*Critical Reviews in Plant Sciences*, 24:1–21, 2005

ISSN: 0735-2689 print / 1549-7836 online
DOI: 10.1080/07352680590910393

# Bioenergy Crops and Carbon Sequestration

**R. Lemus and R. Lal**
*Carbon Management and Sequestration Center, The Ohio State University, Columbus, OH*

**Referee: Professor E. Charles Brummer, Forage Breeding and Genetics, Raymond F. Baker Center for Plant Breeding, 1204 Agronomy, Iowa State University, Ames, IA 50011-1010**

## Table of Contents



Address correspondence to R. Lemus, Carbon Management and Sequestration Center, The Ohio State University, 2021 Coffey Road, Columbus, OH 43210, USA. E-mail: Rocky_Lemus@TAMU-Commerce.edu

**Greenhouse gas (GHG) emissions constitute a global problem. The need for agricultural involvement in GHG mitigation has been widely recognized since the 1990s. The concept of C sinks, C credits, and emission trading has attracted special interests in herbaceous and woody species as energy crops and source of biofuel feedstock. Bioenergy crops are defined as any plant material used to produce bioenergy. These crops have the capacity to produce large volume of biomass, high energy potential, and can be grown in marginal soils. Planting bioenergy crops in degraded soils is one of the promising agricultural options with C sequestration rates ranging from 0.6 to 3.0 Mg C ha$^{-1}$ yr$^{-1}$. About 60 million hectares (Mha) of land is available in the United States and 757 Mha in the world to grow bioenergy crops. With an energy offset of 1 kg of C in biomass per 0.6 kg of C in fossil fuel, there exists a vast potential of offsetting fossil fuel emission. Bioenergy crops have the potential to sequester approximately 318 Tg C yr$^{-1}$ in the United States and 1631 Tg C yr$^{-1}$ worldwide. Bioenergy crops consist of herbaceous bunch-type grasses and short-rotation woody perennials. Important grasses include switchgrass (*Panicum* virgatum L.), elephant grass (*Pennissetum* purpureum Schum.), tall fescue (*Fetusca arundinacea* L.), etc. Important among short-rotation woody perennials are poplar (*Populus* spp.), willow (*Salix* spp.), mesquite (*Prosopis* spp.), etc. The emissions of $CO_2$ from using switchgrass as energy crop is 1.9 kg C Gj$^{-1}$ compared with 13.8, 22.3, and 24.6 kg C Gj$^{-1}$ from using gas, petroleum, and coal, respectively. Mitigation of GHG emissions cannot be achieved by C sinks alone, a substantial reduction in fossil fuel combustion will be necessary. Carbon sequestration and fossil fuel offset by bioenergy crops is an important component of a possible total societal response to a GHG emission reduction initiative.**

**Keywords** biofuel, switchgrass, poplar, willow, biomass, greenhouse effect, soil C dynamics, C sequestration

## I. INTRODUCTION

The atmospheric $CO_2$ concentration has increased by 30 percent since the industrial era (IPCC, 2001). Enhanced carbon sequestration and energy cropping could have the potential to offset 1000 to 2000 Mt C yr$^{-1}$ (Cannell, 2003). To understand the amount of C that can be sequestered in biomass and soils, it is necessary to understand that carbon emitted from fossil fuel is about 420 Gt C since the industrial revolution (IPCC, 2001).

Interests in mitigating the threats of global climate change warrant evaluating crops capable of producing high biomass for energy generation and soil carbon (C) sequestration. Bioenergy crops have the potential to supply a significant portion of U.S and global energy needs while reducing the rate of enrichment of atmospheric $CO_2$. These are mainly perennial crops (herbaceous or woody) which can improve soil quality, enhance nutrient cycling, and sequester C. Perennial crops are also capable of producing large quantities of high C content biomass. Most energy crops are characterized by their perenniality, less maintenance and input, and adaptation to marginal soils.

The fossil fuel dependency and the agricultural and industrial contributions to greenhouse gas (GHG) emissions require identification of species that can transfer atmospheric $CO_2$ into the biomass and soil C pools. Energy crops capture an amount of C in the harvested biomass that is usually equivalent to the C released during combustion and therefore, a C-neutral energy source for that component of the crop (Hansen, 1993). The potential of perennial crops to offset $CO_2$ emissions through soil C sequestration depends on the rate of soil C additions, the long-term capacity of soil C storage, and the stability of C sequestered over time (McLaughlin *et al*., 2002).

In an effort to better understand the role of soils in the global C cycle, several studies have measured and characterized soil C inventories along forest chronosequences and under bioenergy crops (McLaughlin *et al*., 1998; Ma *et al*., 2000a; Pacala *et al*., 2001; Zan *et al*., 2001; Baer *et al*., 2002; Cannell, 2003; Hall and House, 2004). The objective of this review is to collate, review, and synthesize the available information, and assess the potential impacts of growing bioenergy crops on soil C sequestration to mitigate the greenhouse effect.

## II. WORLD ENERGY USE

There has been a worldwide increase in energy consumption and $CO_2$ emissions during the twentieth century (Table 1). At present, there is no shortage of fossil fuels, but long-term availability and desirability of their use are concerns since the rapidly and expanding world economy is expected to increase fossil fuel combustion (Hawken, 1993). The world energy consumption is expected to increase 54 percent by 2025, with much of the energy growth occurring in rapidly expanding economies (International Energy Outlook, 2004).

Bioenergy crops are one of the renewable energy sources for the future. Fossil fuels (oil, coal, and natural gas) supply 85 percent of the total primary energy used in the world

TABLE 1
World energy consumption and carbon dioxide emissions by region, 1990–2001 (Adapted from International Energy Outlook, 2004)

| | Energy consumption (Quadrillion Btu) | | | $CO_2$ emissions (Million Metric Tons or Tg) | | |
|---|---|---|---|---|---|---|
| Region | 1990 | 2000 | 2001 | 1990 | 2000 | 2001 |
| Africa | 9 | 12 | 12 | 656 | 811 | 843 |
| Asia | 75 | 108 | 113 | 5,274 | 7,235 | 7,568 |
| Central and South America | 14 | 21 | 21 | 703 | 961 | 964 |
| EE/FSU | 76 | 52 | 53 | 4,902 | 3,094 | 3,148 |
| North America | 101 | 119 | 116 | 5,769 | 6,731 | 6,613 |
| Middle East | 13 | 20 | 21 | 846 | 1,262 | 1,299 |
| Western Europe | 60 | 67 | 68 | 3,412 | 3,442 | 3,465 |
| Total World | 348 | 399 | 404 | 21,562 | 23,536 | 23,900 |

33.1 MBtu = One metric ton of C.
1 million metric ton of $CO_2$ = 0.273 million metric ton of C.
1 Quadrillion = $10^{15}$ or million-billion.

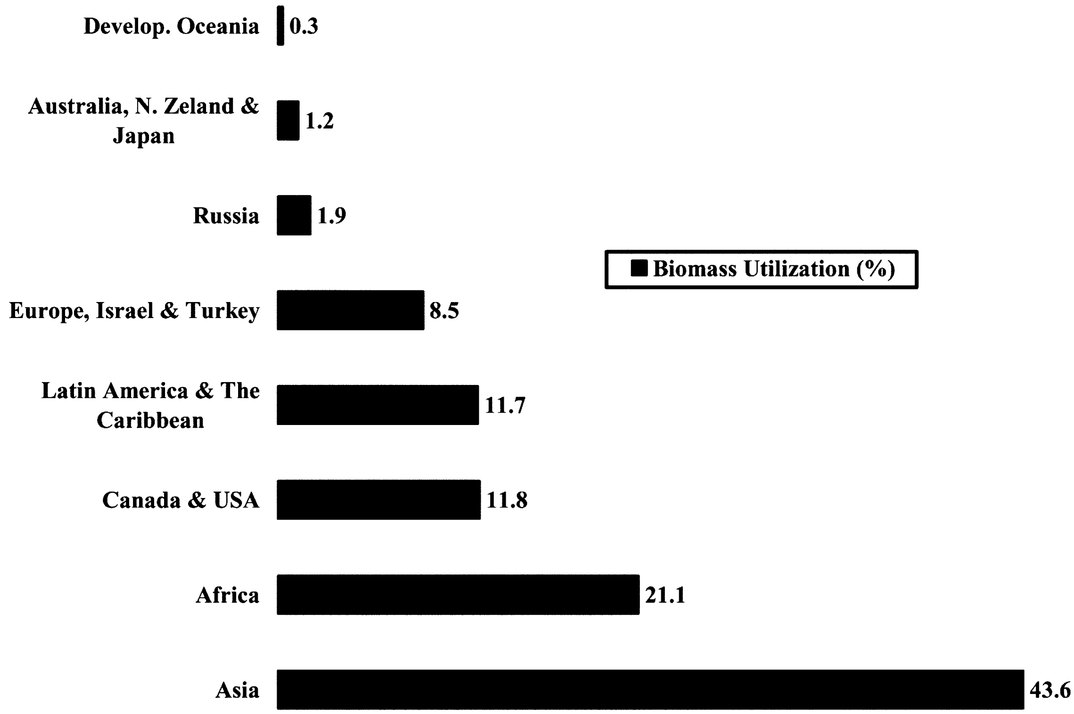


FIG. 1. Utilization of biomass as a primary energy source in the world (Redrawn from Biomass, 2004).

compared with 15 percent supplied through the biomass (Johanson and Lundqvist, 1999). Bioenergy crops are already the fourth largest energy source (>55 EJ $yr^{-1}$) with a large number of developing countries and regions depending on biomass utilization as the principal energy source (Figure 1) (Hall and House, 2004).

An efficient production of bioenergy crops, along with modern conversion technologies, can supply a considerable amount of energy at a large scale while diminishing the net $CO_2$ emissions. In the United States, 4 percent (3.2 EJ) of the total energy is produced from bioenergy crops with an electric generating capacity of about 9,000 MW (Johansson *et al*., 1993), but could easily sustain 20 percent more due to available land that can be converted from intensive agriculture to bioenergy crops (Biomass, 2004).

The $CO_2$ emissions from using perennial crops for energy generation are about 7, 12, and 13 times lower than those produced by gas, petroleum, and coal, respectively (Ma *et al*., 2000a). Specifically, $CO_2$ emissions from switchgrass is 1.9 kg C $GJ^{-1}$ compared with 13.8, 22.3, and 24.6 kg C $GJ^{-1}$ for gas, petroleum, and coal, respectively (Turhollow and Perlack, 1991).

To understand the extent of biomass potential to offset $CO_2$ emissions and maintain energy production, it is necessary to look at the energy flow from biomass production compared to fossil fuel (e.g., coal) (Figure 2). One kilogram of biomass utilized for electricity generation contains approximately 18.5 GJ of energy (McLaughlin *et al*., 1996). Biomass production also involves the utilization of fossil fuel for cultivation, fertilization, and transportation of biomass to the power plants for generation of electricity. An emission rate of 0.5 to 2.0 kg C $GJ^{-1}$ of energy derived from the power plant has been estimated (Cannell, 2003). This means that utilization of 500 kg C is equivalent to emission of about 10 to 40 kg C. This reaction will also consume 0.5 to 1.5 GJ of the energy produced. In this case, the amount of C fixed by photosynthesis is returned to the atmosphere during the generation of electricity. On the other hand, fossil fuel combustion indicates that while we are using the same energy production for coal, the amount of $CO_2$ in the atmosphere is not recycled into the system.

In the United States, 4 percent (3.2 EJ) of the total energy is produced from bioenergy crops with an electric generating capacity of about 9,000 MW (Johansson *et al*., 1993), but could easily sustain 20 percent more due to available land that can be converted from marginal agricultural land to bioenergy crops (Biomass, 2004).

## III. BIOENERGY CROPS

Assessing the land resources available for biomass production must take into account species to be grown and their biomass productivity, which are the essential components to provide alternative energy sources. Bioenergy crops are defined as any plant material used to produce bioenergy, but those grown specifically for the purpose are characterized by the capacity to produce large volumes of biomass, have high energy potential, and are adapted to marginal soils.

The U.S. Department of Energy (U.S. DOE) has identified species capable of alleviating energy constraints and reducing

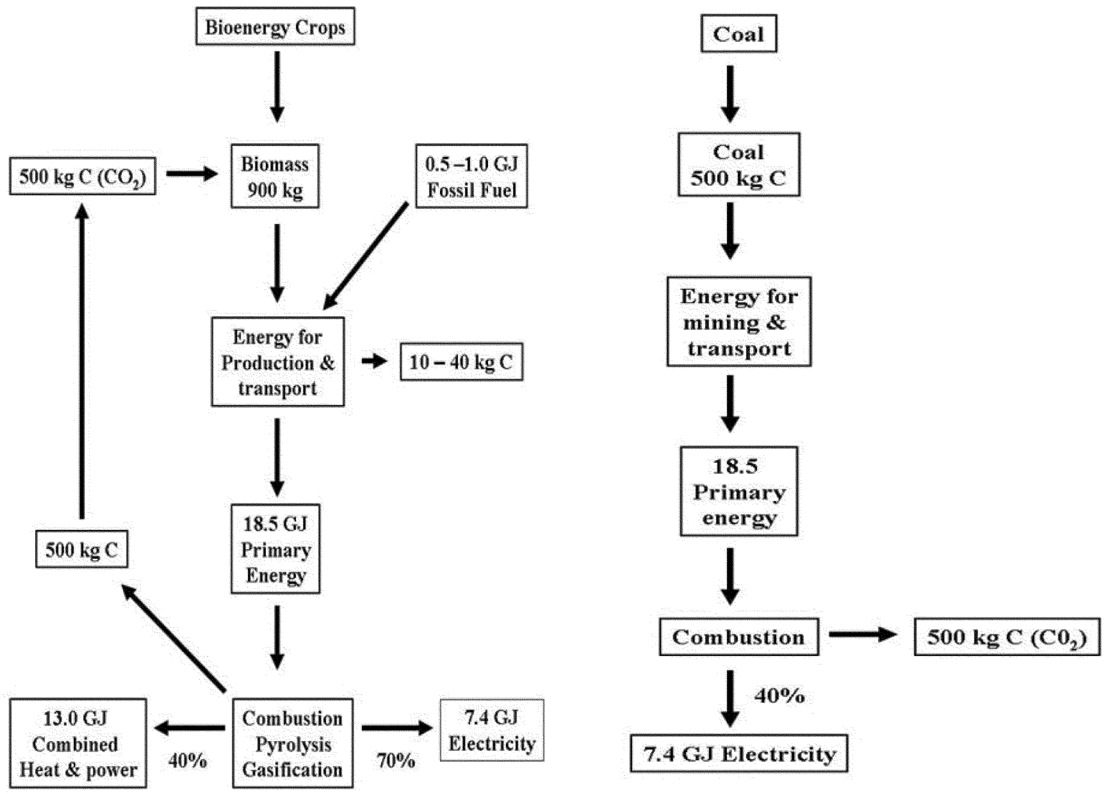


FIG. 2. Flows of energy and carbon when producing electricity from biomass and coal (Redrawn from Cannell, 2003).

$CO_2$ levels (McLaughlin and Walsh, 1998). These species include perennial herbaceous crops and short-rotation woody crops (SRWC).

### A. Herbaceous Energy Crops

Herbaceous crops are plants that have little or no woody tissue, and mostly comprise bunch-type grasses generally harvested like hay at the end of the growing season when important nutrients (especially nitrogen) have been translocated to roots (Lemus, 2004). Different grasses, such as elephantgrass (*Pennisetum purpureum* Schum.), kleingrass (*Panicum coloratum* L.), buffalograss (*Buchloe dactyloides* Nutt.), switchgrass (*Panicum virgatum* L.), miscanthus (*Miscanthus* spp.), reed canarygrass (*Phalaris arundeneacea* L.), tall fescue (*Fetusca arundinacea*), eastern gamagrass (*Tripsacum dactyloides*), and big bluestem (*Andropogon gerardii*) have been identified as promising species for biofuel production (Madakadze *et al*., 1999). These grasses regrow from their roots and do not require replanting for long periods of time (>15 years).

### B. Short-Rotation Woody Energy Crops (SRWC)

The SRWC are fast growing woody plants with a great range of adaptability and good disease resistance. The SRWC considered as bioenergy crops include hardwood species such as poplar (*Populus* ssp.), willow (*Salix* spp.), cottonwood (*Populus fremontii* L.), sweetgum (*Liquidambar styraciflua*), sycamore (*Platanus occidentalis*), black locust (*Robinia pseudoacacia*), silver maple (*Acer saccharinum* L.), and *Eucalyptus*. The SRWCs can be grown for other uses also such as paper production and the waste can be utilized for energy. Some of the species of considerable regional importance in the United States are alders (*Alnus* spp.), mesquite (*Prosopis* spp.), and the Chinese tallow (*Sapium sebiferum*) (Brown, 2003).

## IV. LAND SUITABLE FOR BIOENERGY CROPS

Bioenergy crops can be grown on marginal and highly degraded agricultural soils which must be protected against soil erosion. Reduced levels of soil organic carbon (SOC) in agricultural soils are associated with accelerated erosion and degradation caused by long-term cultivation or drastic disturbance by mining (Mann, 1986). Globally, there are 1,965 Mha of land affected by soil degradation of which a large fraction is affected by soil erosion, and 62 percent are prone to moderate and extreme forms of degradation (Oldeman, 1994).

The major causes of soil degradation are deforestation (29.4%), overgrazing (34.5 percent), and intensive agriculture (28.1%) (Table 2) (Oldeman, 1994). Losses of SOC pool by 30 percent or more have been reported due to cultivation (Li *et al*., 1994). Bioenergy crops can be grown on such marginal soils with low productivity. There are 10.8 Mha of severely eroded soils in the United States which may benefit from growing bioenergy crops and adoption of conservation-effective practices (Table 3). The

TABLE 2
Global and continental land area, distribution of eroded land, and degraded land by deforestation, overgrazing, and intensive farming

| | Land area[a] (Mha) | Soil degradation[b] (Mha) | | | | | Erosion[c] (Mha) |
|---|---|---|---|---|---|---|---|
| | | Deforestation | Overgrazing | Cropland | Total | % | |
| Africa | 2964 | 58 | 238 | 113 | 409 | 14 | 254 |
| Asia | 4376 | 292 | 194 | 200 | 686 | 16 | 311 |
| Australia | 811 | 13 | 83 | 8 | 104 | 13 | 86 |
| Europe | 1002 | 83 | 46 | 60 | 189 | 19 | 87 |
| N. America | 1832 | 17 | 38 | 92 | 147 | 8 | 76 |
| S. America | 2053 | 100 | 63 | 59 | 222 | 11 | 62 |
| Total | 13038 | 563 | 662 | 532 | 1757 | 13 | 900 |

[a]Oldeman, 1994.
[b]FAO, 1997.
[c]Middelton and Thomas, 1997.

TABLE 3
Severely eroded (T = soil loss tolerance) cropland and mineland in need of rehabilitation in the U.S.
(Adapted from Lal *et al*., 2004)

| State | 1000 ha | | | | Land fraction (%) | State | 1000 ha | | | | Fraction of land (%) |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Land cover | Severely eroded (>4T) | Mined land | Total | | | Land cover | Severely eroded (>4T) | Mined land | Total | |
| Alabama | 13391.6 | 41.7 | 49.2 | 90.9 | 0.68 | Nevada | 28635.4 | 29.8 | 75.7 | 105.5 | 0.37 |
| Arizona | 29526.2 | 84.4 | 62.2 | 146.6 | 0.50 | New Hampshire | 2403.2 | 0.4 | 34.3 | 34.7 | 1.44 |
| Arkansas | 13775.5 | 7.2 | 41.3 | 48.5 | 0.35 | New Jersey | 2016.9 | 9.5 | 35.0 | 44.5 | 2.21 |
| California | 41105.1 | 57.2 | 147.2 | 204.4 | 0.50 | New Mexico | 31492.7 | 172.1 | 112.3 | 284.4 | 0.90 |
| Colorado | 26959.7 | 557.5 | 72.4 | 629.9 | 2.34 | New York | 12719.0 | 28.3 | 103.7 | 132.0 | 1.04 |
| Connecticut | 1299.7 | 3.1 | 34.8 | 37.9 | 2.92 | North Carolina | 13641.4 | 115.5 | 78.5 | 194.0 | 1.42 |
| Delaware | 529.5 | 0.0 | 0.9 | 0.9 | 0.17 | North Dakota | 18312.0 | 90.4 | 42.7 | 133.1 | 0.73 |
| Florida | 15194.0 | 1.5 | 294.4 | 295.9 | 1.95 | Ohio | 10704.3 | 55.8 | 121.9 | 177.7 | 1.66 |
| Georgia | 15257.7 | 59.8 | 59.5 | 119.3 | 0.78 | Oklahoma | 18118.7 | 44.0 | 82.7 | 126.7 | 0.70 |
| Idaho | 21643.3 | 72.7 | 53.6 | 126.3 | 0.58 | Oregon | 25142.0 | 32.3 | 40.5 | 72.8 | 0.29 |
| Illinois | 14593.4 | 273.3 | 101.2 | 374.5 | 2.57 | Pennsylvania | 11734.9 | 75.4 | 443.3 | 518.7 | 4.42 |
| Indiana | 9372.1 | 82.7 | 62.7 | 145.4 | 1.55 | Rhode Island | 314.0 | 0.0 | 5.3 | 5.3 | 1.69 |
| Iowa | 14575.4 | 516.4 | 74.8 | 591.2 | 4.06 | South Carolina | 8058.2 | 9.8 | 26.7 | 36.5 | 0.45 |
| Kansas | 21310.0 | 111.1 | 103.3 | 214.4 | 1.01 | South Dakota | 19973.1 | 44.1 | 24.2 | 68.3 | 0.34 |
| Kentucky | 10466.1 | 102.3 | 253.6 | 355.9 | 3.40 | Tennessee | 10915.3 | 168.1 | 83.5 | 251.6 | 2.31 |
| Louisiana | 12367.7 | 5.4 | 43.5 | 48.9 | 0.40 | Texas | 69103.4 | 1671.2 | 300.8 | 1972.0 | 2.85 |
| Maine | 8615.7 | 0.0 | 43.3 | 43.3 | 0.50 | Utah | 21989.0 | 48.6 | 25.2 | 73.8 | 0.34 |
| Maryland | 2709.2 | 23.4 | 19.8 | 43.2 | 1.59 | Vermont | 2490.0 | 1.4 | 17.7 | 19.1 | 0.77 |
| Massachusetts | 2145.6 | 1.1 | 63.2 | 64.3 | 3.00 | Virginia | 10558.6 | 51.6 | 46.9 | 98.5 | 0.93 |
| Michigan | 15158.6 | 37.7 | 153.6 | 191.3 | 1.26 | Washington | 17647.9 | 180.2 | 65.8 | 246.0 | 1.39 |
| Minnesota | 21860.2 | 325.1 | 238.8 | 563.9 | 2.58 | West Virginia | 6276.0 | 3.8 | 81.2 | 85.0 | 1.35 |
| Mississippi | 12351.6 | 72.9 | 59.1 | 132.0 | 1.07 | Wisconsin | 14543.7 | 84.9 | 95.0 | 179.9 | 1.24 |
| Missouri | 18051.7 | 289.9 | 139.2 | 429.1 | 2.38 | Wyoming | 25332.8 | 180.6 | 161.4 | 342.0 | 1.35 |
| Montana | 38085.1 | 320.4 | 130.6 | 451.0 | 1.18 | | | | | | |
| Nebraska | 20035.1 | 60.5 | 27.3 | 87.8 | 0.04 | Total | 782502.1 | 6394.2 | 4434.1 | 10828.3 | 1.38 |

Midwest and the Southeast regions of the United States have been targeted as those where bioenergy crops may be competitive with traditional crops (Walsh *et al.*, 1999). Soils of the southern regions have a greater potential for SOC sequestration due to their low SOC content caused by a combination of long history of crop production and exposure to high temperatures and precipitations (Tolbert *et al.*, 2002). A systematic assessment of the roles of bioenergy crops in the C cycle is important to evaluating the potential of SOC sequestration.

## V. LAND DEGRADATION AND CARBON LOSSES

The SOC pool is a function of the dynamic equilibrium between C gains and losses from the system under a specific land use. Since the 1990s, inappropriate agricultural practices have contributed to the degradation of 562 Mha of the 1.5 billion of cropland worldwide (United Nations, 1996). The global SOC pool is about 1500 Gt (1 gigaton = $10^{15}$ g = billion Mg) in the surface meter of soil (Cole *et al.*, 1996) with C losses between 20 to 50 percent when soils are converted to agricultural land use (Davidson and Ackerman, 1993; Neill *et al.*, 1998).

Soil degradation (deforestation, overgrazing, and intensive agriculture) results in the disruption of the soil structural elements increasing erosion and runoff, and therefore, depletion of the SOC pool by exposing most of the C in the organic matter to oxidative processes (Follett, 2001). The global extent of soil degradation by erosion and other processes is shown in Table 2. The loss of the SOC pool is due primarily to three factors: (1) the reduction in plant roots and residue return, (2) the increase in biological activity as soil aeration is increased by cultivation and soil temperature, and (3) increase in soil erosion that removes carbon-rich materials (Sampson, 2000; Franzluebbers *et al.*, 2001). Information on the extent of land area affected by soil degradation and erosion provides an estimation of how much land can be converted to production of bioenergy crops, restoring the SOC pool and increasing the potential of $CO_2$ mitigation.

The global extent of soil at risk of degradation should bring awareness to policy and decision makers of the dangers resulting from inappropriate land use and soil mismanagement. Erosion rate has decreased in the last decade, but it still affects over 900 Mha worldwide, making up 51 percent of the total land area affected by soil degradation (Table 2). Most of the soil degradation by deforestation and inappropriate agriculture occurs in Asia while soil degradation by overgrazing occurs in Africa. A significant portion of the globally degraded soils are characterized by decrease in productivity and the attendant decline in SOC pool. Rehabilitation of degraded soil can be done effectively if bioenergy crops that are suitable for these regions can be established to improve soil productivity and restore the SOC pool. There is a pressing need to study the feasibility of C sequestration in these regions and the introduction of bioenergy species that can restore SOC pool.

Excessive cultivation of soil is an erosion hazard. Only 3 percent of the land in the United States is considered suitable for continuous cropping with minimal concerns to erosion potential, and 40 percent of the remaining land has row crop potential with limitations to continuous cropping (Kort *et al.*, 1998). Highly erodable land covers a large area in the United State (Table 3). The use of bioenergy crops can restore the SOC at the soil surface and also stabilize the soil with its deep root system.

## VI. BIOENERGY CROPS AND CARBON SEQUESTRATION

Soil C sequestration is a process in which plants remove $CO_2$ from the atmosphere and incorporate it into soil C pool along with other nutrients (N, P, and S). It is estimated that about 75 to 80 percent of the lost C can be re-sequestered in world soils, but ecological factors and management practices limit the rate of SOC sequestration (Wojick, 1999). Conversion of degraded agricultural soils to perennial crops can improve soil quality by increasing C sequestration due to their perenniality, high biomass production, and deep root systems (Ma *et al.*, 2000a). Replacing fossil fuels with bioenergy crops has a potential to reduce the rate of enrichment of atmospheric $CO_2$ because of the cumulative effects due to high biomass accumulation (Bransby *et al.*, 1998). However, bioenergy crops provide net gains in C sequestration only if they replace annual row crops.

Bioenergy crops are the link between sink (biomass and SOC) and the source (fossil fuel combustion). They are the sink/source transition since the C incorporated into their biomass and root system has a high potential for being incorporated into the SOC pool. Bioenergy crops can be used as a good option to sequester atmospheric $CO_2$ by increasing biomass productivity which can be incorporated into existing energy alternatives to improve energy use efficiency. One of the advantages of bioenergy crops is that aboveground biomass can be used to produce energy through combustion without increasing net $CO_2$ emissions. The net $CO_2$ produced comes from the fossil fuel in the production and process of biomass because the C in the aboveground biomass is recycled (Zan *et al.*, 2001).

### A. Biomass Production and Carbon Sequestration

Perennial crops are highly productive, have a high capacity to sequester C from the atmosphere, cause minimal soil disturbance during their growing season, and accumulate SOC over a 40–60-year period (Potter *et al.*, 1999). Assessment of how much of the C in biomass can be sequestered into the soil is important since most of the C stored in the aboveground biomass is to be utilized for energy production. This means that the release of $CO_2$ from co-firing of biomass does not contribute to the net global $CO_2$ levels since the $CO_2$ released during its utilization was recently removed from the atmosphere (Kort *et al.*, 1998).

The SOC is added to the soil mainly by deposition and decay of plant material on the surface and by root growth and senescence below the surface. There have been indications that the amount of C sequestered depends on biomass production and decomposition rate, and the fraction of the biomass that enters the long-term storage capacity (Quian and Follett, 2002). The massive and deep rooting systems in perennial crops allow for direct movement of C into the soil and make it less available for removal by harvest. Climate also affects the aboveground biomass productivity and $CO_2$ mitigation, but most perennial bioenergy crops can survive droughty conditions due to large nutrient reserves in their root system (Ingram and Fernandes, 2001).

### 1. *Aboveground Biomass*

Vegetative residues are important sources of replenishing SOC. The potential for soils to sequester C depends on the rates of biomass productivity relative to C exports controlled by microbial activity (Williams *et al*., 2004). Residue cover has a soil cooling effect and influences decomposition rates by moderating biological activity that could influence the incorporation of C into the organomineral complexes (Ingram and Fernandes, 2001). Cook and Beyea (2000) estimated biomass production of 5.4 Mg C ha$^{-1}$ yr$^{-1}$ by corn (*Zea mays*), 7.4 Mg C ha$^{-1}$ yr$^{-1}$ by switchgrass, and 8.0 Mg C ha$^{-1}$ yr$^{-1}$ by SRWC in a 3-year or 10-year rotation. The amount of biomass production by these cropping systems may reduce $CO_2$ emissions from fossil fuels by sequestering 400 kg C Mg$^{-1}$ of biomass in switchgrass, 500 kg C Mg$^{-1}$ in willow, and 600 kg C Mg$^{-1}$ in poplar, compared with only 300 kg C Mg$^{-1}$ by corn residue. Therefore, the amount of $CO_2$ sequestered in the aboveground biomass (in immobile organic to labile inorganic forms) can have a great impact on the amount of C sequestered in the soil, depending on how much is left on the soil surface.

### 2. *Belowground Biomass*

The prolific root system of perennial crops strongly influences C sequestration by adding significant quantities of organic matter into soil. The organic material containing soil C serves many roles including enhancing soil's capacity to retain and provide water and nutrients to plants. Switchgrass has four to five times more belowground biomass than corn with additions of 2.2 Mg C ha$^{-1}$ yr$^{-1}$ (Zan *et al*., 1997). Thus, removal of biomass of switchgrass or other bioenergy crops may not severely exacerbate the erosion hazard nor adversely affect the SOC. Perennial crops maintain considerable biomass below the typical cutting height and also their fibrous root network close to soil surface aids in soil stabilization and SOC sequestration (Kort *et al*., 1998). Nonetheless, region-specific information is needed to characterize C fluxes in land cultivated to bioenergy crops and how soil respiration affects $CO_2$ exchange in the ecosystem (Mielnick and Dugas, 2000).

The C and N pools are affected by the amount of biomass that is produced in both above- and belowground biomass (Table 4). Differences in root biomass as well as soil C vary with depth with larger percent of root biomass concentrated in the upper 35 cm of the soil profile (Table 5) (Tufekcioglu *et al*., 2003). Most of the root fractions differ among vegetation types and soil depths. Switchgrass has greater live fine roots than other vegetations. Root mass distribution of three switchgrass cultivars (Cave-in-Rock, Alamo, and Kanlow) has been reported up to a 3.3- m depth, with more than 50 percent of the root biomass concentrated in the top 30 cm of the soil profile (Figure 3) (Ma *et al*., 2000b). Differences in root mass distribution among cultivars are related to differences in growth habits and soil types (Figure 4) (Ma *et al*., 2000b). Fluctuations in SOC are usually larger in the top soil layers, probably due to greater effect of precipitation, soil temperature, larger root biomass, and microbial activity (Garten and Ma *et al*., 2000).

Differences in root biomass seem to indicate that cultivar selection along with soil type can affect the rate of SOC sequestration. Major C sequestrations occur belowground, where perennial root biomass, ephemeral roots, and decayed litter act as C sinks to mitigate atmospheric $CO_2$ enrichment (Table 4). Several studies on the assessment of root biomass pool indicate that switchgrass live fine root biomass decreases with depth (Ma *et al*., 2001), with the possibility that the amount of SOC sequestered follows the same pattern. Aboveground biomass and dead roots play an important role in C sequestration and its

TABLE 4

Carbon and nitrogen partitioning in buffers and adjacent crop fields along Bear Creek in central Iowa (Recalculated from Tufekcioglu *et al*., 2003)

| | Carbon pool | | | | | Nitrogen pool | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Aboveground litter | Aboveground biomass | Dead root | Live root | Total | Aboveground litter | Aboveground biomass | Dead root | Live root | Total |
| | | kg ha$^{-1}$ | | | | | kg ha$^{-1}$ | | | |
| Poplar | 1,667 | 17,500 | 417 | 3750 | 23,334 | 5,000 | 10,000 | 1,667 | 6,250 | 22,917 |
| Switchgrass | 8,333 | 1,667 | 417 | 3750 | 14,167 | 4,583 | 2,083 | 833 | 5,625 | 13,124 |
| Cool-season grass | 1,458 | 833 | 417 | 167 | 2,875 | 3,333 | 2,708 | 1,458 | 3,958 | 11,457 |
| Soybean | 625 | 2,708 | 208 | 208 | 3,749 | 833 | 4,792 | 625 | 833 | 7,083 |
| Corn | 1,042 | 417 | 208 | 417 | 2,084 | 1,250 | 2,917 | 625 | 833 | 5,625 |

TABLE 5
Root biomass distribution up to a 125 cm depth in buffer zones and adjacent crop fields along Bear Creek in Central Iowa (Recalculated from Tufekcioglu *et al*., 2003)

| Root fraction* | Vegetation type | | | | |
|---|---|---|---|---|---|
| | Poplar | Switchgrass | Cool-season grass | Soybean | Corn |
| | kg ha$^{-1}$ | | | | |
| 0–35 cm depth | | | | | |
| Live fine root | 5,822 | 8,880 | 6,455 | 6,83 | 917 |
| Dead fine root | 1,549 | 1,248 | 1,943 | 6,75 | 623 |
| Live small root | 561 | 1,899 | 364 | 1,99 | 114 |
| Dead small root | 123 | 225 | 96 | 74 | 64 |
| Coarse root | 3,119 | 0 | 0 | 305 | 1,124 |
| Total | 11,174 | 12,252 | 8,858 | 1,936 | 2,842 |
| 0–125 cm depth | | | | | |
| Live fine root | 7,620 | 13,487 | 8,177 | 894 | 1,209 |
| Dead fine root | 1,854 | 1,411 | 2,163 | 861 | 800 |
| Live small root | 848 | 1,918 | 427 | 199 | 118 |
| Dead small root | 167 | 225 | 97 | 74 | 65 |
| Coarse root | 3,932 | 0 | 0 | 305 | 1,124 |
| Total | 14,421 | 17,041 | 10,864 | 2,333 | 3,316 |

*Fine root = 0–2 mm, small root = 2–5 mm, coarse root = >5 mm.

turnover due to their C and N contents (Tufekcioglu *et al*., 2003). The C and N contents in the aboveground litter differed significantly among vegetation types (Table 4) and a higher percent of dead roots in the two crops (corn and soybean) due to high rate of root turnover during the winter. Tofekcioglu *et al*. (2003) observed that larger percentage of root biomass was concentrated in the upper 35 cm of the soil profile (Table 5).

Assessment of C sequestration in different ecosystems (switchgrass, willow, and corn) in southwestern Quebec (Figure 5) indicated that willow and switchgrass had greater SOC and N contents than corn (Mehdi *et al*., 1998). Switchgrass had a higher root C below 30 cm depth than corn or willow. Despite the similar high root biomass C in switchgrass and willow, soil C accumulation was higher under willow. Expected higher rate of litterfall from the SWRC can maintain high soil C levels in the system. On the other hand, when switchgrass is harvested, most of the biomass is removed. Therefore, only roots and small fraction of litter returned to the soil during the harvest process contributes to SOC in the system (Bransby *et al*., 1998). The SOC present under bioenergy crops (willow and switchgrass) is related to great root mass in the soil profile when compared to traditional crops like corn (Figure 6). Carbon

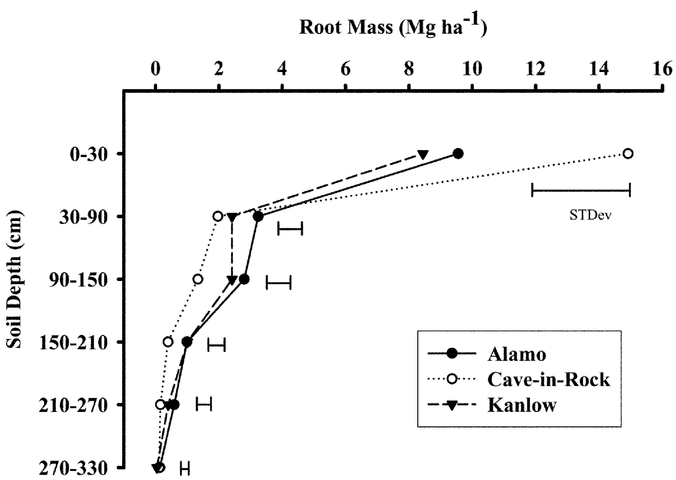


FIG. 3. Root distribution of switchgrass cultivars to soil depth at 330 cm (Redrawn from Ma *et al*., 2000b).

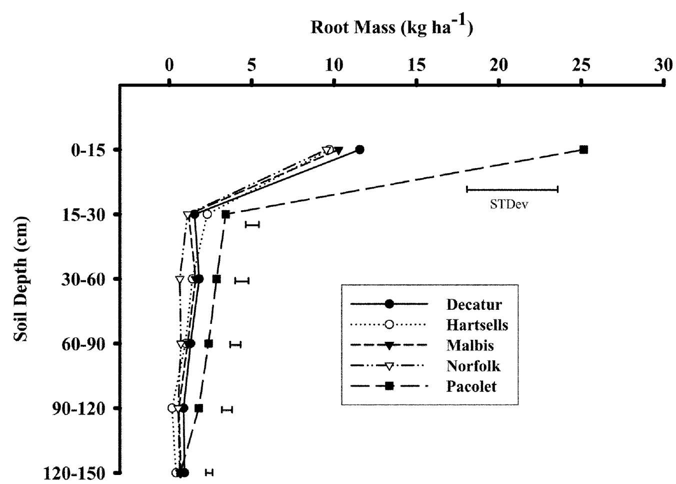


FIG. 4. Root mass distribution of switchgrass in various soils up to 150 cm depth (Redrawn from Ma *et al*., 2000b).

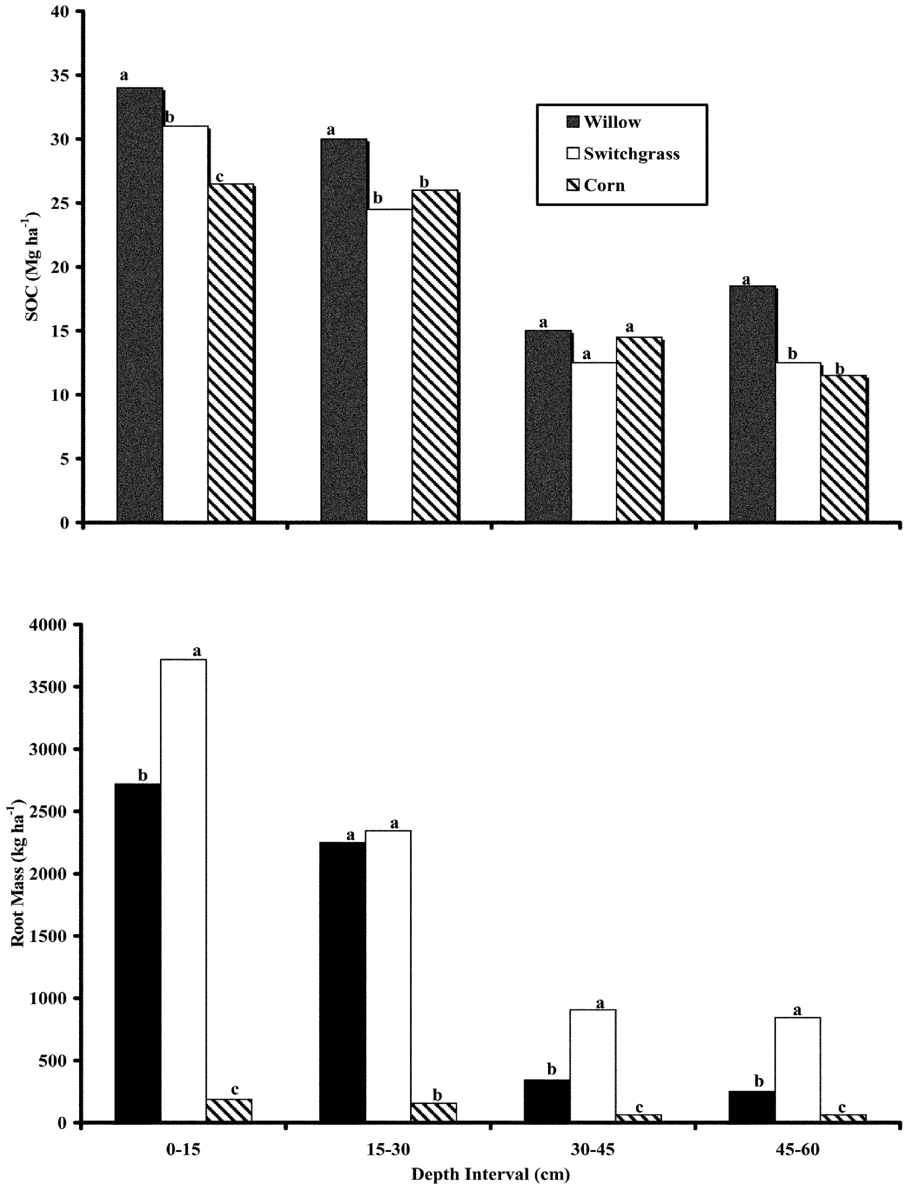


FIG. 5. Soil organic C and root mass in willow, switchgrass, and corn in southwestern Quebec (Adapted from Mehdi *et al.*, 1998). Data averaged over two locations.

oxidation decreases with depth due to reduction in microbial activity; deeper root allocations are more likely to have a stable SOC pool (Grigal and Berguson, 1998). It has been indicated that corn had greater aboveground biomass C returned to the soil while willow sequestered significantly more C in the soil (Table 6) (Zan *et al.*, 2001). The proportion of the total system C (biomass + root + SOC) under the willow was 14.4 and 15.6 percent more than in switchgrass and corn, respectively. Using corn as a reference or baseline for assessing soil C sequestration indicated that switchgrass needs a longer period of time to accrue significant gains, perhaps 10 years or more. Although these species exhibit distinct trends in C allocation, a baseline is needed for making better estimations of the true C being sequestered.

Root biomass is a critical component of total SOC and the soil C dynamics, but changes in root biomass and their contributions to SOC have not been extensively quantified (Gale and Cambardella, 2000). The dynamic portion of the belowground biomass is usually represented by fine roots (Buyanovsky *et al.*, 1987). Perennial warm-season grasses are characterized by well-developed root systems with a great root mass pool that is comparable to the annually produced aboveground biomass (McLaughlin and Walsh, 1998). Most of the C is generated by rhizosphere deposition and fine root turn over from the large active root pool averaging 3 Mg ha$^{-1}$ yr$^{-1}$ (Lynch and Whipps, 1991). The large pool of root biomass is not only a good sink for nutrient reserve, but also a sink to increase the SOM.

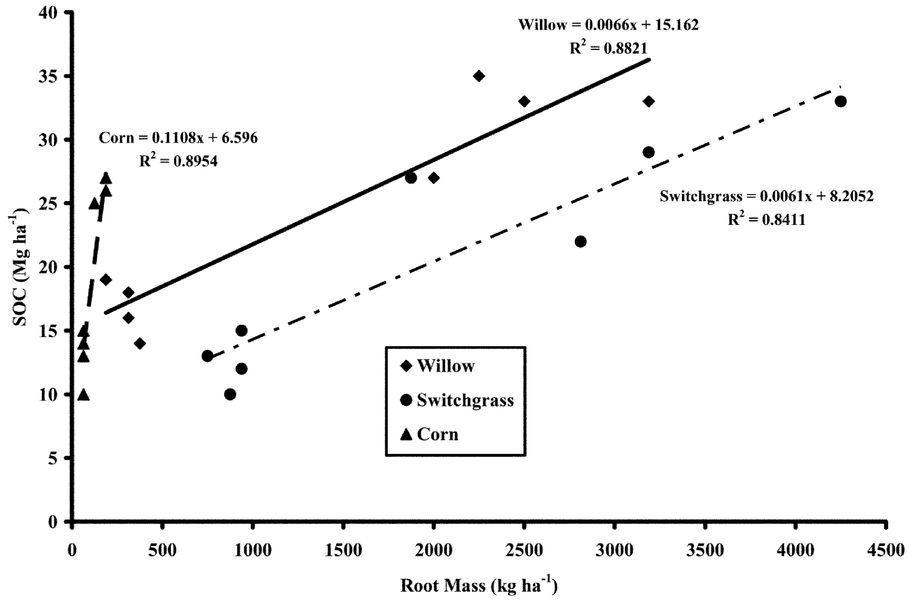


FIG. 6. Relationship between SOC and root mass of various land uses in southwestern Quebec (Adapted from Mehdi *et al*., 1998).

## VII. EFFECT OF SOIL PROPERTIES ON THE SOIL CARBON POOL

There is a strong need to assess how existing organic matter pools affect soil's capacity to sequester C under bioenergy crops. Changes in SOC pool affect soil quality and long-term sustainability (Doran and Parkin, 1994). The potential, attainable, and actual capacity of soils to sequester C is usually limited by specific factors that influence soil properties, biomass production, and management practices (Figure 7). In energy cropping systems soil texture, bulk density, and type of vegetation have high impact due to the relationship of these properties to SOC pool and aggregate stability.

TABLE 6
Rates and changes in biomass and soil C sequestration of three managed systems in southwestern Quebec, Canada over a 3-year period. Data were averaged over two locations (Adapted from Zan *et al*., 2001)

| | Species | | |
|---|---|---|---|
| Ecosystem pool | Willow | Switchgrass | Corn |
| Rate of C sequestration | Mg C ha$^{-1}$ yr$^{-1}$ | | |
| Biomass | 1.71 | 1.92 | 2.38 |
| Root* | 1.25 | 1.06 | 0.21 |
| Total | 2.96 | 2.98 | 2.59 |
| Soil* | 0.04 | 0.03 | 0.03 |
| Change with reference to corn | | | |
| Total biomass | +0.37 | +0.39 | — |
| Soil | +0.01 | 0.00 | — |

*Root and soil samples were collected to a 60 cm depth.

Soil texture affects both amount and retention of SOC from the residue of bioenergy crops (McConkey *et al*., 2003). The stability of SOC is determined by its adsorption on clay and silt particles indicating that soils with high clay and silt contents may enable the formation of micro- and macro-aggregates to further protect SOC (Hassink, 1997). Fine-textured soils have higher SOC contents than coarse-textured soils for the same level of organic inputs (Ingram and Fernandes, 2001). Also, research has indicated that crop residues decompose at higher rates in sandy than in clayey soils (Ladd *et al*., 1985). Fine silt and coarse clay particles contain the highest SOC per unit mass compared to the fine clay fraction (Anderson *et al*., 1981; Zhang *et al*., 1988). The particle's high surface area enhances the formation of organomineral complexes protecting the C from microbial oxidation (Grigal and Berguson, 1998). Therefore, soil texture controls the amount of C from crop residue that is retained in the soil (Liang *et al*., 1998). McConkey *et al*. (2003) also reported that the relative annual increase in SOC under no-till was 0.2 to 1.2 percent yr$^{-1}$, mainly depending on soil clay content. In addition to texture, potential C sequestration is also influenced by other soil properties such as redox potential, CEC, and concentrations of $Ca^{+2}$, $Fe^{+3}$, and $AL^{+3}$ (Grigal and Berguson, 1998). High cation concentrations (*e.g.*, $Ca^{+2}$) protect SOC against oxidation (Oades, 1988).

Soil bulk density also determines the potential of root biomass formation and turnover. In the upper 40 cm of the soil profile, it has been reported that soil bulk density in prairie land is usually lower than that in cropland and restored grassland, with no significant differences among the two cropping systems (Table 7) (Potter *et al*., 1999). The native prairie had the highest SOC concentration at each depth, and that in the restored grassland was intermediate to those in the prairie land and cropland. The SOC

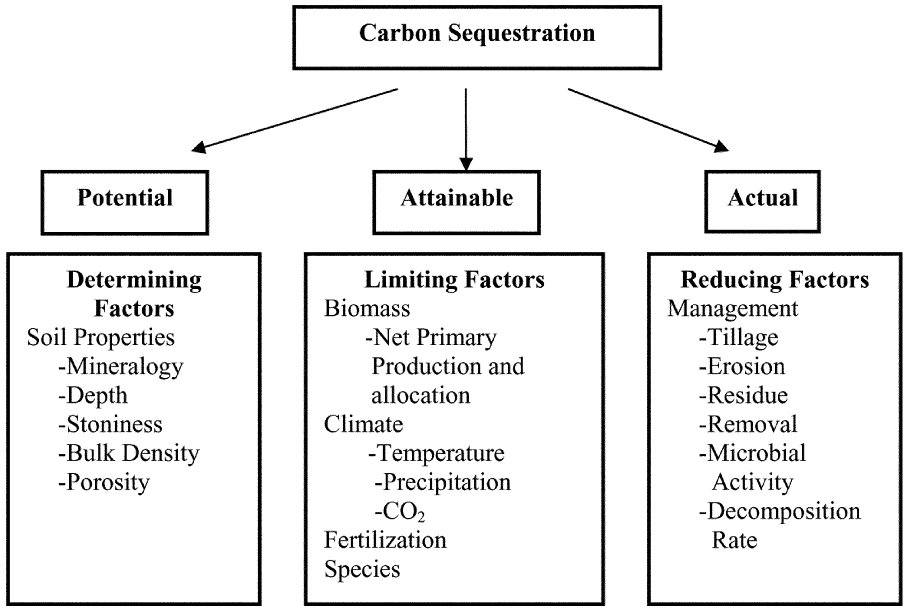


FIG. 7. Factors affecting soil organic C sequestration (Redrawn from the concept by Ingram and Fernandes, 2001).

concentration in the surface 120 cm of the cropland was 8 and 38 percent lower than those of the restored grass- and prairie land, respectively. Soil bulk density was also negatively correlated with SOC concentration (Figure 8), indicating that bulk density may increase with depth due to decrease in SOC concentration and root biomass. The SOC in the upper 15 cm was approximately equal in the grass, prairie, and cropland ecosystems. In a similar study, Baer *et al.* (2002) reported that soil bulk density (in the top 10 cm) decreased with time while total C (greatest change in surface 5 cm), microbial biomass C, and C mineralization rates increased.

TABLE 7
Soil bulk density and SOC of cropland, restored grasslands, and prairie land averaged across three locations in a Udic Haplusters soil in Texas (Modified from Potter *et al.*, 1999)

| | Bulk density ($Mg\ m^{-3}$) | | | Soil organic carbon pool ($Mg\ ha^{-1}$) | | |
|---|---|---|---|---|---|---|
| Depth (cm) | Cropland | Grass | Prairie | Cropland | Grass | Prairie |
| 0–5 | 1.19 | 1.06 | 0.88 | 10.4 | 15.1 | 23.2 |
| 5–10 | 1.21 | 1.25 | 1.04 | 10.4 | 12.4 | 17.1 |
| 10–15 | 1.36 | 1.32 | 1.15 | 10.3 | 11.6 | 16.4 |
| 15–20 | 1.39 | 1.36 | 1.22 | 10.0 | 11.1 | 15.2 |
| 20–30 | 1.39 | 1.41 | 1.32 | 18.1 | 20.0 | 26.6 |
| 30–40 | 1.41 | 1.44 | 1.38 | 15.8 | 17.9 | 23.8 |
| 40–60 | 1.45 | 1.47 | 1.46 | 27.3 | 27.1 | 38.4 |
| 60–80 | 1.52 | 1.52 | 1.52 | 21.9 | 21.8 | 30.1 |
| 80–100 | 1.60 | 1.59 | 1.57 | 17.6 | 17.2 | 26.1 |
| 100–120 | 1.66 | 1.65 | 1.61 | 12.4 | 12.6 | 21.3 |
| Total | — | — | — | 154.2 | 166.8 | 238.2 |

The vegetation type also affects the rates of SOC sequestration. Perennial grasses increase SOC storage since SOM is relatively stable and has long turnover time (Williams *et al.*, 2000). In riparian buffer settings, rates of C accumulation in plant and litter biomass in switchgrass and poplar stands averaged 2960 and 820 kg C $ha^{-1}$ $yr^{-1}$ (Tufekcioglu *et al.*, 2003). Differences among land uses (corn, grass, and prairie) in the upper 5 cm were inconsistent across locations with $C_4$ plants having the lowest SOC at most locations (Table 8). Although the C distribution varies among different grassland types, the relative distribution of SOC is very consistent within perennial grassland ecosystems. This is in contrast to some studies in which total SOC under forest increased with age (Zak *et al.*, 1990; Cook and Allan, 1992). In a study in Australia, annual grasses contained 42 percent less SOC than perennial grasses in the top 10 cm of soil (Ash *et al.*, 1995).

## VIII. RATES OF CARBON SEQUESTRATION

The rate at which C accumulates in the soil varies with biomass productivity, site history, management practices, and physical and biological properties of the soil (Post and Kwon, 2000). Estimates of maximum rates of C sequestration during early conversion to perennial vegetation are generally less than 100 g C $m^{-2}$ $yr^{-1}$ (Post and Kwon, 2000). The average rate of C accumulation can be similar between grassland and forestland (33.2 and 33.8 g C $m^{-2}$ $yr^{-1}$, respectively). Carbon sequestration rates of approximately 0.74 Mg C $ha^{-1}$ $yr^{-1}$ have been reported in land under the Conservation Reserve Program (CRP), 0.36 Mg C $ha^{-1}$ $yr^{-1}$ under conservation tillage row cropping systems,

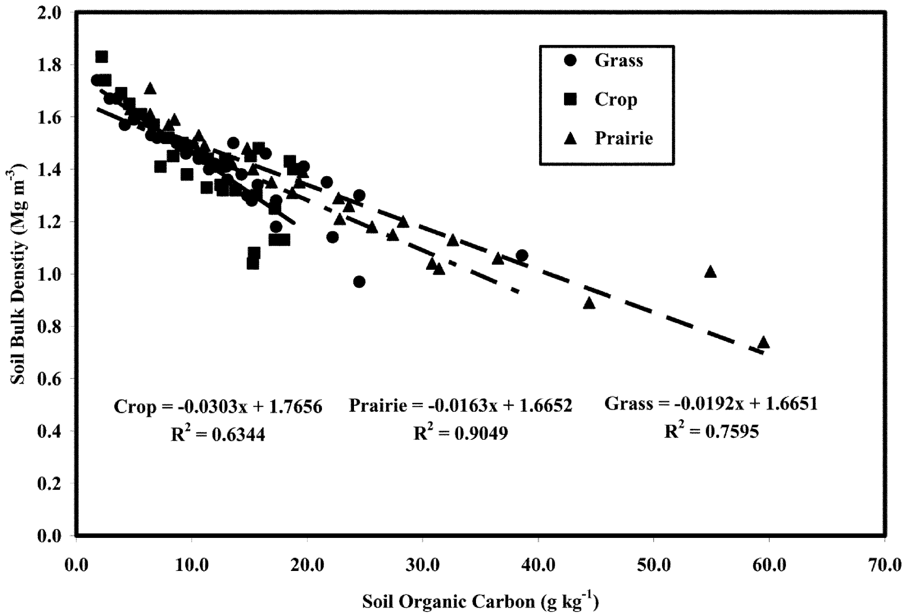


FIG. 8. Relationship between C concentration and soil bulk density of various land uses across different soil depths (Redrawn from Potter *et al*., 1999).

and 7.1 Mg C $ha^{-1}$ $yr^{-1}$ under new tree plantations (Eggers, 2000). Robertson and Shapouri (1993) estimated that grazing systems (>30 Mha) can sequester 30 to 110 Tg C $yr^{-1}$. These results showed that adoption of biofuel production systems reduce the use of fossil fuels and offset other fuel by converting atmospheric $CO_2$ to unharvested perennial biomass and SOC.

Perennial grasses may be more suitable for C sequestration since SRWCs take time for canopy closure, making soil more

TABLE 8
Total SOC at the 0–5 cm soil layer under forest, C3, and C4 grasses (Modified from Corre *et al*., 1999)

| Soil type | Vegetation | Species | Age (yr) | Total SOC (Mg C $ha^{-1}$) | Change (Mg C $ha^{-1}$ $yr^{-1}$) |
|---|---|---|---|---|---|
| Fine-loamy, mixed, mesic Typic Fragiudalf (PA) | Mixed Forest | White pine and Red maple | ∼ 30 | 16.60 | |
| | $C_3$ | Reed canarygrass | Unknown | 21.46 | — |
| | $C_4$ | Switchgrass | 15 | 17.81 | +0.64 |
| Fine-loamy, mixed, mesic Typic Dystrochrept (PA) | Mixed Forest | Red maple and Wild black cherry | ∼ 60 | 23.38 | |
| | $C_3$ | Orchardgrass | Unknown | 19.76 | — |
| | $C_4$ | Swtichgrass | 16 | 18.42 | +0.76 |
| Coarse-silty, mixed, mesic, Typic Dystrochrept (NY) | Mixed Forest | Red maple and Wild black cherry | 60 | 48.48 | |
| | $C_3$ Mix | Ryegrass, Kentucky bluegrass and Orchardgrass | Unknown | 35.16 | — |
| | $C_4$ | Switchgrass | 18 | 39.03 | +1.36 |
| Coarse-silty, mixed, mesic, Typic Fragiaqualf (PA) | Mixed Forest | White oak and Wild black cherry | 60 | 30.73 | |
| | $C_3$ | Bromegrass | Unknown | 35.19 | — |
| | $C_4$ | Switchgrass | 9 | 28.49 | +2.65 |
| Loamy-skeletal, mixed, mesic Typic Dystrochrept (PA) | Mixed Forest | White elm & Wild black cherry | 60 | 42.61 | |
| | $C_3$ | Orchardgrass | Unknown | 28.77 | — |
| | $C_4$ | Switchgrass | 15 | 28.28 | +1.16 |

prone to SOC loss (Harmon *et al.*, 1990). Carbon accumulation during the early aggrading state of grassland establishment on disturbed lands range from 0.33 to 0.54 Mg C $ha^{-1}$ $yr^{-1}$ (Post and Kwon, 2000; Conant *et al.*, 2001). The SOC sequestration rates ranged from 33.2 to 33.8 g C $m^{-2}$ $yr^{1}$ when changing land use from agriculture to grassland or forest, respectively (Post and Kwon, 2000). The differences between grassland and forest systems were not that large indicating that for the short-term economic return, perennial grasses may be advantageous.

## IX. MANAGEMENT OF BIOENERGY CROPPING SYSTEMS AFFECTING CARBON SEQUESTRATION

Two main concerns associated with the effects on the C cycle are the increased atmospheric $CO_2$ due to the use of fossil fuels and the loss of C and soil productivity from agricultural systems (Bransby *et al.*, 1998). Soils lose about 15 to 40 Mg C $ha^{-1}$ of their original C pool and can sequester 60 to 70 percent of the depleted C pool with the adoption species capable to produce high biomass (Lal, 2002). Improvement in soil quality under bioenergy crops depends on species capable of high productivity, fertilization, and harvest management.

### A. Species

Plant species differ in biomass production, and soil C storage is primarily controlled by two fundamental processes: net primary productivity (NPP) and decomposition. Increase in NPP results in an increased C storage, whereas increased decomposition has an opposite effect (Yang and Hsieh, 2002). Maintaining plant species with good vegetation cover and deep root systems such as perennial grasses are important to increasing SOC pool in deeper soil layers (Sommer *et al.*, 2000). Since SOC storage is driven by biological processes, increased mitigation of atmospheric $CO_2$ also depends on bioengineered species that have an increased plant C content, greater root biomass development, increased biomass residues, faster growing rates, greater nutrient use efficiency, more efficient microbial processes, and greater adaptation to highly degraded soils.

Management of degraded areas with more perennial crops (grasses and SRWC) can enhance soil quality and improve SOC sink capacity by improving plant productivity, soil structure, pH, and nutrient pool, and increasing the amount of biomass returned to the soil and incorporated into the SOC pool (Lal, 2003). Understanding how bioenergy crops can reduce erosion and increase C sequestration is crucial to mitigating enrichment of GHGs in the atmosphere.

### B. Fertilization

Soil fertility affects the amount of biomass produced. The SOC pool depends on the amount of biomass returned to the land and the amount of N present in the soil. Several studies have shown that regular application of fertilizers for many years leads to an increase in SOC (Schuman *et al.*, 2002; Rice, 2000; Reeder *et al.*, 1998). Most degraded lands utilized for bioenergy crops are usually deficient in N and can increase biomass production and water-use efficiency in response to N fertilization (Lemus, 2000). Fertilization stimulates biomass production and, therefore, enhances C accumulation (Schuman *et al.*, 2002). Rice (2000) reported that N fertilization increased biomass production in a tallgrass prairie with a 1.6 Mg C $ha^{-1}$ sequestered in the soil over a 5-year period. A 4-year-old CRP lot seeded with cool-season grasses and fertilized with 34 kg N $ha^{-1}$ also showed an increase in SOC in the top 10 cm of a Phiferson sandy loam soil (Reeder *et al.*, 1998). Management practices such as N fertilization, row spacing, and harvest frequency do not affect C sequestration by switchgrass (Ma *et al.*, 2000a), which may be related to the sampling time after the switchgrass establishment (two years).

Unless fertilizers are added, soil fertility declines with the production of bioenergy crops because of a large fraction of biomass removed from the land, thereby altering the C-N ratios. Improving fertilizer use efficiency in bioenergy crops is a key element in increasing the SOC pool. Fertilization rates for herbaceous crops usually range from 50 to 100 kg N $ha^{-1}$ depending on species, region, soil nutrient status, and climate. Fertilization is required to maintain the rapid growth rates in SRWCs, especially after the second rotation (three years after planting).

### C. Harvest Management

The NPP and plant growth capacity of bioenergy crops increase with harvest management. An increase in biological capacity has a higher demand for atmospheric $CO_2$ which may be incorporated into different plant components. Herbaceous perennial crops (*e.g.*, switchgrass) are usually harvested once a year in late fall when most leaves are still intact. Large switchgrass biomass production can significantly increase C sequestration with C concentration of 39-41% (Tufekcioglu *et al.*, 2003). These concentrations can be higher under multiple harvests where new growth requires more sequestered C for tissue formation (Lemus, 2004). Harvest management also affects C sequestration since cutting may shift the allocation of C from active root biomass to regrowth of leaves and in this way alters the SOC profile distribution associated with root activity (Ma *et al.*, 2000a). Therefore, SOC sequestration in herbaceous crops largely depends on root dynamics (Bransby *et al.*, 1998).

Most SRWCs grown for bioenergy purposes are harvested on a 3–10-year rotation and are generally harvested during the dormant season (winter) when significant quantity of nutrients are translocated to the roots and most of the leaf deposition contributes to SOC sequestration. Willows are more efficient at sequestering C on an annual basis than trees grown in native temperate forests, and production per ha can exceed that of the row crops (Ranney *et al.*, 1991). Rapid juvenile growth and high SOC sequestration rates can be sustained if the stands are harvested in short cycles.

There is no conclusive evidence of how harvest management affects SOC sequestration since most of the biomass under biofuel production is removed from the field with small return of organic matter to the soil. New studies which incorporate how these management strategies affect soil quality (especially C sequestration and soil degradation) are necessary because SOC is affected by the balance between inputs from NPP and outputs through decomposition (Conant and Paustian, 2002).

## X. POTENTIAL FOR CARBON SEQUESTRATION: LAND USE CHANGES

Estimating potential of SOC sequestration is more difficult in bioenergy crops than in grain crops. In order to quantify and manage C sequestration under bioenergy crops, it is necessary to deal with a large variability in soil types, landscapes, and management practices that over time may exert secondary effects on SOC sequestration. To date, long-term C responses under bioenergy crops have not been studied extensively as have been in croplands, and only few studies under selected conditions have been documented. Modifying current agricultural management practices by putting highly erodable land (HEL) into perennial crops as means of stabilizing the soil and sequestering C is a relatively cost-effective way to offset GHG emissions (McLaughlin and Walsh, 1998).

The U.S.-DOE (1999) and the IPCC (2000) have estimated the annual global SOC storage and C sequestration potential of changing land use towards more sustainable practices (Table 9). These studies show that productive capacity of the soil can be restored over the next 25 to 50 years by changes in soil management. These reports concluded that the largest SOC gain occurs through the conversion of arable land to agro-forestry and bioenergy crops since these two systems lead to accumulation and incorporation of litter, with large amounts of NPP allocated to root growth. Therefore, conversion to a sustainable agricultural system like bioenergy crops leads to an increase in annual SOC sequestration.

The information on how long it might take for C sequestration to reach equilibrium after bioenergy crops is more hypothetical. The new equilibrium level of SOC may take as long as 50 years (Lal *et al*., 1998). One problem with these assumptions is that interannual fluctuations in SOC contents are large enough to mask the influence of bioenergy crops on SOC pool estimates over a short period of time. Estimates of C sequestration may vary with time due to dynamics of C contained in the SOM and changes in soil bulk density (Markin *et al*., 1996). Surface residue, root mass, and soil bulk density vary within-season (Markin *et al*., 1996). Failure to consider these variable factors in the short term may affect the long-term estimates (Yang and Wander, 1999).

### A. Changes in Land Uses

Changes in SOC pool occur following deforestation and changes in land use. Carbon sequestration potentials are higher in humid temperate area (90 to 454 kg C ha$^{-1}$ yr$^{-1}$) than in semi-arid and tropical areas (45 to 182 kg C ha$^{-1}$ yr$^{-1}$) (Pretty and Ball, 2001). Depletion of the SOC up to 50 Mg C ha$^{-1}$ over a 100-yr period has been reported in agricultural systems of the eastern Amazonian region of Brazil (Sommer *et al*., 2000). These losses are attributed to decreases in organic C inputs, soil redistribution, and leaching of soluble organic C (Slobodian *et al*., 2002). An experiment in north-central India showed that different tree species of SRWCs can sequester up to 45 Mg C ha$^{-1}$ over an 8-yr period (Garg, 1998). Measurements of SOC stocks, losses, and rate of accumulation in Brazil, Cameron, and Indonesia indicated that SOC accumulation is much higher in the aboveground biomass (1.8 Mg C ha$^{-1}$ yr$^{-1}$) than in soils (182 to 545 kg C ha$^{-1}$ yr$^{-1}$) (Palm *et al*., 2000). On the basis of the data on land degradation (Table 2), estimated biomass production of 4 to 8 Mg ha$^{-1}$ yr$^{-1}$ and a SOC sequestration rate

TABLE 9
Estimates of soil C storage and C sequestration in various land use systems which may be sustained over the next 50 years (U.S DOE, 1999; IPCC, 2000; NRI, 2004)

| System | Land area (Mha) | Soil C storage (Pg yr$^{-1}$) | Accumulated C under improved management within land use (Mg C ha$^{-1}$ yr$^{-1}$) | Accumulated C with land use change (Mg C ha$^{-1}$ yr$^{-1}$)* |
|---|---|---|---|---|
| Cropland | 150 | 0.85–0.90 | 0.3 | — |
| Biofuel croplands | 13 | 0.50–0.80 | 0.6–1.2 | 0.1–0.4 |
| Grasslands | 47 | 0.50 | 0.5–0.7 | |
| Rangelands | 164 | 1.20 | — | — |
| Forest | 164 | 1.00-3.00 | 1.0–3.0 | — |
| Degraded lands | 109 | 0.80-1.30 | 0.8–3.0 | — |
| Agroforestry | 21 | — | 0.3–0.5 | 3.1 |
| Grassland | | — | — | 0.8 |

*Land use change from arable land.

TABLE 10
Potential of bioenergy crops for $CO_2$ mitigation in the U.S.

| | | Bioenergy crops | | | | |
|---|---|---|---|---|---|---|
| | | | SRWC | | | |
| Properties | Units | Switchgrass | Poplar | Willow | Total | References |
| Biomass | | | | | | |
| Production | Mg $ha^{-1}$ $yr^{-1}$ | 15.0 | 11.3 | 9.1 | 35.4 | Lemus (2004); Turhollow and Perlack (1991); Tuskan (1998); Volk *et al*. (1999) |
| Carbon | % | 46.0 | 50.0 | 49.4 | — | Lemus (2004); Heller *et al*. (2004) |
| Carbon content | Mg C ha $yr^{-1}$ | 6.9 | 5.7 | 4.5 | 17.1 | |
| Combustion efficiency* | Mg C ha yr-1 | 5.2 | 4.3 | 3.4 | 12.9 | |
| SOC | Mg C $ha^{-1}$ $yr^{-1}$ | 0.8 | 1.1 | 0.9 | 2.8 | Ma *et al*. (2000a); Cook and Beyea (2000); Sanchez *et al*. (2003) |
| Total C (biomass + soil) | Mg C $ha^{-1}$ $yr^{-1}$ | 6.0 | 5.4 | 4.3 | 15.7 | |
| Land availability | | | | | | |
| Severely eroded + Mineland | Mha | 6.0 | 2.4 | 2.4 | 10.8 | Lal *et al*. (2004) |
| Highly eroded | Mha | 23.8 | 11.9 | 11.9 | 47.6 | Lal *et al*. (2004) |
| Total | Mha | 29.8 | 14.3 | 14.3 | 58.4 | |
| Potential biofuel production | Tg $yr^{-1}$ | 447.0 | 162.6 | 130.1 | 739.7 | |
| Potential SOC sequestration | Tg C $yr^{-1}$ | 23.8 | 15.7 | 12.9 | 52.4 | |
| Potential sequestered C | Tg C $yr^{-1}$ | 178.8 | 77.2 | 61.5 | 317.5 | |
| $CO_2$ Emission from fossil fuel combustion | Tg C $yr^{-1}$ | | 1576.8 | | | U.S.-EPA, 2003 |
| $CO_2$ offset | % | 11.3 | 4.9 | 3.9 | 20.1 | |

*Combustion efficiency is based on 75% direct combustion from coal (Taftan Data, 1998).

of 0.2 to 0.5 Mg C $ha^{-1}$ $yr^{-1}$, the potential of world biofuel production can be 7 to 16 Tg C $yr^{-1}$ with a total SOC addition of 0.3 to 0.9 Tg $yr^{-1}$.

In the U.S., bioenergy crops have a biofuel potential of 740 Tg $yr^{-1}$ with a total SOC addition of 52 Tg $yr^{-1}$ (Table 10). Carbon losses due to land conversion in the Great Plains range from 15 to 30 percent (Janzen *et al*., 1998). Tolbert *et al*. (2002) observed that a 3-year conversion of cropland to switchgrass and no-till corn for grain production increased SOC by 0.40 percent and 0.33 percent in upper 10 cm layer, respectively. Garten and Wullschleger (2001) predicted that average SOC sequestration rate on land converted to switchgrass is 0.78 Mg C $ha^{-1}$ $yr^{-1}$ during a 10-year period and 0.53 Mg C $ha^{-1}$ $yr^{-1}$ over a 30-year period and with higher gains in the south-central region (Figure 9). If 10.8 Mha of severely eroded land (Table 3) is converted to switchgrass, it has the potential to sequester 8.4 Tg C $yr^{-1}$ during the first 10-year period. Soils under pastures have more C and N stocks in the upper 20 cm than those under transitional vegetation and forest (Figure 10) (Garten and Wullschleger, 2000), indicating that land cover is a major determinant of C and N distribution in the soil profile. Perennial herbaceous crops can contribute to more efficient recoveries of SOC in areas where it has been depleted by a long-term cultivation.

The SRWCs enhance the terrestrial C pool by SOC sequestration (Hansen, 1993) and in long-lived wood products (Marland and Marland, 1992). Changes in C storage and soil quality have been observed over a 1- to 10-year period by soil management with SRWC (Grigal and Berguson, 1998). In some cases, however, the SOC pool may decline in the early years of SRWC establishment due to mineralization of SOM in the upper soil profile, but SRWCs quickly become a net C sink. Hansen (1993) reported that a 12–18-year-old plantation of hybrid poplar increased SOC pool at the rate of 1.6 Mg C $ha^{-1}$ $yr^{-1}$, when compared to an adjacent cropland. Studies of SOC changes over time in abandoned agricultural lands and aggrading forests shows a substantial net increase in SOC pool over a 40- to 50-year period relative to the antecedent SOC pool under agricultural land (Johnson, 1992). Grigal and Berguson (1998) hypothesized that SRWC may increase C pool by 10-15 Mg $ha^{-1}$ in a 10–15-year rotation.

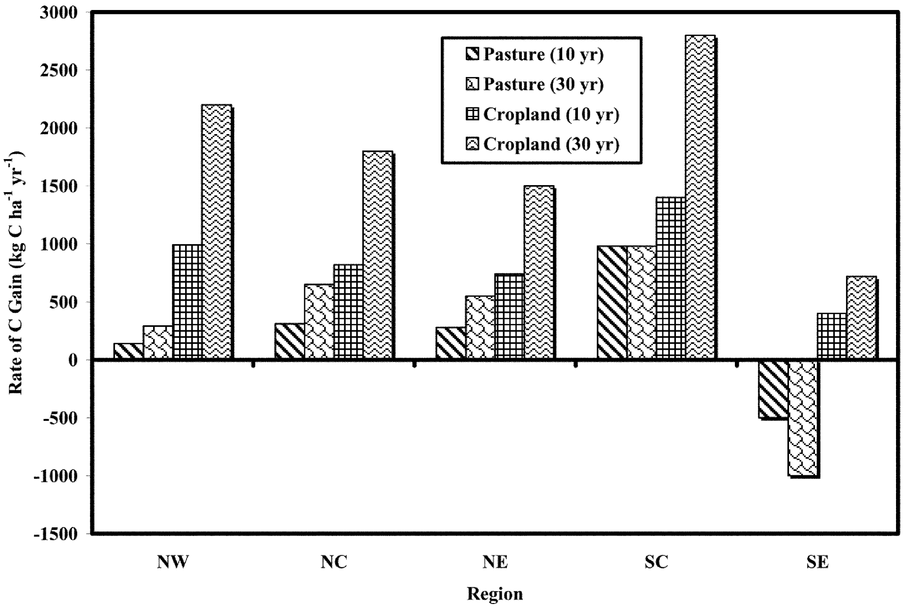


FIG. 9. Predicted regional (NW = northwest, NC = north-central, NE = northeast, SC = south-central, and SE = southeast) gains in soil organic carbon with land conversion from pasture or cropland to switchgrass in a 10- or 30-yr period (Modified from Garten and Wullschleger, 2001).

## XI. MODELING

Modeling C sequestration in bioenergy crops involves the knowledge of several factors (Figure 11) including their temporal and spatial variability, changes in management practices, and species composition. Donigian *et al.* (1998) reported that in a study involving 60 to 70 percent of the U.S. cropland, C sequestration rate can be about 1 to 2 Pg C by 2030. These estimates are based on the assumption that crop yield may increase by 0.5 percent per year, although conservation practices may be highly variable across regions. The model predicted a 15 percent increase in SOC with reduced tillage practices and up to 50 percent with no-till. However, new studies that confirm/support these findings are essential since the data base is rather limited. Ecosystem models can provide a good source of information about the C sink produced by energy crops in the United States. Simulation models (Li *et al.*, 1994) have reported that variations in soil texture and temperature impact the long-term SOC pool, which increases with decrease in temperature and increase in the clay content. However, these models do not include changes in land use such as shifting from row crops to bioenergy crops or to no-till systems, and such changes play a very important role in the C inventory analysis.

Carbon sequestration is also determined by plant growth, senescence, and decomposition (Houston and Marland, 2003). Plant growth determines the rate of NPP, and senescence determines how long plants live, how much biomass accumulation occurs and how much C is stored. Biomass decomposition determines the amount of plant residue left on the ground and how much of it is incorporated into the SOC pool. Ma *et al.* (2000b) indicated that C mineralization, microbial biomass C, and percent microbial biomass C increased after two years of switchgrass establishment. Carbon mineralization increased by 112 and 254 percent at 0–15 and 15–30 cm depths; microbial biomass increased 168 percent in the top 15 cm of soil, and C turnover increased by 116 and 255 percent in the 0–15 and 15–30 cm depths. The data also indicated that C sequestration is positively related to root biomass and plant growth rates as determined by climate and soil nutrient status. Fertilization and other management practices also affect the size and distribution of biomass production.

Models are useful as long-term predictors of changes in the SOC pool, but most of them fail to take into account changes in climate, land use, and management practices that take place over time, as well as soil physical and chemical properties. Considering the sink's future for C allocation due to shift in land use, resource allocation, changes in $CO_2$ concentration, N fertilization, climatic changes, or soil types are important factors (Pacala *et al.*, 2001). Future studies targeted to introduce a model that incorporates land use by bioenergy crops, traditional crops, and pasture areas are essential to understanding C sinks, how land use alters C storage, and to determine the functional aspects of energy crops from the environmental point of view. The new areas of investigation must include modeling of long-term trends in C sequestration under bioenergy crops, comparison with crop-based rotations, management practices, changes in expected biomass and residue, and species composition.

## XII. POLICY OPTIONS

A large potential of C sequestration lies in restoring degraded soils. Adoption of recommended management practices (*e.g.*, conservation tillage, integrated nutrient management, and

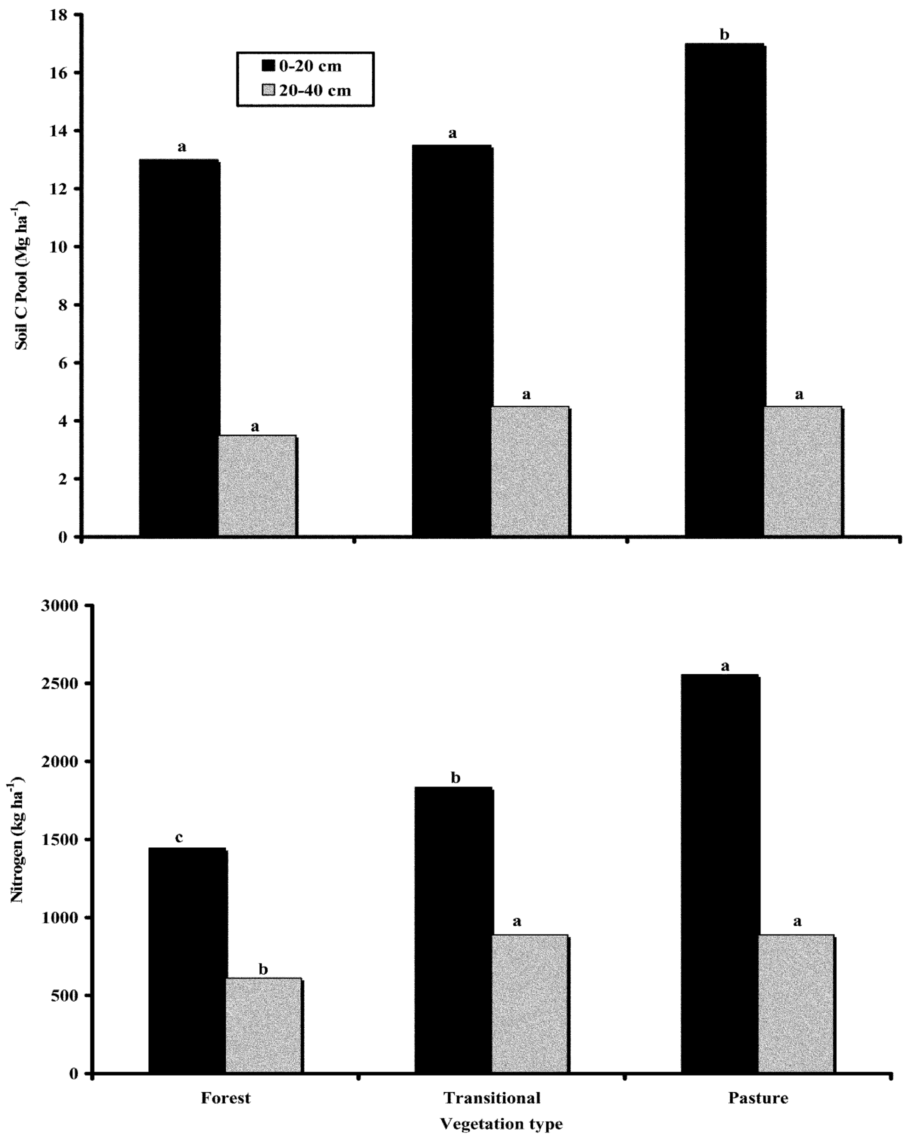


FIG. 10. Mean soil C and N stocks under three types of vegetation (forest, transitional, and pasture) and two soil depths (Redrawn from Garten and Ashwood, 2002).

replacement of annuals with perennials) can enhance terrestrial C pool at the rate of 3 Mg C ha$^{-1}$ during a 10-year period (Janzen *et al.*, 1998). Conversion from annual to perennial crops (herbaceous and SRWC) can buffer negative impacts such as decreasing soil erosion, increasing SOC pool, improving soil quality, and reducing GHG emissions (Cook and Beyea, 2000).

Land-management practices directed to bioenergy crops can increase the SOC stock and earn C credits toward national emissions targets. Sequestering $CO_2$ from the atmosphere also can be used to implement a C credit trading system, which may provide economic incentives to producers involved in CRP (Marland *et al.*, 2001). Such estimates are needed by policymakers, if global problems such as GHGs emissions, soil degradation, and $CO_2$ enrichments are to be effectively addressed. Policy changes must also focus on finding new avenues to divert public support to farmers in the form of stewardship or green payments as well as making progress in encouraging the success of private C trading systems.

Although some conservation programs offer tax benefits to farmers, most producers are reluctant to take their land out of production for a long period of time. Policies that set an even price on C emissions encourage the energy sector to devote land that can be used for the establishment of bioenergy crops and gain C credits for achieving C reductions. Thus, there is a need to estimate stand age, accumulative biomass production and the antecedent SOC levels to determine proper C storage capacity.

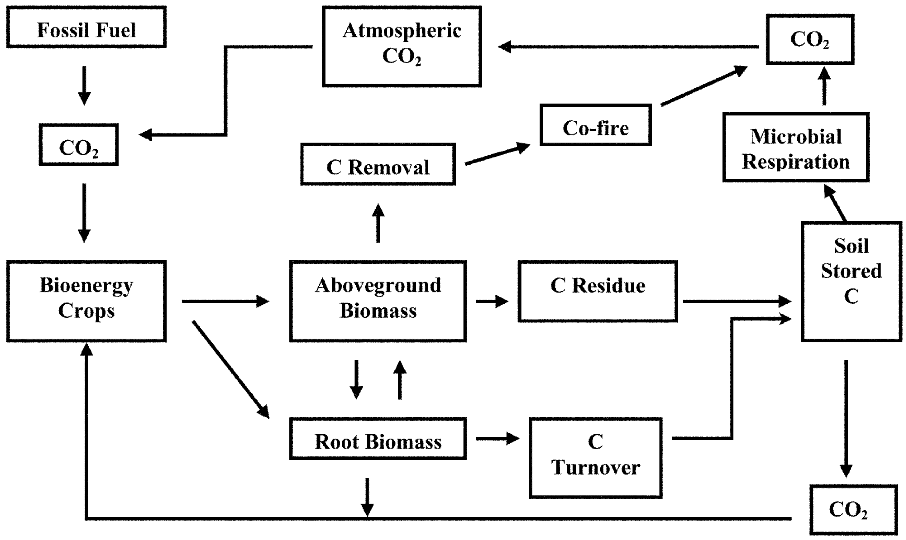


FIG. 11. Carbon partitioning and translocations in bioenergy crop production system.

These measures can increase the amount of C stored, either above and/or below ground, while others can decrease the loss of C from the biosphere. A number of bioenergy related strategies are also possible, such as longer bioenergy crop rotations, improved management of existing bioenergy crops, and C retention enhancing biomass harvest practices.

A balanced system of taxes and subsidies needs to be in place to allow bioenergy crops to be planted on a large scale to decrease C losses and compete fairly in the energy market. The United States already has a policy of land conversion through the CRP, which reduces erosion and enhances SOC pool through the use of perennial covers crops (warm-season grasses and SRWC). Those HELs which cannot be protected against erosion with recommended management practices (RMPs) are incorporated into this program. The government shares the cost of converting cropland to alternative uses such as woodland and bioenergy crops, giving an opportunity to farmers to preserve their land by increasing the SOC pool while at the same time producing a crop that helps to meet future energy needs.

While there is no nationwide commitment to reduce $CO_2$ emissions, policies such as federal C credits or state-level initiatives exist to promote C sequestration and renewable energy production (Hassol and Udall, 2003). The SOC sequestered in these soils can also have a large economic value, which requires establishing guidelines for granting C credits in CRP, and methodologies for obtaining reliable C inventories to determine temporal changes in the SOC pool.

For a strong impact of bioenergy crops on SOC sequestration, sink must become permanent. The important policy questions must focus on how to establish permanent or indefinite sinks, how to prevent C losses, how to agree in measures and whether the cost of implementation can be justified through the potential of C sequestration. In a sense, bioenergy crops represent an important new source of income for farmers, as well as helping to encourage farmers to adopt a wide variety of sustainable practices to decrease C losses.

## XIII. CONCLUSIONS

The C sink capacity in soils is large, both in the United States and globally. Replacing fossil fuels with bioenergy crops can reduce the net flow of $CO_2$ to the atmosphere. The efficiency of this substitution must be based on reduced emissions per unit of used land or biomass. Available land and yield potentials in bioenergy crop production have been subjected to different interpretations. Bioenergy crops have the potential to sequester 317.5 Tg C $yr^{-1}$ or 20% of the total annual U.S. emissions (Table 10) based on biomass yields, the land area dedicated to crop production, the estimated C sequestration potential, and the conversion efficiency. The C mitigation per unit of land is very large with bioenergy crops specifically grown to decrease the C emission from fossil fuel. Converting cropland to bioenergy crops may increase C sequestration in SOM and contribute to atmospheric $CO_2$ mitigation strategies.

As the national and international markets for C trading continues to grow, the sequestered C represent an important new source of income for farmers while helping to preserve the land by adopting a wide range of sustainable practices. Understanding soil C dynamics in bioenergy crops is important since C sequestration can influence biomass production, ecosystem sustainability, soil fertility, and soil structure. Determining the C pool in bioenergy crops can indicate their importance to SOC sequestration. Baseline data along with cropping history are needed for determining these parameters.

Assessing SOC dynamics requires measurements over decades since changes in SOC occur very slowly and annual changes are rather small. Most experiments fail to meet these criteria because they focus on individual and short-term sampling times in which changes in SOC are very small and hard to detect. Although short-term studies indicate that SOC can be significantly improved with perennial biomass production, there is the need of a regional evaluation program for biomass production to determine the long-term soil improvement for various soil types and previous land use characteristics.

Determining the capability of perennial bioenergy crops for SOC sequestration necessitates new studies which combine physical properties, management practices, and biological interaction, as well as plant species to obtain a more precise estimate of the benefits offered by the new bioenergy era. Long-term experiments which address SOC dynamics in soil managed for biomass feedstock production are needed to improve the understanding and capability of bioenergy crops to sequester C over short- and long-term time scales.

More research that involves integration of C data, models, and their relationship to conservation policies is necessary to offer farmers the opportunity of C credits in exchange for conservation initiatives. For perennial crops, there is the need to assess SOC sequestration baselines to perform a cost-benefit analysis to support sustainable development of bioenergy crops because changes in SOC sequestration (in soil and biomass) are necessary to develop future design and monitoring protocol.

A broad base of knowledge of general and qualitative effects of soil management on SOC sequestration, and quantitative estimations of rates of C sequestration in bioenergy crops need to be greatly improved. More information is needed about the type of sequestered C and how stable those compounds are over time. There must be a better understanding of the accumulation and degradation of the SOC pool, and an understanding of how improved N use efficiencies may affect C sequestration.

## REFERENCES


Anderson, D. W., Saggar, S., Bettany, J. R., and Stewart, J. W. B. 1981. Particle size fractions and their use in studies of soil organic matter: I. The nature and distribution of forms of carbon, nitrogen, and sulfur. *Soil Sci. Soc. Am. J.* **45**: 767–772.

Ash, A. J., Howden, S. M., and McIvor, J. G. 1995. Improved rangeland management and its implications for C sequestration. **In**: *Proceedings of the Fifth International Rangeland Congress*, July 23–28, pp. 19–20. Salt Lake City, Utah.

Baer, S. G., Kitchen, D. J., Blair, J. M., and Rice, C. W. 2002. Changes in ecosystems structure and function along a chronosequence of restored grasslands. *Ecol. Applic.* **12**(6): 1688–1701.

*Biomass*. 2004. On line: http://www.fae.sk/Dieret/Biomass/biomass.html

Bransby, D. I., McLaughlin, S. B., and Parrish, D. J. 1998. Soil carbon changes and nutrient cycling associated with switchgrass. *Biomass and Bioenergy* **14**: 379–384.

Brown, R. C. 2003. *Biorenewable Resources: Engineering New Products from Agriculture*. Iowa State Press, Ames, IA. pp. 59–108.

Buyanovsky, G. A., Kucera, C. L., and Wagner, G. H. 1987. Comparative analyses of carbon dynamics in native and cultivated ecosystems. *Ecology* **68**(6): 2023–2031.

Cannell, M. G. R. 2003. Carbon sequestration and biomass energy offset: Theoretical, potential, and achievable capacities globally, in Europe and the UK. *Biomass and Bioenergy* **24**: 97–116.

Cole, V., Cerri, C., Minami, K., Mosier, A., Rosenberg, N., Sauerbeck, D., Dumanski, J., Duxbury, J., Freney, J., Gupta, R., Heinemeyer, O., Kolchugina, T., Lee, J., Paustian, K., Powlson, D., Sampson, N., Tiessen, H., van Noodwijk, M., and Zhao, Q. 1996. Agricultural options for mitigation of greenhouse gas emissions. **In**: *Climate Change 1995. Impacts, Adaptations, and Mitigation of Climate Change: Scientific-Technical Analyses.* IPCC Working Group II. pp. 745–771. Cambridge, Cambridge University Press.

Conant, R. T., and Paustian, K. 2002. Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochem. Cycles* **16**(4): 90/1–90/9.

Conant, R. T., Paustian, K., and Elliott, E. 2001. Grassland management and conversion into grassland: Effects on soil carbon. *Ecol. Applic.* **11**(2): 343–355.

Cook, B. D., and Allan, D. L. 1992. Dissolved organic carbon in old field soils: Total amounts as a measure of available resources for soil mineralization. *Soil Biol. Biochem.* **24**: 585–594.

Cook, J., and Beyea, J. 2000. Bioenergy in the United States: Progress and possibilities. *Biomass and Bioenergy* **18**: 441–445.

Corre, M. D., Schnabel, R. R., and Shaffer, J. A. 1999. Evaluation of soil organic carbon under forests, cool-season, and warm-season grasses in the northeastern U.S. *Soil Biol. Biochem.* **31**: 1531–1539.

Davidson, E. A., and Ackerman, I. L. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry* **20**: 161–164.

Donigian, A. S. Jr., Patwardhan, A. S., Chinnaswamy, R. V., and Barnwell, T. O. 1998. Modeling soil carbon and agricultural practices in the Central U.S.: An update of preliminary results. **In**: *Soil Processes and the Carbon Cycle,* pp. 499–518. Lal, R., Kimble, J. M, Follet, R. F., and B. A. Stewart, B. A., Eds., CRC Press, Boca Raton, FL.

Doran, J. W., and Parkin, T. B. 1994. Defining and assessing soil quality. **In**: *Defining Soil Quality for a Sustainable Environment,* pp. 3–21. Doran, J. W., Coleman, D. C., Bezolicek, D. F., and Steward, B. A., Eds., Soil Sci. Soc. Spec. Publ. 35, Soil Sci. Soc. Am. and Am. Soc. Agron., Madison, WI.

Eggers, T. 2000. *Carbon Sequestration. Farm Economics.* On line: http://www.exnet.iastate.edu/feci/carbon/homepage.html

FAO. 1997. *Main Causes of Dryland Soil Degradation by Region.* Online: http://www.fao.org/NEWS/FACTFILE/FF9710-E.HTM

Follett, R. F. 2001. Soil management concepts and carbon sequestration in zin cropland soils. *Soil Tillage Res.* **61**: 77–92.

Franzluebbers, K., Franzluebbers, A. J., and Jawson, M. D. 2001. Bermudagrass management in the Southern Piedmont USA: I. Soil and surface residue carbon and sulfur. *Soil Sci. Soc. Am. J.* **66**: 254–262.

Gale, W. J., and Cambardella, C. A. 2000. Carbon dynamics of surface residue- and root-derived organic matter under simulated no-till. *Soil Sci. Soc. Am. J.* **64**: 190–195.

Garg, V. K. 1998. Interaction of tree crops with a sodic soil environment: Potential for rehabilitation of degraded environments. *Land Degrad. Dev.* **9**: 81–93.

Garten, C. T., and Ashwood, T. L. 2002. Landscape level differences in soil carbon and nitrogen: Implications of soil carbon sequestration. *Global Biogeochem. Cycles* **16**(4): 61/1–61/14.

Garten, C. T., and Wullschleger, S. D. 2000. Soil carbon dynamics beneath switchgrass as indicated by stable isotope analysis. *J. Environ. Qual.* **29**: 645–653.

Garten, C. T., and Wullschleger, S. D. 2001. *Soil Carbon and Nitrogen Dynamics under switchgrass: Integration of Field Data and Modeling to Ensure Sustainability.* Bioenergy Feedstock Development Programs: Subcontractors and Collaborators Meeting, Memphis, TN, November 7–9, 2001. Oak Ridge National Laboratory, Oak Ridge, TN. On line: http://bioenergy.esd.ornl.gov/02workshop/garten.html

Grigal, D. F., and Berguson, W. E. 1998. Soil carbon changes associated with short rotation systems. *Biomass and Bioenergy* **14**(4): 371–374.

Hall, D. O., and House, J. I. 2004. *Biomass Energy Development and Carbon Dioxide Mitigation Options.* Biomass Energy Development. Online: http://www.uccee.org/CopenhagenConf/hall.htm

Hansen, E. A. 1993. Soil carbon sequestration beneath hybrid poplar plantations in the north central United States. *Biomass and Bioenergy* **5**: 431–436.

Harmon, M. E., Ferrell, W. K., and Franlkin, J. F. 1990. Effects of carbon storage of conversion of old-growth forests to young forests. *Science* **247**: 699–702.

Hassink, J. 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. *Plant and Soil* **191**: 77–87.

Hassol, S. J., and Udall, R. 2003. A change of climate. *Issues Sci. Technol.* **19**(3): 39–46.

Hawken, P. 1993. *The Ecology of Commerce: A Declaration of Sustainability.* Harper Business, New York, p. 21.

Heller, M. C., Keoleian, G. A., Mann, M. K., and Volk, T. A. 2004. Life cycle energy and environmental benefits of generating electricity from willow biomass. *Renewable Energy* **29**: 1023–1042.

Houston, M. A., and Marland, G. 2003. Carbon management and biodiversity. *J. Environ. Manage.* **67**: 77–86.

Ingram, J. S. I., and Fernandes, E. C. M. 2001. Managing carbon sequestration in soils: Concepts and terminology. *Agric. Ecosyst. Environ.* **87**: 111–117.

*International Energy Outlook (IEO)*. 2004. Energy Information Administration. Office of Integrated Analysis and Forecasting. U.S. Dept. of Energy. Washington, D.C. On line: http://www.eia.doe.gov/oia/ieo/index.html

IPCC. 2000. *IPCC Special Report on Land Use, Land-Use Change, and Forestry.* A special report for the Intergovernmental Panel and Climate Change. Approved at IPCC Plenary, Montreal, May 1–8, 200. On line: http://www.ipcc.ch

IPCC. 2001. *Climate Change 2001: Impacts, Adaptation and Vulnerability.* IPCC Third Assessment Report. Intergovernmental Panel in Climate Change. On line: http://www.ipcc.ch/

Janzen, H. H., Campell, C. A., Izaurralde, R. C., Ellert, B. H., Juma, N., McGill, W. B., and Zenter, R. P. 1998. Management effects on soil C storage on the Canadian prairies. *Soil Tillage Res.* **47**: 181–195.

Johanson, J., and Lundqvist, U. 1999. Estimating Swedish biomass energy supply. *Biomass and Bioenergy* **17**: 85–93.

Johansson, T. B. J., Kelly, H., Reddy, A. K. N., and Williams, R. H. 1993. Renewable fuels and electricity for a growing world economy. **In**: *Renewables for Fuels and Electricity.* Chapter 1. Johansson, T. B. J., Kelly, H., Reddy, A. K. N., and Williams, R. H., Eds., Island Press, Washington, D.C.

Johnson, D. W. 1992. Effects of forest management on soil carbon storage. *Water Air Soil Pollut.* **64**: 83–120.

Kort, J., Collins, M., and Ditsch, D. 1998. A review of soil erosion potential associated with biomass crops. *Biomass and Bioenergy* **14**(4): 351–359.

Ladd, J. N., Amato, M., and Odes, J. M. 1985. Decomposition of plant material in Australian soils. III. Residual organic and microbial biomass C and N from isotope-labelled legume material and soil organic matter, decomposing under field conditions. *Aust. J. Soil. Res.* **23**: 603–611.

Lal, R. 2002. Soil carbon dynamics in crop and rangeland. *Environ. Pollut.* **116**: 353–362.

Lal, R. 2003. Global potential of soil carbon sequestration to mitigate the greenhouse effect. *Crit. Rev. Plant Sci.* **22**(2): 151–184.

Lal, R., Kimble, J. M., and Follet, R. F. 1998. *Management of Carbon Sequestration in Soil,* pp. 365–379. CRC Press, Boca Raton, FL.

Lal, R., Sobecki, T. M., Livari, T., and Kimble, J. M. 2004. *Soil Degradation in the United States,* pp. 53–94. Lewis Publishers, Boca Raton.

Lemus, R. 2000. *Cultivar and Fertility Effects on Switchgrass Biofuel Production in Southern Iowa.* M.S. Thesis. Iowa State University, Ames, Iowa.

Lemus, R. 2004. *Switchgrass as an Energy Crop: Fertilization, Cultivar, and Cutting Management.* Ph.D Dissertation. Virginia Tech. Blacksburg, VA.

Li, M., Frolking, S., and Harris, R. 1994. Modeling carbon biogeochemistry in agricultural soils. *Global Biogeochem. Cycles* **8**(3): 237–254.

Liang, B. C., Gregorich, E. G., MacKenzie, A. F., Schnitner, M., Voroney, R. P., Monreal, C. M., and Beyaert, R. P. 1998. Retention and turnover of corn residue carbon. *Soil Sci. Soc. Am. J.* **62**: 1361–1366.

Lynch, J. M., and Whipps, J. M. 1991. Sunstrate flow in the rhizosphere. In: *The Rhizosphere and Plant Growth,* pp. 15–24. Keister, D. L., and Cregan, P. B., Eds., Kluwer Academic, Dordrecht.

Ma, Z., Wood, C. W., and Bransby, D. I. 2000a. Soil management on soil C sequestration by switchgrass. *Biomass and Bioenergy* **18**: 469–477.

Ma, Z., Wood, C. W., and Bransby, D. I. 2000b. Impacts of soil management on root characteristics of switchgrass. *Biomass and Bioenergy* **18**: 105–112.

Ma, Z., Wood, C. W., and Bransby, D. I. 2001. Impact of row spacing, nitrogen rate, and time on carbon partitioning of switchgrass. *Biomass and Bioenergy* **20**: 413–419.

Madakadze, I. C., Stewart, K.A, Peterson, P. R., Coulman, B. E., and Smith, D. L. 1999. Switchgrass biomass and chemical composition for biofuel in eastern Canada. *Agron. J.* **91**: 696–701.

Mann, L. K. 1986. Changes in soil carbon storage after cultivation. *Soil Sci.* **142**: 279–288.

Markin, K. R., Ward, A. P., and Boone, K. M. 1996. Quantifying changes in soil physical properties from soil and management: A survey of experts. *Trans. ASAE* **39**: 2065–2074.

Marland, G., and Marland, S. 1992. Should we store carbon in trees? *Water Air Soil Pollut.* **64**: 181–195.

Marland, G., McCarl, B. A, and Schneider, U. 2001. Soil carbon: Policy and economics. *Clim. Change* **51**: 101–117.

Marland, G., and Schlamadinger, B. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitive analysis. *Biomass & Bioenergy* **13**: 389–397.

McConkey, B. G., Liang, B. C., Campbell, C. A., Curtin, D., Moulin, A., Brandt, S. A., and Lafond, G. P. 2003. Crop rotation and tillage impact on carbon sequestration in Canadian prairie soils. *Soil Tillage Res.* **74**: 81–90.

McLaughlin, S. B., De La Torre-Ugarte, D. G., Garten, C. T., Lynd, L. R., Sanderson, M. A., Tolbert, V. R., and Wolf, D. D. 2002. High-value renewable energy from prairie grasses. *Environ. Sci. Technol.* **36**: 2122–2129.

McLaughlin, S. B., Samson, R., Bransby, D., and Wiselogel, A. 1996. Evaluating physical, chemical, and energetic properties of perennial grasses as biofuels. **In:** *Proceedings of the Seventh National Bioenergy Conference*: *Partnerships to Develop and Apply Biomass Technologies,* September 15-20, Nashville, TN.

McLaughlin, S. B., and Walsh, M. E. 1998. Evaluating environmental consequences of producing herbaceous crops for bioenergy. *Biomass and Bioenergy* **14**(4): 317–324.

Mehdi, B., Zan, C., Girouard, P., and Samson, R. 1998. *Soil Carbon Sequestration under Two Dedicated Perennial Bioenergy Crops.* Research Reports. Resource Efficient Agricultural Production (REAP)-Canada. Online: http://www.reap-canada.com/Reports/reportsindex.htm

Middelton, N., and Thomas, D. 1997. *World Atlas of Desertification*. Published for UNEP by Arnold Publ. 2nd Edition. London, 182 pp.

Mielnick, P. C., and Dugas, W. A. 2000. Soil CO2 flux in a tallgrass prairie. *Soil. Biol. Bioechem.* **32**: 221–228.

Natural Resource Inventory (NRI). 2004. *Land Use.* Natural Resources Conservation Service (NRCS). United States Department of Agriculture, Washington, D.C. On line: http://www.nrcs.usda.gov/technical/land/nri02/nri02lu.html

Neill, C., Cerri, C. C., Melillo, J. M., Feigl, B. J., Steuder, P. A., Moraes, J. F. L., and Piccolo, M. C. 1998. Stocks and Dynamics of soil carbon following deforestation for pasture in Rondônia. **In**: *Soil Processes and the Carbon Cycle,* pp. 9–28. Lal, R., Kimble, J. M., Follet, F. R., and Stewart, B. A., Eds., CRC Press, Boca Raton, FL.

Oades, J. M. 1988. The retention of organic matter in soils. *Biogeochemistry* **5**: 35–70.

Oldeman, L. R. 1994. The global extent of soil degradation. **In**: *Soil Resilience and Sustainable Land Use,* pp. 99–118. Greenland, D. J., and Szabolcs, I., Eds., CAB International, Oxon, UK.

Pacala, S. W., Hurtt, G. C., Moorcroft, P. R., and Caspersen, J. P. 2001. Carbon storage in the U.S. caused by land use change. **In**: *Present and Future of Modeling Global Environmental Change: Toward Integrated Modeling,* pp. 145–172. Matsuno, T. and Kida, H., Eds., TERRAPUB, Tokyo, Japan.

Palm, C. A., van Noordwijjk, M., Woomer, P. L., Alegre, J., Castilla, C., Cordeiro, D. G., Hairiah, K., Kotto-Same, J., Moukam, A., Njomgang, R., Ricse, A., and

Rodrigues, V. 2000. Carbon losses and sequestration potential of alternatives to slash and burn agriculture. **In**: *The Biology and Fertility of Tropical Soils*. TSBF, Nairobi.

Post, W. M., and Kwon, K. C. 2000. Soil carbon sequestration and land-use change: Processes and potential. *Global Change Biol*. **6**: 317–327.

Potter, K. N., Tobert, H. A., Johnson, H. B., and Tischler, C. R. 1999. Carbon storage after long-term grass establishment on degraded soils. *Soil Sci*. **164**: 718-725.

Pretty, J., and Ball, A. 2001. *Agricultural influences on carbon emissions and sequestration: A review of evidence and the emerging trading options*. Centre for Environment and Society Occasional Paper 2001-03. Univ. of Essex, UK. On line: http://www.environmentalobservatory.org/ library/uploadedfiles/Agricultural_Influences_on_Carbon_Emissions_an.pdf

Quian, Y., and Follett, R. F. 2002. Assessing soil carbon sequestration in turf grass systems using long-term soil testing data. *Agron. J*. **94**: 930–935.

Ranney, J. W., Wright, L. L., and Mitchell, C. P. 1991. Carbon storage and recycling in short-rotation energy crops. **In**: *Proceedings of a seminar organized by International Energy Agency/Bioenergy Agreement and National Energy Administration of Sweden on Bioenergy and Greenhouse Effect, Vol. 1*, pp. 39–59. Mitchell, C.P, Ed., NUTUK B, Stockholm.

Reeder, J. D., Schuman, G. E., and Bowman, R. A. 1998. Soil C and N changes on conservation reserve program lands in the Central Great Plains. *Soil Tillage Res*. **47**: 339–349.

Rice, C. W. 2000. Soil organic C and N in rangeland soil under elevated $CO_2$ and land management. **In**: *Proceedings of Advances in Terrestrial Ecosystem Carbon Inventory, Measurement, and Monitoring*, October 3–5, 2000. pp. 83. USDA-ARS, USDA-FS, USDA-NRCS, US Dept. Energy, NASA, and National Council for Air and Stream Improvement.

Robertson, T., and Shapouri, H. 1993. Biomass: An overview in the United States of America. **In**: *Proceedings of the Biomass Conference of the Americas*, 30 August–2 September, pp. 1–17. Burlington, VT.

Sampson, R. N. 2000. Carbon sequestration: What's the best approach? **In**: *Carbon: Exploring the Benefits to Farmers and Society*. The Sampson Group, Inc., Alexandria, VA. On line: http://www.sampsongroup.com/acrobat/carbon.pdf

Sanchez, P. G., Carter, E. A., and Klepac, J. F. 2003. Enhancing the soil organic matter pool through biomass incorporation. *Biomass & Bioenergy* **24**: 337–349.

Schuman, G. E., Janzen, H. H., and Herrick, J. E. 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. *Environ. Pollut*. **116**: 391–396.

Slobodian, N., Van Rees, K., and Pennock, D. 2002. Cultivation-induced effects on belowground biomass and organic carbon. *Soil Sci. Soc. Am. J*. **66**: 924–930.

Sommer, R., Denich, M., and Vlek, P. G. L. 2000. Carbon storage and root penetration in deep soils under small-farmer land-use systems in the eastern Amazon region. *Plant Soil* **219**: 231–241.

*Taftan Data*. 1998. Combustion of fossil fuel, using thermo utilities, MS Excel add-ins. On line: http://www.taftan.com/xl/fossil.html

Tolbert, V. R., Todd, D. E., Mann, L. K., Jawdy, C. M., Mays, D. A., Malik, R., Bandaranayake, W., Houston, A., Tyler, D., and Pettry, D. E. 2002. Changes in soil quality and below-ground carbon storage with conversion of traditional agricultural crop lands to bioenergy crop production. *Environ. Pollut*. **116**: S97–S106.

Tufekcioglu, A., Raich, J. W., Isenhart, T. M., and Schultz, R. C. 2003. Biomass, carbon, and nitrogen dynamics of multi-species riparian buffers within an gricultural watershed in Iowa, USA. *Agroforestry Syst*. **57**: 187–198.

Turhollow, A. F., and Perlack, R. D. 1991. Emissions of $CO_2$ from energy crop production. *Biomass and Bioenergy* **1**(3): 129–135.

Tuskan, G. A. 1998. Short-rotation woody crop supply systems in the United States: What do we know and what do we need to know? *Biomass and Bioenergy* **14**(4): 307–315.

United Nations. 1996. Population Division, *World Population Prospects 1950-2050* (The 1996 Revision), on diskette (U.N., New York, 1996).

U.S. Department of Energy (DOE). 1999. *Carbon Sequestration*. Office of Fossil Energy, U.S. Dept. of Energy. On line: http://www.fe.doe.gov/ coal_power/sequestration/index.html

U.S. Environmental Protection Agency (EPA). 2003. *Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2001*. Washington, D.C.

Volk, T. A., Edick, S., Brown, S., and Downing, M. 1999. Community outreach and education: Key components of the Salix Consortium's willow biomass project. **In**: *Proceedings of the Fourth Biomass Conference of the Americas. Biomass: A Growth Opportunity in Green Energy and Value-Added Products*. August 26–29. pp. 1733–1738. Overend, R. P. and E. Chornet, E., Eds., Oakland, CA.

Walsh, M. E., Ince, P. J., De la Torre Ugarte, D., Alig, R., Mills, J., Spelter, H., Skog, K., Slinsky, S. P., Ray, D. E., and Graham, R. L. 1999. Potential of short rotation wood crops as a fiber and energy source in the U.S. **In**: *Proceedings of the Fourth Biomass Conference of the Americas*, pp. 195–206. Elsevier Science Ltd., Oxford, UK.

Williams, M. A., Rice, C. W., and Owensby, C. E. 2000. Carbon dynamics and microbial activity in tallgrass prairie ecosystem exposed to elevated $CO_2$ for 8 years. *Plant Soil* **227**: 127–137.

Williams, M. A., Rice, C. W., and Owensby, C. E. 2004. Carbon and nitrogen pools in a tallgrass prairie soil under elevated carbon dioxide. *Soil Sci. Soc. Am. J*. **68**: 148–153.

Wojick, D. E. 1999. *Carbon Storage in Soil: The Ultimate No-Regrets Policy?* A report to Greening Earth Society. Online: http://www.greeningearthsociety.org/Articles/1999/carbon1.htm

Yang, W., and Hsieh, Y. P. 2002. Uncertainties and novel prospects in the study of soil carbon dynamics. *Chemosphere* **49**: 791–804.

Yang, X. M., and Wander, M. M. 1999. Tillage effects on soil organic carbon distribution and storage in a silt loam soil in Illinois. *Soil Tillage Res*. **52**: 1–9.

Zak, D. E., Grigal, D. F., Gleeson, S., and Tilman, D. 1990. Carbon and nitrogen cycling during old-field succession: Constrains on plant and microbial productivity. *Biogeochemistry* **11**: 111–129.

Zan, C. S., Fyles, J. M., Girourard, P., and Samson, R. A. 2001. Carbon sequestration in perennial bioenergy, annual corn, and uncultivated systems in southern Quebec. *Agric. Ecosyst. Envrion*. **86**: 135–144.

Zan, C., Fyles, J. M., Girourard, P., Samson, R., and Doan, M. 1997. Carbon storage in switchgrass and short-rotation willow plantations. **In**: *Making a business from biomass in energy, environment, chemicals, fibers, and materials*. Proceedings of the Third Biomass Conference of the Americas: Energy, Environment, Agriculture, and Industry. August 24-29, Montréal, Québec, Canada. pp. 355–361. Overend, R. P. and Chornet, E., Eds., Pergamon, New York.

Zhang, H., Thompson, M., and Sandor, J. A. 1988. Compositional differences in organic matter among cultivated and uncultivated Arguidolls and Hapludalfs derived from loess. *Soil Sci. Soc. Am. J*. **52**: 216–222.