulate at rates ranging from near 0–7 Mg ha^{-1} year^{-1}. In temperate regions gains range from 0.2–0.6 and average about 0.3 Mg ha^{-1} year^{-1} (Jenkinson, 1971). Gains in tropical and subtropical forests are greater, ranging from 1.0–7.4, with an average of 2.0 Mg ha^{-1} year^{-1} (Ramakrishnan and Toky, 1981).

The Kyoto protocol establishes the concept of credits for C sinks (Article 3.3) but allows credits for only a limited list of activities, including afforestation and reforestation (Article 3.4). The protocol does not allow credits for sequestration of C in soils except perhaps (indeed, this is not yet clear) for carbon accumulating in afforested and reforested land. Sequestration in agricultural soils is not permitted to produce C sequestration credits under the Kyoto protocol, but it is mentioned specifically in Article 3.4 for possible inclusion at a later time. A major objection to the inclusion of agricultural soil C sequestration voiced in the Kyoto negotiations is the perceived difficulty in verifying that C is actually being sequestered and maintained in soils.

The prospect opened by the IPCC findings and the Kyoto protocol that carbon may become a tradable commodity has not gone unnoticed in the agricultural and forestry communities. If credit toward national emissions targets could be gained by increasing the stores of carbon on agricultural lands, various land-management practices might be encouraged. The nature of a carbon trading system that encompasses soil sequestration will not be explored in this paper. Utilities and other emitters of greenhouse gases that are anticipating a future regime in which reductions in CO_2 emissions become mandatory will be searching for cost-effective ways to offset or otherwise meet the limits imposed. Direct payments from the emitter to the sequesterer or a market in tradable permits could open up new ways for farmers to augment their incomes.[2]

Marland *et al.* (1999) have speculated on both positive and negative "externalities" of programs involving agriculture to cap and trade C emissions. Such programs would likely increase the commitment of land to reduced tillage practices. They might provide incentive for restoration of degraded lands and for retirement of agricultural lands into permanent grass or forest cover. They might encourage continuation and/or expansion of Conservation Reserve programs and lead to improved management of residues in agricultural harvests. All of these have the potential of reducing soil erosion and its negative consequences for water quality and sedimentation. In addition, reduced tillage could increase soil organic matter content and soil water-holding capacity, leading to a reduced need for irrigation water. Expanded conversion of agricultural lands to grasslands or forests could stimulate wildlife populations. Reduced soil disturbance and, possibly, diminished use of fertilizer could alter the volume and chemical content of runoff from agricultural lands. This would in turn reduce water pollution and improve water quality and the general ecology of streams, rivers, lakes, and aquifers in these regions for use by nonagricultural water consumers.

But negative "externalities" are also possible. Adams *et al.* (1992) and McCarl (1998) show, for example, that programs designed to move agricultural lands into forestry could negatively affect the traditional forest sector, leading to either deforestation of traditional parcels or reduced levels of management and lessened C sequestration that offset much of the total amount of C being sequestered. Expanded use of agricultural lands for C sequestration might compete with the use of agricultural lands for traditional food and fiber production. The result might be decreased food and fiber production, increased consumer prices for crops, meat, and fiber, and decreased export earnings from agriculture. Reductions in intensity of tillage, coupled with the increased amount of plant material left on the soil surface, have been commonly found to require additional use of pesticides for weed, fungus, and insect management. This may have deleterious effects on ecological systems, runoff, and water quality. Conversion of croplands to grasslands tends to decrease emissions of nitrous oxide (N_2O), and also increases the soil sink for methane (CH_4), another greenhouse gas (Mosier *et al.*, 1997).

It is not the purpose of this paper to review all that is known about soil C dy-

[2]As this paper was being completed, Trans Alta Corporation, a member of GEMCo (the Greenhouse Emissions Management Consortium), announced a major agreement to purchase up to 2.8 million tonnes of carbon emission reduction credits (CERCs) from farms in the U.S. According to the Trans Alta news release (October 19, 1999) the IGF insurance company will solicit the CERCs from elegible farmers or landowners, initially from Iowa and ultimately nationwide.

namics. There are many good sources of information on that subject (e.g., Batjes, 1999; Lal *et al.*, 1998a; Stevenson and Cole, 1999). It is the purpose of this paper to emphasize the research and technology needed to facilitate implementation of soil carbon sequestration programs and to identify a particular situation in which the practice can help achieve sustainable development. This paper draws heavily on the proceedings of a conference, Carbon Sequestration in Soils: Science, Monitoring and Beyond (Rosenberg *et al.*, 1999), held in St. Michaels, Maryland, in December 1998 and sponsored by USDOE, USDA, USEPA, NASA, and Monsanto. The paper deals with the following three issues which seem especially pertinent at this early stage of shaping opportunities for implementation of soil carbon sequestration programs:

1. *New science:* The potential for carbon sequestration in all managed soils is large and progress can be made using proven crop, range, and forest management practices. The potential might be even greater if ways can be found to restore more than the two-thirds of the carbon that has been lost from conversion to agriculture, and perhaps even to exceed original carbon contents in some soils and regions. This would involve a search for ways to effect greater, more rapid, and longer-lasting sequestration. New lines of research that promise to improve understanding of soil carbon dynamics and lead to improved sequestration methods are suggested. There is, however, "no free lunch," even in the case of such an apparently benign activity as soil carbon sequestration. Legitimate questions have been raised (e.g., Schlesinger, 1999) as to whether the energy costs of increasing sequestration could significantly reduce the apparently favorable net carbon-balance benefits that might otherwise be realized. New science opportunities are addressed in Section III.
2. *Monitoring and verification:* International agreements allowing soil sequestration to figure into a nation's carbon balance will require agreed-upon means of verification. Current methods may be costly and time consuming and are often not sensitive enough to distinguish year-to-year changes in soil carbon. Improved methods for monitoring changes in soil organic carbon might involve spatial integration based on process modeling and geographic information systems, application of high-resolution remote sensing, and continuous direct measurements of CO_2 exchange between the atmosphere and terrestrial ecosystems and/or the development of "carbon probes." Monitoring issues are addressed in Section IV.
3. *The soil carbon sequestration/desertification linkage:* The potential for carbon sequestration is probably greatest on desertified and degraded lands. It is estimated that there are approximately 2 billion hectares of such lands worldwide, 75% of them in the tropics, with degradation most severe in the dry tropics. The linkages between loss of soil carbon and the desertification process are not fully understood, nor are the mechanisms whereby restoration of carbon in soils can help to halt or even reverse desertification. The potential for increased sequestration on degraded lands through erosion control, agricultural intensification, forest establishment in dry regions, and biomass cultivation is explored in Section V.

Before addressing the three issues of this review, we provide, in Section II, some additional information on the nature and dynamics of the soil organic and inorganic C pools. We describe the mechanisms through which C has been lost from agricultural fields and the practices that make it possible to restore it. We also briefly assess how movements toward conservation tillage and organic farming and the relatively new phenomena of genetically modified crops and precision farming can facilitate efforts to sequester C in soils.

II. PAST, PRESENT, AND FUTURE

A. Causes of Carbon Loss From Soil

Carbon is a component of two important soil compartments or "pools": soil organic C (hereafter SOC) and soil inorganic C (hereafter SIC). Carbon in the SOC fraction appears in complex mixtures of nonhumic (e.g., carbohydrates, proteins, and amino acids) and humic (products of secondary synthesis) substances. Carbonates (CO_3^{-2}) and bicarbonates (HCO_3^-) make up the SIC fraction. The SOC pool is the predominant form of C in soils of humid and subhumid regions. SIC is found predominantly in soils of arid and semi-arid regions. Both SOC and SIC can release C from soil in the form of CO_2.

1. SOC Dynamics

The balance between carbon additions via decomposition of photosynthetic products and losses via microbial respiration determines the amount of organic C stored in soil. Soils are said to reach steady state when carbon additions balance carbon losses. The capacity of soils to store organic carbon varies widely, from less than 1% in sandy soils to 10% or more in poorly drained soils (Stevenson and Cole, 1999). What determines this capacity? Hans Jenny and co-workers were among the first to construct a hierarchical model to explain the relative importance of soil-forming factors on the accumulation of total nitrogen (N) in loamy soils. They arranged these factors in descending order of importance as climate, vegetation, topography, parent material, and age.

It is well established that converting native ecosystems into agricultural fields for food and fiber production generally leads to a decline in SOC (Jenkinson and Ayanaba, 1977; McGill *et al.*, 1988; Mann, 1986; Davidson and Ackerman, 1993; Ellert and Gregorich, 1996). In exceptional cases, the SOC pool can increase in ecosystems in which biomass production is severely constrained by some soil factor (e.g., Al toxicity, P deficiency, species with low root:shoot ratio). Results from numerous long-term experiments have demonstrated SOC losses in temperate climates of up to 60% occurring within 30–50 years of land use conversion (Donigian *et al.*, 1994; Parton *et al.*, 1988) and 5–10 years in tropical ecoregions (Nye and Greenland, 1960; Lal, 1996). Decreases in the SOC pool may be caused by three, often simultaneous, processes: mineralization, erosion, and leaching.

1. *Mineralization:* Most of the biomass produced in natural ecosystems is returned to the soil. However, the rate of mineralization in agricultural ecosystems often exceeds the rate of C accretion occurring through addition of roots and biomass. Further, conversion of natural to agricultural ecosystems can drastically alter soil moisture and soil temperature regimes. Higher soil temperature increases the rate of mineralization of the SOC pool (Jenny and Raychaudhary, 1960; Jenkinson and Ayanaba, 1977). The rate of mineralization can double with every 10°C increase in ambient temperature.

2. *Soil erosion:* Conversion of natural ecosystems to agricultural use generally leads to significant increases in the rates of soil erosion by both water and wind. Soil erosion can be described as a three-stage process: (i) detachment or breakdown of aggregates, (ii) transport of detached particles and other light fractions, and (iii) deposition of the material whenever the velocity of wind or runoff slows sufficiently. Breakdown of aggregates exposes the C, hitherto encapsulated within the aggregate and protected even from microbial breakdown, to microbial activity and facilitates mineralization. Soil erosion is a selective process and involves preferential removal of the finer components comprising humus and clay fractions. In general, the ratio of C content of water- and wind-borne sediments to that of the contributing soil (the C enrichment ratio) is greater than one. Lal (1976) observed in Nigeria that eroded sediments contained 3–5 times more C than the original soil. In southwest Niger, Sterk *et al.* (1996) reported that the wind-blown material trapped at 2 m contained 32 times more C than the topsoil. The C content in a dust sample was 5.4% compared with 0.15% in the topsoil. High enrichment ratio of wind-blown sediments is also reported for soils in Texas by Zobeck and Fryrear (1986).

Thus, the detachment of aggregates and redistribution of C-rich sediments over the landscape may accentuate loss of C from soil to the atmosphere. Conversely, sedimentation and downslope deposition may lead to deep burial of C or its translocation into lakes, reservoirs, and other aquatic ecosystems where it may be sequestered over geologic time (Stallard, 1998).

3. *Leaching:* The soluble fraction of the SOC pool, called dissolved organic carbon (DOC), can be leached out of the soil profile with seepage water (Moore, 1998). While a fraction of the DOC transported into the ground water may be precipitated and sequestered, a large portion may be mineralized and released into the atmosphere as CO_2.

2. SIC Dynamics

Accelerated soil erosion can also lead to exposure of the subsoil rich in calciferous materials. Exposed calciferous horizons are subject to human perturbations and climatic factors (e.g., plowing, application of acidifying fertilizers, root exudates, acid rains). These factors may lead to dissolution of CO_3^{-2} and emission of CO_2. In contrast, formation of secondary carbonates may also lead to sequestration of C in the form of SIC (Nordt *et al.,* 1999). In systems of partial or complete soil leaching, the major mechanism of SIC sequestration is via movement of HCO_3^- into ground water or closed systems that have limited exchange with ambient environments (Wilding, 1999).

3. Magnitude of C Loss through Soil Erosion

Because of the complexity of the factors involved and scarcity of quantitative information on all processes concerned, it is difficult to precisely assess the loss of soil C due to historic erosion. The rate of global C loss is estimated at 1.14 Pg C year^{-1} from soil erosion by water (Lal, 1995), of which 0.23–0.29 Pg C year^{-1} occurs by erosion in drylands (Lal *et al.,* 1999a). The data in Table II show that the magnitude of SOC lost by water erosion from the world's soils may be 21.2 Pg from 1094 million hectares (Mha) of affected land area. In comparison, the magnitude of C loss by wind erosion may be 3.7 Pg from 549 Mha of affected land area (Table II). Thus, the total loss of C from eroded soils of the world may be 24.9 Pg, most of which can be returned to the soil by actions that foster sequestration.

B. Restoring Carbon: Results of Agronomic Experimentation

Systematic documentation of the impact of nutrient manipulation on crop yields and soil properties began in 1843 in Rothamsted, England, with the fertility studies of J. B. Lawes and J. H. Gilbert (Jenkinson, 1991). The scientific discovery of the effects of farmyard manure on soil fertility and the influence of management on soil organic matter dynamics are two examples that demonstrate the importance of such long-term studies as those at Rothamsted. Knowledge gained in experimentation at Rothamsted was later incorporated in mathematical models of soil organic matter dynamics (Jenkinson and Rayner, 1977). Research was later initiated in America, Europe, and Oceania with the objectives of finding suitable cropping systems and studying the influence of nutrient additions on soil fertility. Some of the earliest agricultural experiments were made on the Morrow Plots in Illinois [1876] (Odell, *et al.,* 1984), the Sanborn Field in Missouri [1888] (Buyanovsky *et al.,* 1996), the Askow experiments in Denmark [1894] (Chris-

Table II

Soil C Depletion Due to Historic Soil Erosion by Water and Wind

Erosion severity	Land Area[a] (Mha)	Rate of SOC depletion (Mg ha^{-1})	Total C loss (Pg)
		Water Erosion	
Light	343	5.0	1.7
Moderate	527	20.0	10.5
Strong and Extreme	224	40.0	9.0
Total	1094		21.2
		Wind Erosion	
Light	269	2.5	0.7
Moderate	254	10.0	2.5
Strong and Extreme	26	20.0	0.5
Total	549		3.7

[a]From Oldeman, L. R. (1994).
Adapted from Lal, R. (1999).

tensen, 1996a), the ABC Rotation in Lethbridge [1911] (Campbell *et al.*, 1990), the Waite Permanent Rotation Trial in Australia [1925] (Grace, 1996), the Breton Plots in Alberta [1930] (Izaurralde *et al.*, 1996), and the Residue Management experiment in Oregon [1931] (Rasmussen *et al.*, 1996).

From these and other experiments we have learned that the level of organic matter in soil is a function of C additions and nutrient balance. This is illustrated in Fig. 2 with wheat yield results from rotation ABC at Lethbridge on an Haploboroll (Janzen, 1995) and the Breton Classical Plots (Izaurralde *et al.*, 1996) on a Cryoboralf. Janzen (1995) attributed the upward yield trend of unfertilized wheat to improved cultural methods and genotypes. Wheat after fallow yielded consistently more than continuous wheat because of the water stored during the fallow period. Yields of unfertilized wheat in the wheat fallow system at Breton were low but did not decline over time (Izaurralde *et al.*, 1996). The addition of nutrients via commercial fertilizers or manure increased yields of wheat grown after fallow or forages. Addition of C to soil via residues, roots, and manure had a strong influence on SOC dynamics (Fig. 3). Soil organic C in rotation ABC decreased initially but appears now to have reached equilibrium concentrations. Organic C concentration in the soil at Breton—with low initial SOC content and in a cold subhumid environment—decreased, remained stable, or increased in response to crop productivity and nutrient additions. Results presented at recent workshops and symposia on long-term experiments and simulation studies have added important information regarding the influence of tillage and cropping intensity (fallow frequency) on SOM dynamics (Powlson *et al.*, 1996; Lal *et al.*, 1998b, 1998c;

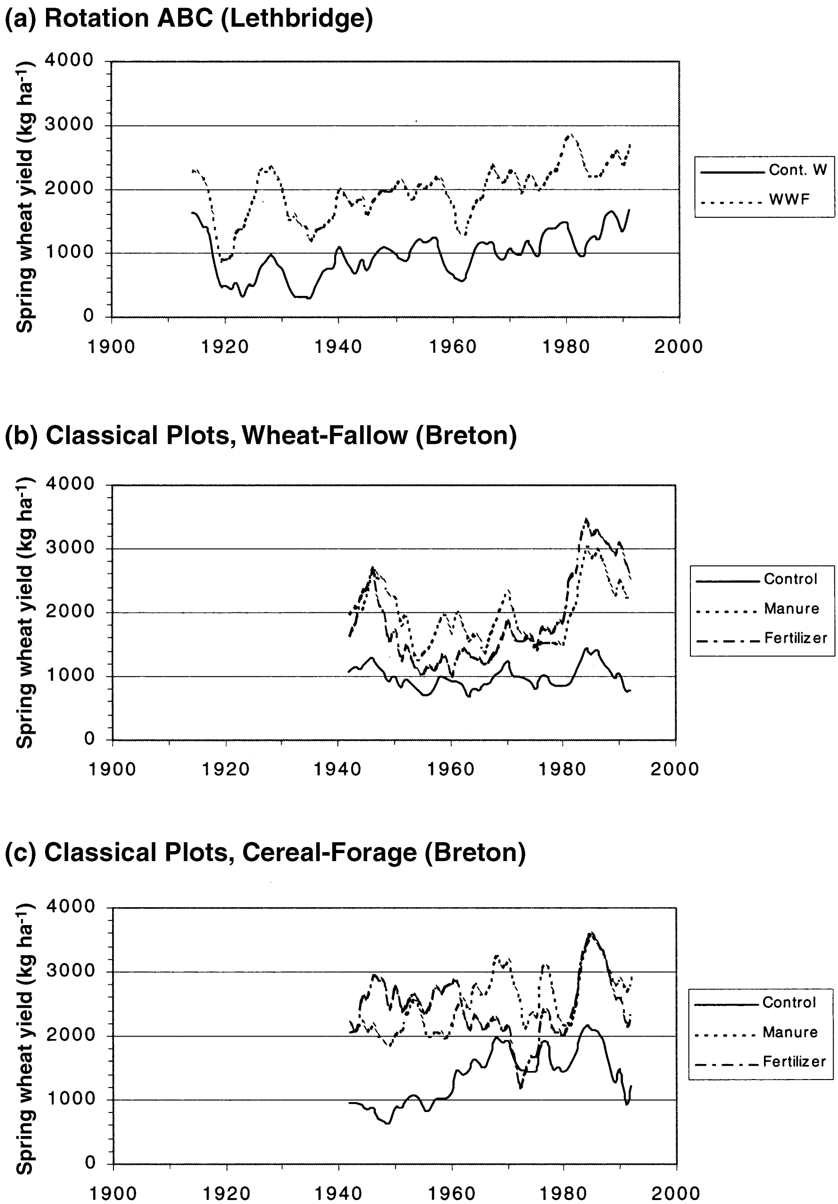

Figure 2 Yields of spring wheat in two long-term crop rotations in Alberta, Canada: (a) Rotation ABC established in 1910 at Lethbridge on a Mollisol (drawn with data from Janzen, 1995); (b and c) Classical Plots established in 1930 at Breton on an Alfisol. (drawn with data from Izaurralde *et al.*, 1996). Abbreviations: continuous wheat (Cont. W), wheat–fallow (WF), wheat–wheat–fallow (WWF), wheat–oat–barley–hay–hay (WOBHH).

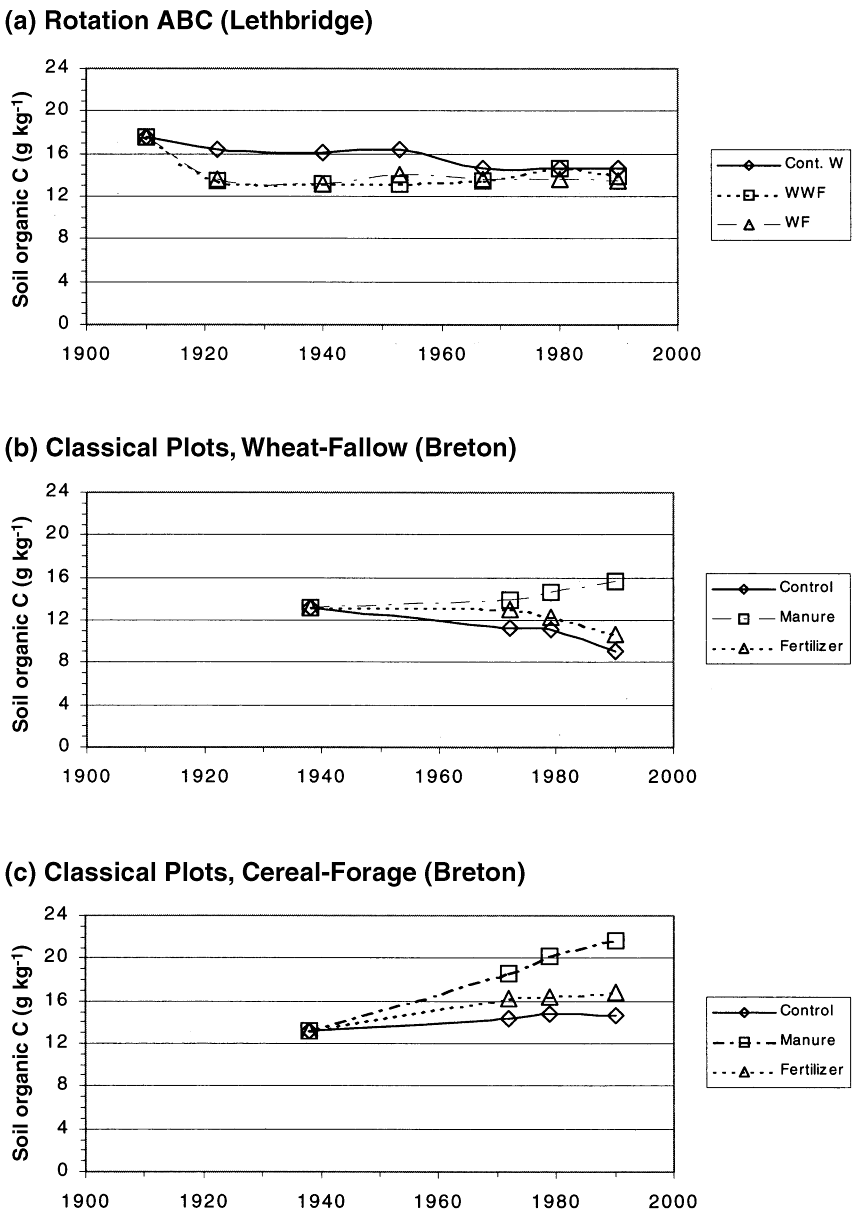

Figure 3 Soil organic C in two long term crop rotations in Alberta, Canada: (a) Rotation ABC established in 1910 at Lethbridge on a Molisol (drawn with data from Monreal and Janzen, 1993), (b) and (c) Classical Plots established in 1930 at Breton on an Alfisol. (Drawn with data from Izaurralde *et al.*, 1996). Abbreviations: Continuous wheat (Cont. W), Wheat—Fallow (WF), Wheat—Wheat—Fallow (WWF), Wheat—Oat- Barley—Hay—Hay (WOBHH).

Paul *et al.*, 1997). Most of these experiments have demonstrated that management for soil carbon sequestration is consistent with the principles of sustainable agriculture (e.g., reduced tillage, erosion control, diverse cropping systems, improved soil fertility).

C. Current and Future Production Trends

1. Producers and Production Technologies

How will agricultural technologies evolve during the next century and what are the implications for soil carbon sequestration? As a wise man once said . . . “It is difficult to forecast, especially the future.” But in the case of 21st century agriculture, at least four technological developments or trends that have already begun and are likely to continue and intensify will impact on soil C sequestration: agricultural biotechnology, conservation tillage, organic farming, and precision farming.

1. *Genetically modified crops:* Rapid adoption of genetically modified (GM) crops attests to the attraction of one of the most innovative—yet highly controversial—technologies introduced to agriculture during the 1990s. While farmers in the Americas (Canada, the U.S., Brazil, and Argentina) have been quick to integrate GM crops with herbicide (e.g., cotton [*Gossypium hirsutum* L.] and maize [*Zea mays* L.] to glyphosate) and pest resistance (e.g., maize to the European corn borer) into their production practices, there is stiff opposition to this technology in Europe and Japan (Lehrman, 1999). European and Japanese consumers appear fearful of potential negative impacts of GM crops on the health of humans and ecosystems. A recent report (Losey *et al.*, 1999) indicates that the pesticide effects of pollen from maize plants incorporating the Bt gene injure the Monarch butterfly. This result can be an early warning that the full consequences of GM crops are not yet understood.

2. *Conservation tillage:* Conservation tillage (CT) is another crop production technology that has come of age during the 1990s. Defined broadly, a field is said to be under CT when, after planting, the residue cover is greater than 30%. CT includes such practices as no-till, ridge-till, and mulch-till. Initially developed with the primary objective of erosion control, CT has evolved over the last 30 years into holistic agronomic systems with many advantages over conventional tillage: (i) economic benefits due to labor and fuel reductions; (ii) better soil quality as a result of increases in SOM with its associated improvements in soil tilth and water-holding capacity; and (iii) benefits to the ecosystem such as improved water and air quality and wildlife habitat. Annual surveys by USDA personnel revealed significant advances in the adoption of CT technologies during the early 1990s ($\sim$21% in 1990; 35% in 1994), but progress has stalled in recent years (CTIC,

1998). It appears from 1998 data that yearly increases of 3.3% in adoption of CT technologies will be required for the U.S. to have 50% of its farmed land under CT by 2002. The CTIC report indicates that producers in Canada, Brazil, and Argentina are adopting CT, particularly no-till, more rapidly than producers in the U.S. CT as a system of choice for achieving production goals and environmental sustainability is already widespread, with success stories from around the world (Carter, 1993a). Its potential as a means of helping to mitigate global-warming provides a new and perhaps key incentive for further increases in the adoption of CT technology. However, the desired effect of enhanced retention of atmospheric C in SOM through the application of CT practices will only be realized through a concerted effort of global dimensions (Cole *et al.,* 1996; Rosenberg *et al.,* 1999).

3. *Organic agriculture:* In response to consumer demand, the quantities of food and fiber produced by organic farming methods continue to grow worldwide. According to the U.S. Organic Trade Association, the organic industry in the U.S. has been growing at annual rates of 20–24%, with yearly sales now of more than $4.2 billion. Although the business volume for organic products is still small compared to that for products produced by conventional methods, it is not difficult to anticipate that organic production technology will continue to evolve and gain more supporters. Organic farming methods rely less on external inputs (e.g., fertilizers and herbicides) than do more conventional production methods. This tighter internal control of nutrient cycling in organic farming has also been shown to affect SOC dynamics. Drinkwater *et al.* (1998) conducted a long-term comparison among three maize-soybean (*Glycine max* (L.) Merr.) rotations: one with chemical weed control and synthetic N additions and two organic methods, one of which included applications of manure. While maize yields in the three systems were remarkably close, marked differences in soil properties developed among the treatments. After 10 years, SOC and total N had increased in the organic treatment receiving manure and declined in the conventional treatment.

4. *Precision farming:* Interpretation of geo-referenced agronomic data collected with satellites and ground equipment allows farmers to practice what is now known as "precision farming." Producers using this technology seek to optimize economic return and minimize adverse environmental impacts by gaining a more precise knowledge of the spatial variability of their soils. This "precision" knowledge is gained with the use of global positioning systems (GPS) and electronics mounted on field equipment (e.g., yield monitors, variable rate fertilizer applicators). There remain, however, many challenges in terms of data collection and interpretation. For example, Hergert (1998) anticipates that wide adoption of precision farming in the next 5–10 years will require significant changes in the methods of soil testing and plant analysis. According to this view, site-specific nutrient recommendations will increasingly rely on information obtained by remote sensors with support from "reference" soil and plant analysis. Tilman (1999) argues that progress toward "greening of the green revolution" can be made not only with find-

ings such as those of Drinkwater *et al.* (1998) but also with those evolving from precision agriculture.

These are but a few examples of current, evolving production technologies with a potential or proven role in increasing soil carbon sequestration. Each emphasizes something special (i.e., molecular biology and genetics, conservation, ecology, and high technology). These technologies are not necessarily competitive with one another but rather, can be synergistic. Their ultimate impact on carbon sequestration will depend not only on the economic benefit realized by individual producers but also on whether society recognizes the value of soil carbon storage to mitigate global warming.

III. NEW SCIENCE APPROACHES

A. Definition of Questions

Many decades of research have been dedicated to discovering the complex and dynamic nature of SOM and to establishing principles for its management. On the basis of this solid footing, soil C sequestration has become an emerging field of research aimed at designing and applying the best management solutions to mitigate greenhouse-gas-induced global warming. Cole *et al.* (1996) identified the opportunity for global C sequestration in agricultural soils. Edmonds *et al.* (1999) evaluated its economic dimensions in the context of a technology-based strategy to stabilize atmospheric CO_2 concentration at a nonthreatening level (Fig. 1). Based on an expert evaluation of the global potential of C sequestration in terrestrial ecosystems (DOE, 1999), Metting *et al.* (1999) have suggested achievable sequestration rates in agriculture to range within 0.85–0.95 Pg C $year^{-1}$. These approximations are based on best estimates of SOC respired by soil and emitted to the atmosphere as CO_2 or transported offsite by erosion as a result of agricultural activities and land use changes. Cole *et al.* (1996) applied point estimates of C losses, summarized previously by Davidson and Ackerman (1993) from a series of 18 field experiments mostly from North America, to project global C losses from soils grouped according to the FAO classification. Unfortunately, actual data on these losses worldwide are scarce, making global estimates subject to error. As discussed in more detail in the next section, information on the history of land use can also help recreate past changes in SOC. Houghton *et al.* (1999) provide an example for the U.S. using census data to estimate a net release of about 27 Pg C between 1700 and 1945. These estimates of past C losses and potential C gains, together with experimental evidence and modeling projections, strongly convey the need for developing a C sequestration science with global applications.

What, then, are the fundamental research questions to be answered in order to advance the science of soil C sequestration? Metting *et al.* (1999) identified three major questions: (1) How do biogeochemical processes control C sequestration in soil, especially under changing climate and land use conditions? (2) Where and for how long can soils act as C sinks? (3) Which are the most effective technologies for realizing or yet enlarging the soil's capacity to sequester C? A corollary derived from these emphasizes the need to evaluate the effectiveness of C-sequestration technologies not only in terms of sustainable-land management principles but also with regard to the net flux of other greenhouse gases (e.g., N_2O). Here we discuss these questions further, emphasizing a model representation of C processes in soil, the role of soil structure in SOC stabilization, the role of roots in SOC dynamics, and the environmental impacts of soil C sequestration.

B. Model Representation of C Processes In Soil

The large number of SOM models currently available exhibit a rich diversity of approaches to representing the cycles of C and other elements in soil. McGill (1996), Parton *et al.* (1995), and Molina and Smith (1998) have conducted comparative analyses of their structure and function. Customarily, C flow in soil is described mathematically by dividing SOC content into compartments or "pools" and treating its dynamic as a first-order process with addition,

$$dC/dt = -kC + A, \tag{1}$$

where dC/dt is the rate of change of C concentration (or mass), k is the decomposition rate constant, C is the concentration (or mass) present at time t, and A is the rate of C addition (e.g., litter, crop residues). The division of SOC content into compartments is required to account for differences in SOM structure and rates of decomposition. The number of compartments used in these models has varied from two (e.g., RothC by Coleman and Jenkinson, 1996) up to eight (e.g., *ecosys* by Grant *et al.*, 1993) according to the complexity needed to represent C flows in soil. Likewise, there is great variation found in ways to describe the addition of organic materials to soil (Molina and Smith, 1998). CENTURY (Parton *et al.*, 1988), a widely used and tested model, divides SOM into four compartments, two of which are reserved for microbial populations with fast turnover rates ($k \sim 6.5$ year^{-1}), a slow SOM compartment ($k = 0.2$ year^{-1}), and a passive SOM compartment ($k = 0.0045$ year^{-1}). CENTURY also divides organic materials (litter) added to soil into metabolic or structural groups according to the speed at which the litter decomposes.

Estimates of C sequestration potential have been based on a fixed proportion of the C lost from agricultural soils (Cole *et al.*, 1996). There is a need to discover when, where, and under what management lie the best opportunities for seques-

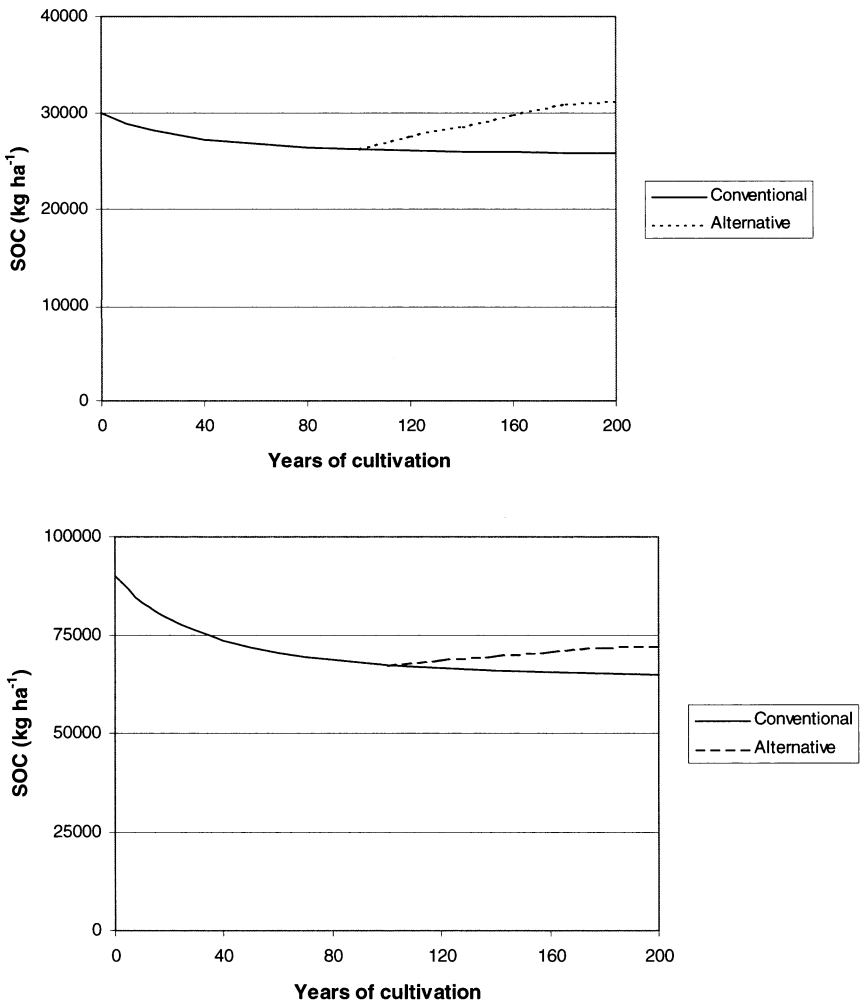

Figure 4 Simulated C dynamics in two soils with different initial contents of organic C under conventional and alternative (C sequestering) practices.

tering atmospheric CO_2. Is it possible to exceed the C level achieved under natural conditions? If so, where do the opportunities exist? We illustrate these questions with results from a simple analytical model of SOM based on first-order kinetics (Fig. 4). The SOM model is composed of three C compartments (microbial biomass (b), active (a) and passive (p)) with decomposition rate constants of k_b = 0.8000 year^{-1}, k_a = 0.0300 year^{-1}, and k_p = 0.0007 year^{-1}. Under conventional practice, soil A receives C inputs of 800 kg ha^{-1} while for soil B the rate is 1600 kg ha^{-1}. Under the alternative practice, the rate of C additions to both soils is assumed to increase by 20%. The C fractions flowing through the microbial biomass, active, and passive compartments are 0.60, 0.39, and 0.01, respectively. The fractional distribution of C compartments is $C_b = 0.0268$, $C_a = 0.4637$, and $C_p = 0.5096$. Another assumption besides the increase in C productivity (i.e., yield) is that it would be possible to increase the C allocation to the passive pool by 1%.

The two soils differing in C content and productivity have been under cultivation for about 100 years. During this period, the soil with high SOC content loses more C (~25%) than the soil with low SOC content (~13%). Alternative practices are then implemented to allow for an increase in C productivity of 20% in both cases. In addition, it is assumed possible to increase the C allocation into the passive compartment by 1%. In this case, the soil with low SOC content not only recovers but also even exceeds its original level before reaching steady-state conditions. In contrast, the soil with high SOC is able to recover only part of the C lost during the first century of use. This type of analysis is especially needed to evaluate the impacts of soil warming on the value of rate constants. Increasing the decomposition rate constants by 4% for the example discussed above lowers the amounts of SOC recoverable with the C-sequestering practice (Fig. 5).

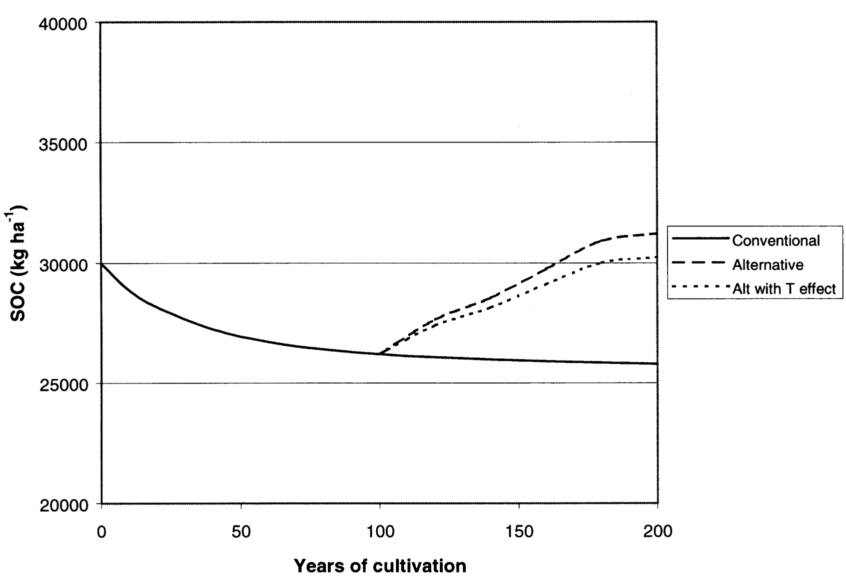

Figure 5. Simulated C dynamics in a soil with low initial content of SOC with C sequestering practice applied after 100 years of cultivation, with and without the influence of temperature on the decomposition rate constant of SOM.

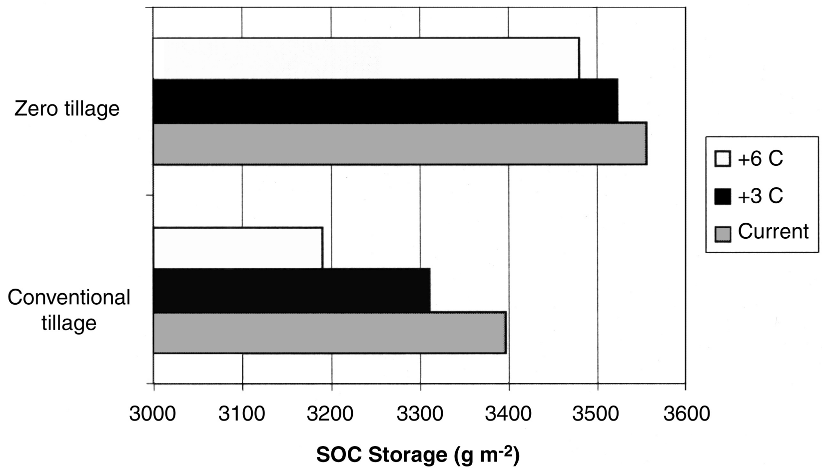

Figure 6 Simulated changes in organic C in conventionally and zero tilled soils under current climate and projections of +3 and +6°C with double CO_2 concentration. (Drawn with data from Grant *et al.*, 1998.)

Using the RothC model, Jenkinson *et al.* (1991) simulated a reduction in global SOC stocks of about 60 Pg by the year 2050 with temperature increasing by 0.03°C per annum. This positive feedback was calculated by holding net primary productivity (NPP) unchanged. Schimel *et al.* (1990) used climate change projections from the Goddard Institute for Space Studies general circulation model as input to the CENTURY model to simulate changes in SOM for the U.S. Great Plains. In spite of simulated increases in NPP, CENTURY predicted a decrease in C storage as a result of decreases in net ecosystem productivity (NEP). Grant *et al.* (1998) used *ecosys*, a high-temporal-resolution model with detailed treatment of transformations and transfers of water, heat, carbon, and nitrogen, to simulate tillage and surface residue effects on soil C storage under ambient and elevated CO_2. The model successfully reproduced the SOC changes observed during an 11-year period under conventional tillage, but underestimated those under zero tillage. Increases in air temperature and [CO_2] led to increases in C fixation by plants and improved water relations. Barley (*Hordeum vulgare* L.) growing under zero tillage allocated more C to roots than to shoots. Under the hypothesized global warming, more C would be lost from soils under conventional than under zero tillage (Fig. 6).

Kirschbaum (1995) used results from laboratory experiments to examine the influence of temperature on the decomposition rate constant of SOM. Interpretation of the data indicated that the decomposition rate constant increases with temperature with a Q_{10} at 0°C $\cong$ 8. The temperature sensitivity of organic matter decomposition decreased with increasing temperature (Q_{10} = 4.5 at 10°C and Q_{10} = 2.5 at 20°C). Kirschbaum concluded that SOM losses could be greater in colder regions than in more temperate or tropical zones. Boone *et al.* (1998) produced evidence from a mixed temperate forest experiment suggesting that root respiration might exhibit even greater temperature sensitivity ($Q_{10} = 4.6$) than bulk soil ($Q_{10} = 3.5$).

Clearly, an increased understanding of the dynamic impact of rising temperatures on the net balance between NPP and organic matter decomposition is essential to advance the field of soil C sequestration.

C. Soil Structure and Soil Organic Matter Stabilization

Soil organic matter has a primary role in the creation and maintenance of soil structure. The connection between SOM and soil structure has been known to scientists and farmers for a long time but only recently has knowledge of this interaction become crucial for the understanding of C accumulation and stabilization in soil. Recent reviews of the subject include those by Christensen (1992, 1996b), Kooistra and van Noordwijk (1996), Tisdall (1996), and Gregorich and Janzen (1996). The 1996 reviews were prepared as part of a special edition of *Advances in Soil Science* edited by Carter and Stewart (1996).

Separating soil aggregates according to size by wet sieving or with ultrasound has proven useful for studying their nature, organization, and function (Tisdall, 1996). Soil aggregates can be broadly classified into *microaggregates* and *macroaggregates* using the mesh diameter of 250 μm as the boundary. The setting of this boundary, however, is far form being capricious. The strong bonds connecting the elemental blocks of microaggregates confer resistance against the disruptive energy contained in water (e.g., rain) or produced by ultrasound. Macroaggregates, which are made by the grouping of microaggregates, may break with ease when submerged in water or shaken. Connecting bonds are created by various sorts of organic materials (e.g., microbial and plant debris, roots, and polysaccharides), organic polymers, polyvalent cations, and electrostatic forces. The latter three prevail as bonding agents in smaller aggregate fractions (<0.2 μm) while debris from microbes and plants, roots, and polysaccharides control the formation and hierarchy of larger aggregates (Tisdall and Oades, 1982). This type of hierarchical organization of soil structure has been observed in Mollisols and Alfisols, where organic materials control aggregate stability, but not in Oxisols, where Fe and Al oxides act as stabilizing agents (Oades and Waters, 1991).

The life cycle of aggregates is controlled by dynamic and complex interactions between organic materials and mineral components. Soil management exerts a strong influence on the formation and persistence of aggregates, especially macroaggregates (Tisdall, 1996; Angers *et al.* 1993; Perfect *et al.,* 1990). Soil organic matter stabilization may arise as a result of chemical recalcitrance (incorporation of aliphatic or aromatic molecules into polydisperse, polyaromatic structures), physicochemical stabilization (sorption of organic substrates onto clay minerals, sesquioxides), stabilized organic constituents, or physical protection (substrates "hide" from decomposers behind physical barriers such as minute pores) (Christensen, 1996b). Generally, the C concentration and C/N ratio decrease as soil aggregates become smaller but the amount of C held in them increases with decreasing aggregate size (Baldock *et al.,* 1992; Cambardella and Elliott, 1993).

The turnover of SOM is determined by biological, chemical, and physical interactions and regulated by environmental factors. Plant residues (e.g., litter, products of root decomposition) represent the main source of C required to maintain or increase SOM. This organic material in transition to becoming stable organic matter is a sensitive and early indicator of SOM changes (Gregorich and Janzen, 1996). This young, low-density organic matter appears in soil relatively unbound to other constituents so it can be extracted directly by density fractionation or sieving (Gregorich and Ellert, 1993). Because of its sensitivity to management, the light fraction organic matter (LFOM) has been used to characterize the early stages of SOC transformations arising from changes in management (Dalal and Mayer, 1987; Bremer *et al.,* 1995; Nyborg *et al.,* 1998; Solberg *et al.,* 1998).

Studies on physical fractionation of SOM have brought a new dimension to the study of SOM turnover. Physical separates are concrete, reproducible, and even "observable" entities. Thus, the dynamics of C and N within these fractions can be studied approximating at least the physical location where these transformations take place. To meet their potential, however, measures of physical entities must be accompanied by measures of their dynamics, and not just quantities at a few points in time. Only if discrete turnover rates can be found associated with physical entities will they be viable substitutes for the current kinetic compartments of simulation models.

Considerable progress has been made by incorporating isotopic methods to study C turnover in soil physical fractions. Gregorich *et al.* (1995) used ^{13}C natural abundance data to calculate a rapid turnover of LFOM in eastern Canadian soils that had been under corn cultivation for 25 years. Jans-Hammermeister *et al.* (1998) used ^{14}C labeling to study the influence of pulsed (e.g., organic amendments) versus daily (e.g., root exudates) additions of C on microbial growth and aggregate formation. The proportion of pulsed ^{14}C recovered in macroaggregates was greater than that in microaggregates. Instead, most of the ^{14}C daily additions were found associated with microaggregates <53 μm.

Although there is an obvious connection between the physical separates and model compartments, current models make little, if any, use of these concepts. Cambardella and Elliott (1993), for example, suggested similarities between particulate organic matter separations with the "slow" compartment in CENTURY. The initialization of SOM compartments then becomes a critical task to ensure the accurate reproduction of SOM trends. Should one model the measurable, or measure the modelable? (Elliott *et al.*, 1996). This question in itself defines a new paradigm in SOM modeling and research. There are already conceptual models of this new approach to SOM modeling (Christensen, 1996b). Future progress in this direction will contribute immensely to the understanding of carbon cycling in soil in the context of production, climatic change, and environmental quality.

D. The Role of Roots In C Sequestration

Agricultural soils have the potential to become a major battleground in the fight against global warming. Understanding below-ground processes is essential to relating them to atmospheric and surface processes. The picture could not be complete without considering the role of roots in C cycling and stabilization. Roots are hard to measure, but progress has been made toward understanding their contribution to SOC content and sensitivity to management practices. Balesdent and Balabane (1996) in France used natural $\delta\ {}^{13}C$ techniques to calculate the contribution of maize roots to SOC storage. To their surprise, they found that roots had incorporated 57 g C m^{-2} $year^{-1}$, a rate 58% greater than incorporation by leaves and stalks together. Xu and Juma (1993) used ${}^{14}C$ pulse labeling to establish possible differences in root growth characteristics and C stabilization between two barley cultivars. Although Samson, a six-row semi-dwarf feed cultivar, produced fewer roots, it led to a greater C stabilization in soil than did Abee, a two-row medium-height feed cultivar (Xu and Juma, 1992). Swinnen *et al.* (1995) also used ${}^{14}C$ pulse labeling to study rhizodeposition of winter wheat and spring barley grown with conventional and integrated farming methods. Use of this technique allowed the authors to discern that management did not affect the growth of wheat roots but did influence that of barley. Contrary to their hypothesis, conventional farming led to greater root growth, root respiration, and rhizodepositional fluxes than those measured under integrated farming. Total rhizodeposition accounted for 7–15% of net assimilation (450–990 kg C ha^{-1} $year^{-1}$), which represented twice as much as the mass of roots left at harvest. Further research in this area is needed to improve our understanding of the role that roots play at the local and global scales (Jackson *et al.,* 1997).

E. Environmental Impacts of Soil C Sequestration

Carbon sequestration occurs when a given practice leads to a positive balance in the flow of C to soil. Inputs such as fertilizers, pesticides, and irrigation needed to grow crops and deposit C in soils have what Schlesinger (1999) calls “hidden costs” in that CO_2 is released in their manufacture or delivery. He asserts that, with the exception of conservation tillage, a full accounting of C emissions associated with sequestration practices would lead to a much less optimistic view of the net C savings than is conveyed, for example, in the foregoing pages. Another potential offset of C sequestration benefits might occur on irrigated lands because of CO_2 released from waters rich in Ca. According to Schlesinger, the application of manure may result in little or no net C sequestration because of the C costs of its production.

These arguments appear not to have considered that fertilization, irrigation, and manuring are employed to increase crop, pasture, or wood production and that either optimization or adoption of these "inputs" or practices might be the only requirement to encourage soil C sequestration. Data from agronomic studies clearly suggest that optimization of nutrient additions alone (e.g., Solberg *et al.*, 1998) or in combination with reduced soil disturbance (e.g., Nyborg *et al.*, 1995; Halvorson *et al.*, 1999) can lead to significant increases in soil C storage. There is little dispute, however, that the real climate-change avoidance benefits of soil C sequestration can be known only with a full accounting of the C costs associated with practices that are believed to further that end.

The environmental cost of C sequestration—its impact on N_2O emissions—needs to be considered in such accounting. There are reports that growing-season N_2O emissions increase under zero tillage (Aulak *et al.* 1984; Linn and Doran, 1984), while other studies contradict such findings. For example, N_2O fluxes from soils receiving fertilizer, manure, and legume-fixed N under conventional and zero tillage in spring under conventional tillage equaled or exceeded those measured under zero tillage (Lemke *et al.*, 1999).

There is a need for comprehensive analyses of experimental and modeling results that bear on this question. A long-term view is required since the real costs (including C costs) of practices aimed at increasing soil C will depend on local conditions of climate, soil type, crop rotation, tillage and irrigation system, water quality, and, certainly, economics.

F. Summary

There has been significant progress toward understanding which biological, physical, and chemical phenomena control C sequestration in soil. Future progress will depend on rigorous mechanistic understanding of the processes operating and regulating C flows and stabilization in soil. To summarize this section, the diagram in Fig. 7 attempts to describe the kind of convergence needed between our understanding of kinetic (i.e., knowledge of turnover times) components and discrete (i.e., measurable) entities. The vertical arrow indicates the degree of stability of turnover times of the components or entities described. The kinetic components are represented toward the left side of the diagram by microbial biomass, slow

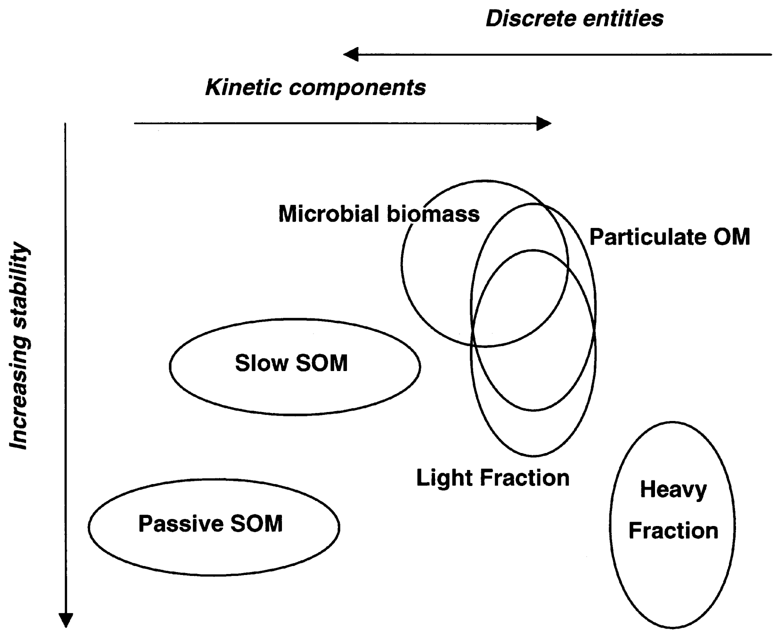

Figure 7 Conceptual representation of the relationships between kinetic components and measurable fractions of SOM and of how they might converge.

SOM, and passive SOM—three SOM "pools" commonly used in the current generation of SOM models. Microbial biomass, particulate organic matter, light fraction, and heavy fraction are viewed as discrete entities because they can be measured reproducibly. Microbial biomass meets the requirement of being both a discrete entity and a measurable kinetic characterization. Drawn to the left of microbial biomass are conceptual components that are well described kinetically but not identified as discrete entities. The opposite situation is represented to the right of microbial biomass. The orientation of the main axis of the ellipses used to represent components and entities suggests the direction in which less is known about the process. The vertical overlap among the discrete groups suggests that, although they are reasonably discrete, they remain kinetically different. The horizontal axis of the slow and passive components suggests that they are characterized by turnover rates, but lack any discrete definition. The challenge is to either obtain discrete entities with well-characterized turnover rates or treat kinetics as a continuum. Only if discrete turnover rates can be found associated with physical entities will they be viable substitutes for the current kinetic compartments of simulation models.

IV. MONITORING AND VERIFICATION

Earth scientists have been measuring soil organic matter for many years to discover its structure and dynamics and to learn how to relate these to ecosystem stability and function. Most farmers and agronomists know that SOM holds the key to soil fertility and is, probably, the single property that best equates with soil quality (Arshad and Coen, 1992; Doran *et al.*, 1994; Acton *et al.*, 1995; Romig *et al.*, 1995; Warkentin, 1995). Soil organic matter content is customarily expressed on a concentration basis (g kg^{-1}), but calculation of the actual quantity of organic matter in the soil requires knowledge of soil bulk density. Bulk density itself is a property influenced by soil organic matter content (Adams, 1973), as well as by tillage and management practices.

Soils may provide a long-term global repository for atmospheric CO_2. Soil carbon-sequestration programs, however, will require that the "sale" and "purchase" of a ton of carbon be based on a system that can monitor and verify that the commodity traded—whether locally, nationally, or internationally—has actually been delivered (i.e., sequestered). Through work in controlled experiments, soil surveys, and monitoring programs, soil scientists have gained considerable experience in measuring quantities of carbon in the soil. However, the methods they have used will have to be applied to the thousands or hundreds of thousands of fields that must eventually be monitored. Accurate but practical methods will be needed, such as those methods being developed under the Canadian Prairie Soil Carbon Sequestration Project, an industry–farmer–government partnership (Izaurralde *et al.*, 1998; A. Donnelly, personal communication[3]). Progress toward this objective was also made at a workshop on methods of assessing soil C pools held in Columbus, Ohio, in November 1998 (Lal *et al.*, 2000).

At the St. Michaels, Maryland, workshop of December 1998, Post *et al.* (1999) presented a comprehensive review of direct and indirect methods for the measurement of carbon sequestration in soils. Drawing on this presentation, we describe current methods for measuring and estimating SOC changes. We also analyze possible sources of error that could be associated with monitoring activities. We classify the various monitoring methods into the categories of *measurements* (direct and indirect) and *estimates*. We recognize, however, that any plan to monitor and verify SCS will necessarily involve both measurements and estimates. Finally, we identify needed research and database development to improve field sampling, laboratory measurement, simulation modeling, and remote sensing activities for monitoring changes in soil.

[3]A. Donnelly, President, Greenhouse Emissions Consortium Management (GEMCo), 1998.

A. Direct Methods

1. Field and Laboratory Measurements

Direct methods for detecting management-induced changes of SOC in time and space are based on field sampling and laboratory determinations. The concentration of organic C (g kg^{-1}) in soil can be determined with relative accuracy by using dry-combustion methods on dry, sieved, and homogeneous samples (Carter, 1993b; Klute, 1986). Carbonates contained in soils can be eliminated from the sample by acid treatment prior to the SOC determination. Conversion of SOC concentrations (g kg^{-1}) to mass per unit volume (kg m^{-3}) or area (kg m^{-2}) requires that soil bulk density (Mg m^{-3}) also be measured. In Alberta, as in many other regions, cultivation over the past 100 years has often reduced both concentration and mass of SOC. In a study designed to assess the status of SOM in Alberta, field sampling on 72 farms across major soil zones showed that all cultivated horizons (Ap) had a lower concentration and mass of organic C than did virgin horizons (Ah) (McGill *et al.,* 1988). Losses of C from A horizons were greater, on average when expressed in concentration terms (48%) than when calculated on a mass basis (24%).

As we know, management practices influence both SOC concentration and soil bulk density. Thus, it is essential that temporal changes in SOC be determined with samples containing the same soil mass. The importance of this matter is illustrated by the work of Ellert and Bettany (1995). They recalculated soil C mass from data in 15 published experiments by adjusting the depth of each layer to contain equal soil mass (equivalent-soil mass method). Use of the traditional method—calculation of soil mass without adjustment for variable depth—revealed that, in 12 of 15 experiments, cultivated soils had lost C. However, the results of the equivalent-soil mass method suggested that C had actually increased in two of the experiments. Agreement between methods was close in only one of the other experiments. In experiments where C loss was observed, the equivalent-soil mass method showed the average loss to be 9% greater with respect to the control treatment than did the traditional method. In six no-tillage experiments where SOC calculated by the traditional method appeared to have increased, the the equivalent-soil mass method indicated increases to be smaller. The Ellert–Bettany analysis demonstrates that the equivalent-mass method should be used when comparisons of volumetric mass of SOC are to be made in soils with changing bulk densities—that is, essentially, in all soils.

The successful application of carbon-sequestering practices (e.g., no-till) to soils previously under long-term cultivation can induce significant changes in both the flux and the storage of carbon. Determination of management-induced short-term changes in soil organic C is challenging because these changes are small (e.g., 1–4 Mg C ha^{-1}) in comparison to the large existing soil C stocks (e.g., 20–80 Mg C ha^{-1}). Detection of treatment-induced changes in C storage requires that measurements made a few years after adoption of the new management be compared with the same measurements on some sort of control. But how does one define an appropriate spatial–temporal "control" or "baseline." Is it the condition of the field at initiation of the new carbon-sequestering practice? Is it an adjacent field under conventional practice at the end of the period of measurement? These two types of control would likely yield different answers depending on whether the soil is under steady-state conditions. If the rate of change in carbon fluxes (C additions—C losses) is zero—i.e., steady state—then both types of control will yield the same answer. There is long-term experimental evidence that the steady state may prevail in many cultivated soils of the North American Great Plains (Janzen *et al.*, 1998). However, if the soil in question had been gaining C under conventional management practice, then the measured difference in C content over an identified time period would overestimate C gains arising from the new practice. Alternatively, if the soil had been losing C at initiation of the new practice, the change in SOC over time would not detect an "avoided loss" of SOC (McGill *et al.*, 1996). Use of a control for side-by-side comparison of conventional and improved management would, at the end of the measuring period, allow for estimation not only of the C gains due to sequestration but also of the avoided loss. The actual gains and avoided losses are important from the point of view of C emissions to the atmosphere offset by sequestration (McGill *et al.*, 1996). Use of both types of control described above is the best way to account for non-steady-state conditions in defining the reality of changes in soil C.

The C transported through erosion is a necessary component of annual soil C budgets that must also be estimated. But soil transport and deposition by wind and water add complexity to efforts to monitor changes in SOC storage. Erosional processes (e.g., detachment, transport, and deposition) can be either measured or estimated in research environments (Lal, 1994). However, because erosion measurements are necessarily long term, complex and costly, they cannot be used practically for large-scale monitoring of soil C sequestration programs. Models such as RUSLE (revised universal soil loss equation; Renard *et al.*, 1991), EPIC (erosion productivity impact calculator; Williams, 1995), WEPP (water erosion prediction project; Laflen *et al.*, 1991), and WEPS (wind erosion prediction system) may provide a practical alternative for estimating erosion effects. Campbell *et al.* (1999a), for example, used EPIC to estimate loss of soil and organic C by wind and water erosion in a 30-year crop rotation experiment in southwestern Saskatchewan. Their estimates were then used to correct changes in SOC projected by the SOM model CENTURY (Parton *et al.* 1988) and by a simplified equation of Voroney *et al.* (1989).

The isotope ^{137}Cs has also been used to estimate total erosion or soil transport over long periods (de Jong *et al.*, 1983). It is strongly adsorbed to clay and organic particles and is, therefore, essentially nonexchangeable. Detected differences in ^{137}Cs activity between native and cultivated sites are attributed to soil erosion (water, wind, tillage) and deposition processes. Using ^{137}Cs techniques in combination with the RUSLE, Busaca *et al.* (1993) were able to estimate soil transport and deposition in Palouse landscapes. Soil samples were taken to depths of 0.4 and 0.6 m from a 46-ha cultivated watershed with its outlet blocked to avoid loss of soil mass using a grid pattern intensive enough to allow the application of geostatistical techniques. For the 26-year period between the ^{137}Cs fallout in 1963 and the sampling in 1989, soil erosion averaged -12 Mg ha^{-1} year^{-1} while soil deposition averaged 19 Mg ha^{-1} year^{-1}. Soil erosion estimated with ^{137}Cs was only 61% of that estimated with RUSLE (-31 Mg ha^{-1} year^{-1}). Busaca *et al.* (1993) placed more confidence in the ^{137}Cs measurements since these were closer than the RUSLE to the limited number of actual erosion measurements available. Cao *et al.* (1993) also used a grid-sampling approach to study ^{137}Cs activity in soil samples of Ap horizons taken from two benchmark sites in Quebec. Using a baseline value of 3100 Bq m^{-2} obtained from a site under native forest, they calculated erosion rates as great as 96 Mg ha^{-1} year^{-1} and deposition rates as great as 109 Mg ha^{-1}year^{-1} within 4.8-ha cultivated fields. Because the benchmark sites shared textural and topographic features, the authors concluded that the distinct management applied to the fields (tillage and silage corn production in one; hay and small grain production in the other) had had a strong influence on erosion rates. It is questionable, however, whether this isotopic method can be used to detect significant differences in SOC over short periods of time (e.g., 5–10 years). Likely, erosion estimates made with simulation models, coupled with information on C content in sediments and sediment enrichment ratios, will remain the most practical way of estimating changes in SOC due to erosion and deposition processes.

Spatial variability is another factor that may affect our ability to detect temporal changes in SOC. The purpose of sampling is to obtain information about a complex population so that meaningful statements can be made with a given degree of confidence (Burrough, 1991). The distribution of soil attributes in a landscape usually lacks homogeneity and, therefore, exhibits spatial dependence. A soil-sampling plan should be designed to incorporate knowledge of this spatial dependence, which can be obtained through field surveys, aerial photo interpretation, and geostatistical techniques. Mahinakbarzadeh *et al.* (1991) investigated one- and two-dimensional spatial variability of SOC distribution in a central Massachusetts soil. They took soil samples at 15-m intervals along a 1200-m transect laid out across several map units and created a grid pattern of 8 rows by 40 columns in a cultivated plot of land in the study area. Their analyses, using both classical statistics and geostatistics, revealed that for short and long transects—within and between map units—the use of classical statistical methods was adequate for estimating SOC variability. Conversely, for locations including several soil map units, the presence of strong autocorrelation trends in SOC distribution indicated geostatistics as the best method for evaluating the spatial dependence of SOC. Knowledge of spatial patterns of soil attributes could lead to sampling designs by landscape position (e.g., knolls, back slopes, foot slopes). Location of pilot areas within landscape units could help reduce sampling errors associated with spatial variability. Ellert *et al.* (1999) devised a detailed plot layout design to reduce errors induced by spatial variability that involves returning to the same location time after time.

Laboratory processing of soil samples and analytical methods deserve attention since they can strongly influence results. Coarse roots are removed in standard soil sample preparation by passing the sample through a 2-mm sieve (USDA-NRCS, 1996), but material from fine roots and dead root fragments is often retained in the sieved material. Root treatment during sample processing may influence treatment comparisons. This is demonstrated by Richter *et al.* (1990) with results from a 7-year experiment in Michigan documenting tillage impacts on soil C storage. Roots were separated from soil materials retained on sieves of 6, 2, and 0.5 mm. Carbon storage to a depth of 0.15 m was reduced by 24% by tillage. Most of this reduction was associated with root biomass. In the tilled soil, the mass of C contained in the root material between 2 and 6 mm in size represented about 10% of soil C. Differences in surface-layer root biomass induced by tillage reduced below-ground C (root + soil) in tilled soils from that in untilled soils by 6.8 Mg ha^{-1} (Fig. 8).

Considerable progress has been made in standardizing and documenting labo-

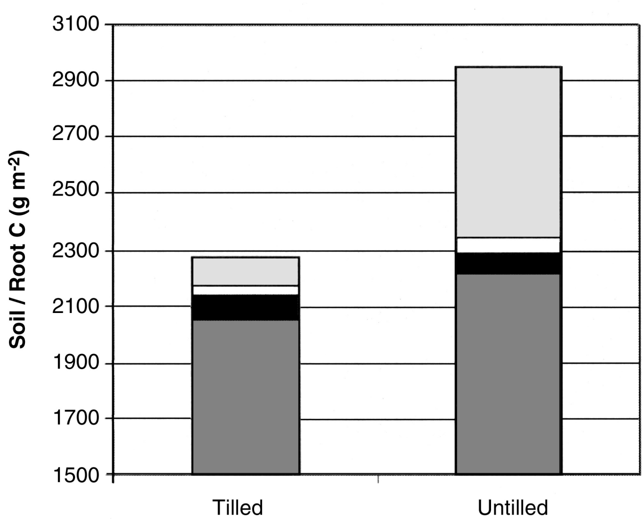

Figure 8 Organic carbon distribution in roots and soil in tilled and untilled (grassland) plots in summer 1986. (Adapted from Richter *et al.,* 1990.)

ratory methods to estimate soil content of inorganic C, organic C, and organic C fractions (i.e., light and heavy fractions) (Carter, 1993b; Culley, 1993; Klute, 1986; Page *et al.,* 1982; Magdoff *et al.,* 1996; USDA-NRCS, 1996). Information on light-fraction C has proven useful as an early indicator of bulk soil C changes otherwise undetectable over short periods (e.g., Janzen *et al.*, 1992; Solberg *et al.*, 1998). Incubation is another commonly used technique to estimate labile pools or light-fraction C. Early indication of changes in SOC induced by management practices can also be obtained by incubating fresh soil samples and measuring CO_2 evolution (Elliott *et al.,* 1994). This may also be accomplished with chloroform fumigation–incubation (Parkinson and Paul, 1982) or chloroform fumigation–extraction (Vance *et al.,* 1987) to determine microbial biomass. There is consensus among scientists that these methods will be essential components of any scheme to monitor C sequestration in operational projects. However, widespread use of these methods appears unlikely since they are extremely laborious and time consuming. Another technique, ^{13}C NMR (nuclear magnetic resonance) spectroscopy, has promise, but is not extensively used in soil science. This technique could contribute not only to elucidation of the complex nature of SOC but also to its quantification (Kinchesh *et al.*, 1995).

2. Indirect Measurements of Ecosystem C by Eddy Covariance

In principle, the changes in C stocks observed in soil and vegetation can also be deduced indirectly from measurements of the CO_2 exchange between the atmosphere and the land surfaces. The net CO_2 exchange due to photosynthetic and respiration processes can be calculated from the covariances between vertical wind (eddies) and CO_2 concentration (Businger, 1986; Baldocchi *et al.*, 1988; Verma, 1990). Until recently, available technology allowed for integration of these fluxes only over short periods of time. However, improvements in software and hardware—including reliable sonic anemometers and rapid response gas analyzers—now allow for the integration of CO_2 fluxes over entire growing seasons.

A substantial number of eddy flux measurements are being made as part of an increasingly larger global network of flux-measurement sites such as AMERIFLUX in North America, EUROFLUX in Europe, and OZFLUX in Australia. These measurements are complemented by a comprehensive collection of plant and soil observations that enrich interpretation of the eddy-covariance data. Salient observations include above- and below-ground plant biomass, leaf area index, litter, soil temperature, soil water content, soil bulk density, soil CO_2 flux, soil organic C, total soil N, and N mineralization. Thus, the indirect measurements of net ecosystem exchange of CO_2 by eddy covariance can lead to an improved understanding of the C cycle at seasonal and annual scales, especially when these complement direct SOC measurements. Combining eddy-covariance flux measurements with terrestrial biosphere models (e.g., Sellers *et al.*, 1996) could provide an alternative method for identifying components of net ecosystem exchange associated with SOC changes.

3. Estimating SOC Stocks and Changes at Site, Regional, National, and Global Levels with Databases, Remote Sensing, and Simulation Models

Soil Databases and Landscape Information

Despite the methodological difficulties discussed above, direct measurement of SOC remains the most concrete way of documenting SOC changes induced by management or land-use practice. Direct measurements of SOC are essential to any protocol for monitoring and verifying C sequestration. Extrapolation of C changes recorded at the field scale to the regional scale, however, requires that the protocols make use of geographic information systems (GIS) and associated databases, computer models, and remote sensing (Kern and Johnson, 1993; Donigian *et al.*, 1994; Izaurralde *et al.,* 2000).

Digital soil databases are essential building blocks for the development of sound C monitoring and verification protocols. In the U.S., for example, such databases have evolved from soil mapping programs conducted by the National Cooperative Soil Survey. The landscape and soil databases that these programs have generated are essentially static, i.e., for the duration of a mapping cycle (approximately 30 years). One example is the State Soil Geographic (STATSGO) national database (USDA-NRCS, 1994). STATSGO maps are made by generalizing detailed soil survey data and are represented at a 1:250,000 scale, except for Alaska where a scale of 1:1,000,000 is used. The minimum size of a map unit is about 625 ha. STATSGO maps are linked to attribute databases with information on the proportions and properties of the soils contained in each map unit. The attribute database contains information on soil properties (e.g., water-holding capacity, soil reaction) and soil use interpretations (e.g., flooding). The new Soil Survey Geographic (SSURGO) database of USDA-NRCS (1995) duplicates the original soil survey maps with mapping scales ranging from 1:12,000 to 1:63,360. This detailed level of mapping is intended for landowners and planners skilled in the use and interpretation of soils data. SSURGO data are available for selected counties in the U.S. As with STATSGO, SSURGO maps are linked to attribute databases containing information on soil properties.

Soil Landscapes of Canada (SLC) is another digital soil database developed by recompiling existing soil survey maps at a scale of 1:1,000,000 (Shields *et al.* 1991). The map units (or polygons) are described in terms of soil and landscape attributes (e.g., surface form, slope, and water table depth). SLC polygons usually contain one or more distinct soil landscape components including small but contrasting inclusions, but the exact locations of these components within the polygon are not defined. Originally, the development of the SLC database was seen as a way to standardize information on plant growth, land management, and soil degradation. These data have become the basis of the hierarchical approach to ecological land classification in Canada (Ecological Stratification Working Group, 1995).

These databases have been used to estimate SOC stocks by means of methods such as those described by Bliss *et al.* (1995). For example, Lacelle *et al.* (1999) have constructed a soil C map of North America using information from Canadian, U.S., and Mexican databases. Sombroek *et al.* (1993) and Eswaran *et al.* (1995) have derived global SOC maps from information contained in the FAO Soil Map of the World.

These static SOC maps constitute the basic layer of information needed to begin analysis of SOC dynamics at site and regional levels. The inclusion of additional layers derived from digital elevation models (DEM), land cover and land use (LCLU) data, census information, and remote sensing can improve our understanding of landscape C dynamics. Information on annual statistics from crop reporting districts can contribute to estimating the amounts of plant C returned to soil. Izaurralde *et al.* (2000) used crop yield data aggregated at the ecodistrict level as input to SOCRATES, a simple SOM model by Grace and Ladd (1995), to describe the cumulative impact of agricultural management on C sequestration. Campbell *et al.* (1999b) have used an alternative approach to models to obtain first approximations of SOC changes by calculating the fraction of residue C converted into soil C using data from long-term Canadian experiments.

Landforms exert major control on soil development by creating unique microclimates above the soil and by regulating the lateral flow of mass, water, and nutrients to and through it. It is now possible to construct highly detailed DEM with remotely sensed data and global positioning systems. These DEMs should be particularly useful in improving our understanding of the spatial distribution of soil properties (e.g., SOC). MacMillan *et al.* (2000) used DEMs in combination with heuristic rules and fuzzy logic to develop automatic procedures for segmenting landforms into landform elements. This high level of detail need not necessarily translate into more complex information. Although the procedure allowed for the identification of up to 15 morphologically defined landform facets, the authors concluded that consolidation of these elements to three or four units was adequate to obtain relevant landform separations at the farm field scale.

Descriptions of Land Cover and Land Use

Availability of land cover and land use information extracted by remote sensing for use in environmental research and modeling is increasing. Recently, Loveland *et al.* (1999) released a 1-km resolution global land cover database prepared with 1992–1993 imagery from the advanced very high resolution radiometer (AVHRR). Detailed land use information is required in certain applications to describe or monitor human activities. A land cover map of the U.S. with a 30-m spatial resolution should become available during 1999 (Vogelmann *et al.*, 1998).

Information on land use history is also required for reconstructing SOC changes. Houghton *et al.* (1999) using historical data for the period 1700–1990 calculated a C budget for the U.S. on when and where land has been cleared for agriculture, abandoned, harvested for wood, or burnt. These data, in conjunction with a terrestrial C model, allowed them to identify periods of net release and net uptake of C and to estimate magnitude of the fluxes. An estimated 27 Pg of C had been released prior to 1945. Since then, a significant net C uptake has been occurring because of fire suppression, forest growth, and abandonment of agricultural land. From a different perspective, Viglizzo *et al.* (2000) used agricultural census data from 1888 to the present to reconstruct land use history of the Argentine Pampas and to analyze the energy balance of farming systems that prevailed there at various times. Sustained efforts toward reconstructing the location and timing of changes in land use–land cover will lead to a better understanding of the processes currently affecting soil C and how these may evolve in the future.

Another rich source of data in the U.S. is the National Resource Inventory (NRI). These data have been collected at 5-year intervals since 1982 to document trends in resource conditions. This comprehensive dataset, gathered at more than 800,000 sample sites, is considered statistically reliable for national, regional, state, and substate analysis. The NRI data collected include information on farmsteads, urban, and built-up areas; windbreaks; streams and water bodies; land ownership; soil information (e.g., soil classification, soil properties, and soil interpretations); land cover and land use (e.g., cropland, pasture land, rangeland, forest land); cropping history; irrigation; erosion; wetlands; and conservation practices.

Future Remote Sensing Capabilities and Geographic Data

Remote sensing products soon to become available could prove very helpful in evaluating soil organic carbon, especially in regions that lack detailed geographical information. These products include high-resolution DEM and land cover characterizations. A new global elevation dataset will be developed from the Jet Propulsion Laboratory's Shuttle Radar Topographic Mission, launched into space in February 2000. Within 2 years of the shuttle flight the mission is expected to create 30- and 100-m DEMs of the entire globe between 60° N and 60° S latitudes (http://www.jpl.nasa.gov/srtm/). The MODerate-resolution Imaging Spectroradiometer (MODIS) sensor, to be deployed on the Terra platform of NASA's Earth Science Enterprise, was launched in late 1999. This sensor may provide additional capabilities for spatial and spectral resolution that are not possible with AVHRR, the source of current global land cover datasets. The MODIS sensor should enable interpretations of the remotely sensed images in terms related to biophysical processes of C accumulation on the landscape [e.g., leaf area index (LAI), the fraction of photosynthetic active radiation (fPAR), and net primary productivity (NPP)].

Process Models

An understanding of SOC dynamics is based largely on long-term field observations in managed and natural ecosystems. Since the pioneer SOM model developed at Rothamsted by Jenkinson and Rayner (1977), a good number of models have been developed that take advantage of the knowledge that emerges from these long-term experiments. McGill (1996) and Molina and Smith (1998) have described the features of many of these models and analyzed the fundamental principles that govern their functioning. There are numerous examples in the literature describing tests of SOM models against short- or long-term datasets at the plot or field scale (e.g., Paustian *et al.*, 1992; Parton and Rasmussen, 1994; Grant *et al.;* 1993). When the details of management are known, the models generally do well in reproducing the trends in SOC changes (Smith *et al.*, 1997). Use of these models has gradually expanded from research mode, site-specific conditions to more generalized, regional scale applications. There are, however, few examples of work comparing SOM models and applying them at regional scales.

Donigian *et al.* (1995) used the CENTURY model (Parton *et al.*, 1988) to calculate area-based soil C changes induced by agricultural management practices in the central U.S. with production systems data and policy alternatives taken from the resource adjustment modeling systems (RAMS) (Bouzaher *et al.*, 1992). RAMS is a linear programming model of agricultural production based on climatic divisions that generates production patterns across spatial units (e.g., crop mixes, tillage, production inputs). The simulation results led the authors to conclude that up to 1 Pg C could be sequestered by 2030 across the study area which represented about 65% of total U.S. cropland.

In Canada, Izaurralde *et al.* (2000) compared six simulation models in order to select one that best fit data from long-term experiments and to scale up C storage simulations to the regional level. Models specifically designed to simulate soil organic matter (RothC, CENTURY, DNDC, and SOCRATES) or that included soil organic matter dynamics as part of the soil-plant-atmosphere continuum (*ecosys*, EPIC) were included in the comparison. Characteristics of four of these models are presented in Table III.

The Canadian study selected the model SOCRATES for the aggregation experiments because it reproduced soil organic carbon dynamics best, as determined in a series of long-term experiments in the Prairie Provinces, and met both statistical (e.g., correlation, chi-square, lack of fit) and practical criteria (ease of use, simplicity, usefulness of input and output data). Three aggregation procedures were used to estimate carbon storage in the dominant soils of two ecodistricts having many soil-climate management combinations. The first estimate (E1) was derived from simulating C storage in all of the soils within an ecodistrict (14 for one and 7 for the second) under the dominant management regimes including dairy, cattle, pigs, wheat, oilseeds, and grain. The second estimate (E2) simplified the model-

Table III

Characteristics of Four Process-based Models that Simulate Soil Organic Carbon Dynamics

Model	Century	DNDC	EPIC 5125	RothC
Spatial scale	Plot to field	Plot	Plot to catchment	Plot to field
Time-step	Week	Hour and day	Day	Month
Time period for soil organic matter simulation	Months to centuries	Days to centuries	Years to centuries	Years to centuries
Soil profile	Uniform—20 cm depth	Surface layers—50 cm depth	Soil layers—up to 10	Uniform—23 cm
Soil organic carbon pools	Three	Three	Two	Three
Texture effect	Yes	Yes	No	Yes
Weather data	Precipitation and temperature	Precipitation and temperature	Radiation, precipitation, temperature	Precipitation, temperature, open pan evaporation
Plant growth simulation	Cereals, grasses, trees	Cereals, grasses	Cereals, grasses, trees	No
Rotations	Multiple crops per year, unlimited rotations	Multiple crops per year; rotations limited to 20 year	Multiple crops per year; unlimited rotations	None
Nutrients and fertilizers	N, P, S	N, P	N, P	None
Tillage	Many options	Many options	Many options	None
Erosion	No	No	Wind and water	No
CO_2 effect	Yes	Yes	Yes	No

Adapted from Izaurralde *et al.* (2000).

ing by hypothesizing that it is possible to simulate C storage using the dominant soil of each ecodistrict and by applying all the management systems characteristic of that ecodistrict. The third approach (E3) simplified the estimation of storage even further by assuming a dominant soil in each ecodistrict and one management that represented the major crops grown within the district. In one ecodistrict with an area of 650,000 ha, estimates of annual soil organic carbon storage from the three aggregation methods were similar (35 Gg C year^{-1} for E1, 31 Gg C year^{-1} for E2, and 29 Gg C year^{-1} for E3). However, discrepancies among the estimates for the second ecodistrict of 750,000 ha were large (1 Gg C year^{-1} for E1, -4 Gg C year^{-1} for E2, and -12 Gg C year^{-1} for E3). Each aggregation method was evaluated for accuracy, sensitivity to nonnormal distributions of soil-climate management combinations, and effort involved.

Models of SOC describe processes of organic matter transformation, protection, and mineralization with different levels of detail and with different dependence on environmental conditions. Consequently, each model has strengths and weaknesses and these should be weighed when selecting a model to meet a specific objective. These circumstances vary, not only with physical and biological conditions in the region under study but also with the amount of experimental experience and richness of climate, land use, and geographical information available for the analysis. There is opportunity to scale regional models by extracting spatially explicit information on LCLU change from high-resolution remote sensing data and using it to model SOC dynamics within a GIS environment. Finally, regional dynamics of primary production can be extracted from remote sensing data not only to extend site and landscape level model outputs but also to serve as an independent source of information for model validation.

B. Summary and Research Needs

Scientifically sound and accurate methods are available for monitoring and verifying changes in soil C. International collaborations should lead to codifying a set of methods into protocols that can be used to account for carbon sequestration chartered in public agreements or private trading contracts. The level of precision will vary with the purposes for which the measurements are applied. These purposes may include (a) determining compliance with local, regional, and national laws or treaties regulating CO_2 emissions; (b) joint implementation projects; and (c) carbon credits and offsets. Current methods are effective for evaluating changes in soil organic carbon at relatively low precision (20–50% error) and at widely spaced time intervals (minimum 3–5 years) with levels of effort that are reasonably affordable. Since relatively small amounts of C sequestered in soils could significantly reduce the rate of increase of atmospheric CO_2 and effectively buy time until low-cost methods are available to reduce CO_2 emissions from power-generating plants, industry, and transportation, there will be considerable pressure to increase the reliability and precision of monitoring soil organic carbon changes on even shorter time scales.

Two basic approaches for monitoring and verifying C changes in soil have been noted in this paper. The first is based on indirect estimation of soil C stock changes over a specified period. The second is based on direct measurements of soil C pools and or fluxes of CO_2 between the atmosphere and the land surface. Although both approaches are built on solid scientific principles, there is an urgent need to improve their operational accuracy and to make them cost effective enough to encourage their widespread use. Verification methods will have to include, at least temporarily, estimates based on incomplete knowledge of some fluxes and/or some temporal interpolation of anticipated long-term trends derived from experiments that may not directly apply to the situation at hand. Clearly, monitoring and

Table IV

Current and Future Technologies for Monitoring Soil C

Technology C measurements	Current (1999–2001)	Mid term (2002–2007)	Long term (2008–2020)
Soil	Reduce sampling errors, improve root estimates	Non-destructive field measurement (exp.)	Non-destructive field measurement (routine, low cost)
Eddy flux	60 sites world wide	Expand to characterize significant landcover types	Routine, part of automated stations (when weather stations satisfy "upwind fetch" requirements; S. Verma, personal communication) (low cost)
Remote sensing	Low resolution LULC, absorbed PAR, hyperspectral (exp.), SAR (exp.)	High resolution, satellite-based hyper spectral, SAR, models (exp.)	High resolution, hyper spectral, SAR, models (routine)
C modeling	Models linked to databases; model intercomparisons	Models driven by RS input (exp.)	Real time simulation of land processes driven by RS
C accounting	Databases, maps, census, models (exp.)	Databases, maps, census, models, new sensors (refinement)	Databases, maps, census, models, new sensors (operational)

Note: RS = Remote Sensing, LULC = Land Use and Land Cover, SAR = Synthetic Aperature Radar.

From Post, W. M. *et al.* (1999). With permission.

verification will have to rely somewhat on methods that fill in information in time and space that cannot be readily observed. Table IV summarizes our understanding and vision with respect to current and future technologies for measuring soil C at the field level and making spatial and temporal projections of soil C changes.

V. A SPECIAL ROLE FOR DESERTIFIED LANDS

A. Extent and Rate of Desertification

It is difficult to obtain exact estimates of the land area affected by desertification, because of the lack of quantitative criteria for defining it and of suitable methods for assessing it. There are discrepancies in various estimates that have been made because of different criteria used. According to Oldeman and Van Lynden (1998) the land areas subject to soil degradation in arid regions range from 1016–1137 Mha, with about 139 Mha subject to severe and extreme degradation. UNEP (1991) distinguished between land degradation and vegetation degradation. Vegetation in rangeland can undergo degradation with or without soil degradation. The data in Table V show that the area affected by land degradation is 1016 Mha, with an additional area of vegetation degradation of 2576 Mha.

Estimates of current rates of degradation also vary widely. UNEP estimates range from 21 M ha year^{1} (UNEP, 1984b) to 27 M ha year^{-1} (UNEP, 1987). Mainguet (1991) estimated the annual rate of desertification at about 5.8 Mha, with 55% occurring on rangelands and 43% on rainfed cropland (Table VI). The annual rate of degradation is greatest on rainfed cropland (~0.44%), followed by irrigated land (~0.10%), and rangeland (~0.09%).

Table V

GLASOD[a] Estimates of Desertification (e.g., Land Degradation in Dry Areas Excluding Hyper-Arid Areas)

UNEP (1991)		Oldeman and Van Lynden (1998)[b]	
Land type	Area (Mha)	Type of soil degradation	Area (Mha)
Degraded irrigated lands	43	Water erosion	478
Degraded rainfed cropland	216	Wind erosion	513
Degraded rangelands (soil and vegetation)	757	Chemical degradation	111
Sub-total	1016	Physical degradation	35
Degraded rangeland (vegetation alone)	2576	Total	1137
Total	3592	Light	489
Total land area (excluding hyper-arid regions)	5172	Moderate	509
% degraded	69.5%	Severe and extreme	139
		Total	1137

[a]GLASOD, Global Assessment of Human and Induced Sol Degradation, UNEP, 1991.

[b]The estimate by Oldeman and Van Lynden does not include the vegetation degradation on rangeland.

Table VI

Estimate of Annual Rate of Land Degradation

Land use	Total land area (Mha)	Rate of land degradation (Mha year^{-1})	% of total
Irrigated land	131	0.125	0.095
Rangeland	3700	3.200	0.086
Rainfed cropland	570	2.500	0.439
Total	4401	5.825	0.132

Note: Calculated from Mainguet, 1991; UNEP, 1991.

B. Strategies For Desertification Control

Productivity of drylands with traditional low-input management systems is extremely low. Consequently, some important strategies for improving productivity are: (1) growing appropriate plant species, (2) enhancing water-use efficiency, (3) controlling erosion and restoring degraded soil, (4) managing and enhancing soil fertility, and (5) adopting improved cropping systems.

1. Appropriate Plant Species

The adaptation of plants to drylands requires attention to many adaptive mechanisms. Consideration must be given to the nature of the root system, growing season duration, canopy structure, and leaf and stomatal characteristics. Most dryland plants close their stomata during daytime to reduce transpiration and conserve water. Such plants have developed special photosynthetic mechanisms more efficient than the normal Calvin cycle or the C_3 mechanism. These mechanisms include the C_4 photosynthetic C fixation (C_4) and the crassulacean acid metabolism (CAM) pathways (Salisbury and Ross, 1992).

Plants with C_4 metabolism grow in hot environments and their leaf anatomy is different from that of C_3 plants. The C_4 plants have a distinct advantage in water and nutrient use efficiencies and tolerate high temperatures (Brown, 1994). C_4 grasses can produce new leaves at lower levels of plant available N than can C_3 grasses (Wilson and Brown, 1983). Some C_4 and CAM plants are listed in Table VII.

The CAM mechanism, an adaptation to desert environments, exists in several families, including cacti, euphorbias, pineapple, and agave. CAM plants have a high water-use efficiency (WUE) and lose 50–100 g of water for every g of CO_2 assimilated compared to 250–300 g for C_4 plants and 400–500 for C_3 plants. The CAM mechanism is similar to that of the C_4 cycle, except that the stomata are open at night and closed during the day.

Table VII

Plant Species with C_4 and CAM Photosynthetic Metabolism

C_4	CAM
Monocots	*Agave Family*
Blue grama (*Bouteloua gracilis*)	American agave (*Agave americana*); century plant (*Agave parryi*)
Mexican teosinte (*Euchlaena mexicana*)	Lechuguilla (*Agave lecheguilla*)
Sorghum (*Sorghum bicolor*)	
Sudan grass (*Sorghum sudanese*)	
Corn (*Zea mays*)	
Sugarcane (*Saccharum officinarum*)	
Dicots	*Succulent plants*
Amaranth (*Amaranthus* spp.)	Saguaro cactus (*Cornegiea gigantea*)
Australian salt brush (A*triplex semibaccata*)	Barrel cactus (*Ferocactus acanthodes*)
	Desert prickly pear cactus (*Opuntia phaeacantha*)
	Beavertail cactus (*Opuntia basilaris*)
	Hedgehog cactus (*Echinocereus triglochidiatus)*
	Yucca plants
	Soaptree yucca (*Yucca elata*)
	Mohave yucca (*Yucca schidigera*)

The choice of appropriate plant species is critical to enhancing WUE. Growing common cereals (wheat, barley, sorghum) and legumes (chickpea, clovers) is less effective than making use of the wide range of grain crops and legumes adapted to drylands. Some promising ones are grain amaranth, quinoa, and triticale among cereals and bambara groundnut, marama bean, tepary bean, and narrow leaf lupin among legumes. In addition to food crops, there are several promising plants that, in addition to being efficient water users, can improve farm income because of the industrially useful materials they produce (e.g., resins, rubber, oil, and fodder) (Table VIII). Many xerophytic plants can be grown for production of fruits, fodder, feedstock, and fuel wood (Table IX).

2. Enhancing Water Use Efficiency

Frequent drought stress is the principal constraint to plant growth in dry areas because precipitation is limited and erratic and a considerable proportion of it is lost to runoff and evaporation. Thus, saving water by reducing runoff and evaporation is critical to enhancing biomass productivity. Experiments conducted in the west African Sahel (Table X), in northern Africa (Table XI and XII), and elsewhere have demonstrated that adoption of improved technologies (e.g., stone bunds, microcatchment, appropriate tillage methods, improved crop rotations, adoption of new species, applications of organic amendments, and judicious use of fertilizers

Table VIII

Perennials and Low-Water-Use Plants of Industrial Importance

Name	Botanical name	Attributes	Usefulness
Grindelia (gum weed)	*Grindelia camporum*	Adaptable to stressful environment	Aromatic resins for use in inks, adhesives, varnishes
Rubber rabbitbrush (gum weed)	*Chrysothamnus* sp. *C. nauseosus* *C. albicaulis* *C. hololeucus* *C. consimilis*	Adaptable to drought and aridity, cold winter and hot summers, nutrient poor soils (C-3 plant). Native to western U.S.	Natural rubber, resins and oils useful for natural pesticides
Guayule	*Parthenium argentatum*	Semi-desert perennial shrub, not very cold and salt-tolerant, need irrigation	Rubber of high quality
Milk weeds	*Asclepias erosa* *A. speciosa*	Herbaceous perennial	Latex and rubber chemical feed stock and fuel, fiber
Neem	*Azadirachta indica*	Drought tolerant, tolerant to saline/sodic soils, fast growing	Biocidal properties, oil of industrial use
Jojoba	*Simmondsia chinensis*	Drought and heat tolerant (up to 50°C), not tolerant to frost	Source of oil and waxes
Vernonia sp.	*Vernonia anthelmintica* *V. galamensis*	Short rainy season of 3–4 months	Epoxy fatty acids, oil, (Vernolie acid)
Bladderpod	*Lesquerella fendleri*	Adapted to dry, carbonate-rich soils; tolerant to heat/drought and cold	Hydroxy fatty acids and oils
Mesquite	*Prosopis juliflora* *P. chilensis* *P. africana* *P. tamarugo* *P. cineraria*	Leguminous tree with deep root system, anti-erosion and dune stabilization and BNF capacity, drought/heat tolerant. Tolerant to saline soils and water (6000 ppm)	Firewood, fodder, mulch materiaL
Gum trees	*Eucalyptus* spp.	Drought tolerant, fast growing	Timber, fire wood, industrial oil, pharmaceuticals
Prickly pears	*Opuntia* spp.	CAM metabolism, high WVE	Specialty crop fruit, fodder, and forage

Adapted from Hinman, C. W., and Hinman, J. W. (1992); H. N. Le Houérou (1975), D. Stiles (1988); and S. G. Patil *et al.* (1996).

Table IX

Some Xerophytic Plants of Economic Importance

Name	Species	Important characteristics	Usefulness
Prickly pears	*Opuntia ficus-indica* *O. streptacantha* *O. fusicaulis* *O. robusta* *O. lindheimeri* *O. inermis* *O. phaeacantha* *Mammillaria microcarpa* *Nopalea cochenillifera*	Special photosynthetic pathway with high CO_2-fixation efficiency (CAM),[a] high WUE for arid regions	Speciality crop fruits (cacti) vegetable crop, and as fodder and forage
Buffalo gourd	*Cucrbita foetidissima*	Perennial plant adapted to wastelands with little rainfall; native of western North America	Seeds rich in oil and protein, and large roots rich in edible starch; fruits can be harvested mechanically
Ye-eb	*Cordeeauxia edulis*	Perennial shrub adapted to hot and dry poor soils	Fruits of high quality
Acacia	*Acacia albida* *A. amara* *A. aneura* *A. ligulata* *A. cyanophylla* *A. salicina* *A. victoriae* *A. nilotica* *A. millifera* *A. auriculiformis* *A. ferugenia*	Drought tolerant, soil fertility improvement soil salinity control	Fodder shrubs fuel wood, gums, and pharmaceuticals .
Henna	*Lowsonia spp.*	Drought tolerant	Shampoos, soaps, cosmetics
Myrrh	*Commiphora spp.*	Drought tolerant	Incense, pharmaceuticals, perfumes and flavoring, paper.
Vetiver	*Vetiveria spp.*	Widely adapted	Aromatic oil, erosion control, ground cover.

[a]CAM = Crassulacean acid metabolism.
Adapted from Stiles, 1988; Hinman and Hinman, 1992; Le Houerou, 1975; Patil *et al.,* 1996.

(especially for phosphorus) can greatly improve WUE, biomass production, and income. Water harvesting techniques offer another opportunity to increase available water for plant growth and improve the edaphic environment. Niemeijer (1998) reported that in eastern Sudan the indigenous *teras* water harvesting system offers greater production security and minimizes risks of crop failure.

Supplementary irrigation is an important means of enhancing biomass produc-

Table X

Effects of Water Harvesting by Stone Bunds on Biomass and Grain Yield of Pearl Millet in Northern Burkina Faso in 1986

Treatment	Biomass yield[a] (Mg ha^{-1})	Grain yield[a] (Mg ha^{-1})
Traditional	2.52	0.233
Traditional + stone bunds	3.01	0.406
Traditional + stone bunds + tillage	4.64	0.837

[a]Both biomass and grain yields could have been greatly increased had the use of water harvesting techniques been coupled with judicious use of fertilizers, especially of P.

Modified from Lamachère, J. M., and Serpantie, G. (1991).

tion in arid and semi-arid regions. With proper management, irrigation can improve productivity and soil quality. With improper management, however, irrigation can lead to waterlogging and salinization. Vast areas of irrigated lands have been degraded due to rise of the water table and buildup of salts in the soil profile. The WUE of existing irrigated lands can be greatly enhanced by the adoption of improved systems (FAO, 1997). Hillel (1997) proposed the HELPFUL irrigation system (high-frequency, efficient, low-volume, partial area, farm unit, low cost). Flood irrigation, the system most commonly used throughout the developing world, is wasteful and most inefficient and can lead to severe land degradation.

In addition to improving efficiency on existing irrigated lands, there is the possibility of expanding irrigation in some regions. Hillel (1997) estimated the total potential irrigable land area in sub-Saharan Africa to be 39 Mha. Rather than large-scale irrigation schemes based on large dams, the emphasis needs to be placed on

Table XI

Effect of Improved Management Systems on Erosion Control and Crop Yield on a Vertisol (12% slope) at Ouzera, Algeria

Treatment	Maximum runoff (% of rainfall)	Erosion (Mg ha^{-1} $year^{-1}$)	Yield (Mg ha^{-1})	
			Grain	Straw
Traditional system (wheat)	7–16	0.2–0.3	0.7	0.2
Improved alfalfa pasture	0–9	0.05–0.3	—	2.2
Improved wheat-legume rotation	1–8	0.1–0.2	6.5	3.1 (5 beans)
Bare fallow	7–86	2.7–6.0	0	0

Modified from Arabi, M., and Roose, E. (1992).

Table XII

Effects of Improved Management[a] on Runoff Control in Ouzera, Algiers

Management	Runoff (% of rainfall)	Income equivalent (U.S. \$ ha^{-1})
1. Agropastoral System		
(i) Traditional	16	89
(ii) Improved	8	1279
2. Sylvopastoral (brown soil)		
(i) Degraded	25	–
(ii) Reforested	3	–
(iii) Grassed	7	–
3. Orchard (red ferra litic soil)		
(i) Traditional	12	357
(ii) Improved	9	1507
4. Vineyard (brown colluval soil)		
(i) Traditional	8.3	1226
(ii) Improved	2.7	2334

[a]Improved systems involved appropriate tillage methods and recommended cropping systems. Modified from Roose, E. J. (1996).

small-scale irrigation projects. Appropriate small-scale irrigation schemes may involve use of ground water, runoff storage, water harvesting techniques, microcatchment farming, and other cost-effective and simple watershed management techniques. Essiet (1990) reported that in Kano State of Northern Nigeria there were more adverse effects in soils under large-scale irrigation than in those under small holder operations.

In addition to salt-affected soils, some water reserves are saline (brackish) with high salt content ranging from 5000–40,000 ppm. Some halophytic plants can be grown by irrigation with saline water (Table XIII). Experiments reported by Glenn *et al.* (1993) show that halophytes irrigated with seawater can produce biomass yield of 17–35 Mg ha^{-1} $year^{-1}$ with a net C uptake of 4–8 Mg ha^{-1} $year^{-1}$ (Table XIV). A considerable part of the biomass C produced has a long residence time because of the slow decomposition rate in drylands (Gifford *et al.,* 1992).

3. Erosion Control and Soil Restoration

a. Erosion Control

There is an urgent need to control erosion and restore productivity of degraded soils. Accelerated soil erosion severely impacts soil fertility (Leys and McTainsh, 1994). Therefore, erosion control is essential to maintaining, improving, and enhancing productivity. Some traditional erosion control measures can be effective under site-specific conditions (Van Dijk, 1997). But widespread adoption of im-

Table XIII

Some Useful Halophytes

Common name	Botanical name	Important characteristics	Usefulness
Pickle weed	*Salicornia* spp. (SOS-7)	Tolerant to saline water (35,000–40,000 ppm). It wilts when irrigated with fresh water. Biomass production capacity of 15–20 Mg ha^{-1} $year^{-1}$.	High grade fodder, and oil seed, 30% of seed is high grade oil. Straw contains 40% salt which after rising can be used as fodder.
Salt grass (wild wheat grain)	*Distichlis palmeri*	Perennial with no gluten in grains. Important for erosion control. Can be grown in intertidal zone with salt content of 40,000 ppm.	Edible grains, and forage grains rich in bran and fiber.
Ny Pa forage	*Distichlis* spp.	7–10 Mg/ha of dry matter per cutting with 4 or more cuttings per year. Salt tolerant forage with no accumulation of salt. Tolerant to saline water up to 2000–15,000 ppm of salt.	Forage as good as alfalfa. Protein content in leaves is 12–25%. Reclamation of salt-affected soils.
Salt bushes	*Atriplex nummularia* *A. halimus* *A. canescens* *A. lentiformis* *A. semibaccata* *A. glauca* *A. vesicaria*	Perennials can be irrigated with saline or brackish water, C-4 metabolism, drought tolerant with deep root system. Biomass production potential of 8–11 Mg ha^{-1} $year^{-1}$.	Useful forage crops, can be grown and harvested to remove salts.
Spirulina	*Spirulina geitleri*	These are algae adapted to saline/alkaline lakes and ponds in arid regions.	High protein food and feed.

proved practices is essential for desertification control and restoration of degraded soils. Engineering techniques of erosion control and runoff management can be made more effective when used in conjunction with biological control measures (e.g., *Vetiveria zizanioldes* hedges). Vegetal cover (percentage of the land area covered by vegetation) is important in controlling erosion by wind and water, gullying and sand dune migration (Lal, 1990; Roose, 1996). Conservation of water in the root zone and soil fertility improvement are important in establishing protec-

Table XIV

Mean Annual Biomass Yield and C Sequestration Rate by Halophytes Irrigated with Sea-water at Puerto Penasco, 1990–1992

Specie	Biomass yield ($Mg\ C\ ha^{-1}\ year^{-1}$)	C sequestration rate[a] ($Mg\ C\ ha^{-1}\ year^{-1}$)
Batis maritima	34.0	8.2
Atriplex linearis	24.3	6.7
Salicornia bigelovii		
Year one	22.4	5.6
Year two	17.7	4.3
Suaeda esteroa	17.2	4.3
Sesuvium partulacastrum	16.7	4.2

[a]Based on net primary production.
Modified from Glenn, E., Squires, V., Olsen, M., and Frye, R. (1993).

tive vegetal cover (Sanchez and Buresh, 1997; Johnston and Syers, 1996). In the southwestern U.S., Bedunah and Sosebee (1986) conducted a 3-year study to eradicate mesquite (*Prosopis gladulosa*) and control soil erosion. Eradicating mesquite increased herbaceous cover of Klein grass (*Panicum coloratum*) and other vegetation, reduced soil erosion, and improved SOC content and soil quality. Other important grass species for improving ground cover and enhancing forage quality of rangelands in the southwestern U.S. include black grama (*Bouteloua eriopoda*), blue grama (*B. gracilis*), and tobosa grass (*Hilaria mutica*) (Seitlheko *et al.*, 1993).

The establishment of shrubs and trees is also an important strategy for desertification control, restoration of eroded soils, and improvement in vegetation cover. The strategy is to plant multipurpose trees for shelterbelts or windbreaks (*Neem* or *Acacia* spp.), for reinforcing riverbanks (*Eucalyptus* or *Populus*), for animal fodder (*Leucaena*, *Acacia*, *Balbergia*), and for fuel wood. Bravo *et al.* (1995) observed that strip cropping controlled wind erosion and enhanced soil quality in Argentina. The soil under a 20-year old weeping lovegrass (*Eragrastic curvula*) pasture strip contained aggregates with 21% greater mean weight diameter and SOC levels 29% higher than did soil in the cropped strip.

Controlled grazing and low stocking rate are important strategies for managing vegetal cover and reducing soil erosion (Abel and Blaikie, 1989). Heavily grazed areas are prone to compaction with a resultant decline in infiltration rates (Seitlheko *et al.*, 1993). In the Pampa region of Argentina, Dreccer and Lavado (1993) observed that the preferential flow paths of a soil with a clayey matrix horizon were decreased by the trampling effect of cattle. Experiments conducted by Wiggs *et al.* (1994) on sand-dune surfaces in the southwest Kalahari Desert showed that destruction of the vegetation canopy increased near-surface wind velocity. Denudation by burning, grazing, or drought led to a threefold increase in dune surface activity. Kumar and Bhandari (1992, 1993) observed severe decline in vegetal cover on grazed sand dune areas in the Rajsthan desert of India. Construction of a barrier to exclude cattle reduced wind erosion, leading to an increase in the growth of palatable species. In contrast to the "conventional wisdom," however, soil and vegetation degradation was observed in northwestern Kenya to occur even in the absence of grazing, especially when plant species growing at the time were dependent on grazing-induced perturbation Oba (1992).

b. Soil Salinity Control

There is about 930 Mha of salt-affected soils in the world (Sumner *et al.*, 1998; Sumner and Naidu, 1998). Therefore, reclamation of salt-affected soils is an important facet of soil quality improvement and increasing C sequestration in below- and above-ground biomass and in the SOC pool. There are many proven methods for reclaiming salt-affected soils (Gupta and Abrol, 1990). The strategy is to enhance SOC content, improve soil structure and infiltration rate, and replace Na^+ adsorbed on clay minerals with Ca^{2+} and Mg^{2+}. Emerson (1995) observed in southern Australia that soil water retention at low suction increased approximately linearly with SOC content, independent of clay content. Creating and maintaining preferential flow paths are important to leaching soluble salts out of the root zone. Translocation of CO_3 to the subsoil with infiltrating irrigation water has an important ameliorative effect. Other options for reclaiming salt-affected soils are the planting of salt-tolerant plants, applications of manure, and use of gypsum as a soil amendment.

Growing salt-tolerant (*halophytic*) plants can improve above- and below-ground biomass production and increase SOC content. Singh (1989) observed that, among several fuelwood species evaluated, *Prosopis juliflora* was the best adapted to alkaline soils and produced the most biomass. *Sesbania sesban* and *Tamarix dioca* also exhibited good adaptability. Singh *et al.* (1994) observed that growing salt-tolerant woody species improved soil quality. The species *Prosopis juliflora, Acacia nilotica, Casuarina equisetifolia, Tamarix articulata, Leptochloa fusca*, and others caused a notable increase in SOC content. The data in Table XV show an increase in SOC content following planting of *Prosopis* alone or with grass. In comparison with the original SOC content of 0.24%, SOC contents in the depths 0–15 cm and 15–30 cm were, respectively, 0.85 and 0.55% for *Acacia nilotica,* 0.66 and 0.33% for *Eucalyptus tereticornis,* 0.93 and 0.58% for *Prosopis juliflora,* 0.86 and 0.58% for *Terminalia arjuna,* and 0.62 and 0.47% for *Albizzia labbek.*

In addition to providing fuel wood, some fruit trees are adaptable to highly alkaline soils. Adaptable fruit trees for northwestern India include jamun (*Syzygrum cuminii*), tamarind (*Tamarindus indica*), ber (*Zizyphus mauritiana*), and guava

Table XV

Increase in SOC Content of an Alkali Soil Due to Reclamative Effect of *Prosopis juliflora–Leptochloa fusca* Silvipastoral System

Time (months)	0–0.15 m depth		0.15–0.30 m depth	
	Prosopis	Prosopis + grass	Prosopis	Prosopis + grass
0	0.18%	0.19%	0.13%	0.12%
22	0.20%	0.28%	0.12%	0.16%
52	0.30%	0.43%	0.19%	0.21%
74	0.43%	0.58%	0.29%	0.36%

Adapted from Singh, G., Singh, N. T., and Abrol, I. P. (1994). With permission.

(*Psidium guajava*) (Singh *et al.,* 1997). Tree species with beneficial and reclamative effects on alkali soils include *Eucalyptus tereticornis, Acacia nilotica, Albizia lebbeck,* and *Terminalia arjuna.* Improvements in SOC content can set in motion restorative trends leading to increased soil microbial activity and thus biomass C (Ragab *et al.,* 1990). In north central India, Garg (1998) observed significant improvements in SOC content of a sodic soil over an 8-year period following planting of salt-tolerant tree species. The SOC pool in the top 0.6-m layer increased with time after trees were planted (Figure 9). The rate of increase was low for the first 2–4 years, high (exponential) between 2 and 6 years, and stabilized at a low value between 6 and 8 years.

Salt-tolerant trees and forage species have also proven useful in reclaiming sodic soils of Pakistan (Qadir *et al.,* 1996). In addition to improving structural properties, trees affect salt balance by lowering the water table, which leads to a net downward leaching. In southwestern Australia, Farrington and Salama (1996) observed that revegetation by trees and shrubs controlled salinity on dryland.

Applications of manure and gypsum are also important for improvement of soil structure and reclamation of sodic soils. Use of manure and compost is facilitated by integrating livestock with the cropping system (Chaudhary *et al.,* 1981; Haque *et al.,* 1995; Harris, 1995; Pieri, 1995). More (1994) reported that application of farm by-products and organic manures in Maharashtra improved quality of sodic vertisols, enhanced SOC content, and increased crop yields. Batra *et al.* (1997) observed the impacts of growing karnal grass (*Leptochloa fusca*) and of applying gypsum on the reclamation of an alkaline soil. There were significant improvements in soil quality within 3 years of applying these treatments. Singh *et al.* (1988, 1989) and Singh and Singh (1995, 1996) observed that application of gypsum and farmyard manure enhanced the survival rate of mesquite (*Prosopis juliflora*) on a highly alkaline soil.

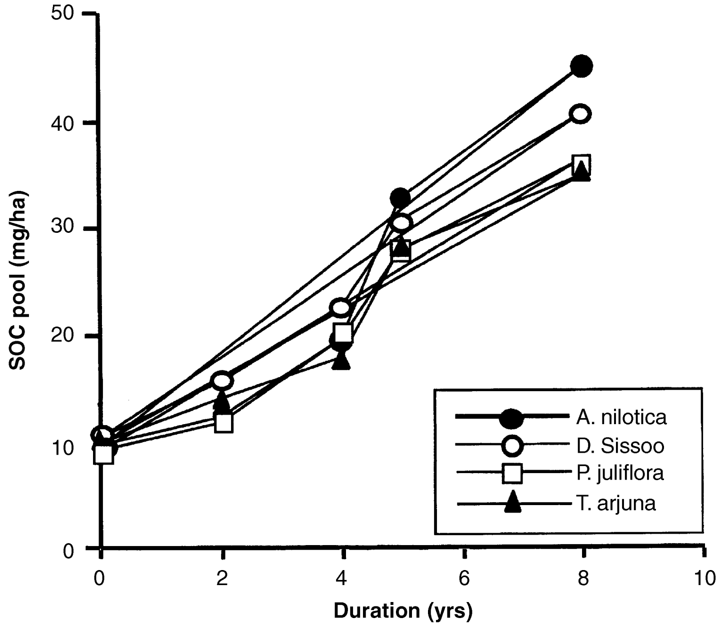

Figure 9 C sequestration through reclamation of salt-affected soils in northern India (recalculated from Garg, 1998).

4. Soil Fertility Management

The improvement of soil fertility is an important aspect of soil quality enhancement and C sequestration in soil and biomass. Fertilizer formulations may be different for different soils and land-use patterns, but low rates of fertilizer application are usually recommended for dry areas where rainfall is uncertain. Where feasible, a combination of organic manures and inorganic fertilizer is better in such regions than inorganic fertilizer alone. Several long-term experiments in dry regions have demonstrated the importance of judicious use of fertilizer, compost, and nutrient management (Traoré and Harris, 1995; Singh and Goma, 1995; Pieri, 1995; Miglierina *et al.*, 1996; Laryea *et al.*, 1995). Diaz *et al.* (1997) monitored the impact of urban biosolids on soil quality in semi-arid areas of Spain. Plant cover and biomass increased substantially, and urban biosolids proved an effective amendment to regenerate the plant cover on degraded soils.

Nambiar (1995) summarized data from several long-term soil fertility management experiments conducted in India. For sandy soils of Ludhiana, Punjab, the SOC content of unmanured plots increased from 0.20% in 1971 to 0.25% in 1989. With application of NPKS and manure, however, the SOC content increased from about 0.20% in 1971 to 0.40% in 1989. For a clayey soil at Jabalpur, central India, the SOC content increased from about 0.6%–1.1% with recommended soil fertility management. Jangir *et al.* (1997) observed the yield of pearl millet (*Pennisetum glaucumu*) in an arid region of Jodhpur, Rajsthan, to increase with application of N fertilizer up to 40 kg N ha^{-1} under rainfed conditions and 80 kg N ha^{-1} with supplemental irrigation. In Gurdaspur and Hissar, Mishra *et al.* (1974) reported that application of manure at rates of 9–30 Mg ha^{-1} year^{-1} led to a significant increase in SOC content. Similar observations were made by Ruhal and Singh (1982). In Madras, Muthuvel *et al.* (1989) reported that application of manure to a cotton-pearl millet rotation on a dryland Vertisol increased SOC content and crop yield. Chaudhary *et al.* (1981) reported that SOC contents were increased by ~0.03, 0.04, and 0.14% by applications of 30 kg of P_2O_5, 60 kg P_2O_5, and 15 Mg of manure per hectare to a pearl millet-wheat rotation.

The beneficial impact of soil fertility enhancement on SOC content has been observed elsewhere in dry areas prone to desertification. In the Ethiopian Highlands, Wakeel and Astartke (1996) recommend intensification of agricultural practices on Vertisols to minimize risks of land degradation. Michels *et al.* (1995) observed in Niger that surface application of millet residue mulch at 2 Mg ha^{-1} significantly reduced wind erosion and increased SOC content and cation exchange capacity of the surface soil layer. For Kalahari Desert soils of Botswana, Perkins and Thomas (1993) reported that SOC content of soils near wells ranged from 4.8–7.6% compared with 0.4–0.6% about 1000 m away (Table XVI). With the needed inputs of organic matter, therefore, SOC content can be increased by an order of magnitude even in harsh environments.

Table XVI

SOC Content of Soils of Three Ranches (A, B, C) in the Kalahari Desert, Botswana, in Relation to the Distance from the Well

Distance from borehole (m)	SOC content (%) A	B	C	Mean (%)
0	6.4	4.8	5.0	5.4 ± 0.9
25	2.0	7.6	1.4	3.7 ± 3.4
50	2.0	2.2	0.4	1.5 ± 1.0
200	0.4	0.6	0.6	0.5 ± 0.1
400	0.4	0.6	0.6	0.5 ± 0.1
800	0.6	0.6	0.6	0.6 ± 0.0
1500	0.8	0.6	0.4	0.6 ± 0.2

Modified from Perkins, J. S., and Thomas, D. S. G. (1993).

5. Improved Farming Cropping Systems

The productivity of drylands can be greatly improved by replacing traditional farming/cropping systems with improved systems.

a. Crop Rotations

Crop rotations are important to soil quality enhancement. Galantini and Rosell (1997) reported that rotations of mixed pasture (5.5 years) and annual crops (4.5 years) in Argentina maintained 17.3 Mg ha^{-1} of SOC compared with 11.2 Mg ha^{-1} in continuous cultivation with a wheat-sunflower rotation. In another study, Miglierina *et al.* (1993, 1996) observed a high SOC content in wheat-grassland and wheat-alfalfa rotations especially with a conservation tillage system. Barber (1994) observed in eastern Bolivia that subsoiling and incorporation of cover crops in rotation enhanced quality of degraded soils. Shahin *et al.* (1998) observed in Saudi Arabia that introducing alfalfa in rotation with wheat on a sandy soil decreased salinity. A threefold increase in SOC content compared with the control occurred as well. In Maharashtra, India, Lomte *et al.* (1993) reported that intercropping sorghum with legumes and application of manure increased SOC content and aggregation.

b. Fallowing

Fallowing, taking land out of agricultural production and permitting natural vegetation to grow, is a common practice for restoring degraded soils. Bush fallowing is most widely practiced in tropical and subtropical agriculture (Nye and Greenland, 1962). In northern Nigeria, Abubakar (1996) monitored the impact of duration of fallowing on changes in SOC content. The mean SOC content of the surface soil was 0.94% for 2-year fallow, 1.13% for 5-year fallow, 1.42% for 10-year fallow, and 1.44% for 15-year fallow. There were also improvements in SOC content of the subsoil. The SOC content reached 95% of the equilibrium level within 10 years of fallowing. The data from a 30-year fallowing experiment conducted in eastern Spain showed that SOC content stayed more or less constant at a low level for several years. Subsequently, there occurred a significant increase in SOC content in the top 0.1m-layer and a slight increase in the 0.2 to 0.3-cm depth. The data showed a notable increase in SOC content to occur after 20 years of fallowing. Although natural vegetative regeneration can enhance soil quality (Ruecker *et al.,* 1998), fallowing efficiency can be greatly improved by the cultivation of appropriate cover crops (Barber and Navarro, 1994).

c. Residue Mulch

Application of biomass (crop residue, tree leaves, and branches etc.) to degraded soils can enhance activity of soil fauna and improve soil quality. Termite activity is important to improving soil structure in arid and semi-arid regions. Mando (1997) monitored the effectiveness of mulch and termite activity in the rehabilitation of soil structure in Burkina Faso. Termite activity improved structure, soil-water regime, biomass production, and SOC content. In Niger, Sterk *et al.* (1996) observed that application of Zai (using pits filled with compost) and mulching with crop residues or tree branches decreased risks of wind erosion and improved productivity. Residue mulches are effective against rill/interrill erosion on gentle slopes but not against mass movement on steep slopes. Badanur *et al.* (1990) reported that application of sorghum stubbles and safflower stalks on Vertisols in Karnataka, India, improved soil structure and increased SOC content. In degraded red soils of south China, Li *et al.* (1994) observed that applications of organic material to degraded soils enhanced SOC content.

d. Forestry Measures for C Sequestration In Dry Regions

Widespread deforestation for fuel wood and other domestic uses has exacerbated the impact of harsh dry environments (Boahene, 1998). Therefore, afforestation is an important strategy. There are several multipurpose trees which can grow under the harsh environments of dryland regions. Growing trees can improve soil quality, albeit at a slow rate, and sequester C in soil and biomass. Kair (*Capparis decidua*), one such tree, is adaptable to the drylands of northwest India (Gupta *et al.,* 1989). Mesquite (*Prosopis* spp.) has been useful in reclaiming salt-affected soils in India.

Lack of fuel for cooking is a major problem among rural communities in dry areas. Deforestation for fuel wood has led to denudation of the landscape and exacerbated risks of erosion and desertification. The establishment of fuelwood plantations is critical to desertification control and restoration of degraded soils. Some promising species for fuel wood production, soil quality improvement, and desertification control are listed in Table XVII. In Nigeria, Jaiyeoba (1998) monitored changes in soil properties following conversion of Savannah woodland into pine (*Pinus oocarpa*) and *Eucalyptus camaldulensis* plantations. The SOC content of 0–0.15-m depth declined during initial stages of tree establishment and increased thereafter to a steady equilibrium value, attained in about 16 years. The initial decline was apparently due to the near absence of ground cover and low biomass production, emphasizing the need to grow grass or a cover crop in association with trees. Garg (1998) monitored changes in properties of a sodic soil under four different tree species in north central India. Results showed a marked improvement in biogeochemical characteristics in general but in SOC content in particular. The SOC pool increased from less than 10 Mg ha^{-1} to about 45 Mg ha^{-1} over an 8-year period. Dry soil bulk density of the 0–0.15 m layer of the unplanted site was 1.8 Mg m^{-3} compared with 1.61 Mg m^{-3} under *Acacia nilotica*, 1.50 Mg m^{-3} under *Dalbergia sissoo,* 1.43 Mg m^{-3} under *Prosopis juliflora,* and 1.55 Mg m^{-3} under *Terminalia arjuna.*

Table XVII

Tree Species Suitable for Firewood and Biofuel Production in Dry Areas

Common Name	Botanical name
Tamarisk	*Tamarix* spp.
Gum trees	*Eucalyptus* spp.
Leucaena	*Leucaena* spp.
Cypress	*Cupressus* spp.
Casuarinas	*Casuarina equisetifoli*a and other spp.
Mesquite	*Prosopis* spp.
Neem	*Azadirachta* sp.
Acacia	*Acacia* spp (see Table IX)
Teak	*Tectona grandis*
Casia sp.	C. *Siamea*
Other species	*Dalbergia* spp.
	Khaya spp.
	Albizia spp.
	Cassia spp.
	Parkia spp.
	Terminalia arjuna
	Pongamia pinnata
	Sesbania sesban
	Morus alba
	Populus deltoides

Adapted from Le Houérou, H. N. (1975), Mainguet, M. (1991), Singh, G., Singh, N. T., and Abrol, I. P. (1994), and Patil, S. G., Hebbara, M., and Devarnvadagi, S. B. (1996).

C. Options For C Sequestration In Dry Areas

Three principal options for C sequestration in dry areas are: (1) agricultural intensification on prime agricultural land, (2) restoration of degraded or desertified lands, and (3) biofuel production.

1. Agricultural Intensification for SOC Sequestration

Adoption of improved/recommended practices on agricultural and pastoral lands is an important strategy for agricultural intensification. Many recommended practices for sustainable use of soil and water resources have proven successful under on-farm conditions and are awaiting adoption by farmers in dry areas. Recommended practices involve soil-water conservation and management, irrigation management, soil fertility enhancement including inorganic fertilizers and organic amendments, residue management and tillage methods, improved varieties and associated cropping systems, and integrated pest management including effective weed control measures. Improving soil quality is the principal strategy for C sequestration. Increasing biomass production, decreasing losses of soil, water, and nutrients from the ecosystem, fostering soil biodiversity (activity and species diversity of soil flora and fauna), and strengthening nutrient cycling mechanisms would improve soil quality, increase aggregation, and increase SOC content in the stable fraction with relatively longer turnover time.

Programs of agricultural intensification can be applied to prime agricultural land–land with little or no degradation. The area of land meeting this definition is estimated at 427.3 Mha. Because of their inherently high soil quality and favorable biomass production, these soils can undergo agricultural intensification through better management. Further, plants with the C-3 photosynthetic pathway generally have increased C fixation rates when CO_2 levels in the atmosphere are increased. Enhanced WUE due to partial stomatal closure in CO_2-enriched environments may also lead to high biomass production in both C_3 and C_4 species (Ojima *et al.,* 1993). Assuming that adoption of improved land use and farming/cropping systems may lead to an increase in SOC content of 30–50 kg ha^{-1} $year^{-1}$, the total potential of C sequestration in such soils is 0.013–0.021 Pg C $year^{-1}$.

2. Soil Restoration and Desertification Control

Desertification control is an important global strategy for C sequestration in soils (Squires *et al.,* 1995). Restoration of desertified soils involves (a) eroded land, (b) physically/chemically degraded soils, and (c) salt-affected soils.

a. Restoration of Eroded Soils for C Sequestration

Land area affected by strong and extreme soil erosion is 104 Mha and an additional 424 Mha is moderately eroded. The strongly and extremely degraded areas need to be taken out of agricultural/pastoral land uses and put under restorative management. Trees and shrubs can be grown on these lands to prevent further soil erosion and to produce biofuel. Establishing suitable tree species would have three distinct benefits: (a) provide the much needed ground cover and root system for protecting the soil against erosive forces of wind and water, (b) produce biomass that can be used as fuel, and (c) enhance SOC content and sequester C in soil. Suitable species are those with deep root systems that anchor the soil and protect the seedlings against drought stress. Some of the below-ground biomass produced will be converted to humus. Assuming a low rate of increase of SOC content at 40–60 kg C ha^{-1} $year^{-1}$, the potential of C sequestration in soil is 0.004–0.006 Pg C $year^{-1}$.

There are 424 Mha of moderately eroded lands in dry areas. Recommended agricultural management practices need to be adopted on these lands for soil erosion control and soil quality enhancement. Such practices include installation of engineering devices (e.g., diversion channels, drop structures, infiltration ditches, stone filters or gabions, terraces) and adoption of appropriate soil (e.g., tillage methods, soil fertility management, residue management, soil amendments) and crop management systems (e.g., rotations, agroforestry techniques, integrated pest management). Adoption of improved practices would control soil erosion, improve crop yield, enhance soil quality, and sequester C as the result of increased SOC content. Enhancing SOC content in dry climates is a challenging task, and the rate of increase in SOC may be low even under ideal conditions. Assuming the potential rate of increase in SOC content at 80–120 kg C ha^{-1} $year^{-1}$, the total potential of C sequestration in moderately eroded soils is 0.03–0.05 Pg C $year^{-1}$.

The total potential of C sequestration through soil erosion management is shown in Table XVIII. Adoption of erosion control measures and recommended management practices would decrease C emissions from erosion-displaced sediments by as much as 50–75% of the estimated emission (0.13–0.20 Pg) (Lal, 1995). Restoring vegetative cover to strongly and extremely eroded lands can offset fossil fuel emissions of 0.14–0.21 Pg C $year^{-1}$ and SOC sequestration of 0.004–0.006 Pg C $year^{-1}$. Adoption of recommended agricultural practices on slightly and moderately eroded lands (Oldeman, 1994) has the potential to sequester 0.06–0.10 Pg C $year^{-1}$. Thus the total potential of C sequestration through soil erosion management in dry areas is about 0.33–0.52 Pg C $year^{-1}$.

b. Physical and Chemical Degradation

In addition to erosion, drylands also are prone to physical and chemical forms of soil degradation. Soil physical degradation involves decline in soil structure leading to crusting, compaction, hard-setting, and exposure of plinthite that eventually hardens into a laterite (Lal and Miller, 1989). Mechanisms of soil chemical degradation include salinization, fertility depletion or nutrient imbalance, and acidification. Land area affected by strong and extreme forms of physical and

Table XVIII

Potential of C Sequestration Through Restoration of Degraded Soils

Technological options	C sequestration potential (Pg C $year^{-1}$)
1. Restoration of eroded soils	
(a) Erosion control (50–75% of emission reduction)	0.13–0.20
(b) Restoration of strongly/extremely eroded soils	0.004–0.006
(c) Adoption of recommended practices on slightly/moderately eroded soils	0.06–0.10
2. Fossil fuel off-set	0.14–0.21
Total	0.33–0.52

chemical degradative processes is 34 Mha (UNEP, 1991). As with the severely eroded lands, these lands may be taken out of production and reverted to restorative land use through the planting of appropriate trees and shrubs. In addition to production of biofuel (see the following section), soil restoration would also improve SOC content, albeit at a low rate of about 40–60 kg ha^{-1} year^{-1}. Therefore, the potential of SOC sequestration in these soils is 0.001–0.002 Pg year^{-1}.

In addition, there are 46 Mha of moderately degraded soils. Assuming that adoption of recommended practices on these soils may lead to C sequestration at the rate of 80–120 kg ha^{-1} year^{-1}, the potential of SOC sequestration on these lands is 0.004–0.006 Pg C year^{-1}. Therefore, total potential of C sequestration through restoration of physically and chemically degraded soils is 0.005–0.008 Pg C year^{-1}.

c. SOC Sequestration through Reclamation of Salt-Affected Soils

There are 930 Mha of salt-affected soils in arid and semi-arid regions of the world. Adoption of reclamative measures on these soils can increase above- and below-ground biomass production and increase SOC content. Assuming that the rate of SOC increase through adoption of reclamative measures is 200–400 kg C ha^{-1} year^{-1} (the rate was 3–4 Mg C ha^{-1} year^{-1} in some soils in north central India), the potential for C sequestration is 0.186–0.372 Pg C year^{-1}.

Calculations made in this section are tentative due to possibilities of double accounting. For example, soils prone to physical and chemical degradation are also susceptible to accelerated soil erosion. However, the database is not precise enough to assess areas specifically affected by each soil degradative process.

3. Biofuel Offset through Desertification Control

Strongly and extremely degraded soils should be taken out of agricultural and pastoral land uses and planted to tree, shrub, or grass species that can be used as biofuel. The biomass can be used for domestic purposes or for power generation. Trees established on 104 Mha of severely eroded lands can also be used as biofuel. Once established (2–3 years after planting/sowing), the above-ground biomass production potential in these dry environments is 2–3 Mg ha^{-1} year^{-1} with a total production of 208–312 $\times$ 10^6 Mg year^{-1}. With a fuel efficiency of 0.7 for direct use (relative to 1 for fossil fuel), the biofuel C offset for these lands is 0.14–0.21 Pg C year^{-1}. Total area of strongly and extremely degraded lands (erosion, physical and chemical degradation) in dry areas is 138 Mha. Growing appropriate biofuel crops on such lands is an important strategy. Once established (2–3 years after planting/sowing), the above-ground biomass production potential in these environments is 2–3 Mg ha^{-1} year^{-1} with a total production of 275–413 $\times$ 10^6 Mg C year^{-1}. With a fuel efficiency of 0.7 for direct use, the biofuel C offset for these lands is 0.19–0.29 Pg C year^{-1}.

D. C Sequestration in Secondary Carbonates in Drylands

The role of soil inorganic C (SIC) in C sequestration is less well understood than that of SOC. SIC may be a sink or a source or have no effect upon C sequestration depending on site-specific conditions. In systems of partial or complete soil leaching, the major mechanism for sequestered SIC is via movement of HCO_3 into groundwaters or other closed systems isolated from the ambient environment. Reconstructions of carbonate fluxes in soils formed in strongly calcareous parent materials over geological time periods suggest that this mechanism could account for upwards of 1 Mg ha^{-1} year^{-1} of SIC. While this mechanism may appear of limited importance on degraded drylands, certainly it has implications when groundwaters undersaturated with $Ca(HCO_3)_2$ are used for irrigation. Enhanced biomass primary productivity and salinity control strategies (e.g., gypsum amendments, organic wastes) can result in increased leaching of $Ca(HCO_3)_2$ if land is irrigated with waters that are not already saturated with respect to bicarbonates.

Dissolution of exposed carbonates in soil systems by acid rain, nitrogenous fertilizers, oxidation of iron sulfides, or organic acids may result in a source of atmospheric CO_2 if subsequent precipitation of pedogenic carbonates occurs. However, if the dissolved bicarbonates are either leached through the soil or removed by overland flow without subsequent precipitation of carbonates, then this mechanism creates a transient sink of intermediate to long residency rather than a source of atmospheric CO_2. The rate of C sequestration through formation of secondary carbonates is not well established. Some researchers argue that the rate is slow (3–5 g m^{-2} year^{-1}) and of little significance (Schlesinger, 1997). Others, however, support the idea that rate of sequestration of atmospheric C may be much higher (Nordt *et al.*, 1999), with a maximum rate of 114–124 kg C ha^{-1} year^{-1}. Formation of secondary carbonates is accentuated by biotic activity (e.g., root growth, termites) (Monger and Gallegos, 1999).

Table XIX

Estimates of C Sequestration Through Formation of Secondary Carbonates

Ecoregion	Land area (10^6 km^2)	Potential rate of C sequestration[a] (kg ha^{-1} year^{-1})	Total sequestration potential (Pg C year^{-1})
Arid	25.5	0–1	0–0.0026
Semi-arid	23.1	3–114	0.0069–0.2633
Sub-humid	13.0	1–124	0.0013–0.1599
Total			0.0082–0.4258

[a]Data courtesy of Dr. L. P. Wilding, Texas A & M University, College Station, TX.

The data in Table XIX show the potential of C sequestration through formation of secondary carbonates with a range of 0.008–0.426 Pg C year^{-1}. The wide range is indicative of the great variation and large uncertainty due to differences in soil profile characteristics, moisture and temperature regimes, land uses, and ecoregional characteristics.

E. Desertification Control: How Much Carbon Can Be Sequestered?

The total potential for restoration of degraded lands and desertification control to sequester C is shown in Table XX. The total potential appears to range from 0.9–1.9 Pg C year^{-1}, with a mean of about 1.4 Pg C year^{-1}. The options with high C sequestration potential include erosion control (36%), biofuel C offset (29%), and reclamation of salt-affected soils (21%). Secondary carbonate formations might account for another 14%. It is apparent that restoration of eroded and salt-affected soils and erosion control are important strategies. Squires *et al.* (1995) estimated that management of drylands through desertification control has the C sequestration potential of 1.0 Pg C year^{-1}. These high estimates are in contrast to the overall low C storage potential of world soils estimated by Schlesinger (1990). Estimates presented in Table XX are crude, tentative, and merely suggestive of the high potential that exists in adoption of judicious land use measures in drylands. Uncertainties are high and may be on the order of 30–50%. Further estimates of potential for different strategies are not additive, and the data need to be used in due consideration of site-specific conditions.

Table XX

Potential of Desertification Control and Land Restoration to Sequester C

	Potential to sequester C		
	Pg C year^{-1}		% of total
Process	Range	Mean	Potential
Emission reduction through erosion control	0.2–0.3	0.25	18
Restoration of eroded lands	0.2–0.3	0.25	18
Restoration of physically and chemically degraded soils	<0.01	<0.01	–
Reclamation of salt-affected soils	0.2–0.4	0.3	21
Agricultural intensification on undegraded soils	0.01–0.02	0.015	–
Fossil fuel C offset through biofuel production	0.3–0.5	0.4	29
Sequestration as secondary carbonates	0.01–0.4	0.2	14
Total	0.9–1.9	1.4	100

VI. SUMMARY AND CONCLUSIONS

Feeding the world's burgeoning population will be one of the most serious challenges that science and society face in the new century. Adding carbon to soil in the form of organic matter is one means of sustaining the productivity of agriculture so that it can do that job. It now turns out that practices that increase the content of C in soil can also help in the solution of another major environmental challenge by slowing the increase of the radiatively active greenhouse gas, CO_2, in the atmosphere. The combustion of fossil fuels and land use changes, especially tropical deforestation, are the proximate causes of the rising atmospheric concentration of CO_2. The organic matter content of most agricultural soils is significantly lower than it was prior to cultivation. These soils are estimated to provide a sink for between 40 and 80 Pg of carbon that may be satisfied by application of appropriate management practices in the course of the next 50–100 years. Vast areas of degraded soils not currently in agricultural use, if reclaimed and properly managed, can provide an even larger sink for carbon.

However, soil carbon sequestration, is no panacea. As Fig. 1 shows, far more carbon is likely to be emitted in the 21st century than can ever be sequestered in soils. Other measures for stabilizing atmospheric CO_2 will require (a) that fossil energy be used more efficiently, (b) that new, non-CO_2-emitting energy sources be deployed, and (c) that technologies be developed for removing CO_2 at the smokestack and depositing it in the deep oceans or in depleted oil wells or aquifers. The new energy sources and C-capturing technologies are not yet fully developed and are likely to be very costly for some time after they are. Conversely, methods for increasing soil C content are well established, are immediately deployable, and are known to have mostly beneficial effects on the environment. Thus, it makes sense to emphasize the value of immediate efforts to sequester C in soils while the new energy and disposal technologies are being readied.

In this paper we have emphasized three issues: how to increase sequestration and retention of C in soils; how to monitor or otherwise estimate the actual C sequestration accomplished in operational programs; and how best to further C sequestration as a means of restoring degraded lands of the world.

Significant progress in the field of soil C sequestration should be expected from a more mechanistic understanding of the processes operating and regulating C flows and stabilization in soil. There is opportunity for unifying our knowledge of SOM in terms of kinetic components with what we know by treating it as discrete elements. The challenge is to learn how to characterize these discrete entities—such as light fraction and particulate organic matter—by their turnover rates. This will ensure the creation of viable substitutes for the current kinetic compartments of simulation models. To accomplish this will require continued efforts toward a mechanistic understanding of soil structure formation and stabilization, soil organic matter-structure interactions, and the role of roots and organisms in the flow of belowground carbon.

The emergence of soil C sequestration as a potential strategy for controlling greenhouse warming and the trading in C that will follow may require that changes in SOC be monitored for many years in many thousands of fields worldwide. Currently available methods for monitoring changes in SOC are labor and cost intensive, relatively imprecise, and unable to detect changes over periods shorter than 3–5 years. Research and development will be needed to provide methods that are nondestructive, more accurate and precise, and reasonably priced to support operational monitoring programs. There is a need for sensors that can measure C at fixed sites for experimentation and/or be used to probe at fixed or random points within a field contracted to sequester C. Expanded networks of C eddy flux measurements are needed to provide verification for models of soil C change over a wide range of land uses, but they cannot practically be used to monitor individual farm fields. High-resolution remote sensing imagery of land use/land cover can also provide background information for verification of other measurements or models. Future satellites equipped with hyperspectral sensors are expected to provide farm- and field-scale imagery that would allow detailed monitoring of C-sequestration practices (e.g., conservation tillage, irrigation). Simulation modeling of SOC and agroecosystem processes will necessarily play an important role in the early years after operational sequestration programs come into effect.

There are vast areas of degraded and desertified lands throughout the world, many in developing countries where improvements in rangeland management, dryland farming, and irrigation can add C to soil. These improvements will begin the essential process of stabilizing the soil against further erosion and degradation with concomitant improvements in fertility and productivity. These lands now provide a special opportunity to link the objectives of two United Nations Conventions—the Framework Convention on Climate Change and the Convention to Combat Desertification.

Our analysis indicates that three campaigns for restoration of degraded lands and desertification control, namely erosion control, biomass production to offset fossil fuels, and reclamation of salt-affected soils together, have the potential to sequester from 0.7–1.6 Pg C year^{-1}. Well-proven land management practices and a wide variety of adapted species are available for the task. The formation of secondary carbonates in irrigated soils might sequester another 0.1–0.3 Pg C year^{-1}.

The growing interest in soil C sequestration indicates that agricultural soils are likely to play an important role in stabilizing the atmospheric concentration of CO_2 and to do so by means that are consistent with sustainable agriculture.

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