

Terrestrial Biological Carbon Sequestration: Science for Enhancement and Implementation

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The purpose of this chapter is to review terrestrial biological carbon sequestration and evaluate the potential carbon storage capacity if present and new techniques are more aggressively utilized. Photosynthetic CO₂ capture from the atmosphere and storage of the C in aboveground and belowground biomass and in soil organic and inorganic forms can be exploited for safe and affordable greenhouse gas (GHG) mitigation [Watson *et al.*, 2000]. Nevertheless, C sequestration in the terrestrial biosphere has not been seriously pursued since its introduction in the Kyoto Protocol over a decade ago. Concerns have been raised that C sequestration in the biosphere is finite and not permanent, that it is difficult to measure and monitor, that there would be “carbon leakage” outside of the mitigation activity, and that any attention paid to environmental sequestration would be a distraction from the central issue of reducing GHG emissions from energy production and use. International accord and success in reducing emissions from the energy system are not coming easily, and concerns about climate change are growing. It is time to reevaluate all available options that might not be permanent yet have the potential to buy time, bridging to a future when new energy system technologies and a transformed energy infrastructure can fully address the climate challenge. Terrestrial sequestration is one option large enough to make a contribution in the coming decades using proven land management methods and with the possibility that new technologies could significantly enhance the opportunity.

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1. INTRODUCTION

Fossil fuel combustion and land use change have elevated atmospheric CO₂ concentrations from 280 ppmv at the beginning of the industrial era to more than 381 ppmv in 2006. Carbon dioxide emissions from fossil fuels and cement rose 71% during 1970–2000 to a rate of 7.0 pg C year⁻¹ [Marland *et al.*, 2007]. Canadell *et al.* [2007] estimated that CO₂ emissions rose at a rate of 1.3% per year during 1990–1999, but since 2000, it has been growing at 3.3% per year. Emissions reached 8.4 pg C year⁻¹ in 2006. It is likely that the current 2-ppm annual increase will accelerate as the global economy expands, increasing the risk of climate system impacts. Terrestrial sequestration is one option large enough to make a contribution in the coming decades using proven land-management methods and with the possibility that new technologies could significantly enhance the opportunity. Here we review progress on key scientific, economic, and social issues; postulate the extent to which new technologies might significantly enhance terrestrial sequestration potential; and address remaining research needs.

Since the 1950s, oceans and terrestrial ecosystems have taken up, in net, about 45% of all fossil fuel emissions. There are, however, indications that this fraction may be decreasing because natural sinks may not be able to keep pace with the increase in fossil fuel emissions due to ocean acidification and as some portions of the terrestrial biosphere reach saturation [Canadell *et al.*, 2007].

Both the accelerating greenhouse gas (GHG) emission rate and failure of environmental sinks to keep pace make stabilization of atmospheric CO₂ during the next century an even larger challenge. Further complicating the problem, atmospheric stabilization at any concentration requires that net emissions level off and eventually drop to near zero. To achieve this stabilization requires transformation of the energy systems worldwide, which will require many decades for development and deployment. Large-scale application of current terrestrial methods can effectively “buy time” [McCarl and Sands, 2007] until the development of new technologies make contributions. Since the costs that societies must bear in taking action on climate change depend on the scalability and costs of the contributing options, including terrestrial sequestration in a portfolio of response options adds early flexibility and decreases the overall cost of attaining a given level of emission reductions [Edmonds *et al.*, 2000; Pacala and Socolow, 2004; Edmonds *et al.*, 2007]. Terrestrial C sequestration, with application of known management methods, could conservatively sequester more than 0.5 pg C year⁻¹ by 2040 and contribute from 6% to 23% of the emissions mitigation necessary by midcentury and accumulating to over 40 pg C by 2100 to contribute to

atmospheric stabilization [Thomson *et al.*, 2008]. We suggest that, with the development and implementation of selected technologies, terrestrial sequestration could plausibly be enhanced severalfold. Finally, we review the progress made on what are perceived as impediments to implementation and how they can be solved.

2. OVERVIEW OF TERRESTRIAL BIOLOGICAL C SEQUESTRATION

Terrestrial sequestration is the storage of C in above-ground and belowground biomass and in soil C as soil organic matter and inorganic C. Carbon stored in forests and soils can persist for decades to centuries. We begin with an overview of the basic present-day approaches to terrestrial carbon sequestration.

2.1. Increasing Forest Biomass

The rate of net CO₂ uptake by forests varies in a predictable way as forests grow from establishment or recover from past disturbance [Pregitzer and Euskirchen, 2004]; understanding the factors that influence the rate of CO₂ uptake is the basis for management [Birdsey *et al.*, 2006, 2007]. Forestry activities that promote C storage include afforestation, improved forest management, and deforestation avoidance. Improved management to enhance productivity, control disease, and manage fires during the forest life cycle would increase C sequestration over that calculated by Thomson *et al.* [2008] in the base scenario of Plate 1. Protecting forests from wildfire increases C stocks but, combined with climate change effects, may also increase the risk of larger releases of stored CO₂ [Westerling *et al.*, 2006]. Afforestation, particularly on abandoned agricultural land and degraded land, generally increases soil C in addition to producing wood [Post and Kwon, 2000; Guo and Gifford, 2002]. Avoiding deforestation and forest degradation preserves existing C stocks that would otherwise be lost to the atmosphere. Full accounting for the effects of different management actions on C storage requires knowledge of changes in ecosystem C pools [Johnson and Curtis, 2001; Echeverria *et al.*, 2004], harvested wood products [Schlamadinger and Marland, 1996], and fossil fuel GHG emissions associated with growing, harvesting, and manufacturing [Schlamadinger *et al.*, 1997].

2.2. Soil C Sequestration

A combination of biological, chemical, and physical processes results in soil C stores about three times as large as in living vegetation. Soil C sequestration connotes both an in-

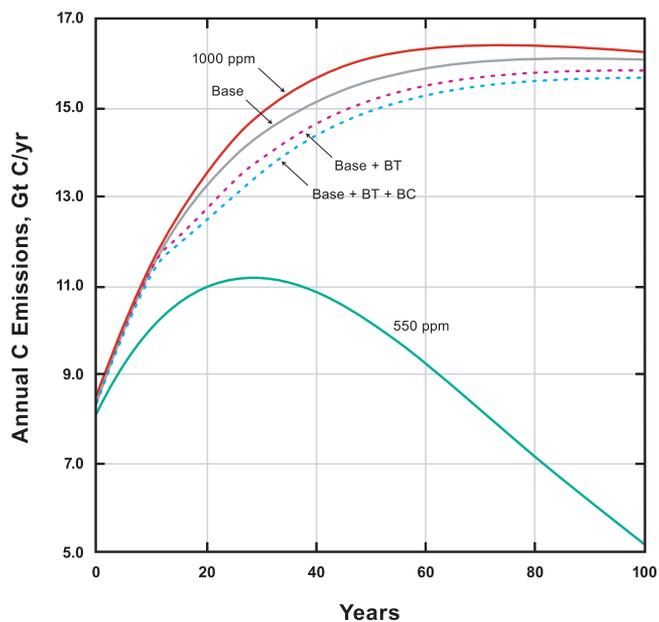


Plate 1. Projected net atmospheric CO₂ emissions in the 21st century for three scenarios involving terrestrial C sequestration and two reference scenarios for stabilization at 1000 (red line) and 550 ppmv (green line). The terrestrial sequestration scenarios include the conservative baseline scenario of *Thomson et al.* [2008] (base, solid gray line), and this scenario enhanced by biotechnology (base + BT, dashed magenta line) or a combination of biotechnology and biomass carbonization (base + BT + BC, dashed blue line). The BT enhancement assumes a productivity increase of 1% of global NPP with a 30-year implementation period and no degradation of the additional C. The BC enhancement assumes carbonization of 1% of global NPP, 50% C conversion efficiency, a C storage half-life of 80 years, no fossil fuel offsets, and full implementation in 30 years.

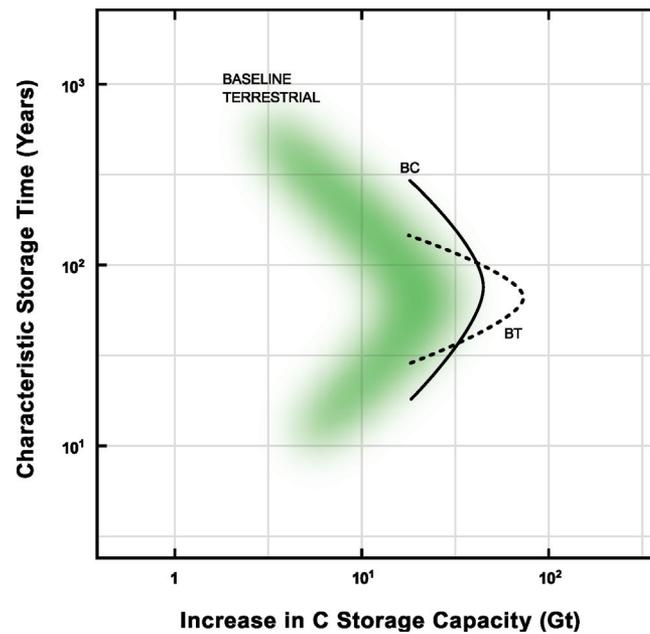


Plate 2. A Lackner plot [Lackner, 2003] showing the projected increase in global storage capacity for terrestrial carbon sequestration achievable by the end of the 21st century with known practices [Thomson *et al.*, 2008] and the associated characteristic storage time (half-life) of the C (baseline terrestrial, green field). Also shown are enhancements to this storage capacity that could be achieved through implementation of biomass carbonization (BC, solid curve) and biotechnology (BT, dashed curve). The BT enhancement assumes a productivity increase of 1% of global NPP with a 30-year implementation period, and no degradation of the additional C. The BC enhancement assumes carbonization of 1% of global NPP, 50% C conversion efficiency, a C storage half-life of 80 years, no fossil fuel offsets, and full implementation in 30 years.

crease in the amount of C stored in soil and the maintenance of C stocks over an extended period. Although inorganic C can be sequestered in some soils [Martens *et al.*, 2005], the rate is slow [van Dam *et al.*, 1997], so we focus on the mechanisms and processes controlling the sequestration of plant C in soil organic matter, including the turnover of plant and mycorrhizal detritus and the incorporation of black C produced by the incomplete combustion of vegetation or fossil fuels. Increased storage of soil organic C (SOC) occurs via increased C inputs, reduced C losses, or both. Carbon lost through decomposition returns CO₂ to the atmosphere, but local losses due to erosion and subsurface transport to groundwater can potentially contribute to C inputs and/or sequestration elsewhere [Jardine *et al.*, 2006; Izaurralde *et al.*, 2007b; Van Oost *et al.*, 2007].

Carbon residence time is a key factor affecting sequestration potential in different soils [Luo *et al.*, 2003]. Even with constant input, conditions or manipulations that increase residence time can effectively sequester C. Thus, strategies to increase both input rates and residence time lead to enhanced sequestration. When the primary control on residence time is limited to decomposition caused by environmental extremes (such as low temperature, low O₂), C inputs may be sequestered seemingly without constraint (e.g., boreal peat deposits). But this C is vulnerable to release from storage if environmental conditions were to moderate [Freeman *et al.*, 2001]. Under more biologically favorable environmental conditions, biochemical alteration and physicochemical protection are the primary mechanisms controlling SOC residence time [Jastrow *et al.*, 2007].

2.3. Nature of Soil C Inputs

SOC inputs are biologically altered to forms that are relatively more resistant to decomposition and, in some cases, are stabilized by sorption on mineral surfaces [Kogel-Knabner, 2002; von Lutzow *et al.*, 2006]. Plant and mycorrhizal litter is fragmented into particulate organic matter, which is then further decomposed. Throughout this process, C assimilated by soil biota and used for growth is similarly transformed after these organisms expire, and their wastes and extracellular products are decomposed. New, larger polymers are then assembled from the smaller molecules released during decomposition. The resulting compounds, historically and collectively known as humus, do not have well-defined composition or structure, but increasing evidence suggests they may consist of dynamic clusters of chemically altered and unaltered compounds held together loosely by hydrophobic interactions and hydrogen bonding [Piccolo, 2002; Sutton and Spósito, 2005]. Many transformations rely heavily on extracellular enzymes excreted by microorganisms. Enzyme

production, activity, and longevity depend on the proximity of substrates, the number of sorption sites on substrates, and enzyme sorption affinities for soil minerals [Shen *et al.*, 2002; Schimel and Weintraub, 2003; George *et al.*, 2005]. Soil pH, redox conditions, and mineralogy are additional factors affecting the reactions [Jastrow *et al.*, 2007].

Many soils also contain significant amounts of pyrogenic C from periodic burning in situ or atmospheric deposition from natural fires and industrial sources [Schmidt and Noack, 2000]. Collectively known as black C, these highly aromatic compounds range from partly charred plant material to char and charcoal to graphite and soot particles. On a relative scale, most black C forms are more recalcitrant than biogenic C in soil, and mean residence times can be hundreds to thousands of years [Glaser *et al.*, 2002; Laird *et al.*, 2008]. However, pyrogenic soil C is by no means inert, and under certain conditions, some forms can be readily oxidized or biochemically degraded [Bird *et al.*, 1999; Hamer *et al.*, 2004].

2.4. Stability and Protection of Soil C

Biochemical alteration of SOC is inhibited, and residence time is significantly increased by interactions with soil minerals. Soil C in various stages of alteration can be protected by an array of molecular associations with mineral surfaces. These largely chemical interactions depend on various factors, including the characteristics of the organic matter, reactivity and specific surface of soil minerals, base-cation status, presence of Fe and Al oxides, pH, and redox conditions [e.g., Sollins *et al.*, 1996; Baldock and Skjemstad, 2000; von Lutzow *et al.*, 2006]. This physicochemical protection is affected by numerous processes, including diffusion of soluble or colloidal C, advection of dispersed particles, mechanical actions of plant and fungal growth, mixing by soil fauna, localized hydration changes, freeze–thaw cycles, and mechanical disturbances, such as tillage [Six *et al.*, 2004; Jastrow *et al.*, 2007]. Potentially greater protection occurs when microbial access to substrates is physically impeded or when soil structural controls on gas exchange and moisture conditions inhibit decomposer activity [Young and Ritz, 2000].

Physical protection of C is intimately tied to processes responsible for creation, turnover, and stabilization of soil aggregates at multiple, often hierarchical, scales [Tisdall and Oades, 1982; Jastrow and Miller, 1998; Six *et al.*, 2004]. Although minimal long-term protection of SOC is afforded by macroaggregates (>250 μm in diameter), their formation slows the mineralization of fresh C inputs, and their turnover rates control the longer-term stabilization of C within microaggregates (50–250 μm) [Six *et al.*, 2004]. Even greater increases in C residence time result from occlusion of C in

silt- and clay-sized aggregates, which likely form within microaggregates [Chenu and Stotzky, 2002; Balabane and Plante, 2004]. Aggregate structure and dynamics also influence soil porosity, which controls the distribution and activity of decomposers and their enzymes [Young and Ritz, 2000; Ekschmitt *et al.*, 2005]. Substrates inside pores with necks $<0.2 \mu\text{m}$ are not accessible to microorganisms, and extracellular enzymes are excluded from nanoscale pores [Zimmerman *et al.*, 2004; Chenu and Plante, 2006]. Interactions between water films and small pore necks can lead to anaerobic spaces within largely aerated aggregates and to localized limitations on decomposer activity [Young and Ritz, 2000].

The various stabilization mechanisms operate at different time scales [von Lutzow *et al.*, 2006], and their interactions lead to a continuum of SOC pools with residence times that can range from less than a year to centuries and even millennia. Further, the effectiveness and relative importance of different mechanisms vary with soil type and other conditions, and current evidence indicates that most stabilization mechanisms can become saturated [Six *et al.*, 2002]. While the unique physical and chemical properties of a given soil may define an overall saturation limit (i.e., additional inputs will not lead to additional sequestration), management methods and environmental conditions likely determine a lower “effective saturation capacity” [Stewart *et al.*, 2007]. The duration and efficiency of sequestration for a given soil are therefore related to its saturation deficit, that is, the difference between initial C stocks and those at its effective saturation capacity [West and Six, 2007].

2.5. Previous Soil Management Results in Present Opportunities

Soils that, over the past decades to centuries, have been depleted of organic matter through inadequate nutrient management, erosion, and excessive tillage provide an opportunity to gain back significant amounts of C with current best-practice management methods, including reduced tillage (especially no-till), erosion control with establishment of perennial vegetation, pasture improvement, introduction of high-yielding crop varieties, crop intensification, crop rotations, weed control, and optimal management of nutrients and water. The analysis by Thomson *et al.* [2008] indicates that terrestrial sequestration with current best practice (base case in Plate 1) can be economically scaled up to achieve sequestration rates of $0.5 \text{ pg C year}^{-1}$ by 2040 with one third accounted for by soil sequestration and the remainder by reforestation. Their global integrated assessment estimates a conservative accumulation of terrestrial sequestration of 23–41 pg C during the next century.

3. PROMISING ADVANCED TECHNOLOGIES FOR C SEQUESTRATION ENHANCEMENT

The contribution of current methods for C sequestration tapers off toward the end of the century in the analysis by Thomson *et al.* [2008], primarily because sequestration methods are adopted on all available land. The only way to boost the terrestrial contribution in later periods would be through development and deployment of advanced technologies. Many new technical developments stand poised to greatly enhance terrestrial sequestration; here we discuss the potential impact of three promising approaches.

3.1. Biotechnology

The postgenomics era provides an unprecedented opportunity to identify genes, enzymes, biochemical pathways, and regulatory networks that underlie rate-limiting steps in C acquisition, transport, and fate—and thereby yield new approaches to enhance terrestrial C sequestration. An investment in plant and microbial biotechnology could harness these new approaches to increase biomass production in agricultural crops and fast-growing trees in managed plantations. Research would focus on targeted improvements in light-use efficiency and photosynthesis [Long *et al.*, 2006], in root growth and acquisition of nutrients [Hirel *et al.*, 2007], and by overcoming constraints imposed on plant productivity by temperature and drought [Tuberosa and Salvi, 2006]. Genome-enabled increases in the production of plant biomass would, all else being equal, translate to enhanced input of C to soils via shoot and root litter, increasing the storage of C in terrestrial ecosystems. Gains in C sequestration are also likely to be realized by understanding how genes and proteins that control the chemical composition of litter could impact the rates and magnitudes of C sequestration.

Attempts to increase rates of photosynthesis through genetic engineering have focused on accomplishing this goal by increasing the total amount of Rubisco in leaves [Suzuki *et al.*, 2007]. Contrary to original expectations, this research has been met with mixed success. Additional research seeks to increase photosynthesis and plant productivity not by modifying the amount but by optimizing the distribution of resources between enzymes of carbon metabolism and/or by altering the kinetic properties of the Rubisco enzyme itself. Theoretical analyses suggest that expressing Rubisco as having either a higher specificity for CO_2 relative to O_2 or a higher maximum catalytic rate of carboxylation per active site could increase photosynthetic carbon gain by 25% or more in C_3 plants [Zhu *et al.*, 2004]. Some have questioned whether substrate specificity of this abominably perplexing

enzyme can be improved [Tcherkez *et al.*, 2006]. As an alternative, similar gains in carbon acquisition could be achieved by altering resource allocation to each of the enzymes involved in the Calvin cycle, photorespiratory metabolism, and sucrose and starch synthesis [Zhu *et al.*, 2007]. In this case, numerical simulations suggest that optimized allocation of resources to specific enzymes could greatly increase carbon gain without an increase in the total nitrogen invested in proteins involved in photosynthetic carbon metabolism. This process illustrates a potential win-win situation; even small gains in plant productivity distributed across a large land area could contribute to meaningful enhancements to carbon sequestration (Plate 2).

Our estimate of the potential enhancement benefit for plant biotechnology to terrestrial carbon sequestration starts with the value for global net primary productivity (NPP), currently about 62 Gt C year⁻¹, of which about 50% is associated with managed tropical forests, temperate forests, and crops [Prentice *et al.*, 2001]. Assuming that half of the managed forest and cropland area benefits from a 10% increase in biomass productivity, the amount of C available for sequestration annually would increase by 1.5 Gt. Further assuming a 30-year adoption period and 70 years at full implementation, the net cumulative increase in C available for sequestration would be about 130 Gt C by 2100 (Plate 1). This is a maximum estimate equivalent to an increase of 2.5% in global NPP. A conservative estimate equivalent to an increase of 1% in global NPP yields a cumulative 53 Gt C by 2100.

Additional C could be sequestered if genome-enabled modification of root turnover times could be achieved. Application of modern system biology approaches and other advanced methodologies to improve fundamental understanding of soil microbial communities and their habitats is another approach that could further the potential for enhanced terrestrial sequestration. Such information could lead to new management practices and production of organic matter materials that optimize microbial activities for the transformation of residues specifically to enhance sequestration.

3.2. Biomass Carbonization

Converting harvestable biomass to more recalcitrant C rather than completely combusting it offers a new approach to terrestrial sequestration as a potential side benefit of bioenergy production. With low-temperature pyrolysis, biomass is carbonized by heating under low-oxygen conditions while producing liquid and gaseous biofuels. Since combustion would not be complete, char-like substances would also be produced. The nature and yield of the solid product obtained

depends on the feedstock and temperatures and pressures employed [Antal and Gronli, 2003; Röhlein, 2006] and ranges from a lignitic material produced under hydrothermal conditions to a series of chars whose porosity increases with temperature from 250° to 800°C. The carbonized biomass could then be incorporated in soil to protect it from further oxidation where, depending on the nature of the product, it may also improve nutrient- and moisture-holding properties while decomposing at a much slower rate than unconverted biomass [Lehmann, 2007a, 2007b].

The contribution of biomass carbonization to C sequestration is potentially large (Plate 2). Four variables need to be considered: sustainable biomass production level, carbonization efficiency, land storage capacity, and characteristic storage time (i.e., the oxidation rate of carbonized C placed in soil). Sims *et al.* [2007] assume that we can sustainably supply 250 EJ year⁻¹ of energy from combustion of biomass. Assuming 80% combustion efficiency, this amount translates to a maximum of 8 Pg C year⁻¹ or approximately 13% of global NPP. A significant fraction of this could be used to produce both energy and char by carbonization processes. Carbonization efficiency ranges from 15% to more than 95% of biomass C and decreases with increasing process temperature. At 400°C, thermodynamic equilibrium calculations suggest theoretical efficiencies of 45–55% [Antal and Gronli, 2003]. The land capacity available to sequester this C can be estimated at between 1.35 and 6.02 Gha, with the lower value representing current cropland [Prentice *et al.*, 2001] and the upper value the land area potentially suitable for cultivation [Eswaran *et al.*, 1999]. Assuming an average soil bulk density of 1.3 Mg m⁻³, an incorporation depth of 0.5 m and an increase in C content of 25 g kg⁻¹ soil, we calculate the total global capacity of soil for char, which is between 235 and 1050 pg C. The characteristic storage time (i.e., half-life) of this C in soil depends greatly on the carbonization process used as well as soil properties, such as temperature and pH, and likely varies for the different forms of C present in the carbonized solid. Nevertheless, it is substantially greater than that of the original biomass [Baldock and Smernik, 2002]. Several lines of evidence [Bird *et al.*, 1999; Cheng *et al.*, 2008] suggest a minimum half-life of 60–100 years for char left at the surface of soil. For chars incorporated into soil, longevity may increase because of lower O₂ levels, based on analogous data from marine sediments [Middelburg *et al.*, 1999; Masiello, 2004].

Soil chars 750–2500 years old have been dated in Terra Preta soils created by aboriginal tribes in the Amazon Basin [Saldarriaga and West, 1986; Glaser *et al.*, 2001]. In work on a substance analogous to a low-temperature char, Chang and Berner [1998, 1999] measured rates of oxidation of a bituminous coal (72% C) in water saturated with

O₂ and obtained a rate constant of 2 pmol O₂ consumed m⁻² s⁻¹. They noted that the CO₂ release rate was only 30–50% of this value, with the remainder of the C going to dissolved organic C. Assuming a 5-m² g⁻¹ specific surface representative of a low-temperature char, a C release rate equivalent to the O₂ consumption rate, and putting it in terms of C only, we obtain a half-life of 132 years. In scenarios we have run that assume a 30-year start-up period for the adoption of biomass carbonization to a level of 1% of global NPP, a characteristic storage time of 80 years yields a net sequestration of 19 pg C in the next century. Fossil C offsets from the energy produced more than double this value. Conversion of a larger fraction of global NPP would increase the net impact proportionally.

3.3. Deep-Soil Sequestration

One third of the 2300 Gt C stored in soils is located at depths greater than 1 m [Batjes, 1996], where because of low oxygen levels and strong stabilization on mineral surfaces, C half-lives are measured in millennia [Trumbore, 1997]. The lower horizons of widespread, mature soils, such as Alfisols, Ultisols, and Oxisols, have a tremendous capacity to adsorb organic C because of their vertical extent, acidic pH, and abundant clay and iron oxide contents [Jardine et al., 1989; Benke et al., 1999; Kong et al., 2005]. Results from sorption experiments [Jardine et al., 2006] suggest that the maximum for increasing this sequestration reservoir is 165 pg C for each meter of soil depth.

Common fertilization methods in agriculture and silviculture may, in fact, already be enhancing the downward vertical spread of organic carbon through the soil profile with the benefit of long-term carbon protection. The application of nutrient sources is typically performed to enhance growth and biomass of crops and trees [Allen et al., 1990]. Forest fertilization is a rapidly expanding management practice being adopted by the forest industry with more than 480,000 ha of commercial forest land in the southeastern United States receiving urea and phosphate amendments in 1998 [North Carolina State Forest Nutrition Cooperative, 1999]. Certain nitrogen sources, particularly urea, not only enhance plant vigor but also interact strongly with soil organic materials [Bengtson, 1970; Ogner, 1972; Kelly, 1981; Kissel et al., 1988; Sen and Chalk, 1994; Ouyang et al., 1999]. Mineralization and dissolution of soil organic matter is stimulated by N fertilizers, which hydrolyze to form alkaline solutions. For example, an extremely high pH (pH 9) sometimes develops in the immediate vicinity of hydrolyzing urea granules, which renders otherwise resistant humus complexes soluble and available for transport through soil horizons. Kelly [1981] observed large increases in mobile dissolved organic

carbon in a forest soil as a result of urea fertilization. Phosphate, another plant-limiting nutrient that may be added to deficient soils, also plays an additive role in the solubilization of soil organic matter. Many of the newer polyphosphates are effective in the degradation of aggregates stabilized by soil organic matter, thus increasing the active surface area of organic carbon [Bengtson, 1970; Kelly, 1981; Fan et al., 1993]. Disaggregation of the soil organic matter by phosphate fertilizers should therefore enhance the effectiveness of urea-induced organic dissolution process. Further, phosphate ions can compete effectively for carbon sorption sites on mineral surfaces, allowing deeper movement of C into the soil profile [Sibanda and Young, 1986; Reemtsma et al., 1999]. Research and demonstration projects are needed to evaluate, reconcile, and optimize the tradeoffs between (1) efforts to stabilize SOM at the surface in order to maintain a sustainable nutrient reserve, soil tilth, and optimum soil structure to support healthy plant growth such as implementation of no-till and afforestation and (2) practices to actively disaggregate and move humus from the surface for the purpose of sending DOC deeper in the profile.

To efficiently use this reservoir, however, ways of moving C from the upper horizons to greater depths are needed. Amendments with lime, urea, and phosphate fertilizers offer a possible approach, although experimental research is needed to prove its ultimate utility. While having been found to be effective in needle leaf forests on Andisol soil [Adams et al., 2005], the effectiveness of amendments to hydrolyze particulate organic matter, is relatively unknown. Second, the energy required to produce the amendments required needs quantification with respect to the amount of C sequestration attained. Finally, the amount of GHGs released under various environmental conditions for each amendment needs determination.

4. IMPLEMENTATION ISSUES

So far, we have summarized research that indicates that there is sufficient potential of terrestrial C sequestration to be economically attractive. Considerable progress has been made on other perceived obstacles. These include saturation and lack of permanence, additionality and leakage, difficulty in measuring and monitoring, and complexity of transactions in carbon trading.

4.1. Saturation and Permanence

C accumulation in the biosphere cannot continue indefinitely, and a new steady-state C content will follow a change in environmental conditions or management. Annual accumulations following reductions in tillage intensity, are

almost always observed in the short term [West and Post, 2002], but may cease after as little as 15 or 20 years. Soil sequestration can continue for much longer following grassland establishment [Shen *et al.*, 2002] and afforestation [Post and Kwon, 2000; Guo and Gifford, 2002; Paul *et al.*, 2002]. Recent literature indicates that even some of the oldest forests continue to sequester C in biomass and soils [Pregitzer and Euskirchen, 2004; Zhou *et al.*, 2006]. The C content of a soil is volatile and is difficult to monitor and measure accurately. It is also heterogeneous and varies over time, even in the absence of changes in land management. Land managers are attracted to the idea of payment for C sequestration but hesitant about liability and concerned about commitment to stewardship in perpetuity.

Dornburg and Marland [2008] have argued that “even temporary sinks put us on a lower path for climate change, a path that will not otherwise be accessible.” Some planned, temporary sinks may turn out to be permanent [Chomitz, 2000], and even if individual sinks turn out to be temporary, the aggregate of sinks will be larger in the presence of incentives than in their absence [Marland *et al.*, 2001]. However, the reversibility of environmental sinks suggests that any economic incentives provided for C uptake must be reversible if the C is subsequently released, and there is now literature [Murray *et al.*, 2007] on accounting approaches focused on discounted credits and credit rental that take reversibility into consideration. West *et al.* [2004] describe an accounting approach that models the average consequences of a change in agriculture methods so that credits could be based on management methods rather than on extensive measurements. These kinds of approaches are all based on providing incentives for protecting the atmosphere and biosphere while preventing land managers from being overwhelmed with program management or gaming the system of incentives. Temporary or rented credits are, of course, less valuable than permanent credits, and Murray *et al.* [2007] and Kim *et al.* [2008], for example, have begun to evaluate the value of temporary credits. They find that temporary credits may be worth less than half as much as permanent credits but that credits in the biosphere may still be inexpensive in comparison to other mitigation options.

4.2. Additionality and Leakage

These issues are concerned with giving mitigation credits to activities that would have occurred in any event or giving credits for an activity in one time or place that is completely or partially offset by activities stimulated elsewhere. As interest and tentative experience with C mitigation activities accumulate, practitioners are beginning to develop means to predict, minimize, and account for these losses

of benefits. The key is establishing appropriate baselines: What would have happened in the absence of a mitigation activity? Brown *et al.* [2007], for example, collect a variety of experiences and analyses of baselines and leakage. Baselines might be either project-specific or regional. Leakage of carbon benefits can be controlled to some extent but varies widely among activity types and can run to nearly 100% of benefits for some forest protection activities, and yet analyses like that of Sathaye *et al.* [2006] suggest that protecting carbon that is already in the biosphere provides one of the most important opportunities. Leakage is likely to be less for changes in agricultural methods, although an increase in agricultural productivity could be reflected in a reduction in the total amount of land being tilled. Although leakage may represent a significant reduction in the net sequestration benefit, it is possible to minimize the effect with good project design [Westerling *et al.*, 2006; Birdsey *et al.*, 2007; Intergovernmental Panel on Climate Change, 2007; Nabuurs *et al.*, 2007]. Given the concerns, Smith and Scherr [2000, 2003] raised the concept of proportional additionality and discussed procedures to (1) estimate the baseline proportion of activity that would have happened in the absence of a mitigation program and (2) provide incentives only for the increment above the baseline [Pacala and Socolow, 2004; Birdsey *et al.*, 2006; Thomson *et al.*, 2008].

The radiative forcing of C in the atmosphere can be balanced or augmented by changes in the surface albedo and latent heat transfer. For an increase in tree cover at high latitudes, especially in areas with extensive snow cover, the warming effect of darkening the surface is suggested to be larger than the cooling effect of taking CO₂ from the atmosphere [Betts, 2000; Bala *et al.*, 2007]. This can be regarded as a form of leakage that must also be considered. Current understanding is that the global average C cycle effect of afforestation in midlatitudes would be attenuated by changes in the surface energy balance, while in the tropics, the effects of afforestation on latent heat transfer would augment the cooling effect of removing CO₂ from the atmosphere. So it matters very much where sequestration takes place. Within the broad latitudinal bands of the global albedo studies, there is significant variability in radiative effects and sequestration potential that has not yet been quantified. Afforestation could be targeted to areas and forest types that resulted in net atmospheric cooling. An important consideration is that C sequestration in soils will have much less impact on surface albedo. The net effect on the surface energy balance of C sequestration in soils is yet to be evaluated. Also, we are becoming increasingly aware of the importance and distinction between changes in the global average climate and in the local and regional climates [Marland *et al.*, 2003].

4.3. Measuring and Monitoring

Soil sampling followed by determination of organic carbon concentration by dry combustion and calculation of SOC mass from bulk density constitutes the standard methodology for measuring and monitoring soil carbon sequestration [Post *et al.*, 2001; Izaurralde and Rice, 2006]. Significant progress has been achieved during the past 10 years toward refining, enhancing, and adapting this method for measuring and monitoring soil carbon sequestration at field and regional scales. It is now possible to measure soil carbon changes as small as 1 Mg C ha⁻¹ in a period of 3 years [McConkey *et al.*, 2000] or estimate it with the use of simple or complex simulation models [Paustian *et al.*, 1997; Smith *et al.*, 2007]. Despite these successes, there is recognition that fast, accurate, and cost-effective methods will be needed to measure and monitor soil carbon sequestration in large-scale projects deployed in different regions of the world. In terms of field technologies, several instruments have been developed or recently modified for in situ measurements of soil carbon. These include laser-induced breakdown spectroscopy [Cremers *et al.*, 2001], inelastic neutron scattering [Wielopolski *et al.*, 2000], and diffuse reflectance IR spectroscopy in the near-infrared and mid-infrared wavelength regions, ranges 400–2500 and 2500–25,000 nm, respectively [McCarty *et al.*, 2002; Christy *et al.*, 2006]. For additional discussion, also see Brown and Sampson, this volume. Comparisons of measurements using these technologies to make soil C estimates at several field sites side by side have been producing results with precision and adaptability to field conditions comparable to sampling followed by laboratory analysis [Izaurralde *et al.*, 2007a].

Measurement and monitoring approaches using current or advanced methods need to be integrated to field-level and regional scales using computer simulation and remote sensing on some dynamic and geographically appropriate basis [Paustian *et al.*, 1997; Smith *et al.*, 2007]. Mooney *et al.* [2004] calculate that the costs of such schemes for agricultural soil C sequestration are approximately 3% of the offset value, or around \$3 per ton C. Additional bureaucratic costs of establishing measurement and monitoring could add another \$4–6 per ton C [Antinori and Sathaye, 2007]. Procedures for monitoring and verification in forests have been developed but are potentially costly because forest ecosystems have multiple C pools. Some pools are relatively easy to measure and monitor, such as tree boles, while understory biomass and litter and forest soils can be more difficult. A variety of approaches are described in forest C accounting protocols for the United States and in pilot projects implemented by nongovernmental organizations [Izaurralde and Rice, 2006].

4.4. Transaction Complexity

A final economic issue relevant to agricultural soils is that individual land parcels are often too small to be meaningful by themselves and so must be aggregated into large-enough groups to represent significant quantities of C. For example, a 100,000-metric ton (t) year⁻¹ tillage-based contract would require about 50,000 ha (about 260 farms at U.S. average farm size) with an average C offset rate of 0.5 t C ha⁻¹ [West and Post, 2002]. Several organizations, Greenhouse Emissions Management Consortium and DeltaCarbon in Canada and the United States, respectively, have been experimenting with voluntary programs through the Chicago Climate Exchange to perform this function. For additional information, Roed-Larsen and Flach, this volume, discuss verification and accreditation schemes for climate-related transactions.

5. ANCILLARY BENEFITS AND RISKS

Environmental issues attendant to other C sequestration approaches, such as deep geologic or ocean sequestration, are less of an issue with terrestrial sequestration. Safety and environmental acceptability are generally understood to be positive aspects of terrestrial sequestration. There are also situations in which co-benefits to C sequestration induce activity. Silvicultural methods to increase C sequestration may be compatible with other ecosystem management goals, such as restoration of habitat and biodiversity. Maintaining and enhancing ecosystem services, such as habitat or water quality, can be considered in strategies to manage forest C. The same holds for agricultural soils for which C sequestration can serve the dual roles of GHG mitigation while helping adapt agroecosystems to climate change. Increased levels of soil organic matter bring a multitude of benefits, including enhanced crop productivity leading to enhanced food security, nutrient storage, soil tilth, infiltration and water-holding capacity, and pH buffering capacity [Zvomuya *et al.*, 2008]. For example, loam soils with increasing levels of SOC from 1% to 4% could increase their total porosity by an average of 0.06 m³ m⁻³, improve soil water retention, and possibly reduce N₂O emissions through the development of a more aerobic soil environment. These improvements will make soil processes more resilient to climate change.

A substantial sink for carbon in soils may derive from the application of the sequestration methods discussed here. Applications of fertilizer, irrigation, are important forestry and agricultural practices, but the complexities of a full accounting of the GHG emissions and sinks associated with these activities must be considered [Schlesinger, 1999].

There are potentially significant risks associated with each of the three advanced technologies discussed above. Issues of concern for biotechnology include the capability of the genetically modified organisms (GMOs) to escape and potentially introduce the engineered genes into wild populations, the persistence of the gene after the genetically modified organism has been deployed, the susceptibility of nontarget organisms (e.g., insects which are not pests) to the gene product, the stability of the gene,; the reduction in the spectrum of other plants including loss of biodiversity, and increased use of chemicals in agriculture. The environmental safety aspects of GM crops vary considerably according to local conditions [Carr and Levidow, 2000; Jank and Gaugitsch, 2001]. The short- and long-term impact of biochar additions and organic matter treatments with hydrolyzing agents are unknown and may have negative impacts on nutrient availability and soil structure.

6. SUMMARY AND RESEARCH NEEDS

Considerable scientific and technical progress has been made during the more than 10 years since the drafting of the Kyoto Protocol making terrestrial C sequestration and soil C sequestration, in particular, more compelling. Terrestrial C sequestration could be an important and immediate option for creditable GHG mitigation. The cost of attaining atmospheric CO₂ stabilization will be significantly reduced even if current techniques of terrestrial C sequestration are more aggressively utilized. Jakeman and Fisher [2006] examined the cost of attainment with and without a terrestrial contribution and found that incorporation of current terrestrial sequestration techniques reduces overall stabilization costs by 30 to 57%. McCarl and Sands [2007] found terrestrial contributions to be especially important in early decades for reducing costs.

Including additional enhanced sequestration technologies, e.g., the three advanced methods presented in this chapter, reduces CO₂ stabilization costs even further [Thomson et al., 2008]. Additional research on process-level understanding, measurement, implementation, and integrated assessment will increase terrestrial C sequestration further and, at the same time, provide significant economic and environmental benefits [Jacobs and Hendrey, 1999]. In particular, understanding of factors controlling C allocation mechanisms in plants and soils is needed to predict the fate of C in different ecosystems. Modern, interdisciplinary system science approaches to plant and microbial biology are needed to develop predictive, mechanistic models of processes responsible for the transfer of C from litter and roots to long-lived soil pools. Basic knowledge of these mechanisms as well as those controlling vertical movement of C in soil, formation

and the role of inorganic C, and landscape-scale erosional processes, among others, would increase confidence for those responsible for project implementation. Inexpensive, innovative measurement technologies are required. Research needs related to implementation and assessment should address economic and social issues, such as permanence, cost and the associated value of environmental services, and improved understanding of the role of terrestrial sequestration over time as part of a portfolio of strategies for GHG mitigation. Here improvement of economic models that build on better understanding of fundamental scientific processes is needed. In the end, however, despite these central research needs, the current body of scientific information should be sufficient to overcome existing implementation issues. Indeed, pursuit of terrestrial C sequestration may give us what Dyson long ago referred to as a way to address “the rise in CO₂ emissions within a few years by a means less drastic than the shutdown of industrialized civilization....” [Dyson, 1977].

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