Managing Soils and Ecosystems for Mitigating Anthropogenic Carbon Emissions and Advancing Global Food Security

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Soil carbon (C) is a dynamic and integral part of the global C cycle. It has been a source of atmospheric carbon dioxide (CO_2) since the dawn of settled agriculture, depleting more than 320 billion metric tons (Pg) from the terrestrial pool, 78±12 Pg of which comes from soil. In comparison, approximately 292 Pg C have been emitted through fossil-fuel combustion since about 1750. However, terrestrial pools can act as a sink for as much as 50 parts per million of atmospheric CO_2 for 100 to 150 years. The technical sink capacity of US soils is 0.288 Pg C per year; Earth's terrestrial biosphere can act as a sink for up to 3.8 Pg C per year. The economic potential of C storage depends on its costs and cobenefits, such as global food security, water quality, and soil biodiversity. Therefore, optimally managing the soil C pool must be the basis of any strategy to improve and sustain agronomic production, especially in developing countries.

Keywords: climate change, environment quality, soil management, food security

Anthropogenic perturbation of the global carbon cycle (GCC) during the carbon (C) age has become one of the major threats to civilization. The atmospheric concentration of carbon dioxide (CO_2), 383 parts per million (ppm) in 2008 and increasing at the rate of about 2.2 ppm per year, is strongly coupled with the climate: CO_2 influences climate through the greenhouse effect, and climate moderates CO_2 concentration by altering the GCC, including its pools and fluxes. The observed and predicted increases in global temperature are attributed more to greater atmospheric abundance of CO_2 and other active gases than to solar variability or other natural factors. Emission of 4 petagrams (Pg, 1 billion metric ton = 1 gigaton) C through anthropogenic activities increases the atmospheric concentration of CO_2 by approximately 1 ppm. Abrupt climate change strongly affects the GCC because it changes plant and animal phenology, and increases the intensity and frequency of extreme weather events such as Hurricanes Katrina and Rita, causing strong impacts on forests, tree regeneration, and species composition; the loss of wetlands; and altering the emission of CO_2 from decomposing biomass.

There are five global C pools (figure 1): (1) oceanic, at 39×10^3 Pg and increasing at 2.3 Pg C per year; (2) fossil fuel, at 5×10^3 to 10×10^3 Pg, mined and combusted at 8 Pg C per year; (3) pedologic, at 2.5×10^3 Pg to 1-meter depth composed of 1.55×10^3 Pg of soil organic C (SOC) and 950 Pg of soil inorganic C (SIC); (4) atmospheric, at 780 Pg and increasing at 4 Pg C per year; and (5) the biotic pool, comprising 560 Pg of live biomass and 60 Pg of detritus material (Canadell et al. 2007a, GCP 2008). In addition, the marine sediments contain 20×10^6 Pg C that is not being circulated. The pedologic and biotic pools combined, called the terrestrial C pool, equals 3120 Pg; this is about four times the atmospheric pool. The pedologic pool (2.5×10^3 Pg) is 3.2 times the atmospheric pool. The terrestrial pools have been drastically altered by human activities (deforestation, biomass burning, soil cultivation, drainage of peatlands) since the dawn of settled agriculture about 10,000 years ago (Ruddiman 2003), with an estimated depletion of more than 320 Pg C. However, in contrast to the fossil-fuel pool, which has been a major source of CO_2 since about 1750, terrestrial C pools can be either a source of or sink for atmospheric CO_2, depending on land use and management. The objective of this article is to describe the potential and challenges to making terrestrial pools sinks for atmospheric CO_2 through understanding and management of controls, underlying processes, and the factors and causes that affect C dynamics in terrestrial ecosystems.

Terrestrial pools as source of atmospheric CO_2

The increase in the human population—from about 2 million to 10 million at the development of agriculture between 10 and 12 millennia ago, to 6.7 billion in 2009, and the projected 9.2 billion by 2050—has resulted in large-scale

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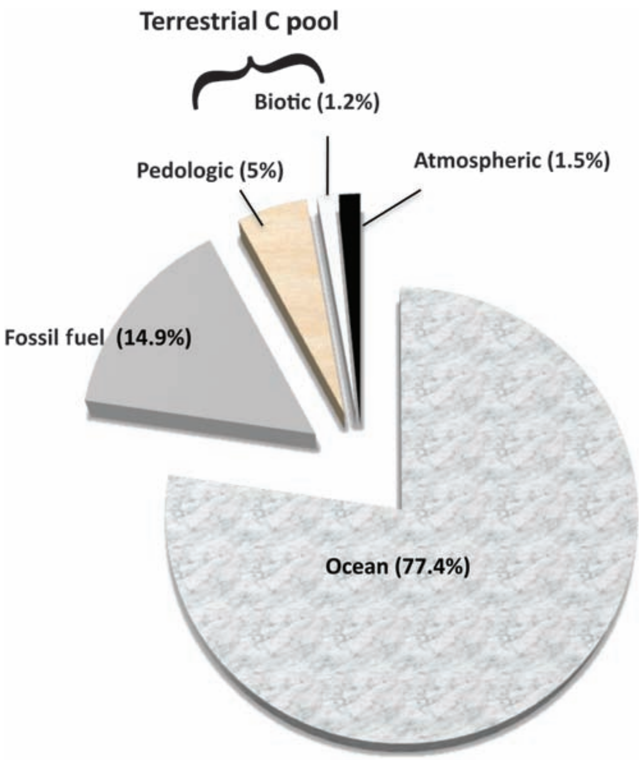

Figure 1. The five interconnected global carbon pools contain a total of 50,400 billion metric tons (Petagrams, Pg) carbon (C). These pools interact with one another, constituting the global C cycle. The terrestrial pool, pedologic and biotic combined, makes up 6.2% (3120 Pg) of the total C in circulation. In addition, marine sediments contain approximately 20 × 10^6 Pg C, and carbonate rocks contain more than 65 × 10^6 Pg, which is not in circulation. The atmospheric C pool is small enough that it is easily overwhelmed by anthropogenic-induced perturbation (burning of fossil fuels) of the terrestrial pools, as has been the case since about 1850.

conversion of natural ecosystems to agricultural land use. Between 1700 and the 1990s, land-use change resulted in the conversion of 1.135 billion hectares (Bha) of forest and woodlands to croplands, and 0.669 Bha of savanna, grassland, and steppe to croplands. The area of cropland has increased from 0.265 Bha to 1.5 Bha, and area of grazing land from 0.580 Bha to 3.45 Bha (Foley et al. 2005). Resulting from drastic land-use conversion, human appropriation of net primary production (NPP) is estimated at 15.6 Pg per year, which is 23.8% of the potential NPP. Of this, 53% is contributed by harvest, 40% by land-use-induced productivity changes, and 7% by human-induced f res (Haberl et al. 2007). Such large-scale conversions of natural ecosystems have already resulted in historically large emissions of C into the atmosphere. Ruddiman (2003) suggested that from the dawn of settled agriculture (6000 years before present) to today, approximately 320 Pg of C may have been depleted through land-use conversion. Canadell and colleagues (2007b) estimated that between 124 and 158 Pg C may have been depleted from land-use conversion since the 1850s. Lewis and colleagues (2005) estimated that between 1750 and 2000, global land-use change released approximately 180 Pg C to the atmosphere, with 60% coming from the tropics. Of the total terrestrial depletion from the biosphere, approximately 78±12 Pg C may have been depleted from Earth's soils (Lal 2004a). In comparison, CO_2 emissions from fossil-fuel combustion from between 1750 and 2002 are estimated at 292 Pg C; comprising 148 Pg from coal, 105 Pg from oil, and 39 Pg from gas (Holdren 2008). Under business as usual scenarios an additional 200 Pg of emissions from fossil-fuel combustion are expected from 2003–2030, 137 Pg of which will be from coal, 61 Pg from gas, and 2 Pg from oil (Holdren 2008).

Soil degradation and desertification

Carbon emissions from terrestrial ecosystems are exacerbated by soil degradation and desertif cation. Soil degradation refers to a decline in soil quality or its capacity to produce economic goods and provide ecosystem services. Desertif cation refers to soil degradation in arid environments. Dregne (1998) estimated that 3.592 Bha of land (soil and vegetation) have been affected by desertif cation, of which 1.965 Bha may be a result of soil degradation (Oldeman and Van Lynden 1998). Eswaran and colleagues (2001) estimated desertif cation tension zones affecting a total land area of 4.32 Bha, of which 1.170 Bha are in regions with a high population density of more than 41 persons per square kilometer. Bai and colleagues (2008) estimated that land degradation affects 3.5 Bha, or 23.5% of gloabal land area. An additional 0.003 Bha of land per year is required to accommodate and provide basic necessities for the annual population growth (70 million to 80 million per year).

Agricultural soils contain 25% to 75% less SOC than their counterparts in undisturbed or natural ecosystems (Lal 2004a). Land-use conversion, drainage, soil tillage, biomass burning, and other farm operations create a negative C budget because C losses exceed gains and attenuate the rate of mineralization of soil organic matter (SOM). Degraded and desertif ed soils have lower SOC than undegraded or slightly degraded soils. Low NPP in degraded soils leads to a low input of biomass C that creates a negative C budget. A lower SOC pool in the root zone degrades soil quality, which exacerbates soil erosion and reduces the NPP.

Erosional processes preferentially remove the light SOC fraction, and the soil C transported into depressional sites and aquatic ecosystems may create a C sink. Of the 1.9 Pg C per year carried into inland waters from terrestrial sources, 0.8 Pg is emitted into the atmosphere, and the remaining C is buried in aquatic sediments (Cole et al. 2007). Factors leading to erosion-induced C sinks include reduced decomposition in depositional sites and dynamic replacement of eroded C (Van Oost et al. 2007). Whether soil erosion is a source or sink of atmospheric CO_2 can perhaps be resolved through an ecogeomorphologic perspective on SOC movement through the landscape. Because degraded soils and desertified lands are highly depleted of their ecosystem C pool, restoration of these ecosystems can create a large C-sink capacity (Lal 2004a).

The carbonization of terrestrial biosphere

It is time to move beyond the Copenhagen Accord (Lal 2010) and reduce the risks of abrupt climate change (ACC) by carbonization of the terrestrial biosphere before it reaches the tipping point. The transfer of atmospheric CO_2 into biomass through photosynthesis and the retention of part of the NPP into the biosphere is called "carbonization." Similarly, conversion of a part of the NPP into stable humic substances and secondary carbonates is called terrestrial sequestration; this method can be compared with burial into oceanic ecosystems (oceanic sequestration) and engineering techniques for the injection of industrially emitted CO_2 into geologic strata (geologic sequestration).

The global maximum or potential terrestrial C-sink capacity is approximately equivalent to the historic depletion of C from the terrestrial biosphere. This sink capacity can be filled through a strategic carbonization of terrestrial ecosystems by enhancing the C pools in soils and biota. A field of corn (*Zea mays* L.) captures about 400 times as much C (12.5×10^{-9} Pg of biomass composed of stover, roots, grains, etc., and containing 5×10^{-9} Pg C per hectare) as there is annual increase (12.5×10^{-12} Pg of C from an increment of 2 ppm of CO_2 per year) of man-made atmospheric CO_2 in the entire column of air above that field from ground to the upper reaches of the atmosphere (figure 2). However, most of the biomass C thus photosynthesized is respired back into the atmosphere. Nonetheless, the capacity of vegetation to photosynthesize atmospheric CO_2 can be used as a strategy to create a positive C budget ($C_{input} > C_{output}$) while also enhancing ecosystem services.

There is a close link between SOC pools and those in the above- and belowground biomass because of the strong

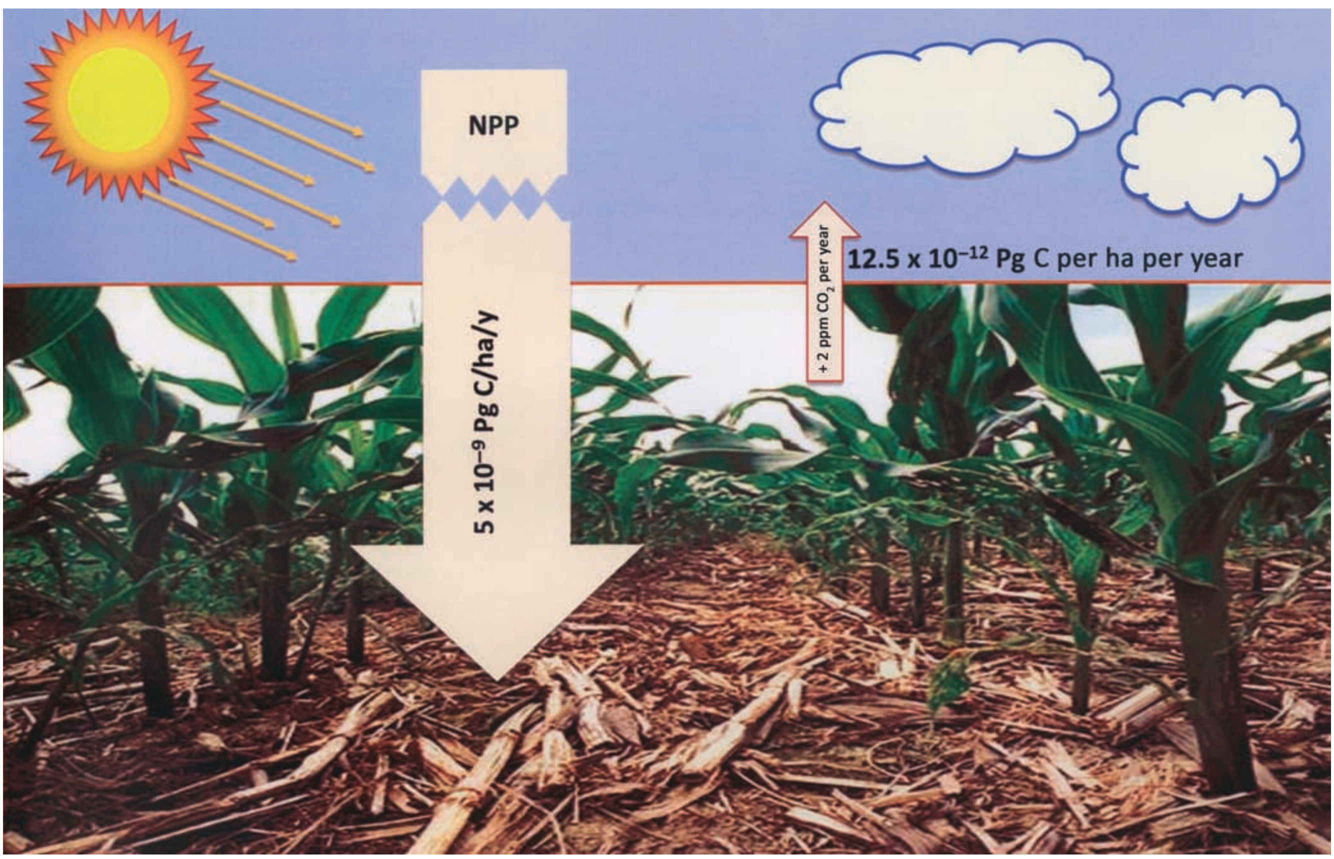

Figure 2. The net primary productivity (NPP) of a field of corn (5×10^{-9} billion metric tons carbon [Pg C] per hectare [ha] per year) is 400 times more than the annual increase (12.5×10^{-12} Pg C per ha per year) in atmospheric concentration of carbon dioxide at the rate of 2 parts per million per year. The NPP of agricultural fields is strongly influenced by management: no-till versus plow tillage, the use of fertilizers and manures or soil amendments, water table management and irrigation, the use of improved or genetically modified craps versus traditional varieties, and cropping and farming systems including agroforestry. The NPP value shown in this figure is not corrected for the C emissions of farm operations, which differ among no-till (shown in this photograph) and plow tillage systems. Refer to table 1 for the life-cycle analyses of both tillage systems.

positive correlation between NPP and the SOC pool. Recarbonization of the terrestrial biosphere necessitates an understanding of the ecosystem properties determined by the interactions between slow- and fast-moving processes, and those that operate across a range of scales from the molecular to the soilscape or watershed level. These processes are nonlinear and highly variable, they strengthen linkages among biotic and abiotic reactions, and they enhance the ecosystem resilience.

Carbonization of terrestrial ecosystems (terrestrial sequestration) can be set in motion by a two-pronged approach: (1) increasing NPP, and (2) sequestering C trapped in NPP.

Restoring degraded and desertified soils. There are several options for increasing NPP, among them, restoring degraded soils and ecosystems has a high technical potential. With 3.5 Bha of degraded and desertif ed lands, increasing NPP and trapping some of the increase in the recalcitrant (resistant to decomposition) form in soil humus and biomass could have an important impact on the atmospheric concentration of CO_2. The goal is to develop land-use and -management strategies that will somehow permanently lock up and remove CO_2 from the natural C-cycling process. Desertif cation control and the restoration of degraded soils and ecosystems are important options for sequestering C and mitigating ACC. Conversion of marginal agricultural lands to forest plantations and other perennial land uses is another option.

Managing peatlands and restoration of wetlands. Peatlands occupy 0.416 Bha worldwide, and 80% of that area is in the temperate cold climate of the Northern Hemisphere. Peatlands contain about 400 Pg of C (MacDonald et al. 2006), have been sinks for atmospheric CO_2, and have contributed to global cooling for millennia. The accumulated peat C is equivalent to about 100 to 200 ppm of atmospheric CO_2, or 25% to 50% of the present atmospheric C pool. Peatlands in the Northern Hemisphere have been a C sink at the rate of 2×10^{-10} to 3×10^{-10} Pg C per hectare (ha) per year for millennia. Up to 26% of all terrestrial C has accumulated in peatlands since the last glacial maximum (Smith et al. 2004). Peat soil prof les, which are anoxic below 0.1 to 0.5 meters (m), are subject to methanogensis. Thus, northern peatlands are also a source of methane (CH_4), with varying estimates of 0.01 to 0.045 Pg CH_4 per year. Limiting the cultivation of peatlands and the restoration of wetlands by inundation and blocking drainage is essential for creating a positive C budget, and reducing methanogenesis may enhance the net benef ts of wetlands.

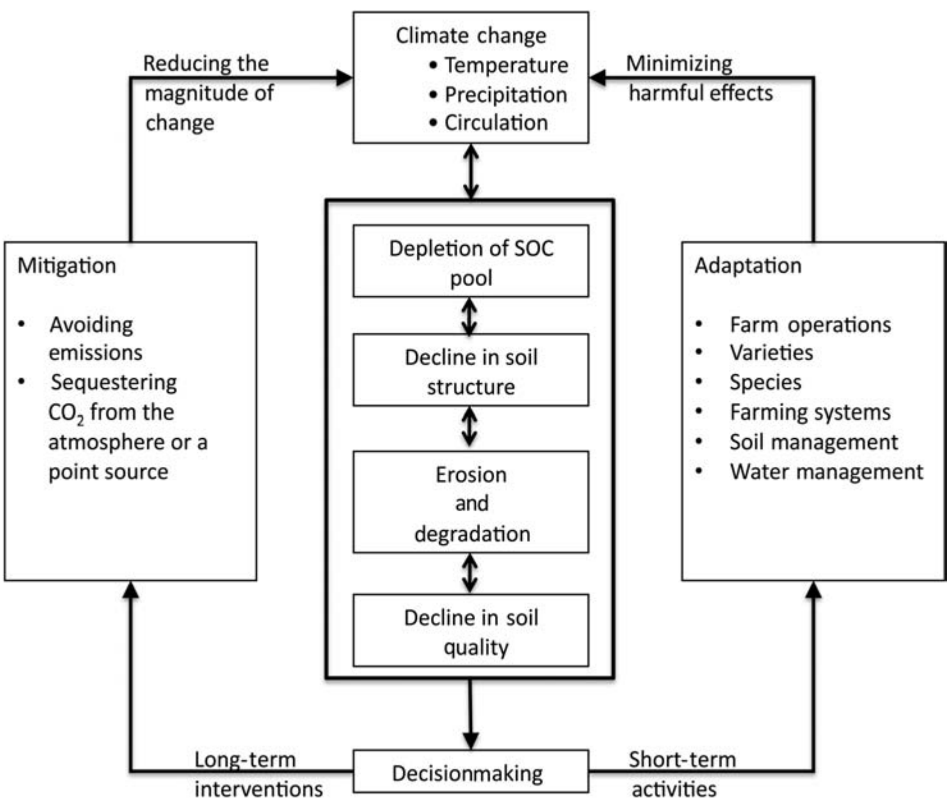

Figure 3. Impact, adaptation, and mitigation of climate change on agricultural ecosystems. Mitigation strategies aim at reducing the magnitude of change by avoiding emissions and sequestrating carbon dioxide. In comparison, adaptation strategies aim at minimizing the harmful effects of abrupt climate change. Adaptation options include recommended and innovative practices of soil and crop management. Mitigation strategies operate over a longer time scale than adaptation strategies. Soil degradation caused by erosion and other processes depletes the soil organic carbon (SOC) pool, and exacerbates emissions of carbon dioxide and other greenhouse gases.

Burial of biomass. Various authors have proposed massive schemes of burying plant products such as wood or crop residues (Karlen et al. 2009). However, neither burial of trees in open mine pits nor burial of crop residues in the deep oceans seems ecologically feasible, economically manageable, or logistically practicable. The energy (C) cost of harvest, transport, and other operations cannot be ignored. The idea of burying crops in the ocean is also based on the assumption that these residues are a waste product, with little if any benef cial impacts on soil quality, but crop residues are important to managing soil quality, recycling nutrients, conserving soil and water, and advancing global food security (Karlen et al. 2009).

Carbon sequestration in cropland soils. Judicious management of agricultural soils is an important strategy for enhancing their C pool (figure 3). World cropland soils cover about 1.5 Bha and have a large C sink capacity created through historic depletion of the SOC pool. Agricultural soils, if not managed with recommended management practices, can be a major source of atmospheric CO_2 and N_2O. The additional land area required to meet the increase in food demand for the growing world population and associated changes in dietary preferences may be drastically reduced if the average cereal grain yield of 2.7 tons per hectare in 2000 can be increased by 63% by 2025, and by 122% by 2050 (Wild 2003). The dramatic increase in agronomic production that occurred between 1960 and 2000 was attributed to yield improvement (78%) from genetic and agronomic management, expansion in cropland area (15%), and an increase in cropping intensity (7%). Similarly, the required increase in agronomic production between 2000 and 2030 may occur through yield improvement (70%), expansion of cropland area (20%), and greater cropping intensity (10%; Vergé et al. 2007). The adoption of biotech crops can reduce the hidden C costs of pesticides and increase NPP. Therefore, management of the SOC pool is an important aim for policymakers trying to achieve adaptation to and mitigation of ACC (Hansen et al. 2008), while also advancing global food security (Lal 2004a).

Cropland soils can be used to mitigate (figure 4) and adapt (figure 5) to ACC. Strategies of emission avoidance include those that increase the use efficiency of inputs, such as conservation agriculture, guided traffic, precision farming, or soil-specific management, and that create a positive C budget. Soil restoration and remediation are also important to ACC mitigation (figure 4). Adaptation to climate change can be enhanced through mulch farming, agroforestry systems, integrated nutrient management (INM), and soil-water conservation. The strategy is to create positive C and nutrient (nitrogen [N], phosphorus [P], sulfur [S]) budgets in conjunction with judicious management of soil water through its conservation, harvesting, and recycling (drip subirrigation [DSI], fertigation, etc.). Conservation tillage, in conjunction with mulching, growing cover crops, and the use of INM, is especially relevant in well-drained and undulating soils that are prone to crusting and accelerated erosion. Already adopted on about 0.1 Bha worldwide and increasing in area, conservation tillage and no-till (NT) farming are appropriately termed "the quiet revolution."

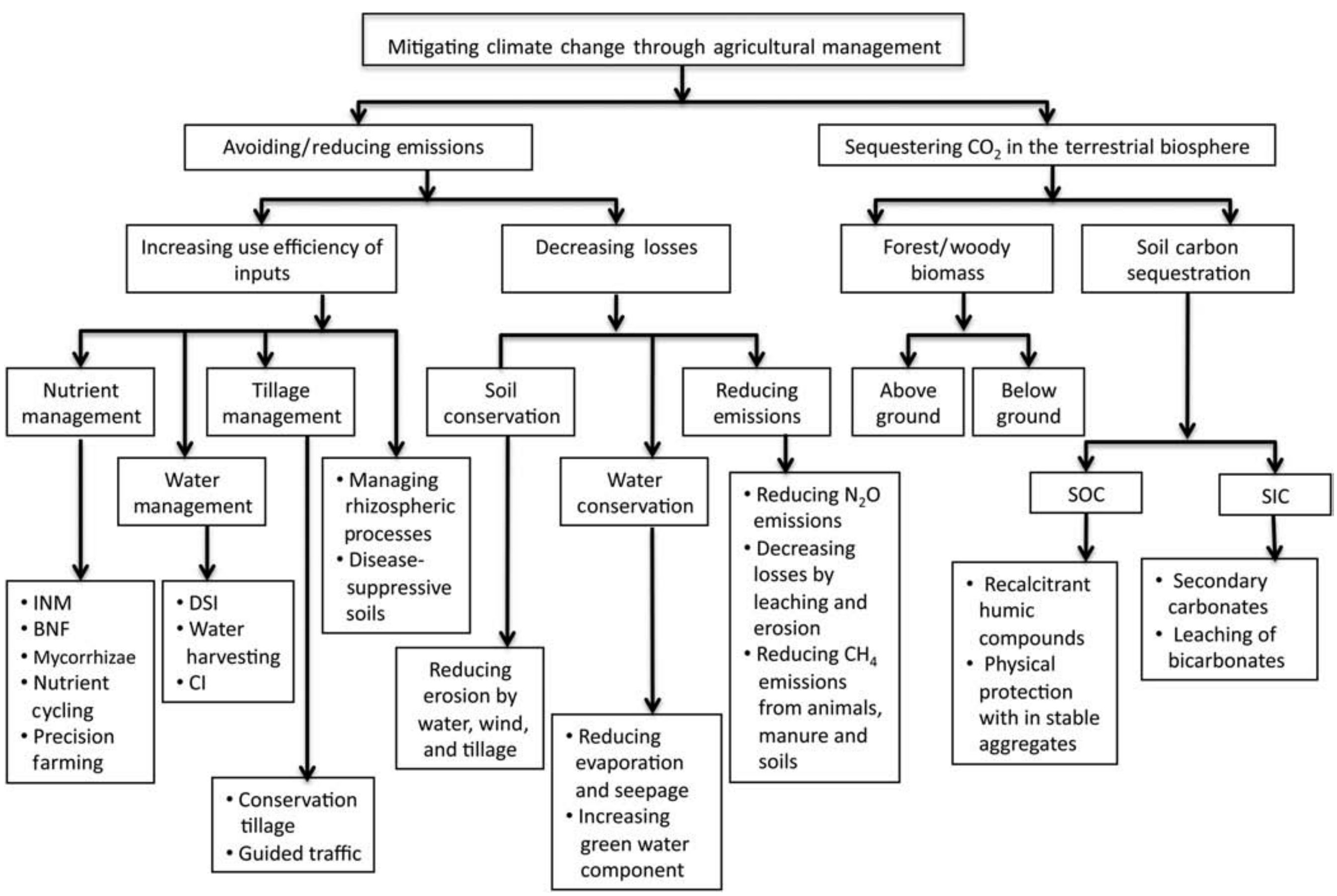

Figure 4. Mitigation of climate change through reduction in emissions from agricultural operations, and sequestration of carbon within the terrestrial biosphere comprising of soils and biota. BNF, biological nitrogen fixation; CI, condensation irrigation; DSI, drip sub-irrigation; INM, integrated nutrient management; SIC, soil inorganic carbon; SOC, soil organic carbon.

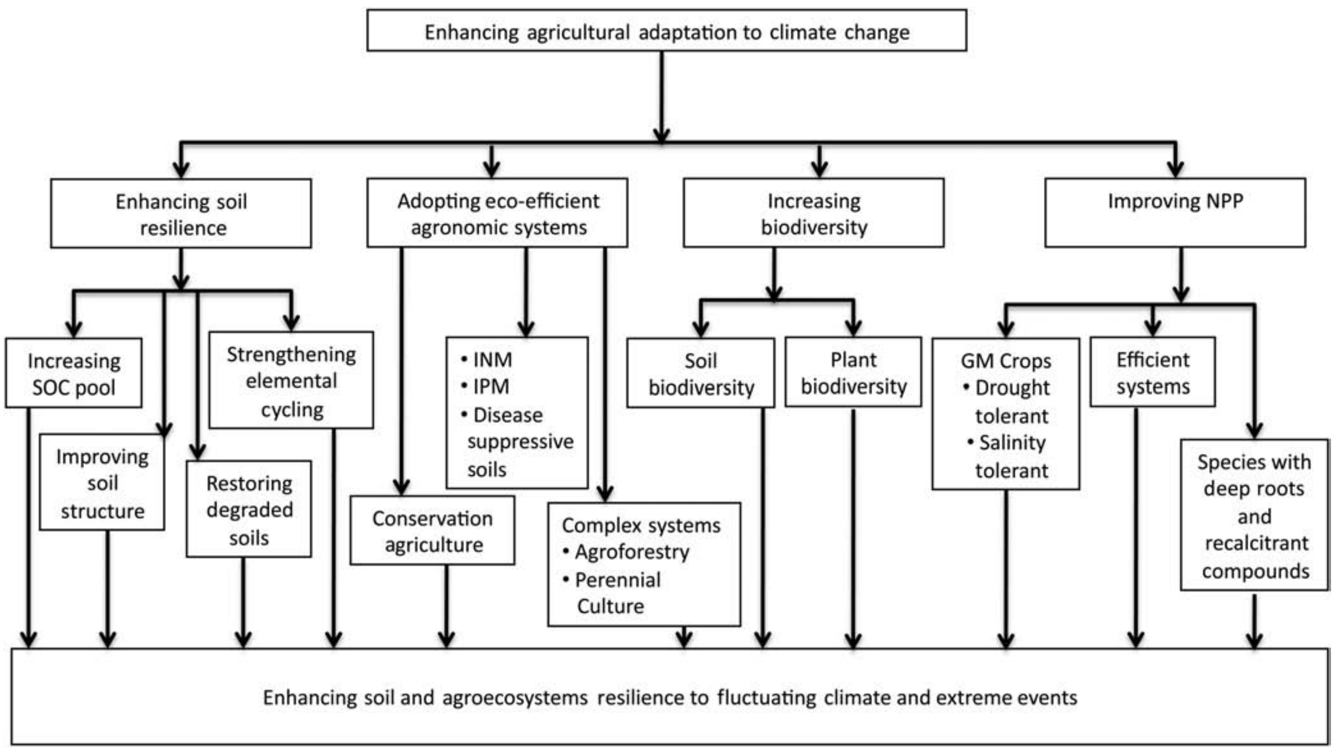

Figure 5. Agricultural strategies for adaptation to climate change include those that produce large biomass even under conditions of higher temperatures, lower effective precipitation, increases in the frequency and intensity of extreme events, and increased pressure of pests and pathogens. GM, genetically modified; INM, integrated nutrient management; IPM, integrated pest management; NPP, net primary production; SOC, soil organic carbon.

Energy-use eff ciency, even with herbicides for weed control, is often higher with NT farming than with conventional tillage. The rate and total magnitude of SOC sequestration, an average of about 0.55×10^{-9} Pg C per ha per year on a global scale (West and Post 2002), depend on residue management and recycling organics; climate regime; N application; and soil properties, including the antecedent SOC pool. No-till management is not effective in some soils and climates, and it may be appropriate to measure the SOC pool beyond the surface 0.2-m depth. Furthermore, the potential to mitigate global warming with NT management is highly variable and complex, and can be realized only when NT farming is practiced over the long term (Six et al. 2004). In some soils, nitrous oxide (N_2O) emission is higher under NT than under plow tillage (PT) (Steinbach and Alvarez 2006); in others, tillage treatment has little effect on N_2O emission.

Increases in temperature and decreases in effective soil moisture as a result of ACC may alter weed competition, growing season duration, and crop (e.g., wheat, rice) sensitivity to high temperatures at the grain-f lling stage, and may increase the soil respiration rate and decomposition of SOM with attendant shifts among labile and recalcitrant C pools. Pacala and Socolow's estimate (2004) of sequestering 1 Pg C per year by converting from PT to NT farming between 2004 and 2050 is rather optimistic; Smith and colleagues (2008) reported lower C sequestration estimates from conversion from PT to NT. The technical potential of about 0.4 to 1.2 Pg C per year on cropland soils (Lal 2004a) is based on the assumption that crop residues are recycled as mulch, and that adequate nutrients are available for the humif cation of biomass C. Using cyanobacteria for direct conversion of CO_2 to isobutanol and isobuteraldehyde (biofuel) is a better strategy than using crop residues for cellulosic ethanol (Sheehan 2009): Hidden C costs come from the application of fertilizers and other farm operations (Lal 2004b), and the application of N can enhance the rate of mineralization in some soils. The goal of soil management is to enhance C gains and reduce C losses through understanding and managing the processes that govern the dynamic equilibrium of the SOC pool (f gure 6).

Seiler and Crutzen (1980) estimated that the large output of elemental C (0.5 to 0.7 Pg C per year) as a result of the incomplete combustion of biomass into charcoal has a signif cant impact on the GCC. The late Wim Sombroek, from Holland, described the use of charcoal by Amazonian tribes to improve soil fertility (*terra preta do Indio*). The f re-induced charcoal (or pyromorphic humus) is composed of rearranged macromolecular substances of weak colloidal properties, with enhanced resistance against chemical and biological degradation and long residence time (Zimmerman 2010). Similarly, the use of biochar has also been proposed for C sequestration (Roberts et al. 2010)

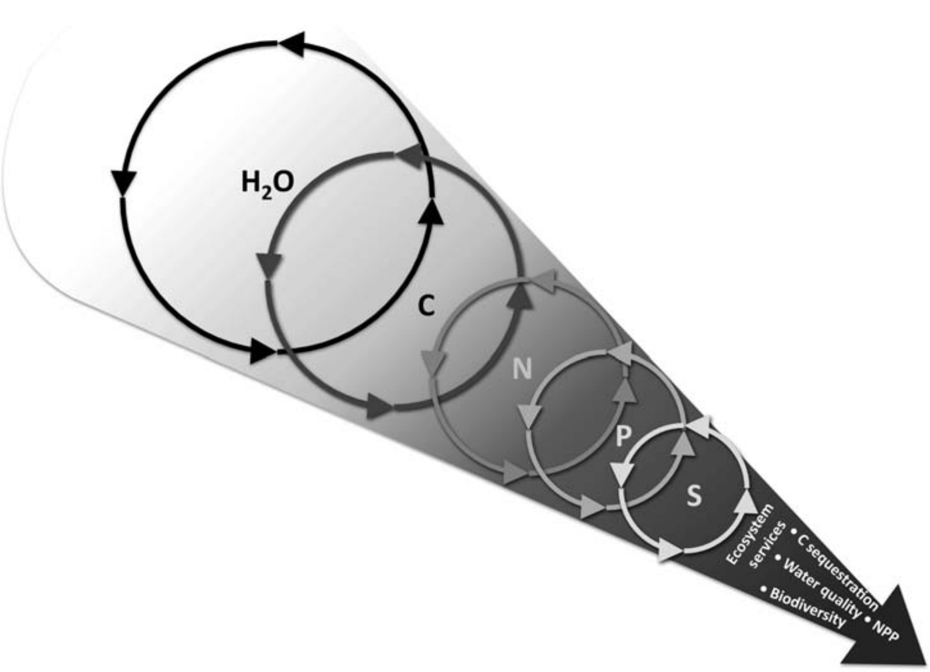

Figure 6. Managing cycling of water, carbon (C), nitrogen (N), phosphorus (P), and sulfur (S) to enhance soil quality and amplify ecosystem services, improve water quality, increase biodiversity, and increase net primary productivity (NPP). Photosynthesis and carbon dioxide fertilization effects can be limited by drought stress and a lack of essential nutrient elements (e.g., N, P, S). Therefore, strengthening coupled cycling of essential elements and that of water would enhance ecosystem C pool and functions.

and as a soil amendment. Hansen and colleagues (2008) estimated that replacing slash-and-burn agriculture with slash-and-char farming and using agricultural and forestry products for biochar production would reduce atmospheric CO_2 by 8 ppm within half a century (equaling approximately 17 Pg C). The potential of biochar application to soil to mitigate atmospheric CO_2 has been estimated at 1 Pg C per year by Sohi and colleagues (2010), 5.5 to 9.5 Pg C per year by 2100 by Lehmann and colleagues (2006), and the removal of 100 ppm of CO_2 by Lenton and Vaughan (2009). However, there are few, if any, field-based studies that provide credible data on the quantitative assessment of diverse soil-management scenarios and that strengthen knowledge about the basic principles involved. Further, a survey of Scandinavian forest soils reveals that pyrogenic C pools have a highly patchy distribution, and a shorter-than-expected lifetime (Ohlson et al. 2009). Wardle and colleagues (2008) suggested that fire-derived charcoal may worsen the loss of forest humus. It is important to recognize that, similar to the option of ocean burial of crop residues, harvesting crop residues and other coproducts for producing biochar is not an option that can be widely recommended. However, the production of biochar (in conjunction with generating energy) is a possibility in site-specific situations such as those from saw dust; rice husk; food packaging; and the processing of coproducts, dairy, and poultry manure.

Carbon sequestration in forest ecosystems. Increased tree growth and the evolution of other large vascular land plants photosynthesized a large amount of atmospheric CO_2 during the mid-to-late Paleozoic. The Paleozoic is the earliest of three geologic eras, spanning from 542 million to 251 million years ago. During the late Paleozoic, great forests of primitive plants thrived on land, forming the great coalbeds of Europe and North America. Today, global forest ecosystems, with a total land area of 4.17 Bha, contain about 1146 Pg of C, two-thirds of which is contained in soils and peat deposits (Dixon et al. 1994). Deforestation depletes the ecosystem C pool, which in low latitudes contributes about 18% of the total anthropogenic emissions (Lal 2005a, 2005b, IPCC 2007). Canadell and colleagues (2007c) estimated current emissions from deforestation to be 1.5 Pg C per year; therefore, sustainable management of forest ecosystems is important for offsetting and creating negative emissions, especially over periods as short as human generations.

Reforestation has a large potential to enhance the terrestrial C pool through photosynthesis into biomass and humification of part of the biomass into the SOC pool (Lal 2005a, 2005b). Many factors affect the rates of C sequestration in forest soils, including C input by litter and roots into different soil horizons, N fertilization, spatial distribution of key soil properties, soil age, moisture regime, site management, frequency and intensity of burning and the addition of charcoal, residue management, and land-use history (Lal 2005a, 2005b). Just as the rise of vascular plants (trees) during the Paleozoic had a strong influence on the concentration of atmospheric CO_2, forest biomes presently have the potential to store 160 to 170 Pg C. Forest biomes are considered an unknown sink for the so-called missing or fugitive C. McKinsey and Company (2009) estimated that by 2030, afforestation can mitigate 0.27 Pg C per year; reforestation, 0.38 Pg C per year; and improved management, an additional 0.08 Pg C per year. Sohngen and Sedjo (2006) suggested that forests may mitigate up to 75 Pg C during the next 50 years or so. While forestry-based C projects are being developed because of their large sink capacity, C dynamics in forest ecosystems are poorly understood. There are numerous unknowns, especially for Amazonian ecosystems (Prentice and Lloyd 1998).

Some of the unknown variables in forest ecosystems include the CH_4 soil-vegetation-atmosphere fluxes in tropical ecosystems, N_2O emissions from tree plantations in the humid tropics, and the trade-offs between C and water use in large-scale afforestation that lead to the depletion of the freshwater supply. Perpetual drought can shift biomes from forests to savannas, with strong impacts on the terrestrial C pool. Nutrient deficits (N, P, potassium [K], S, etc.) also limit C sequestration in forest ecosystems (Lal 2005a, 2005b). Elevated atmospheric CO_2 can cause accumulation in the soil C pool only when N is added at rates well above typical atmospheric N inputs, when other nutrients (P, K, molybdenum) are also added, and when drought stress is not a limiting factor.

Carbon sequestration in grassland ecosystems. Grasslands cover 2.9 Bha globally, including 2.0 Bha under tropical grasslands or savannas and 0.9 Bha under temperate grasslands. Three global regions of savannas are Africa (1.5 Bha), South America (0.21 Bha), and Asia and the Pacific (0.28 Bha). As semiarid lands, grasslands are resource limited, especially in N and water. Thus, soil hydrological properties strongly affect water and C dynamics in grassland ecosystems. Temperate grasslands, dominated by C_3 species, are also limited by C because of high photorespiration rates, reducing the NPP. A large portion of the C assimilated by young plants may be transferred belowground depending upon soil type, rainfall distribution and amount, and management improvements. Relevant management practices for soil C sequestration include fertilization, controlled grazing regime, conversion of degraded croplands and native vegetation to improved pastures, sowing of leguminous and grass pasture species, fire management, and water conservation (Conant et al. 2001, Lal 2002, 2008). Rates of soil C sequestration in grassland ecosystems range from 1.1×10^{-10} to 3.04×10^{-9} Pg C per ha per year with a mean of 5.4×10^{-10} Pg C per ha per year (Conant et al. 2001), with large variation among biomes specially caused by prior land-use changes such as forest to pasture or cropland to pasture.

Unknowns in realizing the terrestrial C sink potential

Despite the important role that the terrestrial biosphere plays in the GCC, there is incomplete and insufficient scientific knowledge to describe the interactions between the components of Earth's systems and the relationship between the GCC and other biogeochemical and climatological processes. We must understand pathways of C in nature and the impacts of microbial processes if we are to identify strategies toward an effective recarbonization of the biosphere. Such knowledge is essential to more accurate predictions of atmospheric concentration of CO_2, and of the fluxes between the terrestrial pools and the atmosphere (Houghton 2007) and the hydrosphere (Lal 2003). Because of coupled cycling, better understanding is also needed for the interaction between C and nutrient elements (N, P, S) and water cycles (figure 6), and the transport of sediments and C into aquatic ecosystems (Cole et al. 2007). Uncertainties about the global energy budget (Trenberth and Fasullo 2010) make it important to link energy budgets with coupled cycling of C and water. Inland freshwater ecosystems (lakes, rivers, and reservoirs) have not been widely studied as important components of the GCC, and the integration of inland water into the terrestrial C budget is an important issue (Cole et al. 2007). Therefore, studying coupled C cycling with cycles water and element is essential. Terrestrial C pools and fluxes are being drastically perturbed by a combination of natural and anthropogenic activities. Although terrestrial ecosystems can be substantial C sinks, there is a trend toward saturation of terrestrial C-sink capacity (Canadell et al. 2007b), probably due to increases in soil degradation (Lal 2009). In this context, the processes affecting the dynamics of soil C pools, at a range of spatial scales (rhizosphere to soilscape and watershed scales), must be understood (figure 7). Specific techniques of mitigating climate change (figure 4) and adapting to ACC (figure 5) must be developed on a soil and biome basis. Rather than just assessing the changes in SOC pools in the root zone,

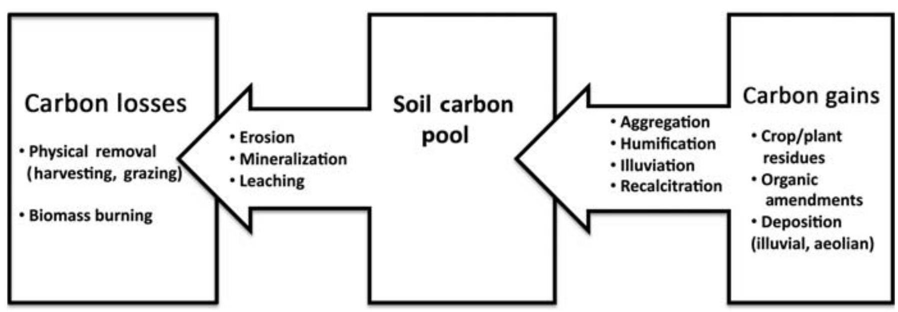

Figure 7. Processes governing the dynamic equilibrium of the soil carbon pool. The dynamic equilibrium of the soil carbon pool depends on the net balance between carbon gains and losses. Carbon gains depend on inputs of biomass, including plant and animal residues, application of organic amendments (e.g., compost, manure, biochar), and deposition of carbonaceous compounds (both organic and inorganic) by the wind and water. Carbon thus added to the soil is stabilized against microbial attack by the formation of structural aggregates, conversion of simple organic compounds into complex humic substances, transfer of carbon into subsoil (illuviation), and its conversion into substances that are resistant to decomposition (recalcitration). Carbon losses from soil are caused by physical removal of the biomass through grazing, harvesting, and fire. Soil carbon is also depleted by accelerated erosion (from wind, water, tillage, and gravity), decomposition, or mineralization and leaching of dissolved carbon compounds. The strategy is to create a positive carbon budget by making carbon gains greater than carbon losses.

Table 1. Carbon footprint of conventional-till and no-till corn (recalculated and revised from Pimentel and Pimentel 2008).

Parameter	Conventional farming		Conservation agriculture	
	Quantity per hectare	Kilograms carbon emitted per hectare	Quantity	Kilograms carbon emitted per hectare
Input (emissions)				
Labor	11.4 hours	44.0	11.4 hours	44.0
Mold board plow	1	15.2	—	0
Discing	2	16.6	—	0
Harrowing	1	2.0	—	0
Seeding	1	3.2	—	3.8
Machinery	55 kg	97.0	20 kg	35.3
Nitrogen	153 kg	198.9	153 kg	198.9
Phosphorus	65 kg	13.0	65 kg	13.0
Potassium	77 kg	11.6	77 kg	11.6
Lime	1120 kg	179.2	2000 kg	320.0
Seeds	21 kg	49.6	25 kg	59.0
Herbicides	6.2 kg	39.1	9 kg	56.7
Insecticides	2.8 kg	14.3	4 kg	20.4
Irrigation	8.1 cm	100	0	0
Electricity	13.2 kwh	3.2	13.2 kwh	3.2
Shredding	0	0	1	4.4
Transport	204 kg	16.1	204 kg	16.1
Total		803.0		786.1
Output (sequestration)				
Grain yield	8655 kg	2969.4	9000 kg	3037.8
Stover yield	8655 kg	3462.0	9000 kg	3600.8
Total		6431.4		6687.8
Soil erosion (20% emission) 10.0 Mg/ha	1×10^4 kg/ha	−60	0	
Carbon sequestration in soil	0	−500		500
Net carbon output		5871.4		7187.8
Ratio of output to input		7.3		9.1

Note: If the yield reduction with no-till is 10%, the output to input ratio is the same.
ha, hectare

we need to do complete life-cycle analysis to evaluate the C footprint of specif c management practices. An example of components for a life-cycle analysis for two tillage systems is shown in table 1. There may also be a positive feedback caused by the negative impact of ACC on terrestrial pools, making permafrost and soils of higher latitudes a net C source (Rodionow et al. 2006). Extreme weather events such as Hurricane Katrina and soil warming (Conant et al. 2008) may also affect the terrestrial C pool. Several studies suggest that terrestrial biomes, rather than being signif cant sinks, may in the future become an important C source (Friedlingstein 2008, Heimann and Reichstein 2008). However, it has also been argued that estimates of SOC sequestration are optimistic, and that temporary C sequestration cannot prevent ACC. Yet soil C sequestration has numerous cobenef ts. It is the proverbial low-hanging fruit; C sequestration in terrestrial ecosystems must be integral to any strategy to address the ACC because of its numerous cobenef ts, especially with regard to advancing global food security.

Soil carbon sequestration and food security

The number of food-insecure people rose from 854 million in 2007 to 1020 million in 2009 (Lal 2010, Lele 2010), primarily because of the increase in the prices of food staples during 2008. World market prices for major food commodities (e.g., grains, vegetable oils) rose sharply to historic highs of more than 60% above 2008 levels. In addition to rising energy prices and higher global demand for biofuel feedstocks (corn, soybeans), the adverse weather conditions of 2006 and 2007 in some major grain and oil-seed producing areas (e.g., Australia) reduced food supplies. Eliminating poverty and hunger of more than 1 billion malnourished people and meeting the food demand of an additional 2.5 billion by 2050 will necessitate increasing global cereal production by 70% (Lele 2010). We may also need to raise production by 2.5 to 3 times in some developing countries in Africa and Asia. Yet the projected ACC may adversely affect the food supply through alterations in agronomic yields (Lobell et al. 2008, Pimentel and Pimentel 2008). Thus, adaptation to ACC through improved soil quality by adoption of recommended management practices is key to advancing global food security. Soil degradation, with its severe adverse impacts on the use efficiency of inputs and agronomic production, must be reversed through restoration of the SOC pool above the threshold level of about 1.1% in the root zone. Improvement in soil quality through C sequestration can substantially increase agronomic productivity and advance food security in developing countries. An increase in the SOC pool by 1×10^{-9} Pg C per ha can increase crop yield by 20 to 70 kilograms (kg) per ha for wheat (*Triticum aestivum*), 10 to 50 kg per ha for rice (*Oryza sativa*), 30 to 300 kg per ha for corn, and 10 to 20 kg per ha for beans (*Phaseolus vulgaris*). Total food production in developing countries can be increased by $24–32 \times 10^{-3}$ Pg per year for grains and $6–10 \times 10^{-3}$ Pg per year for roots and tubers (Lal 2006). Bringing the Green Revolution to small landholders in sub-Saharan Africa necessitates restoring the SOC pool to above the critical threshold level so that soils can respond to the inputs of improved varieties, irrigation, conservation tillage, fertilizers, and other amendments.

Payment for ecosystem services

Ecosystem services are public goods that provide benefits to a large number of people. Rather than subsidies and emergency aids, recommended management practices can be advanced through agricultural policies that promote payment for ecosystem services (PES). Examples of payments include those for sequestering C in soil to mitigate and adapt to ACC, enhancing water availability and quality, strengthening nutrient cycling, controlling floods, decreasing anoxia of coastal ecosystems, increasing biodiversity, and providing habitat for plants and animals. Societal benefits of ecosystem services may be local (e.g., water supply) or global (e.g., C sequestration in soil to mitigate ACC). Biodiversity conservation, closely related to terrestrial C sequestration, is relevant to PES (Barrios 2007). Contracts for soil C sequestration and PES that contribute to household income are important strategies to provide farmers with incentives, especially the resource-poor small landholders in developing countries, to adopt management practices that increase and diversify ecosystem services from agricultural lands. Antle and Stoorvogel (2009) proposed a methodology of PES for agricultural soil C sequestration, and outlined criteria for assessing the societal value of such ecosystem services. The PES strategy can also advance the United Nations Millennium Development Goals of cutting hunger and poverty and empowering women, who account for most of the farming in the developing countries.

In order to implement the PES strategy, we need additional information on the following (Lal 2009)

Gross versus net sequestration. Net C sequestration is a quantitative measure of gains in SOC or the terrestrial C pool after subtracting the emissions used to produce C storage or input from farm activities. Complete life-cycle assessments must be conducted to quantify the carbon losses versus carbon gains to assess the soil and ecosystem C budget for a specific management practice. Carbon inputs to the ecosystem include those from inputs of seed, fertilizer, pesticides, tillage methods, irrigation, and other farm operations (table 1; Lal 2004b, West and Marland 2004). Estimates of the depletion of SOC or an ecosystem pool caused by mineralization and erosion (Cole et al. 2007) are also important to assessing the net C sequestration. Net C gains for a given management practice must also be assessed with reference to a baseline management system implemented over the same time duration.

Measurement of soil C sequestration over a landscape, farm, or watershed scale. Measurements of soil C concentration in a laboratory have been done since the 1860s. The principal objective of these measurements has been the evaluation of soil fertility in the root zone as influenced by agronomic practices such as tillage, crop rotations, manuring, and so on. Both wet and dry combustion methods are used for these measurements. Conventional units of reporting SOC and total C concentrations are percentages or gram-per-kilogram measurements on a dry-weight basis. Uncertainties in the data on measurements of SOC concentrations from croplands are caused by spatial variability in soil properties (e.g., soil bulk density). In the context of evaluating soil C sink capacity of managed ecosystems, the objective is to quantify management-induced changes in the total SOC pool (Pg C per ha) and compute the rate of its change with reference to baseline (Pg C per ha per year) over a landscape, farm, watershed, or regional scale. Since the 2000s, several advancements have made it easier to measure the SOC concentration, including rapid spectroscopic methods, remote sensing measurements, and in-field assessment by laser-induced breakdown spectroscopy, or LIBS. The soil C pool can also be measured directly over a landscape by *in situ* noninvasive soil C analysis based on inelastic neutron scattering technique. Micrometeorological techniques, such as eddy covariance, provide a direct measure of CO_2 and water fluxes between ecosystems and the atmosphere. Eddy covariance techniques can be useful to assess interseasonal variability in C uptake. Satellite normalized difference vegetation index data sets (Raun et al. 2001) and climate data can be used to assess seasonal fluxes in C over a landscape. There is a range of models available to predict the SOC pool, including CENTURY, Roth C, C-Quest, EPIC, and others. Mishra and Lal (2011) report that the SOC pool can be reliably predicted within desired-depth intervals by using the soil and terrain characteristics and climatic parameters. By using a range of spatial autocorrelations in SOC data within a geographic weighted regression framework, Mishra and Lal (2010) reported that credible estimates of SOC pool can be made at large regional spatial scales.

Permanence. To be effective in mitigating the ACC by offsetting emissions through fossil-fuel combustion and other anthropogenic activities, atmospheric CO_2 sequestered as humus must remain in the soil for long periods—at decadal, centennial, and millennial scales. Similarly, C sequestered in the forest biomass must not be released by fire or processes leading to its decomposition. Just as wood C can be preserved, SOC also has a long residence time (permanence). The residence of soil C is enhanced when it is encapsulated within stable microaggregates (physical protection), forms stable organo-mineral complexes with clay (chemical protection), is converted into recalcitrant humic substance (biological protection), or when it has been illuviated (transported) into the subsoil and away from the zone of natural or anthropogenic perturbation (Tisdall 1996). Furthermore, once sequestered, C is likely to remain in the soil as long as the same land-use and management practices are followed. Marland and colleagues (2001) and McCarl and Schneider (2001) suggested that the issue of permanence should not detract from the pursuit of C sinks as a mitigation effort. Terrestrial sinks also play a bridging role in greenhouse gas mitigation policy by reducing the cost of current compliance until technologies in other sectors are developed and eventually take effect. However, several studies express strong concerns about the permanence of biologically sequestered C because terrestrial ecosystems may provide a positive and amplifying feedback in a warming world (Heimann and Reichstein 2003). These studies argue that future warming may reduce the C sink (Friedlingstein 2008). Temperature sensitivity of SOC decomposition is not clearly understood.

Technical versus economic potential and the marginal abatement cost. Technical potential refers to the maximum or theoretical potential C sink capacity of a soil under "best case" situations that ignore economic, social, or institutional constraints (Bangsund and Leisritz 2008). This "upper bound" of soil and tree C storage capacity can reduce atmospheric CO_2 by 50 ppm by the period ending in 2150 (Hansen et al. 2008). In contrast, economic potential refers to the amount of C that can be sequestered under the specific conditions of cost-benefit scenarios. The cost-curve analysis provides information on the magnitude of C that can be sequestered in soil and biota at the costs associated with specific management practices, such as choosing a tree species, selecting site management options for tree plantations, stand management, adopting NT farming, changing crop rotation, growing a cover crop, eliminating summer fallow, using INM, adopting DSI, establishing buffer strips, establishing agroforestry measures, planting appropriate forage species, changing stocking rates, and so on. The technical potential of soil C sequestration is 0.144 to 0.432 Pg (0.288 Pg) C per year for the United States (Lal et al. 2008), 0.4 to 1.2 Pg (0.8 Pg) C per year in cropland soils of the world (Lal 2004a), and 2.55 to 4.96 Pg (3.76 Pg) C for the terrestrial biosphere (table 2). However, the potential depends on the monetary cost of C credits. McCarl and Schneider (2001) presented the marginal abatement cost of C sequestration in US agricultural soils. The economic potential of soil C sequestration increases from 0.042 Pg C per year at price of less than $10 per ton C to 0.075 Pg C per year at price of $65 per ton C. McCarl and Schneider (2001) concluded that the maximum economic potential of C sequestration in US soils is 0.06 Pg C per year. The US Environmental Protection Agency (2009) estimated the economic potential

Table 2. Technical potential of carbon sequestration in the terrestrial biosphere through afforestation, restoration of degraded and desertified soils, and adoption of recommended management practices on agricultural soils (data based on Lal 2004a, 2005a, 2005b, 2009).

Activity	Technical potential (billion metric tons carbon per year)
Afforestation, forest succession, agroforestry, peatland	1.2 to 1.4
Forest plantations	0.2 to 0.5
Savanna and grassland ecosystems	0.3 to 0.5
Cropland management	0.4 to 1.2
Restoration of salt-affected soils	0.3 to 0.7
Desertification control	0.2 to 0.7
Total technical potential	2.55 to 4.96 (3.8)

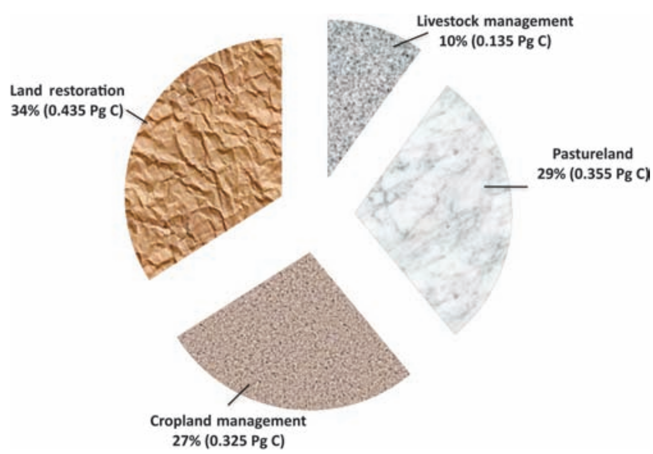

Figure 8. Abatement cost curve of carbon (C) sequestration in agriculture sector (redrawn from McKinsey and Company 2009). Carbon sequestration in the terrestrial biosphere is relatively cost-effective and has numerous cobenefits. Technological options have been widely proven, and are immediately available for wide-scale applications in diverse ecoregions. It is important to identify policies that promote the adoption of appropriate technologies by land managers in developed and developing countries. Pg, billoin metric tons C.

of C sequestration in US soils for the period 2010–2050 for a C price ranging from $1 to $50 per ton. Similar analysis of the marginal abatement cost is needed for the economic potential of global-scale C sequestration, preferably on a soil (Oxisols, Ultisols, Vertisols, Alfisols, etc.) and biome basis (humid tropics, temperate, boreal, etc.; Smith et al. 2008).

McKinsey and Company (2009) computed the greenhouse gas abatement cost curve, and showed that tillage and residue management had a negative cost of about −€65 per ton C compared with a cost of €45 to €60 per ton C for C capture and sequestration technology (1€ = approximately $1.5). On the basis of the global analysis of land suitability for C sink for afforestation and reforestation, total economic potential of C sequestration in the terrestrial biosphere is 1.25 Pg C per year until 2030 (figure 8). Both relative cost and risks are lowest for C sequestration in biomass and highest with geoengineering (geologic) and oceanic sequestration.

Farming carbon. Terrestrial C can be traded through the Chicago Climate Exchange, the BioCarbon Fund of the World Bank, and other markets. The Clean Development Mechanism (CDM) under the Kyoto Treaty allows for some emission reduction credits from afforestation and reforestations; however, soil C is not yet traded through the CDM. Negotiations included agricultural offsets in the Copenhagen meeting in December 2009, but with no progress (Lal 2010). Zomer and colleagues (2008) conducted a global analysis of land suitability for C sinks and estimated that 0.75 Bha are suitable under CDM. Of this, 46% are in South America, and 27% are in sub-Saharan Africa. Factoring out natural and indirect human-induced effects on C sources and sinks from the direct human-induced changes is necessary to trading C credits.

Canadell and colleagues (2007a) outlined an accounting approach to establish a credible and transparent system of computing C credits and debits by factoring out climate viability and CO_2 and N fertilization by (a) selecting a longer accounting or measurement period, (b) using activity-based accounting, (c) using a baseline scenario, and (d) stratifying the landscape into units with distinct average C pool. Rokityanskiy and colleagues (2007) modeled the effects of policies designed to induce landowners to adopt recommended management practices to sequester C in soils and terrestrial ecosystems. For C trading to be an attractive incentive for farmers to adopt recommended management practices, the price of soil and terrestrial C must reflect the societal value. The low price in the Chicago Climate Exchange and other voluntary markets is undervaluing the precious C commodity, and this can lead to even more misuse than has occurred in the past. It is therefore important to determine the C price according to transparent, just, and fair criteria. Although markets determine the C price, it is grossly undervalued because of the absence of an effective cap-and-trade system.

Conclusions

Recarbonization of the terrestrial biosphere by restoring C pools in Earth's soils, trees, and wetlands is an important step toward both mitigation and adaptation to ACC. In comparison with the engineering techniques of geologic and oceanic sequestration, C sequestration in terrestrial ecosystems has numerous cobenefits, such as increasing NPP, advancing food security, improving the quality and quantity of water resources, enhancing biodiversity, and others. Payment for ecosystem services can promote the adoption of recommended management practices and conversion to restorative land use. In order to realize the potential for C sequestration in the terrestrial biosphere we need a better understanding of basic processes at a range of scales, from molecular to watershed, and through life-cycle analyses of the managed ecosystems. In addition to the technical and maximum potential, the economic and realizable potential must be computed through assessment of the marginal abatement cost. There is therefore a strong need for multidisciplinary approaches to addressing this important global issue. Measurement, monitoring, modeling, and verification of the C pool and fluxes at soilscape, landscape, watershed, regional, and national scales are essential for terrestrial C sequestration to be adopted by the UN Framework Convention on Climate Change as an offset for anthropogenic emissions. Regardless of climate change, it is important to realize that the basic needs (e.g., food, feed, fiber, fuel) of Earth's 10 billion inhabitants by 2100 cannot be met without restoring the services of the terrestrial ecosystems, especially soil quality, for which recarbonization of the biosphere and enhancement of soil organic carbon pool are essential prerequisites.

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References cited

Antle JM, Stoorvogel JJ. 2009. Payment for ecosystem services, poverty and sustainability: The case of agricultural soil carbon sequestration. Pages 133–161 in Lipper L, Sakuyama T, Stringer R, Zilberman D, eds. Natural Resource Management and Policy: Payment for Environmental Services in Agricultural Landscapes. Springer.

Bai ZG, Dent DL, Olsson L, Schaepman ME. 2008. Proxy global assessment of land degradation. Soil Use and Management 24: 223–234.

Bangsund DA, Leistritz FL. 2008. Review of literature on economics and policy of carbon sequestration in agricultural soils. Management of Environmental Quality 19: 85–99.

Barrios E. 2007. Soil biota, ecosystem services and land productivity. Environmental Economics 64: 269–285.

Canadell JG, Kirschbaum MUF, Kurz WA, Sanz MJ, Schlamadinger B, Yamagata Y. 2007a. Factoring out natural and indirect human effects on terrestrial carbon sources and sinks. Environmental Science and Policy 10: 370–384.

Canadell JG, et al. 2007b. Saturation of the terrestrial carbon sink. Pages 59–78 in Canadell JG, Pataki DE, Pitelka LF, eds. Terrestrial Ecosystems in a Changing World. Springer.

Canadell JG, Le Quéré C, Raupach MR, Field CB, Buitenhuis ET, Ciais P, Conway TJ, Gillett NP, Houghton RA, Marland G. 2007c. Contributions to accelerating atmospheric CO_2 growth from economic activity, carbon intensity and efficiency of natural sinks. Proceedings of the National Academy of Sciences 104: 18866–18870.

Cole JJ, et al. 2007. Plumbing the carbon cycle: Integrating inland waters into the terrestrial carbon budget. Ecosystems 10: 171–184.

Conant RT, Paustian K, Elliott ET. 2001. Grassland management and conversion into grassland: Effects on soil carbon. Ecological Applications 11: 343–355.

Conant RT, Steinweg JM, Haddix ML, Paul EA, Plante AF, Six J. 2008. Experimental warming shows that decomposition temperature sensitivity increase with soil organic matter recalcitrance. Ecology 89: 2384–2391.

Dixon RK, Brown S, Houghton RA, Solomon AM, Trexler MC, Wisniewski J. 1994. Carbon pools and flux of global forest ecosystems. Science 263: 185–190.

Dregne HE. 1998. Desertification assessment. Pages 441–458 in Lal R, Blum WH, Valentine C, Stewart BA, eds. Methods for Assessment of Soil Degradation. CRC.

Eswaran H, Reich P, Beinroth F. 2001. Global desertification tension zones. Pages 24–28 in Stott DE, Mohtar RH, Steinhardt GC, eds. Sustaining the Global Farm. Selected Papers from the 10th International Soil Conservation Organization Meeting; 24–29 May 1999.

Foley JA, et al. 2005. Global consequences of land use. Science 309: 570–574.

Friedlingstein P. 2008. A steep road to climate stabilization. Nature 451: 297–298.

[GCP] Global Carbon Project. 2008. Carbon Budget and Trends 2007. GCP. (21 July 2010; *www.globalcarbonproject.org*)

Haberl H, Erb K-H, Krausmann F, Gaube V, Bondeau A, Plutzar C, Gingrich S, Lucht W, Fischer-Kowalski M. 2007. Quantifying and mapping the human appropriation of net primary production in Earth's terrestrial ecosystems. Proceedings of the National Academy of Sciences 104: 12942–12947.

Hansen J, Sato M, Kharecha P, Beerling D, Berner R, Masson-Delmotte V, Pagani M, Raymo M, Royer DL, Zachos JC. 2008. Target atmospheric CO_2: Where should humanity aim? Open Atmospheric Science Journal 2: 217–231.

Heimann M, Reichstein M. 2008. Terrestrial ecosystem carbon dynamics and climate feedbacks. Nature 451: 289–292.

Holdren JP. 2008. Meeting the climate change challenge. 8th Annual John H. Chafee Memorial Lecture; 17 January 2008, Ronald Reagan Building and International Trade Center, Washington, D.C.

Houghton RA. 2007. Balancing the global carbon budget. Annual Review of Earth and Planetary Sciences 35: 313–347.

[IPCC] Intergovernmental Panel on Climate Change. 2007. Climate Change 2007: Mitigation. Cambridge University Press.

Karlen D, Lal R, Follett RF, Kimble JM, Hatfield JL, Miranowski JM, Cambardella CA, Manale A, Anex RP, Rice CW. 2009. Crop residues: The rest of the story. Environmental Science and Technology 43: 8011–8015.

Lal R. 2002. Soil carbon dynamics in cropland and rangeland. Environmental Pollution 116: 353–362.

———. 2003. Soil erosion and the global carbon budget. Environmental International 29: 437–450.

———. 2004a. Soil carbon sequestration impacts on global climate change and food security. Science 304: 1623–1627.

———. 2004b. Carbon emission from farm operations. Environmental International 30: 981–990.

———. 2005a. Forest soils and carbon sequestration. Forest Ecology and Management 220: 242–258.

———. 2005b. Soil carbon sequestration in natural and managed tropical ecosystems. Journal of Sustainable Forestry 2: 1–30.

———. 2006. Enhancing crop yield in the developing countries through restoration of the soil organic carbon pool in agricultural lands. Land Degradation and Development 17: 197–209.

———. 2008. Savannas and global climate change: Source or sink of atmospheric CO_2? Pages 81–102 in Faleiro FG, de Farias Neto AL, eds. Savanas, Desafios e Estrategias pare o Equilibrio entre sociedade, agronegocio e recursos naturais. EMBRAPA.

———. 2009. Challenges and opportunities in soil organic matter research. European Journal of Soil Science 60: 158–169.

———. 2010. Beyond Copenhagen: Mitigating climate change and achieving food security through soil carbon sequestration. Food Security 2: 169–177.

Lal R, Follett RF, Stewart BA, Kimble JM. 2008. Soil carbon sequestration to mitigate climate change and advance food security. Soil Science 172: 943–956.

Lehmann J, Gaunt J, Rondon M. 2006. Biochar sequestration in terrestrial ecosystems: A review. Mitigation and Adaptation Strategies for Global Change 11: 395–419.

Lele U. 2010. Food security for a billion poor. Science 326: 1554.

Lenton TM, Vaughan NE. 2009. The radiative forcing potential of different climate geoengineering options. Atmospheric Chemistry and Physics Discussion 9: 2559–2608.

Lewis SL, et al. 2005. Tropical forests and atmospheric CO_2: Current conditions and future scenarios. Pages 147–153 in Schellnhuber HJ, ed. Avoiding Dangerous Climate Change. Cambridge University Press.

Lobell DB, Burke MB, Tebaldi C, Mastrandea MD, Falcon WP, Naylor RL. 2008. Prioritizing climate change adaptation needs for food security in 2030. Science 319: 607–610.

MacDonald GM, Beilman DW, Kremenetski KV, Sheng Y, Smith LC, Velichko AA. 2006. Rapid early development of circumarctic peatland development and atmospheric CH_4 and CO_2 variations. Science 314: 285–288.

Marland G, McCarl BA, Schneider UA. 2001. Soil Carbon: Policy and economics. Climatic Change 51: 101–117.

McCarl BA, Schneider UA. 2001. The cost of greenhouse gas mitigation in U.S. agriculture and forestry. Science 294: 2481–2482.

McKinsey and Company. 2009. Pathways to a Low Carbon Economy. Version 2 of the Global Greenhouse Gas Abatement Cost Curve. McKinsey and Company. (21 July 2010; *https://solutions.mckinsey.com/ClimateDesk/default/en-us/requestfullreport.aspx*)

Mishra U, Lal R. 2011. Predictive mapping of soil organic carbon: A case study using geographic weighted regressional approach. In Clay D, Shanahan J, eds. GIS Applications in Agriculture—Nutrient Management for Improved Energy Efficiency. CRC Press. Forthcoming.

Ohlson M, Dahlberg B, Okland T, Brown KJ, Halvorsen R. 2009. The charcoal carbon pool in boreal forest soils. Nature Geoscience 2: 692–695.

Oldeman LR, Van Lynden GWJ. 1998. Revisiting the Glasod Methodology. Pages 423–427 in Lal R, Blum WH, Valentin C, Steward BA, eds. Methods for Assessment of Soil Degradation. CRC Press.

Pacala S, Socolow R. 2004. Stabilization wedges: Solving the climate problem for the next 50 years with the current technologies. Science 305: 968–972.

Pimentel D, Pimentel MH. 2008. Food, Energy and Society. CRC Press, Taylor and Francis.

Prentice IC, Lloyd J. 1998. C-quest in the Amazon basin. Nature 396: 619–620.

Raun WR, Solie JB, Johnson GV, Stone ML, Lukina EV, Thompson WE, Schepers JS. 2001. In season prediction of potential grain yield in winter wheat using canopy reflectance. Agronomy Journal 93: 131–138.

Roberts KG, Gloy BA, Joseph S, Scott NR, Lehmann J. 2010. Life cycle assessment of biochar systems: Estimating the energetic. Economic and climate change potential. Environmental Science and Technology 41: 827–833.

Rodionow A, Flessa H, Kazansky O, Guggenberger G. 2006. Organic matter composition and potential trace gas production of permafrost soils in the forest tundra in northern Siberia. Geoderma 135: 49–62.

Rokityanskiy D, Benitz PC, Kraxner F, McCallum I, Obersteiner M, Rametsteiner E, Yamagata Y. 2007. Geographically explicit global modeling of land-use change, carbon sequestration, and biomass supply. Technological Forecasting and Social Change 74: 1057–1082.

Ruddiman WF. 2003. The anthropogenic greenhouse era began thousands of years ago. Climatic Change 61: 261–293.

Seiler W, Crutzen PJ. 1980. Estimates of gross and net fluxes of carbon between the biosphere and the atmosphere from biomass burning. Climatic Change 2: 207–247.

Sheehan J. 2009. Engineering direct conversion of CO_2 to biofuel. Nature Biotechnology 27: 1128–1129.

Six J, Ogle SM, Breidt FJ, Conant RT, Mosiers AR, Paustian K. 2004. The potential to mitigate global warming with no-tillage management is only realized when practiced in the long term. Global Change Biology 10: 155–160.

Smith LC, MacDonald GM, Velichko AA, Beilman DW, Borisova OK, Frey KE, Kremenetski KV, Sheng Y. 2004. Siberian peatlands a net carbon sink and global methane source since the early Holocene. Science 303: 353–356.

Smith P, et al. 2008. Greenhouse gas mitigation in agriculture. Philosophical Transactions of the Royal Society B 363: 789–813.

Sohi SP, Krull E, Lopez-Capel E, Bol R. 2010. A review of biochar and its use and function in soil. Advances in Agronomy 105: 47–87.

Sohngen B, Sedjo R. 2006. Carbon sequestration costs in global forests. Energy Journal (special issue): 109–126.

Steinbach HS, Alvarez R. 2006. Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in Pampean agroecosystems. Journal of Environmental Quality 35: 3–13.

Tisdall JM. 1996. Formation of soil aggregates and accumulation of soil organic matter. Pages 57–96 in Carter MR, Steward BA, eds. Structure and Organic Matter Storage in Agricultural Soils. Lewis.

Trenberth KE, Fasullo JT. 2010. Tracking Earth's energy. Science 328: 316–317.

[USEPA] US Environmental Protection Agency. 2009. Climate Change Legislation. US EPA. (21 July 2010; *http://epa.gov/climatechange/wconomics/downloads/WM-EPAAnalysis-Data*)

Van Oost K, Quine TA, Glovers G, De Gryze S, et al. 2007. The impact of agricultural soil erosion in the global carbon cycle. Science 318: 626–629.

Vergé XPC, De Kimpe CD, Desjardins RL. 2007. Agricultural production, greenhouse gas emissions and mitigation potential. Agricultural and Forest Meteorology 142: 255–269.

Wardle DA, Nilsson MC, Zackrisson O. 2008. Fire-derived charcoal causes loss of forest humus. Science 320: 629.

West TO, Marland G. 2004. Net carbon flux from agricultural ecosystems and methodology for full carbon cycle analyses. Environmental Pollution 116: 439–444.

West TO, Post WM. 2002. Soil organic carbon sequestration rates by tillage and crop rotations. Soil Science Society of America Journal 66: 1930–1946.

Wild A. 2003. Soils, Land, and Food: Managing the Land During 21st Century. Cambridge University Press.

Zimmerman AR. 2010. Abiotic and microbial oxidation of laboratory-produced black carbon (biochar). Environmental Science and Technology 44: 1295–1301.

Zomer RJ, Trabucco A, Bossio DA, Verchot LV. 2008. Climate change mitigation: A spatial analysis of global land suitability for clean development mechanism afforestation and reforestation. Agriculture Ecosystems and Environment 126: 67–80.

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