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Agriculture's Role in Greenhouse Gas Mitigation

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Foreword *Eileen Claussen, President, Pew Center on Global Climate Change*

This Pew Center report is the fourth in our series examining key sectors, technologies, and policy options to construct the "10-50 Solution" to climate change. The idea is that we need to tackle climate change over the next fifty years, one decade at a time. This report is also a companion paper to *Agriculture and Forest Lands: U.S. Carbon Policy Strategies*, being published simultaneously.

Our reports on electricity, buildings, and transportation described the options available now and in the future for reducing greenhouse gas emissions from those sectors. Agriculture may be less important than those other sectors in terms of its overall contribution to U.S. greenhouse gas emissions, but it has an important role to play within a strategy to address climate change. Agriculture is important not only because of the potential to reduce its own emissions, but because of its potential to reduce net emissions from other sectors. Agriculture can take carbon dioxide, the major greenhouse gas, out of the atmosphere and store it as carbon in plants and soils. Agriculture can also produce energy from biomass that can displace fossil fuels, the major contributor to greenhouse gas emissions.

Looking at options available now and in the future, this report yields the following insights for agriculture's potential role in greenhouse gas mitigation:

- If farmers widely adopt the best management techniques to store carbon, and undertake cost-effective reductions in nitrous oxide and methane, aggregate U.S. greenhouse gas emissions could be reduced by 5 to 14 percent.
- With technological advances, biofuels could displace a significant fraction of fossil fuels and thereby reduce current U.S. GHG emissions by 9 to 24 percent. Using biomass to produce transportation fuels could also significantly reduce our reliance on imported petroleum.
- Further research is needed to bring down the costs of biofuels and, particularly if agriculture is to participate in a GHG cap-and-trade system, to better assess the impacts of practice changes.
- The level of reductions achieved will strongly depend on the policies adopted. Policies are needed to make it profitable for farmers to adopt climate-friendly practices, and to support needed research.

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Executive Summary

The impact of human activities on the atmosphere and the accompanying risks of long-term global climate change are by now familiar topics to many people. Although most of the increase in greenhouse gas (GHG) concentrations is due to carbon dioxide (CO_2) emissions from fossil fuels, globally about one-third of the total human-induced warming effect due to GHGs comes from agriculture and land-use change. U.S. agricultural emissions account for approximately 8 percent of total U.S. GHG emissions when weighted by their relative contribution to global warming. The agricultural sector has the potential not only to reduce these emissions but also to significantly reduce net U.S. GHG emissions from other sectors. The sector's contribution to achieving GHG reduction goals will depend on economics as well as available technology and the biological and physical capacity of soils to sequester carbon. The level of reductions achieved will, consequently, strongly depend on the policies adopted. In particular, policies are needed to provide incentives that make it profitable for farmers to adopt GHG-mitigation practices and to support needed research.

The agricultural sector can reduce its own emissions, offset emissions from other sectors by removing CO_2 from the atmosphere (via photosynthesis) and storing the carbon in soils, and reduce emissions in other sectors by displacing fossil fuels with biofuels. Through adoption of agricultural best management practices, U.S. farmers can reduce emissions of nitrous oxide from agricultural soils, methane from livestock production and manure, and CO_2 from on-farm energy use. Improved management practices can also increase the uptake and storage of carbon in plants and soil. Every tonne of carbon added to, and stored in, plants or soils removes 3.6 tonnes of CO_2 from the atmosphere. Furthermore, biomass from the agricultural sector can be used to produce biofuels, which can substitute for a portion of the fossil fuels currently used for energy.

Carbon stocks in agricultural soils are currently increasing by 12 million metric tonnes (MMT) of carbon annually. If farmers widely adopt the best management techniques now available, an estimated 70 to 220 MMT of carbon could be stored in U.S. agricultural soils annually. Together with attainable nitrous oxide and methane reductions, these mitigation options represent 5 to 14 percent of total U.S. GHG emissions. The relevant management technologies and practices can be deployed quickly and at costs that are low relative to many other GHG-reduction options. To achieve maximum results, however, policies must be put in place to promote, and make attractive to farmers, practices that increase soil carbon and efficiently use fertilizers, pesticides, irrigation, and animal feeds. It is also important to ensure funding to improve the measurement and assessment methods for agricultural GHG emissions and reductions, including expansion of the U.S. Department of Agriculture's National Resource Inventory.

In particular, this inventory needs to include a network of permanent sites where key management activities and soil attributes are monitored over time. Such sites would provide information vital to helping farmers select the most promising management practices in specific locations.

Profitability of management practices varies widely by region, as does the amount of carbon storage attainable. Initial national-level studies suggest that, with moderate incentives (up to $50/tonne of carbon, or $13 per tonne of CO_2), up to 70 MMT of carbon per year might be stored on agricultural lands and up to 270 MMT of carbon per year might be stored through converting agricultural land to forests. Mitigation options based on storage of carbon in soils would predominate in the Midwest and Great Plains regions; whereas in the Southeast, agricultural land would tend to be converted to forestland. Information on the costs and supply of GHG reductions from reducing nitrous oxide and methane emissions are very limited, and more studies in these areas are needed.

Agriculture can also reduce GHG emissions by providing biofuels—fuels derived from biomass sources such as corn, soybeans, crop residues, trees, and grasses. Substitution of biofuels for fossil fuels has the potential to reduce U.S. GHG emissions significantly and to provide a major portion of transportation fuels. The contribution of biofuels to GHG reductions will be highly dependent on policies, fossil fuel prices, the specific fossil fuels replaced, the technologies used to convert biomass into energy, and per acre yields of energy crops. In a "best-case" scenario, where energy crops are produced on 15 percent of current U.S. agricultural land at four-times current yields, bioenergy could supply a total of 20 exajoules (EJ)—almost one-fifth of the total U.S. year-2004 demand for energy. This corresponds to a 14 to 24 percent reduction of year-2004 U.S. GHG emissions, depending on how the biomass is used. If advanced conversion technologies are not widely deployed, or if yield gains are more modest, GHG reductions would be on the order of 9 to 20 percent. For biofuels to reach their full potential in reducing GHG emissions, long-term, greatly enhanced support for fundamental research is needed.

Application of best management practices in agriculture and use of biofuels for GHG mitigation can have substantial co-benefits. Increasing the organic matter content of soils (which accompanies soil carbon storage) improves soil quality and fertility, increases water retention, and reduces erosion. More efficient use of nitrogen can reduce nutrient runoff and improve water quality in both surface and ground waters. Similarly, improving manure management to reduce methane and nitrous oxide emissions is beneficial to water and air quality and reduces odors. Biofuel use, particularly substituting energy crops for imported petroleum for transportation, has important energy security benefits. However, as biofuel use expands, it will be important to ensure that biomass is produced responsibly, taking both environmental and socio-economic impacts into consideration.

Although challenges remain, agriculture has much to offer in helping to reduce net GHG emissions to the atmosphere, while at the same time improving the environment and the sustainability of the agricultural sector. Further research and development will result in improved assessments of GHG contributions from agriculture, increases in agriculture's contribution to renewable energy for the nation, better ways to manage lands, and design of more efficient policies. Government policy plays an important role in making best management practices and biofuel production economically attractive, and farmers will adopt best management practices for GHG reduction only if they seem profitable. Perceived risks and availability of information and capital play important roles in perceptions of profitability. Thus, risk reduction, availability of information, and access to capital are some of the key issues that must be addressed through policies. With the right policy framework, U.S. farmers will be important partners in efforts to reduce GHG emissions while reaping multiple co-benefits.

I. Introduction

All three of the major greenhouse gases—carbon dioxide (CO_2), methane, and nitrous oxide—are components of the earth's natural cycling of carbon and nitrogen. Agricultural lands, because of their large extent and intensive management,[1] have a significant impact on the earth's carbon and nitrogen cycles, and agricultural activities result in releases of all three of these greenhouse gases (GHGs). While currently a substantial source of GHGs, agriculture has great potential to reduce the buildup of these gases in the atmosphere. Importantly, studies to date suggest that a significant portion of the agricultural practices that could reduce emissions or remove CO_2 from the atmosphere are relatively low-cost.

The impact of human activities on the atmosphere and the accompanying risks of long-term, global-scale climate change are by now familiar topics to many people. Changes in the atmosphere's composition have occurred through the buildup of GHGs, which act to trap outgoing long-wave radiation and emit it back to the earth's surface. The concentrations of these gases have been measured directly over several decades and indirectly (from air bubbles trapped within deep polar ice-cores) over many millennia. This historical record clearly shows an accelerating increase in atmospheric GHG concentrations over the past 150 years, caused by a variety of human activities (IPCC 2001). While most of the increase is due to CO_2 emissions from fossil fuels, land use and agriculture play significant roles. Overall, land use change (predominantly in the tropics) and agricultural activities globally account for about one-third of the warming effect from increased GHG concentrations (Cole et al. 1997).

However, ecosystem processes also act to dampen these GHG increases, primarily through the uptake and storage of CO_2 in plants and soil on land and in oceans. These uptake and storage processes—referred to hereafter as carbon "sinks"—play a significant role in the global CO_2 cycle, so that only about one-half of the CO_2 emitted from fossil fuels accumulates in the atmosphere. The other half is absorbed by the oceans and terrestrial ecosystems (IPCC 2001). Without these sinks, the rate of increase in atmospheric CO_2 concentrations would be roughly twice the present rate.

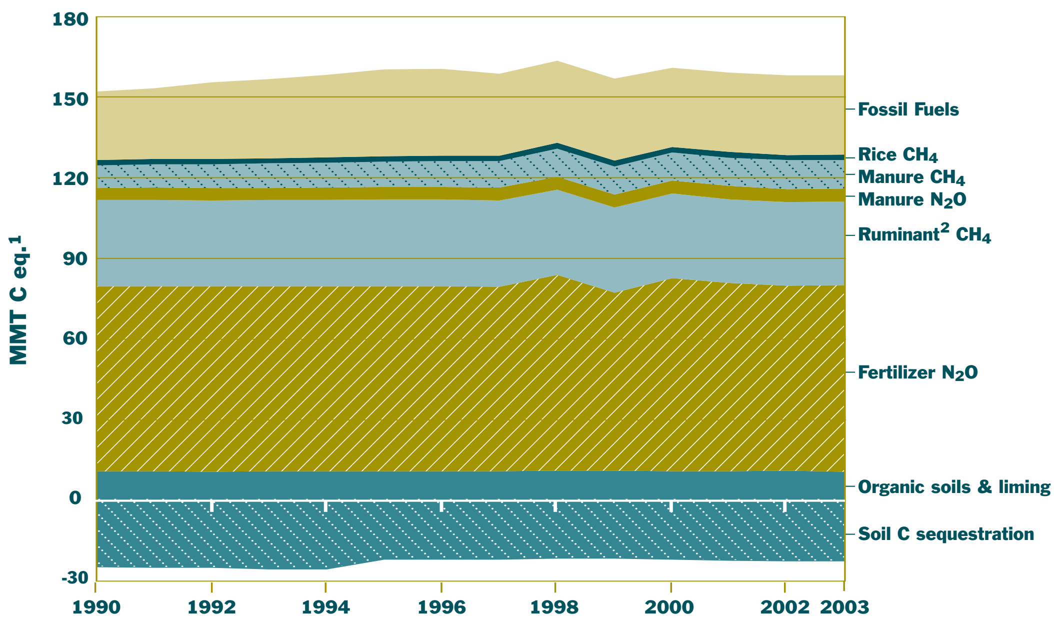
Thus, within a broad-based approach to GHG mitigation, in addition to reducing emissions, there is a need to preserve and enhance the sink capacity of the world's ecosystems, including agriculture. Throughout this report, the term *mitigation* is used in a broad sense, encompassing both GHG emission reductions and GHG removals from the atmosphere by sinks.

GHG emissions and sinks due to U.S. agriculture, as shown in Figure 1, are reported annually by the U.S. Environmental Protection Agency (USEPA) as part of the U.S. commitment to the United Nations Framework Convention on Climate Change. While these estimates of emissions and sinks are based on the best available scientific information, and are derived by using internationally accepted accounting procedures, there is considerable uncertainty in their magnitude.

Figure 1

Sources of U.S. Agricultural GHG Emissions

and Sinks (Shown as Negative Emissions) Since 1990



Sources: USEPA 2005, USDA 2003.

Note: fossil fuel use by agriculture is shown here, although it is reported as part of the energy sector of the U.S. national GHG inventory.

[1]C eq. stands for carbon equivalent. The carbon equivalents for CH_4 and N_2O are calculated by using GWPs and carbon equivalents (see Box 1).

[2]Animals that release methane as part of their normal digestive processes, e.g. cows and sheep.

Over the past decade, U.S. agricultural soils overall have acted as a small net sink of approximately 12 million metric tons (MMT) of carbon per year, mainly due to improved soil management practices and the establishment of conservation reserve lands (USEPA 2006). These practices are helping to sequester about 23 MMT of carbon per year in mineral soils, which make up greater than 99 percent of annual cropland area. However, net carbon emissions of about ten MMT of carbon per year from the small area (about 1.3 million hectares [Mha]) of cultivated organic (i.e., peat and muck) soils[2] offset 40 percent of the carbon gain in non-organic (mineral) soils. Emissions from agricultural liming contribute an additional one MMT of carbon per year, so that—taking into account both soil emissions and sinks—the result is a net sink of about 12 MMT of carbon per year.

CO_2 emissions from U.S. agricultural energy use amount to about 25 to 30 MMT of carbon per year. Nitrous oxide emissions (76 MMT carbon-equivalent per year in 2004) are the dominant agricultural contribution to the greenhouse effect when expressed on the basis of their global warming potential (GWP), a measure commonly used to equate the warming effects of different GHGs. (See Box 1 for explanations of GWP and carbon equivalents.) The main sources of nitrous oxide are nitrogen fertilizers

Box 1. Global Warming Potentials; Carbon and CO_2 Equivalents

While the physics of the greenhouse effect are similar for all gases, each gas differs in its overall effect on the earth's radiation balance, depending on the concentration of the gas, its residence time in the atmosphere, and its physical properties with respect to absorbing and emitting radiant energy. A common measure, termed the *global warming potential* (GWP), is used to equate the effect of different greenhouse gases on a mass basis. By convention, the effect of carbon dioxide (CO_2) is assigned a value of one (1) and the warming potential of other gases are expressed relative to that standard, i.e., on a CO_2–equivalent basis. For the U.S. national inventory, the GWP values used are 310 for nitrous oxide and 21 for methane (slightly different values have been estimated in the report *Climate Change 2001 The Scientific Basis* from the Intergovernmental Panel on Climate Change). This means that one tonne of nitrous oxide is deemed to have the same warming effect as 310 tonnes of CO_2, and one tonne of methane is deemed to have the warming effect of 21 tonnes of CO_2. Despite its lower GWP relative to the other gases, the total contribution of CO_2 to greenhouse warming is much greater, about 60 percent versus 20 percent and 6 percent for CH_4 and N_2O, respectively, owing to the much higher concentration of CO_2 in the atmosphere compared to the other two gases.

Most of the carbon in soils is in the form of complex organic compounds and inorganic carbonates. Consequently carbon in soils is generally quantified in terms of the carbon mass, not CO_2 (although when this carbon is released to—or removed from—the atmosphere it is mainly as CO_2). A mass of carbon can be expressed in terms of its CO_2 equivalent by using the masses of one atom of carbon (12) and two atoms of oxygen (12 + 2 x 16 = 44) and carbon's contribution to a molecule of CO_2. Thus the CO_2 equivalent of one tonne of carbon is 3.66 (44/12) tonnes of CO_2. Similarly CO_2 can be expressed in terms of its carbon equivalent; the carbon equivalent of one tonne of CO_2 is .27 (12/44) tonnes of carbon.

and manure applied to cropland and pastures, leguminous crops, and crop residues. Some nitrous oxide emissions also occur from stored manure. Annual U.S. agricultural methane emissions are approximately 44 MMT carbon-equivalent per year (2004 estimate) and stem mainly from livestock, animal waste, and rice cultivation. In aggregate, agricultural GHG emissions account for roughly 8 percent of total U.S. emissions from all sources (USEPA 2006), on a carbon-equivalent basis.

Historical changes in American agriculture have greatly influenced past trends in GHG emissions. The agricultural activities of Native Americans were small in scale and probably had a negligible effect on GHG emissions,[3] although their use of fire may have affected carbon stocks by promoting the expansion of the prairie grassland (replacing forestlands) in the central United States. With the onset of European settlement, the clearing of forests, prairies, and wetlands for agriculture resulted in a large net source of CO_2 to the atmosphere. In fact, prior to the 1900's, land use was the dominant global source of CO_2 from human activity, and since 1850 an estimated 160 billion metric tons of carbon from biomass and soils have been emitted worldwide as a consequence of land use and land-use changes (Houghton 2003). On farmland, intensive cultivation practices, along with comparatively low productivity, harvesting and removal of residues, and soil erosion depleted the carbon stocks of many agricultural soils by 30 to 50 percent or more compared to their condition under native vegetation. A cycle of soil exhaustion and land abandonment, in part, fueled the westward expansion of agriculture in North America.

Since the 1940's, the emergence of modern agriculture has dramatically altered the relative sources and magnitudes of agricultural GHG fluxes. Steadily increasing productivity, improved cropping practices, erosion control measures, and reduced tillage intensity (see Chapter II) have stabilized and begun to rebuild the organic carbon stocks of many agricultural soils. As a result, on a net basis, U.S. croplands currently remove more CO_2 from the atmosphere than they release to it; i.e., they now act as a net sink for CO_2. Reforestation of marginal agricultural lands, particularly in the eastern United States, has also contributed to the present carbon sink (about 174 MMT of carbon per year) attributed to U.S. forestland (USEPA 2006). On the other hand, the development and growing use of industrial fertilizers over the past 50 years has greatly increased the input of nitrogen to soils, resulting in nitrogen losses to the environment in various forms, including as nitrous oxide emissions. Growth in livestock numbers, particularly within large, confined operations, has increased emissions of methane from livestock and manure over the same period.

Current and future trends in the structure of American agriculture will affect both future emissions and opportunities for GHG mitigation. Crop yields have been increasing 1 to 2 percent per year over recent decades, and increases are likely to continue for the foreseeable future (Reilly and Fuglie 1998). These increases, along with continued adoption of conservation tillage and maintenance of conservation set-aside programs (see Box 2, page 11), are likely to support further increases in soil carbon stocks. Higher crop yields also increase the potential for shifting some land from food production to energy crop production. Nitrogen fertilizer use has tended to level off over the past 15 years, likely reflecting regulatory pressures in some regions as well as higher fertilizer costs and retirement of cropland under the Conservation Reserve Program (See Box 2). Consequently, nitrous oxide emissions from U.S. agriculture have flattened out since 1990 (USEPA 2006). Since 1990, a decline in cattle and sheep populations has been counterbalanced by a rise in swine and poultry populations, resulting in roughly stable agricultural methane emissions since 1990 (USEPA 2006). These trends reflect the current technical, economic, and policy environment, in the absence of explicit management decisions and policies to mitigate GHGs.

As detailed in the following chapters, the current *technical* potential to mitigate GHGs through improved agricultural practices over the next 10 to 30 years is substantial, estimated at approximately 102 to 270 MMT carbon-equivalent per year. This estimate derives from a combination of carbon sequestration (70 to 221 MMT carbon), nitrous oxide reductions (23 to 31 MMT carbon-equivalent), and methane reductions (9 to 18 MMT carbon-equivalent). This mitigation potential equals or exceeds present emissions from U.S. agricultural sources (about 160 MMT carbon-equivalent in 2004) and represents 5 to 14 percent of year-2004 U.S. GHG emissions from all sources and for all gases. In addition, energy produced from agricultural biomass sources, if substituted for fossil fuels, represents a mitigation potential of 510 to 1,710 MMT CO_2-equivalent per year (140 to 470 MMT carbon-equivalent per year) or 7 to 24 percent of total 2004 U.S. GHG emissions (see Chapter IV). However, the mitigation levels that can be achieved economically are likely to be substantially lower than these technical potentials.

As discussed in Chapter III, a variety of economic and social factors will influence the adoption of alternative practices and production systems, although studies to date suggest that a significant portion of agricultural mitigation practices can be characterized as low-cost options (i.e., relative to many mitigation options in energy, industry, transportation, etc). Further, Chapter IV points out that significant research efforts to reduce biofuel conversion costs and increase energy crop yields would be needed for biofuels to reach their full mitigation potential. Finally, changes in land use and management to achieve GHG mitigation can contribute to overall environmental improvements. Hence, a broader consideration of the costs and benefits of improved agricultural practices, beyond the realm of climate change concerns, is merited.

The practices that could be implemented to stimulate GHG mitigation, the resulting economic opportunities, a review of biofuel options, and the policy implications of these opportunities are discussed in the following chapters.

II. Mitigation Opportunities: Increased Sinks and Reduced Emissions

Agriculture offers a diversity of means for addressing greenhouse gases. This chapter covers opportunities to alter agricultural management practices in ways that increase storage of carbon and reduce emissions of carbon dioxide, methane, and nitrous oxide. Section A below delineates opportunities to increase soil carbon content, thereby removing CO_2 from the atmosphere. Section B reviews opportunities to mitigate methane and nitrous oxide emissions and discusses the need for a comprehensive multi-gas approach. Section C describes measurement and monitoring methods essential for guiding policy and implementation and the need for further research and development to improve their accuracy. Finally, Section D highlights added benefits for water, air, and soil quality from practices that reduce GHG emissions.

A. Opportunities to Increase Soil Carbon

Historically, agricultural practices have caused large carbon losses from U.S. cropland soils. If half or more of the original carbon stock of croplands could be regained, tens to hundreds of million metric tons of carbon could be stored (i.e., added to and sequestered) in soils annually over the next several decades. Carbon additions to soil are favored by management practices that increase plant residues. Reducing carbon losses, i.e., by slowing the rate of soil organic matter decay, also increases soil carbon stocks. Land-use changes such as conversion of annual cropland to grassland or forest and restoration of degraded lands can also increase soil carbon. Following adoption of practices or land uses that increase carbon stocks, soil carbon may increase for 20 to 30 years, after which soil carbon contents tend to stabilize and there is no further increase in carbon storage due to the new system (CAST 2004).

1(a). Cropland Management

Numerous field experiments show that soil carbon stocks tend to attain levels that are roughly proportional to the annual rate of carbon added (e.g., see Paustian et al. 1997; Huggins et al. 1998). A number of management practices are available to increase cropland soil carbon inputs. Increasing soil carbon inputs by increasing the productivity of crops is largely in line with farmers' management goals of achieving high productivity. Carbon inputs to soil can also be increased by using crop rotations with high residue yields, by reducing or eliminating the fallow period between successive crops in annual crop rotations, and by making efficient use of fertilizer and manure. On annual croplands, soil carbon losses can be reduced by decreasing the frequency and intensity of soil tillage, in particular through conversion to no-till practices.

Use of high-residue crops and grasses. Annual crops that produce large amounts of residues (plant matter left in the field after harvesting), such as corn and sorghum, typically result in higher soil carbon levels than many other crops. Hay and pasture lands also tend to have high carbon inputs because perennial grasses allocate a large portion of their total carbon assimilation to root growth. For example, long-term experiments at two sites in Ohio (Dick et al. 1998) show about ten tonnes per hectare more soil carbon after 30 years under continuous corn crops or with corn-oat-hay rotations than for corn-soybean rotations, equal to an average gain of 0.3 tonnes per hectare per year (t/ha/yr).[4] Conversion from continuous cereal cropping to cereal-hay rotations was estimated to increase soil carbon by about 1 percent per year, or about 0.5 t/ha/yr, for average European conditions (Smith et al. 1997a).

Reduction or elimination of fallow periods between crops. In semi-arid regions like the Great Plains, summer fallow (a practice where soil is left unplanted for an entire cropping year) was developed as a way of storing soil moisture to improve yields and reduce crop failure. However, summer fallow practices caused high rates of soil carbon loss and soil degradation in large areas of the western United States (Haas et al. 1957). More recently, new cropping systems that combine winter wheat with summer season crops (e.g., corn, sorghum, millet, bean, sunflower) in rotation using no-till practices (see below) have proved successful in both improving soil moisture and increasing soil carbon (Peterson et al. 1998).

In more humid regions, where fields may be left fallow in winter, it is often feasible to grow winter cover crops, usually legumes or annual grasses, and thus maintain vegetation year-round. Cover crops serve several functions, including taking up excess soil nutrients (e.g., nitrogen) to reduce leaching or other losses to the environment, fixing atmospheric nitrogen (e.g., legumes), and controlling weeds; but they also serve to augment the input of plant residues, thereby increasing soil carbon content.

Efficient use of manures, nitrogen fertilizers, and irrigation. As a general rule, promoting the *efficient* use of inputs such as fertilizer, manure, and irrigation will yield the best results for GHG mitigation (Paustian et al. 2000). Efficiency in this context is defined as maximizing crop production per unit of input. If high rates of crop production (with attendant carbon input increases) are achieved primarily through increased nitrogen fertilization and irrigation, increases in other GHG emissions, particularly nitrous oxide (see section B below), can offset part or all of the gains in soil carbon. Tailoring fertilizer and manure applications to satisfy crop nitrogen demands, so that less nitrogen is left behind in the soil, can reduce nitrous oxide emissions while building soil carbon stocks. Efficient use of irrigation water will similarly reduce nitrogen losses,[5] including nitrous oxide emissions, and minimize CO_2 emissions from energy used for pumping while maintaining high yields and crop-residue production.

Use of low- or no-till practices. Reducing soil carbon losses on croplands is primarily accomplished through reducing the frequency and intensity of soil tillage. Soil tillage tends to accelerate organic matter decomposition—including the oxidation of carbon to CO_2—by warming the soil, breaking up soil aggregates (i.e., soil's granular structure), and placing surface residues into the moister environment within the soil (Reicosky 1997; Six et al. 2000). Traditional tillage methods such as moldboard plowing, which fully inverts the soil, cause the greatest degree of disturbance and consequently tend to cause the most degradation of soil structure and loss of soil carbon stocks. In many areas, the trend over the past several decades has been towards reduced tillage practices that have shallower depths, less soil mixing, and retention of a larger proportion of crop residues on the surface.

No-till, a practice in which crops are sown by cutting a narrow slot in the soil for the seed and herbicides are used in place of tillage for weed control, causes the least amount of soil disturbance. Recent reviews (West and Marland 2002; Ogle et al. 2005) have summarized no-till effects on soil carbon. For example, Ogle et al. (2005) analyzed data from 126 studies worldwide and estimated that soil carbon stocks in surface soil layers (to 30 centimeter [cm] depth) increased by an average of 10 to 20 percent over a 20-year time period under no-till practices compared with intensive tillage practices. The relative increases in carbon stocks were higher under humid than dry climates and higher under tropical than temperate temperature regimes. Using U.S. data only, an overall average increase of 15 percent in carbon stocks was estimated. To put this in perspective, many U.S. cropland soils contain 40 to 80 metric tons per hectare (t/ha) of carbon in the top 30 cm, which would imply an increase of 6 to 12 t/ha over a 20-year period (equivalent to 0.3–0.6 t/ha of carbon per year). West and Marland's (2002) estimate of average annual rates of 0.34 metric tons per hectare per year (t/ha/yr) of carbon increase under no-till conditions, based on long-term experiments in the United States, falls within this range. Finally, CO_2 emissions from machinery use are decreased by 40 percent for reduced tillage and

70 percent for no-till, relative to conventional tillage (West and Marland 2002), contributing to further reductions in GHGs from reducing tillage intensity.

1(b). Grazing Land and Hayland Management

Permanent grasslands used as pastures, rangelands, and hayfields can maintain large soil carbon stocks due to several characteristics. Perennial grasses allocate a high proportion of photosynthetically fixed carbon below ground, maintain plant cover year-round, and promote formation of stable soil aggregates. However, the potential for *increasing* soil carbon in such systems is highly dependent on past management and the extent to which plant productivity and carbon inputs can be enhanced through management improvements. Grassland systems that have been degraded in the past or maintained under suboptimal management conditions are most conducive to sequestering additional carbon.

Management activities can include boosting plant productivity through fertilization, irrigation, improved grazing, introduction of legumes, and/or use of improved grass species. Intensive management strategies are usually restricted to more humid regions with high productivity potential or regions where irrigation is used. In semi-arid western rangelands, manipulating grazing systems and altering species composition are the principal management tools available. See the companion paper, Richards et al. 2006, for a discussion of the feasibility of management changes on U.S. rangelands.

Conant et al. (2001) summarized more than 115 studies of grassland management effects on soil carbon and estimated rates of soil carbon increase ranging from 0.1 to 3 t/ha/yr for different management improvements. The highest rates occurred with introduction of deep-rooted African grasses in South American savannas (Fisher et al. 1994), whereas most of the higher rates for temperate grasslands in the United States showed carbon gains of around 1 t/ha/yr. Average rates of carbon increase were approximately 0.3 t/ha/yr for fertilization and improved grazing systems and approximately 0.7 t/ha/yr for introduction of legumes.

2. Land-use Changes to Increase Soil Carbon

Conversion of annual cropland to grasslands or forest and restoration of severely degraded lands offer significant opportunities to increase soil carbon. Conversion can occur through set-asides (see Box 2) or through moving land from annual crop production into use as hay or pasture lands. Land set-asides and restoration projects typically involve the establishment of perennial grasses or trees that mimic

Box 2. Carbon Sequestration in the Conservation Reserve Program

In the United States, the Conservation Reserve Program (CRP) was initiated in 1985 and currently encompasses over 13 million hectares (ha), mainly highly erodible and/or marginally productive annual cropland that has been set aside. Several studies show that set-asides are effective at sequestering carbon (e.g., Gebhardt et al. 1994; Paustian et al. 2001; Conant et al. 2001; Follett et al. 2001). From an extensive review of the literature, Ogle et al. (2003) estimated soil carbon stocks increased under grassland set-asides by an average of 16% in the top 30 cm, over a 20-year period, equivalent to average gains of almost 7 MMT per year on the 13.4 million ha of CRP land in the United States. In general, soil carbon gains under CRP tend to be correlated to productivity and hence are higher in more humid regions, compared to semi-arid areas.

conditions prevalent under native vegetation. In most instances, conversion of agricultural lands to forest will increase soil carbon stocks as well as aboveground woody biomass. Post and Kwon (2000) found mean rates of soil carbon change of 0.35 t/ha/yr following afforestation. Interactions between tree species and soil type seem to be important in governing how soil carbon stocks change. Converting cultivated cropland to grassland typically increases soil carbon at rates of 0.3 to 1.0 t/ha/yr for a period of a few decades (Lal et al. 1998; Conant et al. 2001). Carbon accumulation rates at the upper end of this range are representative of cropland conversion to managed pastures in more humid areas, while grassland set-asides in semi-arid climates, for example, might have carbon increases at the lower end of this range. See the companion paper, Richards et al. 2006, for a discussion of options for increasing aboveground carbon in U.S. forests.

Highly degraded sites, such as severely eroded areas, reclaimed surface mines, and saline soils represent situations with high potential carbon sequestration rates but also higher costs and technical difficulties associated with the reclamation. Where vegetation has been absent and/or topsoil removed, the *initial* carbon contents prior to restoration will be very low. Hence, the capacity for accruing soil carbon is large, provided that a productive plant cover with high rates of carbon inputs from residues can be established and maintained. Lal et al. (1998) cite studies reporting several-fold increases in carbon stocks over a 5 to 15 year period following restoration of severely degraded sites.

Cultivated organic soils represent another land restoration opportunity. These lands are a significant source of agricultural CO_2 emissions, with high rates of up to 10 to 20 t/ha/yr of carbon (Ogle et al. 2003). Hence, wetland restoration may be a mitigation option. However, restored wetlands may emit methane, which would need to be considered in assessing the overall mitigation potential of this type of restoration.

3. Total Agricultural Soil Carbon Sequestration Potential

Carbon sequestration rates vary by climate, topography, soil type, past management history, and current practices. Various global and national estimates for potential soil carbon sequestration have been made. These estimates are usually based on overall carbon gain for a suite of practices and the available area on which these practices could be applied, resulting in estimates of biological or technical potential. Paustian et al. (1998) estimated a global potential from improved agricultural soil management of 400 to 600 MMT of carbon per year, and the Intergovernmental Panel on Climate Change (IPCC 2000) estimated potential rates from improved cropland, grazing land, and agroforestry of 390 MMT of carbon per year by the year 2010 and 780 MMT of carbon per year by 2040, assuming a lag-time in the adoption of improved practices. In the United States, Lal et al. (2003) estimated an overall potential for soil carbon sequestration (excluding forest-related activities) of 70 to 221 MMT of carbon per year for a combination of practices including land set-asides, restoration of degraded lands, conservation tillage, irrigation and water management, and improved cropping and pasture systems. This overall figure represents net increases, taking into account increased GHG emissions associated with the management improvements. A more detailed, regionalized assessment for U.S. annual cropland alone estimated a total soil sequestration potential of 83 MMT carbon per year (see Box 3 and Figure 2).

Box 3. U.S. Cropland Carbon Sequestration Potential

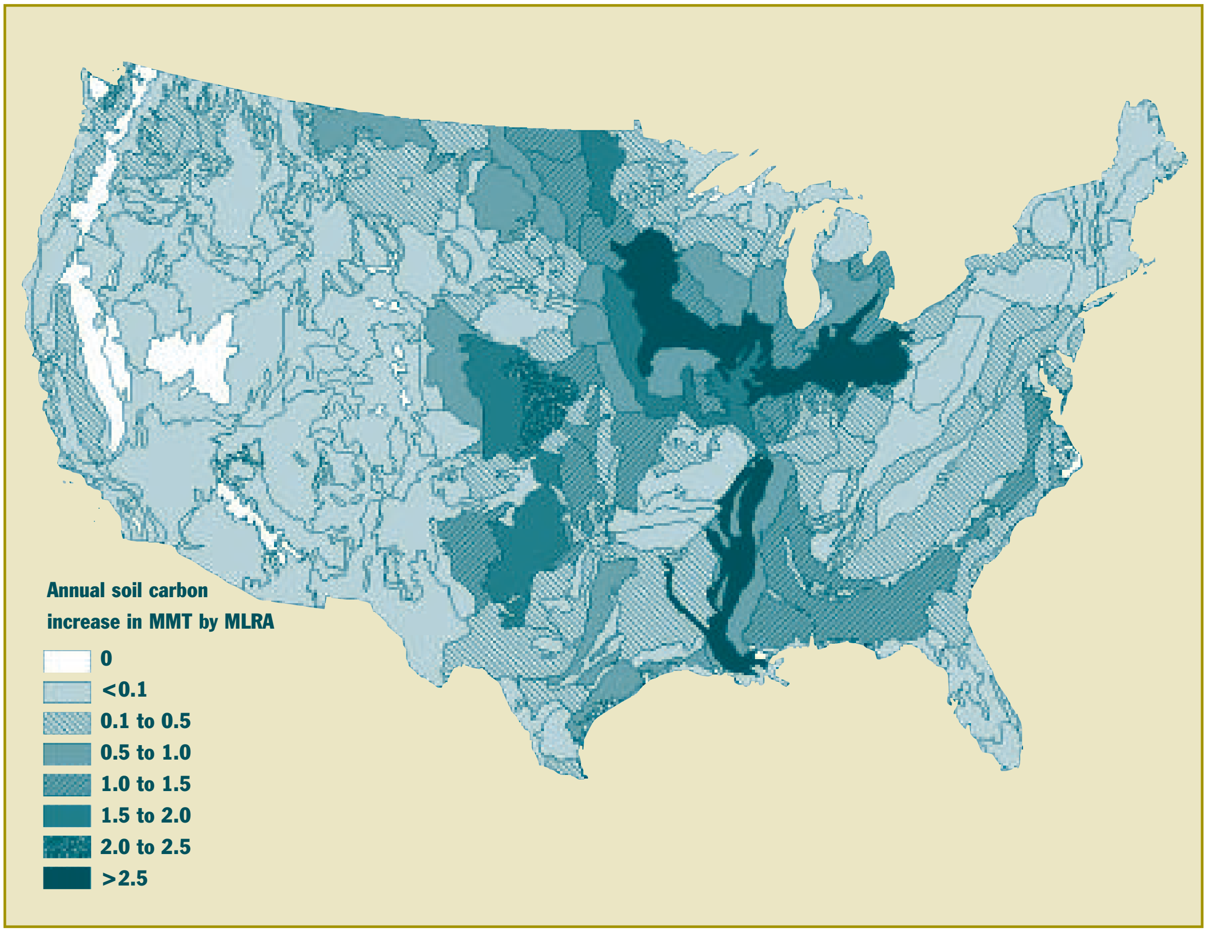
A regional assessment of U.S. soil carbon sequestration potential was carried out by Sperow et al. (2003). Several scenarios entailing widespread adoption of 'best management practices' for carbon sequestration were analyzed. Potential sequestration rates for U.S. cropland ranged up to 83 MMT carbon per year. The highest amounts were attributed to conversion to no-till (47 MMT carbon per year), followed by use of cover crops (22 MMT carbon per year), set-asides (11 MMT carbon per year) and elimination of summer fallow (3 MMT carbon per year). Potential soil carbon sequestration (from combined practices) was highest in the northern Corn Belt region and in the Mississippi delta, with somewhat lower amounts in the Piedmont region of the Southeast and the Northern, Central and Southern Great Plains.

There are numerous uncertainties surrounding such estimates of carbon sequestration potential. On the one hand, development of new technologies specifically targeted at increasing soil carbon (through plant breeding or new soil amendments) could increase potentials. On the other hand, rising temperatures due to global warming will likely stimulate soil organic matter decomposition, which may reduce or eliminate the potential to further increase soil carbon stocks. Finally, the amount of carbon sequestration which is actually *achieved* will depend on economic, social, and policy factors (see Chapter III and the companion Pew report, Richards et al. 2006) and is likely to be substantially less than the biological and technical potential cited above.

Figure 2

Geographic Distribution of Potential Soil Carbon Sequestration

on Agricultural Land with Widespread Adoption of Current Best Management Practices



Source: Sperow et al. 2003.

Note: MLRA = Major Land Resource Area.

B. Reducing Agricultural Nitrous Oxide and Methane Emissions

Nitrous oxide (N_2O) and methane (CH_4) emissions result from both crop and livestock operations and account for approximately 80 percent of U.S. agricultural greenhouse gas emissions on a GWP basis. Despite challenges, there is considerable scope for reducing these emissions. Nitrous oxide constitutes the largest agricultural source of GHG emissions in terms of warming potential (48 percent), and almost 70 percent of total U.S. nitrous oxide emissions are from soils. The best option for reducing these emissions is to use fertilizers more efficiently; adoption of best fertilization practices could reduce agricultural N_2O emissions by 30 to 40 percent (CAST 2004). Livestock are the main source of agricultural CH_4 emissions (see Figure 1, page 2, Ruminant CH_4). Increasing the efficiency of production (meat, milk) per animal can decrease these emissions and also reduce costs. Manure management accounts for 25 percent of U.S. agricultural CH_4 emissions; anaerobic (i.e., oxygen-free) digesters that capture and use the methane as an energy source—thereby displacing fossil fuels—offer a nearly ideal solution for these emissions. Adoption of best practices could reduce total U.S. agricultural methane emissions by 20 to 40 percent (CAST 2004).

1. Reducing Nitrous Oxide and Methane Emissions From Soils

A characteristic of modern agriculture is the huge increase in nitrogen supplied—not only as mineral fertilizer but also through nitrogen-fixing crops (e.g., alfalfa, clover, and soybeans) and animal manure—to boost crop productivity (Mosier et al. 2001). Because N_2O emissions are strongly influenced by the availability of nitrogen in soil, some N_2O emissions are an unavoidable consequence of maintaining highly productive crop and pastureland. However, improved control of the amount, timing, and placement of fertilizer can minimize these emissions. Methane emissions from agricultural soils are mainly associated with flooded soils such as rice-growing areas and wetlands. Most soils are not a major source of CH_4 and, in fact, most non-flooded soils remove some CH_4 from the atmosphere.

Nitrous oxide. Unlike the case for CO_2 and CH_4, there are no significant biological sinks for atmospheric N_2O. Since agricultural N_2O emissions correlate with the amount of nitrogen available in soils, mitigation rests largely on increasing the efficiency of nitrogen use without compromising crop yields.

Using nitrogen more efficiently means better matching its availability to plant needs. However, because of variable weather conditions, it is hard to predict crop nitrogen needs at the start of the growing season when most fertilizer is applied. Since fertilizer is relatively inexpensive in comparison with other farm costs, over-fertilization (applications in excess of plant needs) is common as farmers hedge against inadequate nutrients. Greater than 50 percent of the major cropland area in the United States is rated as having high nitrogen balances, resulting in soils highly susceptible to losses of N_2O to the atmosphere and nitrate (NO_3^-) to water bodies (USDA 2003).[6] Where animal manure is applied, farmers may not adequately account for its nitrogen contributions and, therefore, add too much supplemental fertilizer. Also, with increasing size and concentration of confined animal feeding operations (CAFOs), manure supply and transport costs lead to application of manure at higher than recommended rates on nearby fields (Edmonds et al. 2003).

In addition to application rate, timing is a factor in the efficiency of nitrogen use. Currently about 30 percent of the U.S. area cropped to corn, cotton, and potatoes was fertilized in the autumn (USDA 2003). Fall application of fertilizer (motivated by lower fertilizer costs in the fall and to save time during spring planting) results in high concentrations of mineral nitrogen remaining in soils over a several-month period with no plant uptake, making that nitrogen vulnerable to losses. Applying fertilizer only after the start of the growing season—and ideally as split applications over time—provides better synchrony with plant demands because crop uptake capacity is low at the beginning of the growing season, increases rapidly during vegetative growth, and then drops sharply as the plant nears maturity. Slow-release fertilizers, such as sulfur-coated urea, which delay the release of fertilizer applied at planting time until plant nitrogen uptake capacity is higher, can also be used.

Where and how fertilizer is applied also influences the efficiency of nitrogen use. Surface application of fertilizer and manure is subject to greater volatilization losses, predominantly as ammonia gas, than injected fertilizer. Ammonia is eventually deposited downwind in environments where it may result in N_2O emissions. In 1997, fertilizer was surface-applied on about 30 percent of corn, 60 percent of wheat, and more than 70 percent of cotton and potato acreage (USDA 2003). Injecting fertilizer and manure into the soil, near the zone of active root uptake, both reduces nitrogen losses and increases plant nitrogen use, resulting in less residual nitrogen that can be lost as N_2O.

Methane. Methane is produced in soils by bacteria, termed *methanogens*, which function under strictly anaerobic (oxygen-free) conditions. Consequently, CH_4 emissions from agricultural soils are largely restricted to flooded soils, such as those used for rice cultivation and other cultivated wetland crops (e.g., cranberry bogs), where water-saturated conditions inhibit the diffusion of oxygen into much of the soil. Because the area of flooded rice land and other wetland crops in the United States is small, total CH_4 emissions from these sources in the United States are also relatively small— about 0.4 MMT of CH_4 per year (two MMT carbon-equivalent) from 1.4 million hectares (Mha) of rice lands in California, the Gulf Coast and Mississippi delta region, and Florida (USEPA 2005).[7] However, on a per-hectare basis the emissions are significant. Potential mitigation options include changes in crop breeding and management of water, fertilizer, and residues. Perhaps most promising is the selection and breeding of new rice varieties that are less conducive to transport of CH_4 through the plant to the atmosphere (Aulakh et al. 2002). Because 60 to 90 percent of CH_4 emissions from growing rice occur via transport through the plant tissues,[8] choosing rice cultivars with a high resistance to CH_4 transport could reduce emissions by as much as 50 percent (Sass 1994).

Under aerobic (oxygenated) conditions, other soil bacteria consume CH_4, oxidizing it to CO_2. Because CH_4 has a GWP 21 times greater than CO_2 (see Box 1, page 3), the conversion of CH_4 to CO_2 yields an overall decrease in total GHG warming. Globally, soils eliminate about 20 to 60 MMT of CH_4 per year (115 to 345 MMT carbon-equivalent) through oxidation. The highest rates of CH_4 oxidation occur in undisturbed, native ecosystems. Cultivated soils have much lower rates of CH_4 oxidation—for example, CH_4 oxidation was reduced by 80 percent in annual cropland compared with deciduous forests in southern Michigan (Robertson et al. 2000), and similar reductions (80 to 90 percent) were found when cropland was compared with native prairie in eastern Colorado (Bronson and Mosier 1993). In general, conversion of marginal cropland to permanent set-aside and use of no-till methods on annual cropland are the practices that will be most beneficial to strengthening the CH_4 sink on agricultural soils.

2. Reducing Livestock-Related Methane and Nitrous Oxide Emissions

Livestock-related emissions from digestive processes and animal wastes account for 26 percent of total agricultural emissions. Although enteric (digestive tract) emissions are more significant (70 percent of agricultural CH_4 emissions), emissions from livestock wastes have a greater potential for mitigation. Improving manure-handling facilities, for example by covering animal-waste lagoons and capturing and burning the CH_4, can reduce emissions while providing renewable energy and income. Capture and combustion of CH_4 from animal wastes also reduces other environmental problems, including odors and nitrate pollution. Overall the best option for reducing digestive process emissions is to increase the *efficiency* of livestock production.

Manure storage and management. Manure management in the United States currently accounts for 25 percent of agricultural CH_4 and 6 percent of agricultural N_2O emissions. In addition to GHG production, problems associated with odor and nutrient pollution from animal wastes are widespread. Hence, improvements in manure handling that address both GHG reductions and odor and nutrient problems are of great interest.

Manure produced by livestock can emit N_2O and/or CH_4 during storage and following application to soil. The type and rate of these emissions are highly dependent on the storage conditions and characteristics of the manure. In general, storage under anaerobic conditions (lacking oxygen, such as in waste lagoons) will produce CH_4 while N_2O emissions will be suppressed.[9] Conversely, piled storage and composting of manure will promote largely aerobic decomposition, suppressing CH_4 emissions but promoting N_2O emissions. Both types of handling and storage methods are widely used in the United States. Lagoon storage is frequently used for large dairy facilities and other confined animal operations, while piled storage is typical for smaller farms and for some confined operations such as beef feedlots. Anaerobic digesters[10] in conjunction with lagoon storage systems offer a nearly ideal option—N_2O emissions are suppressed and CH_4 can be used as an energy source, thereby displacing fossil fuels.

Opportunities for mitigating N_2O emissions from stockpiled or composted manure are relatively limited. One way to reduce N_2O emissions is by increasing the ratio of bedding material (straw or sawdust) to manure. This immobilizes more nitrogen by converting it into organic compounds. Perhaps the most effective measure for reducing manure-related N_2O emissions from stockpiled or composted manure is to apply the manure at rates based on crop needs, thus maximizing plant uptake of manure-derived nitrogen, and minimizing the amount of residual nitrogen left in the soil.

Enteric fermentation. Methane is produced in the digestive tract of animals, particularly in ruminants such as cows, sheep, goats, and camels. This source of CH_4 emissions is termed *enteric fermentation*. In the United States these emissions amount to about 5.4 MMT of CH_4 per year (30.7 MMT carbon-equivalent), which represents about 70 percent of agricultural CH_4 emissions and 20 percent of total agricultural GHG emissions on a carbon-equivalent basis.

Because CH_4 emissions from enteric fermentation are influenced by the feed quality and digestive efficiency of the animals, improving these will reduce CH_4 emissions. In simple terms, the more rapidly food is processed and passed through the rumen (first stomach of ruminants), the less time there is for CH_4 production. For most confined livestock in the United States, feed quality and digestibility are already at a relatively high level, and further improvements from conventional changes in feed rations are likely to be modest. However, where diets are not optimal, incorporating more digestible feed such as grain, silage,[11] and legume hay (e.g., clover or alfalfa) in the diet can reduce emissions. One area where substantial improvements are possible is in improving forage quality for grazing animals on smaller livestock operations through better pasture management (DeRamus et al. 2003). Various feed additives such as edible vegetable oils and certain antibiotics can also be used to inhibit the rumen bacteria that produce CH_4 (Teather and Forster 1998).

For a given animal type and food quality, CH_4 production will be roughly proportional to food intake. Thus, increasing the amount of product (meat, milk) per unit of food consumed will effectively reduce CH_4 emissions per unit of product. Ways to increase the production efficiency of individual animals include improved animal genetics (breeding) and animal health.

C. Measurement, Modeling, and Information Needs

Reliable measurements and models are necessary to design and assess policies for mitigating GHG emissions. As the United States intensifies its efforts to include the agricultural sector in efforts to address climate change, it will be important to improve the accuracy and robustness of estimates of the GHG implications of adopting practices described above. In particular, the capacity to estimate methane and nitrous oxide emissions and emission changes needs to be strengthened, and a national monitoring system to provide measurements of soil carbon stocks over time should be established. A combination of field measurements and models provides the best means to estimate national- and regional-scale agricultural emissions and sinks and to forecast changes in emissions due to changes in management practices, environmental and economic conditions, or government policies. To provide better data for GHG inventories and models, additional information on specific management activities should be collected together with measurements of soil attributes as part of a national soil inventory network.

1. Measurement of Greenhouse Gas Emissions and Carbon Sinks

Both emissions and sinks of GHGs can be measured directly through measurement of gas fluxes.[12] To date, however, measuring GHG fluxes from soils has been largely restricted to research applications. In the case of CO_2, net emissions between the atmosphere and the land surface can also be inferred from changes over time in carbon stocks (i.e., the amount of carbon in soils and vegetation) (see below). The carbon content of a soil sample can be determined with a high degree of precision. Thus, the challenges in estimating changes in carbon stocks are not in measuring soil carbon content at a specific location *per se* but rather in designing accurate and cost-efficient *sampling* designs to provide estimates of changes for fields and across larger areas. Measurement techniques available for gas fluxes and for carbon stocks, as well as their relative strengths and limitations, are described below.

Measurement of greenhouse gas fluxes. Rates of GHG emissions from soils and/or uptake of CO_2, and CH_4 can be measured directly (see Box 4). However, measuring flows of these gases over areas and time periods of interest poses significant difficulties because emission rates are highly variable in both space and time. For instance, emissions rates of N_2O can change 100-fold or more following a rainstorm or fertilization (Smith et al. 1997b), and similar changes in CO_2 emission rates occur following tillage (Reicosky 1997). Hence, deriving annual flux rates (see endnote 12 for definition of "flux") requires frequent sampling so that large, short-term fluxes are adequately represented. The high spatial variability in flux rates implies that either several small areas within a field need to be sampled and then averaged or that the measurement technique itself needs to integrate fluxes over a fairly large area.

Box 4. Gas Flux Measurement

Two basic methods can be used to measure fluxes from soils: chambers and micrometeorological techniques. Use of chambers is restricted to low-growing vegetation or bare soils, and their small size (usually less than one square meter) means that several chambers are needed for a given area to account for the high spatial variability of gas fluxes. Chamber methods also generally require highly labor-intensive, repeated sampling to obtain annual flux estimates. A few automated chamber systems have been developed, but they require sophisticated facilities. Micrometeorological techniques use measurements of air movement to and from the land surface, together with measurements of gas concentrations within the air mass. Integrating these measurements provides an estimate of gas fluxes between the land surface and the atmosphere. Applications range in scale from small tower-based measurements that estimate fluxes from tens or hundreds of square meters to aircraft-based measurements that integrate fluxes from tens or hundreds of square kilometers of land area. However, micrometeorological approaches are also subject to limitations and the relative expense and sophistication of the instrumentation largely restrict their application to research purposes, at least at present.

Measurement of soil carbon stocks. Changes in soil carbon stocks can be used as an integrated measure of net soil CO_2 flux over time because the predominant flows of carbon between soils and the atmosphere are in the form of CO_2.[13] Soils that are accumulating carbon represent a net flux of CO_2 from the atmosphere (via the plant) to the soil, and soils that are losing carbon represent a net flux of CO_2 to the atmosphere. Hence, measurements of soil carbon stocks at a set location, over a known time period (e.g., $[C_{yr\,=\,2005} - C_{yr\,=\,2000}]/5$ years = average annual carbon flux between year 2000 and year 2005) provide an estimate of the net CO_2 flux at that location over that period.

Soil carbon content can be accurately measured using modern dry-combustion carbon-nitrogen analyzers, and even older methods (e.g., wet-oxidation) provide acceptable accuracy and precision.[14] Consequently, designing cost-effective sampling schemes is the main challenge in estimating carbon stock changes over larger areas. As with gas fluxes, soil carbon content typically exhibits high spatial variability, showing as much as five-fold differences, even within uniformly managed fields (Robertson et al. 1997). Furthermore, most soils contain substantial amounts of carbon. Typical agricultural soils in the midwestern United States contain 30 to 60 tonnes of carbon per hectare in the upper 20 centimeters, whereas annual changes in stocks rarely exceed one t/ha/yr. Therefore, several years (perhaps five years or more) between sampling times is usually required to reliably detect and measure stock changes. In short, soil-monitoring networks need designs that account for both the spatial variability of soils and the response time of soil carbon stocks. (See Box 5.)

Box 5. Direct Measurement of Soil Carbon

Key elements of an appropriate sampling procedure include: (1) use of precisely geo-referenced sites for repeated sampling; (2) use of sufficiently large diameter cores to enable accurate determination of bulk density and carbon concentration on the same soil sample (see endnote 14); (3) consistent accounting of carbon in surface debris; and standardization of sampling time (i.e., season), equipment, and soil preparation procedures; (4) appropriate use of nested or stratified sampling designs to account for influences of soil type, topographic position, and hydrologic conditions; (5) archiving of samples for subsequent study or reanalysis; and (6) acquisition and use of verified soil standards (i.e., a soil sample where carbon content is very precisely known and which is used to calibrate instruments).

Because much of the variability in soil organic matter content occurs within short distances (i.e., variability is expressed at fine spatial scales), the sampling intensity required to detect an average change in carbon stocks at a specified confidence level[15] diminishes sharply as the area of land being considered increases (Conant and Paustian 2002). While hundreds of sample locations might be required to adequately characterize soil carbon stocks for a single county, several thousand locations might suffice to characterize stocks for the entire country. Options that can significantly reduce sampling requirements include accepting lower statistical confidence levels (e.g., 90 percent vs. 95 percent) and lengthening the resampling period in order to increase the ability to detect change relative to background variability. However, such options necessarily involve tradeoffs between the accuracy and precision[16] of the estimates and the cost of obtaining the information.

2. Modeling and Information Needs

The factors that control GHG fluxes and soil carbon stock changes can be articulated in mathematical models. There are two basic types of models: empirical models and "process-oriented" models. Empirical models use data from field measurements to determine statistical relationships between soil carbon stocks and environmental and management factors (e.g., IPCC 1997; Ogle et al. 2003), whereas more dynamic, "process-oriented" models attempt to simulate the biological, chemical, and physical processes controlling GHG dynamics. Process-oriented models are particularly useful because they can represent many combinations of management practices and soil and climate conditions. A number of dynamic, process-based models have been developed to simulate soil carbon stock changes (reviewed in McGill 1996) and nitrous oxide and methane fluxes from soil (reviewed in Frolking et al. 1998). Previously process models were used primarily for research purposes, but applications to provide estimates for policy and decision makers on soil carbon and GHG mitigation are increasing rapidly (e.g., Paustian et al. 2001, 2002; Brenner et al. 2002; Del Grosso et al. 2005; USEPA 2006).

For regional applications, models require input data on environmental conditions—such as climate, soil type, and topography—and on management activities, and how all these data vary over time and location within the region. In the United States, high-quality spatial data are generally available for climate, soil type, and topography; and the variety of spatial data sets available, especially from remote sensing, is increasing. However, accurate data on land use and management activities are generally less available and are the most limiting data component for model-based estimates. While there is a great deal of aggregate data on agricultural management practices at county, state and national levels, this aggregate data has significant limitations for analyzing relationships between GHGs and management practices. For example, agricultural cropping systems are comprised of crop rotations; and practices such as fertilizer application, tillage, and manuring vary for the different crops within the rotation. Thus, for example, county-level data on total fertilizer use fails to provide adequate detail for models attempting to forecast GHG changes due to specific changes in crop rotations.

The National Resources Inventory (NRI) is a land-use inventory that has collected general data on land use and management (e.g., land cover and crop rotations) at several hundred thousand points on non-federal land across the entire United States since 1982. Hence, the NRI is one of the primary sources of data used in models for estimating national-level agricultural GHG emissions and sinks (USEPA 2006). However, information on many management practices such as tillage, fertilization, and manuring has not been routinely collected in the NRI survey. Thus, available aggregate data on these management practices must be adjusted and interpolated for use in the models, resulting in greater uncertainty in estimates of soil carbon and GHG fluxes than if more detailed management data were available.

Collecting additional information on specific management activities—such as tillage practices and crop-specific fertilizer use on croplands and grazing practices and fertilizer use on pastures—at some or all of the NRI locations would provide a more comprehensive and consistent set of data for estimating GHG fluxes.[17] Furthermore, direct measurements of soil carbon at a small subset of NRI points could form the basis for a soil-monitoring network. Currently, the United States lacks a system of permanent benchmark locations where soil carbon is measured over time in conjunction with collection of data on management practices.[18] Pilot soil carbon inventory systems with permanent benchmark sites have been established in the Canadian prairies (McConkey and Lindwall 1999), and a national system has been instituted in New Zealand (Tate et al. 2003). Over time, such a system in the United States would provide valuable information to improve models and provide more accurate and reliable inventory estimates.

Existing long-term field experiments have played an essential role in developing models to estimate carbon sequestration and GHG emissions (Paul et al. 1997; Ogle et al. 2003; Smith et al. 1996). In addition to the distributed, on-farm network of soil monitoring locations linked to the NRI (described above), more long-term field experiments are needed in which different management systems are analyzed while controlling for the key management variables affecting soil carbon and GHG emissions, such as fertilizer use. This is especially important in underrepresented areas and systems—including sites that measure nitrous oxide and methane fluxes—to improve empirical models and to test and validate process-based models.

D. Additional Benefits of Agricultural Greenhouse Gas Mitigation

Adopting management practices that reduce GHG emissions and/or increase soil carbon will contribute additional environmental benefits. Most of the environmental problems stemming from agricultural activities have a basic underlying cause—inefficient (wasteful) use of resources. For instance, inefficient use of nitrogen fertilizers and animal manure—either applying amounts in excess of crop needs or poor choices in timing, form, or placement—is the key source of high nitrous oxide emissions, as well as nitrate leaching into groundwater, nitrogen loss in runoff, and ammonia volatilization. More efficient nitrogen fertilizer use can reduce pollution of groundwater, lakes, and streams. Methane digesters for treatment of animal waste can provide on-farm energy as well as help reduce odors and pollution from wastes. Hence, the true value of improvements in agricultural management should be measured by considering all associated environmental benefits, including GHG mitigation.

Many of the management practices suggested for carbon sequestration protect the soil surface, dramatically decrease erosion rates, and increase soil organic matter. The importance of organic matter to maintaining healthy and productive soil is well recognized by any weekend gardener. Organic matter in soil performs numerous key functions, acting as a repository for soil nutrients, promoting favorable soil structure for plant rooting and water-holding capacity, and acting as a filter for pesticides and other organic compounds. Soils that are rich in organic matter and are well aggregated, with a granular structure, also tend to be less susceptible to soil erosion from both rain and wind. Reduced sediment runoff translates into improved water quality, reduced costs for water treatment, less dredging for navigation, longer life spans for reservoirs, and healthier aquatic ecosystems. Reduced wind erosion means better air quality, less dust, and a cleaner viewscape. Thus, the buildup of soil organic matter that accompanies carbon sequestration provides many environmental, social, and economic co-benefits.

If all of the above benefits result from more conservation-oriented management practices, why aren't these practices being used everywhere? On the surface, this seems particularly puzzling given that the vast majority of farmers and ranchers view stewardship of the land as an important value. The answer is complex and involves many technical, social, and political factors—in particular, how agricultural activities translate into economic values in the marketplace, as discussed in Chapter III.

E. Summary and Policy Implications

U.S. agriculture has significant technical potential to reduce GHG emissions and sequester carbon in soils using currently available technology. Overall, potential GHG mitigation amounts to 102 to 270 MMT carbon-equivalent per year over the next few decades or 5 to 14 percent of total 2004 U.S. GHG emissions.

Because of the multiple factors affecting emission and sequestration processes, some practices that decrease emissions of one gas may increase emissions of another. The key to reducing net GHG emissions from agriculture is to promote practices that maintain or increase carbon stocks, while at the same time increasing the efficiency of agricultural inputs (e.g., fertilizer, irrigation, pesticides, animal feed, and animal waste). GHG mitigation policies should address both carbon stocks and emissions of nitrous oxide and methane to achieve the best overall mitigation results.

The largest potentials for soil carbon sequestration are associated with adoption of no-till practices, reduced fallow and use of cover crops, and conservation set-asides with perennial grasses and trees on highly erodible cropland. Nitrous oxide emissions from soils constitute the single largest agricultural GHG source. More efficient use of nitrogen fertilizer and manure, as well as additives that inhibit the formation of nitrous oxide in soils, could reduce nitrous oxide emissions by 30 to 40 percent.

Livestock production accounts for most of the methane emitted from U.S. agriculture, largely through enteric fermentation and emissions from stored manure. Manure management systems that capture and combust methane can provide a renewable energy source that is both profitable to farmers and can displace fossil fuels. Improved production technologies (e.g., improved feed quality, methane-suppressing feed additives, and animal breeding) can reduce enteric methane emissions, increase livestock production, and perhaps improve profitability. Adoption of best practices could reduce total U.S. agricultural methane emissions by 20 to 40 percent.

Accurate methods to quantify GHG emissions at multiple scales—from the farm level to the national level—are key to successful policy design, implementation, and evaluation. Agricultural sinks and emissions are highly variable and spatially dispersed (i.e., "non-point"), posing challenges in designing efficient and cost-effective measurement approaches. Combined measurement- and model-based inventory and assessment tools have been developed and are being used to inform managers and policy makers. However, further research and development is needed to improve the accuracy and utility of these tools. Examples are:

- Additional data collection on key management activities (e.g., tillage, fertilization, manuring, and grazing practices on cropland and pastures) as part of land-use inventories such as USDA's National Resource Inventory.

- Establishment of a network of permanent benchmark sites in farmer fields, suitable for periodic measurement of soil carbon and other soil attributes.

- Greater integration of data from remote sensing as well as development and application of new technologies for more rapid, cheaper *in situ* measurement of soil carbon stocks and GHG fluxes.

- Additional field and laboratory research on poorly studied systems, including (1) emissions from drained, organic soils (such as peat soils) converted to cropland; (2) changes in GHG emissions following restoration of drained soils to wetland conditions; (3) net GHG fluxes from no-till and reduced till practices and from systems with only infrequent tillage; and (4) impacts of cover crops on GHG emissions and sinks.

- Rigorous validation and uncertainty analysis of estimation methods to improve our ability to forecast future agricultural GHG emissions and sinks and to analyze the potential outcomes of mitigation policies.

The additional economic and water, soil, and air quality benefits resulting from the practices that reduce GHG emissions should be factored in to broaden policy support. Policy actions can include providing outreach, education, and technical assistance to landowners through existing or new programs, or enhancing and structuring government support programs so that they better target the production of environmental goods, including GHG mitigation. For a discussion of policy options, see the companion Pew Center report, Richards et al. 2006.

III. Economic Feasibility of Agricultural Carbon Sequestration

Studies show that, through the use of available technology, carbon could be accumulated and stored in agricultural soils and afforested lands at costs competitive with other forms of emission reductions. The economically feasible amount of carbon that could be sequestered by agriculture varies across regions and depends on the economic incentives, each region's environmental conditions, the land-use and management options suitable to each region, and farmers' socio-economic characteristics. This chapter reviews the emerging literature on the economic feasibility of GHG mitigation in U.S. agriculture through changes in land-use and management practices. Most of this literature addresses the potential for carbon sequestration. It is important to note that carbon sequestration in agricultural soils occurs until a maximum level is attained, which, in most cases, is estimated to occur 20 to 30 years following a change in management. Carbon accumulation through afforestation spans a longer time period depending on the maturation period of trees, estimated to be from 70 to 150 years for important species (e.g., Loblolly and Ponderosa pine; Stavins and Richards 2005). The lack of adequate, sufficiently detailed data has hampered researchers from estimating the costs of reducing nitrous oxide and methane emissions in agriculture. Section A below discusses the findings of economic, model-based studies of carbon sequestration; Section B reviews the literature covering factors that influence adoption of carbon sequestering practices not analyzed in these studies.

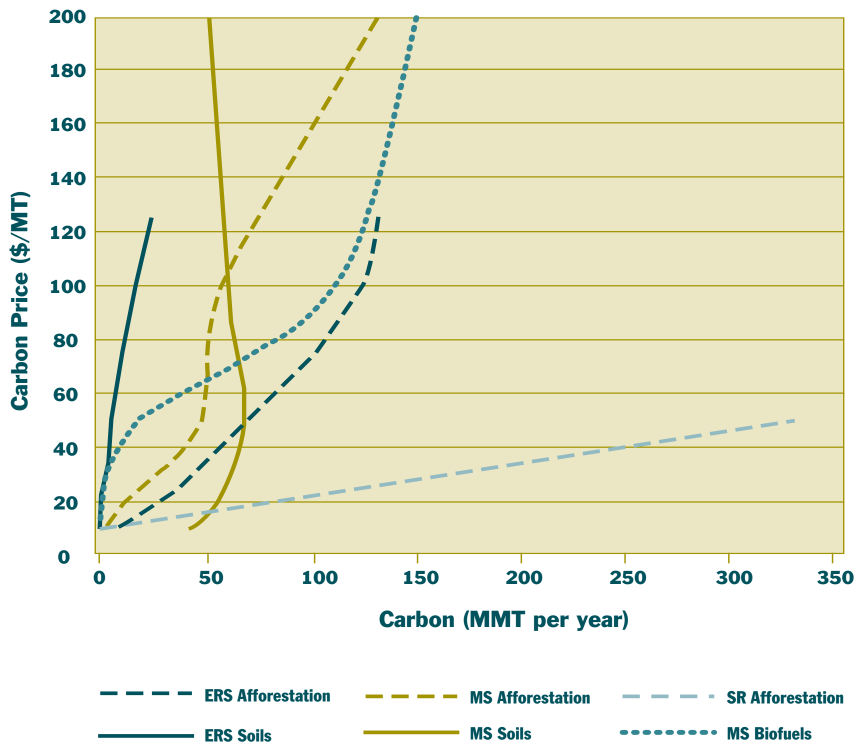
A. Economic Studies of Carbon Sequestration

The economically rational farmer will switch to practices that sequester more carbon only if those practices are profitable. Farms are businesses, and profitability is a key factor in predicting farmers' choice of practices (Griliches 1980; Kislev and Peterson 1982; Uri 1999). The land-use or management practices that sequester the most carbon are not necessarily the most profitable. Therefore, incentives *at least as great as any income lost due to changing practices* will be required to induce some farmers to adopt relevant practices. As the incentives increase, the amount of carbon sequestered in a region will increase and approach the technical potential.

Recent model-based studies of economic potential for carbon sequestration on agricultural lands are listed in Appendix A. These studies indicate that agriculture can sequester carbon at a cost competitive with other forms of GHG reductions. The models show that carbon sequestration rates and the amounts achievable vary regionally, and that costs of achieving sequestration depend on whether the sequestration is achieved through afforestation or cropland management changes. Costs also depend on: (1) whether payments are offered per hectare for all hectares in specified management or land uses, or per tonne of carbon sequestered; (2) whether all farmers using a desired practice or only new adopters are eligible to receive payments; and (3) the length of the payment period.

Figure 3

Carbon Supply Curves for Afforestation, Biofuels and Changes in Crop Management in the United States*



*Based on Lewandrowski et al. 2004 (ERS), McCarl and Schneider 2001 (MS), and Stavins and Richards 2005 (SR).

Note: A key difference between the studies of afforestation appears to be the assumptions made about where trees are likely to be grown. The Lewandrowski et al. and McCarl and Schneider studies assume that little or no afforestation would occur in the plains and mountain states region. In contrast, the studies on which the Stavins and Richards afforestation supply curve is based assume that a significant amount of afforestation would occur on rangelands which are located predominantly in the plains and mountain regions.

The economic feasibility of forest and soil carbon sequestration can be represented with supply curves. These curves are constructed using estimates of: (1) the technical potential for carbon sequestration; (2) the amount of land on which carbon-sequestering practices could be adopted; and (3) farmers' adoption of practices that sequester carbon in response to different incentive levels. Figure 3 presents such carbon supply curves from McCarl and Schneider (2002), Lewandrowski et al. (2004), and Stavins and Richards (2005). As can been seen from Figure 3 and Appendix A, these studies vary considerably in their estimates of how much carbon would be stored in soils annually at any given price for carbon. For example, as prices increase from $10 to $125 per tonne, Lewandrowski et al. (2004) predict soil carbon storage increasing from one to 27 MMT per year, whereas the McCarl and Schneider (2002) study predicts an increase from 44 to 70 MMT per year. At $50 per tonne of carbon, the Lewandrowski study suggests that 10 MMT would be stored annually, while the McCarl and Scneider study gives 70 MMT of carbon per year.

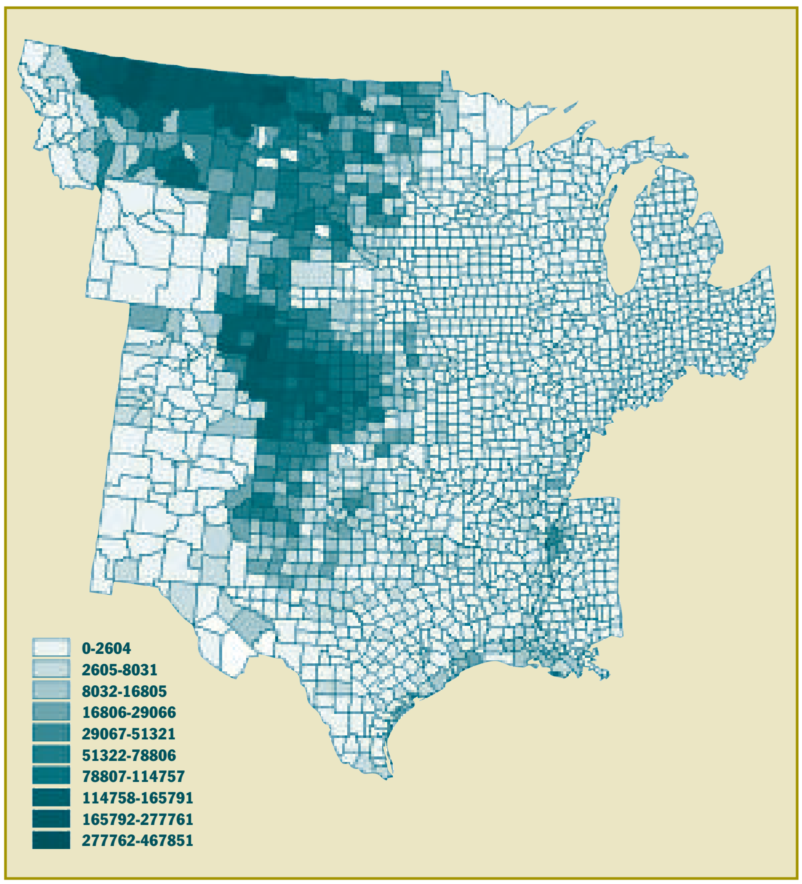
All of the studies shown on Figure 3 and summarized in Appendix A are based on the principle of opportunity cost, i.e., that to obtain more of one thing, something else must be given up. In some cases carbon-sequestering practices are more profitable than conventional practices and in those cases they will usually be adopted without additional incentives. In other cases, when a farmer changes practices to obtain more soil carbon, there will be a reduction in income unless the sequestered carbon has a market value. In these cases, as carbon prices rise, more farmers will switch to carbon-sequestering practices and more carbon will be stored. This relation between carbon price and amount of carbon sequestered is shown in the supply curves in Figure 3.

All of the studies listed in Appendix A and the data shown in Figure 3 should be regarded as preliminary estimates of the economic potential for carbon sequestration. The studies make various assumptions that may limit the generality

Figure 4

Wheat Acreage in Crop Fallow Rotation

in the Central United States, by County



Source: U.S. Census of Agriculture.

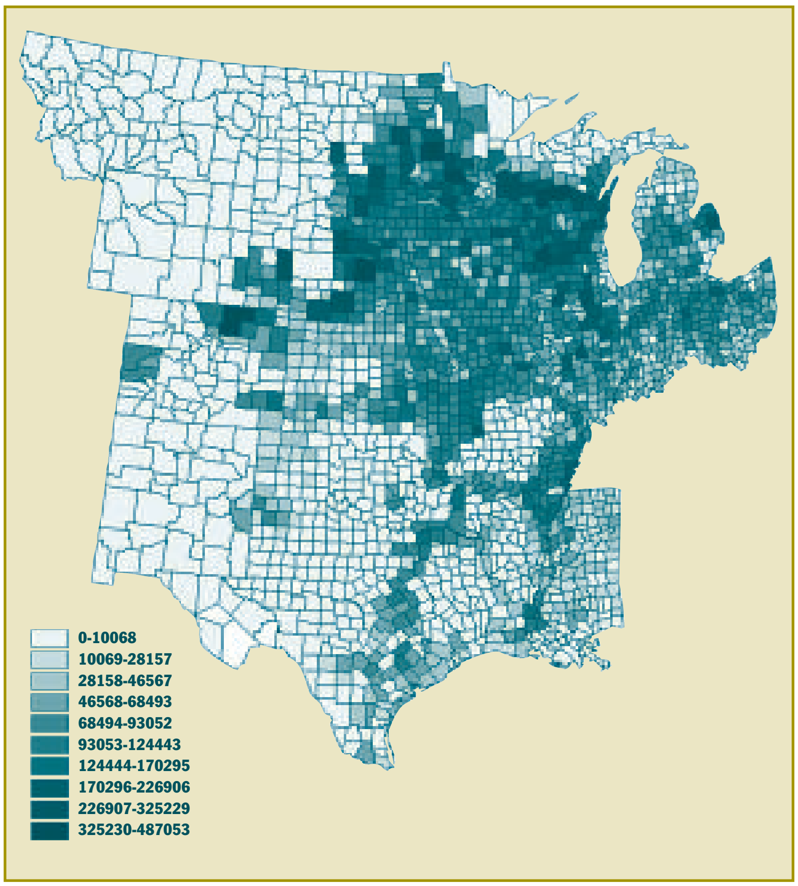
and comparability of their findings. In addition to use of different assumptions, results vary because they are based on different models and use different data sets. The following discussion elaborates further on key factors influencing costs of soil carbon sequestration and on differences among the models used.

1. Regional Differences

The agriculture and forestry studies summarized in Appendix A show that the economic potential for carbon sequestration varies substantially across regions of the United States. These differences in economic potential are explained by several factors. First, the amount of carbon sequestration achievable differs by location, as shown in Figure 2 on page 13, which portrays the different amounts achievable across the United States. Second, amounts of agricultural land available for conversion to new practices also vary by location. For example, Figure 4 shows acres in wheat-fallow rotation in the central United States by county—acres which could be switched to more continuous cropping. Figure 5 shows the acres under conventional tillage in the corn-soybean system in this region—acres that could be switched to conservation tillage. Third, the costs of changing practices vary by crop due to differences in profitability and capital costs.

Figure 5

Corn, Soy, and Feed Acreage Managed under Conventional Tillage in the Central United States, by County



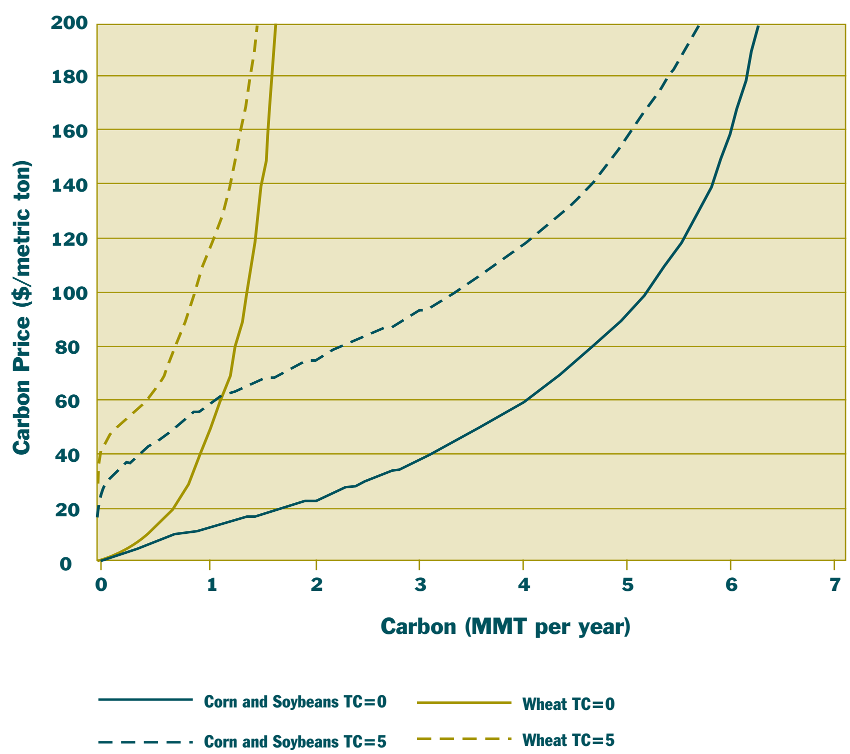
Source: Conservation Tillage Information Center.

Figure 6 summarizes the impact of the above factors on the economic potential of sequestering carbon for the two main cropping systems in the central United States. It shows that, in the central United States, at each price of carbon the corn–soybean system would sequester more carbon than the wheat-based system. The shapes of the supply curves in Figure 6 are determined by the factors mentioned above: sequestration rates, amount of land in relevant crops, and crop-specific production costs and profitability.

Antle et al. (2002a) have shown that carbon supply curves for soil carbon sequestration in the grain-producing region of Montana differ substantially from one sub-region to another. Similarly, Plantinga et al. (1999) and Stavins (1999) show differences in the afforestation carbon supply curves for different regions of the United States. By considering both soil and afforestation opportunities, McCarl and Schneider (2001) find that at moderate carbon prices (less than $50 per tonne), soil sequestration would occur predominately in the corn belt, plains, and mountain states, whereas afforestation would be undertaken predominantly in the Delta states. Other regions would become important at relatively high carbon prices.

Figure 6

Soil Carbon Supply Curves for Major Central U.S. Crop Systems



NOTE: The supply curves shown are based on adoption of conservation tillage in corn and soybean systems and adoption of conservation tillage and fallow reduction in wheat systems, with zero and $5 per hectare transaction costs. TC = transaction cost.

Source: Antle et al. 2005.

2. Payment Options

Practice-based (per hectare) versus per tonne payments. Incentives for farmers to sequester carbon can be provided in two forms: payments for each hectare on which they change practices (called per-hectare or practice-based payments) or payments based on the quantity of carbon sequestered (per-tonne payments).[19] Most of the studies in Appendix A utilized a per-tonne payment in their simulations. The studies by Antle et al. (2003) and Pautsch et al. (2001) show that per-hectare contracts could be up to five times more costly per tonne of carbon sequestered than per-tonne contracts in a given eco-region. This result confirms a well-established economic principle that the most efficient outcome will be achieved when the incentive is targeted directly at the desired outcome. Thus, if the objective is to maximize the amount of carbon sequestered, providing incentives for the quantity of carbon is more efficient than providing incentives for certain practices. This efficiency improvement is particularly important in the case of carbon sequestration because of the substantial differences in carbon sequestration rates and costs of changing practices in different locations. These findings also imply that government programs that provide subsidies for government-specified "best management practices" are likely to result in relatively high costs per tonne of carbon sequestered, compared with programs that pay farmers per tonne of carbon. However, implementation of such results-based approaches will require resolution of a number of issues, including development of cost-effective methods to measure carbon gains. Further discussion of these issues can be found in the companion paper, Richards et al. 2006, Chapter IV, Section E.

Paying all farmers using a practice versus only new adopters. In their study of carbon sequestration resulting from adoption of conservation tillage in Iowa, Pautsch et al. (2001) found that paying all adopters, as currently done in the Conservation Security Program, would roughly double the costs of payments compared with payments only for new adopters. However, many farmers argue that paying only new adopters punishes "early adopters", who already practice good environmental stewardship, and that such a restriction would be unfair. The companion paper (Richards et al. 2006, Chapter IV, Section D) provides further discussion of this issue.

Payment for land-use change versus management-practice changes. Placing land in permanent grass or trees generally sequesters more carbon per hectare than changes in cropland management. However, currently the cost to farmers of switching from marketable crops to permanent grass or trees is high in terms of forgone crop income. Consequently, all of the studies to date that have simulated the conversion of cropland to permanent grass cover have found this approach to be much more costly per tonne of carbon than changing management practices. For example, Antle et al. (2001) found that use of per-hectare payments to induce Montana farmers to change from wheat to permanent grass would sequester relatively small quantities of carbon at a high cost (in the range of $50 to $300 per tonne of carbon), whereas farmers would sequester more carbon at a lower cost (in the range of $10 to $65 per tonne of carbon) by reducing fallow while continuing to grow wheat. In their national study, Lewandrowski et al. (2004) found that conversion of cropland to grassland would not be competitive with other sequestration options for carbon at prices in the range of $10 to $125 per tonne of carbon. Where farmers can derive income from sustainable harvesting of permanent grasses—e.g., for use as animal feed or biofuels—a lower carbon price is needed to induce cropland-to-grassland conversion.

In contrast to conversion of cropland to grassland, afforestation appears to be economically competitive with crop management changes. This is because afforestation typically provides carbon accumulation rates two to four times higher than permanent grass and five to ten times higher than changes in crop-management practices. Therefore, even though the opportunity cost of planting trees is high, the per hectare carbon revenues render afforestation competitive with changes in crop-management practices.

Length of payment period. To the extent that practices that store carbon are less profitable than practices that release carbon, farmers will very likely need incentives such as a carbon payment for the full duration of the time that the carbon sequestering and storing practices are to be maintained. Antle and McCarl (2002) observe that if society would like soil carbon to remain sequestered for 50 years, the cost will be about double the cost of paying for 20 years of soil carbon increases (assuming a 5 percent interest rate). However, carbon payments offered only for several years could induce some farmers to permanently adopt carbon-sequestering practices, particularly if the new practices turn out to be more profitable than previous practices. Farmers may not have made these ultimately more profitable practice changes without incentives due to uncertainty about their future productivity or environmental benefits (Antle and Diagana 2003). Farmers also might decide to permanently adopt a practice if they realize that other valuable environmental benefits are being produced and that they could be rewarded for them, e.g., by being paid for hunting rights (see Box 6). Even if farmers were not being financially rewarded for these additional environmental benefits, they might decide to maintain the practice in order to preserve these co-benefits for personal reasons or because of societal recognition for their environmental stewardship.

Box 6. Value of Co-benefits

Agricultural practices that sequester carbon provide other environmental benefits, such as erosion control, improved water quality, and wildlife habitat, in addition to GHG mitigation (Lal et al. 1998; Ribaudo et al. 1989). However, it is difficult to assign monetary values to such benefits. Plantinga and Wu (2003) estimated that the value of the environmental benefits of reduced soil erosion and increased wildlife habitat from a program to convert 25 percent of cropland to forest in Wisconsin would be about $100 million, compared to program costs of about $100 to $130 million. Hence, the value of these other environmental benefits alone could justify the afforestation, in effect reducing the net cost of using afforestation as a carbon sink. They were not able to value the additional benefits of improved ground water quality or biodiversity. McCarl and Schneider (2001) also found that carbon sequestration practices would generate benefits in the form of reduced levels of erosion and reduced water pollution by agricultural chemicals, but they did not attempt to attach a dollar value to these benefits. Feng and Kling (2005) show in a study of the upper Midwest region that payments that included other environmental services would result in substantially different patterns of adoption of conservation practices than would result from payments for carbon alone.

3. Model Differences

Study results differ in part because they focus on different practices and/or different regions and in part because they are based on different models and data. In particular, some of the studies listed in Appendix A use farm-level data, some use county-level data, and others use national-level models. For example, the Antle et al. 2001 and 2003, and Kurkalova et al. 2003 studies are based on field-level data and models, while the Antle et al. 2005 study is based on county-level data. An important source of different results between the Antle 2005 and Kurkalova et al. 2003 studies is the different estimated carbon sequestration rates used. Kurkalova et al. (2003), using field-level data, report an average carbon sequestration rate for adoption of conservation tillage in Iowa that is about double the rate estimated by Antle et al. (2005) in their cropland model based on county-level data covering the entire central United States. This is a major factor in the much lower marginal costs shown by Kurkalova et al. as compared to Antle. A sensitivity analysis of the carbon supply curves for Montana found that changes in carbon sequestration rates could significantly alter the estimated economic potential for sequestration (Antle et al. 2002b).

The Antle and Kurkalova studies take crop prices as givens and thus do not account for possible changes in commodity prices in response to changes in land use and management induced by carbon payments. In principle, these studies embed all of the factors that determine land use and management practices selected by farmers because they are based on empirical models that rely on observations of actual farmer behavior. In contrast to these studies, the McCarl and Schneider (2001) and Lewandrowski et al. (2004) studies use national-level models that represent the principal production systems in different regions of the United States. These models solve for the land-use and management practices that maximize economic welfare of consumers and producers subject to various constraints (e.g., the total amount of land and other resources available). These models are not directly linked to observed behavior of farmers; they are based on data aggregated to the regional level and, because they represent market supply and demand, they do allow prices to vary in response to land-use and management changes. While the price effects of changes in agricultural practices are likely to be negligible, switching large amounts of land from crop production to trees or permanent grasses could potentially affect agricultural prices. However, both of these national-level studies indicate that for carbon prices below $100 per tonne of carbon, changes in agricultural prices are relatively small.[20]

B. Factors Other than Opportunity Costs

Researchers have documented that factors in addition to opportunity costs are likely to influence farmers' willingness to participate in conservation programs or contracts. The results reviewed in Section A above are all based on the principle of opportunity cost—they assume farmers will adopt the most profitable practices and land uses. The following discussion covers factors other than opportunity cost that may affect farmer decisions to adopt new practices. Such factors include risk and uncertainties; program implementation costs; farmer reluctance to substitute tree growing for crop production; access to financing; and socio-economic factors such as farmer age, education, experience, farm size, and ownership status.

1. Risk and Uncertainty

Farm managers face many risks (defined generally as random events with known probabilities) and uncertainties (defined as random events with unknown probabilities). Production and price risks and uncertainties are considered to be key factors influencing farmers' land use and management decisions (Just and Pope 2002). Studies based on field and county-level data implicitly capture the effects of risk and uncertainty on farmers' behavior, whereas the national models generally do not. Nevertheless, none of the models used thus far for carbon sequestration analysis provides an explicit representation of risk. Consequently, they cannot be used to investigate how changes in risk or risk attitudes would affect carbon sequestration potential. Studies of technology adoption, however, show that farmers' perceptions of risk can lower adoption rates below the levels predicted by opportunity cost analyses (Sunding and Zilberman 2001). Therefore, the studies reviewed above and summarized in Appendix A may overstate the degree to which farmers would be willing to adopt carbon-sequestering practices, thereby understating the costs.

Because agricultural carbon sequestration in response to changes in land use and management occurs slowly over many years, contracts for soil carbon or agroforestry are likely to be long-term. Entering into long-term contracts poses a risk that economic or technological conditions will change, thereby potentially imposing unforeseen costs on adopters in the form of forgone opportunities. These risks may be substantial considering the relative instability of agricultural prices, the high rates of innovation in agriculture, and the degree of uncertainty about various government programs and regulations affecting agricultural producers. In addition, there is the risk that anticipated rates of carbon sequestration specified in a contract might not be realized—for example, due to unanticipated climate shocks or errors in estimating soil properties—thus potentially exposing the farmer to penalties for contract default. In short, the rational farmer will factor risk into the decision to participate in a carbon contract, but quantifying and valuing these risks and their impact on sequestration supply curves is difficult.

Some evidence on the importance of these risk considerations is provided by the study of van Kooten et al. (2002). These authors conducted a survey of agricultural landowners to ascertain their willingness to adopt practices that would sequester carbon, including adoption of reduced tillage, reducing the amount of fallow in dryland cropping systems, and planting trees in plantations. They found that 50 to 60 percent of respondents indicated a willingness to consider changes in agricultural management, but only about 20 percent were willing to consider tree plantations, even if adequately compensated. They also found that most respondents preferred contracts less than 20 years in length. These findings are based on responses to hypothetical questions and therefore should be interpreted with caution. However, the findings do suggest that farmers may be less willing to change practices than opportunity cost calculations would indicate, particularly in the case of substituting trees for crops.

2. Program Implementation Costs

Policies designed to increase agricultural GHG mitigation will involve costs for establishing contracts with farmers. These costs might be borne by taxpayers if carbon sequestration is incorporated into existing USDA programs. However, if carbon sequestration were undertaken in conjunction with a private-market carbon-trading program, such costs would be borne by buyers of the contracts and by farmers. If carbon sequestration is incorporated into existing USDA programs, it should be possible to implement carbon sequestration programs without adding significantly to program administration costs,[21] as long as those programs do not involve significant costs to monitor practices or measure actual changes in carbon. See Richards et al. 2006 for further discussion of costs of federal programs.

Carbon contracts might also be carried out as part of a private-market trading program that establishes a demand for agricultural or forest-sector sequestration and GHG-emission reductions. Contract costs would then be borne by project participants (McCarl 2002; van Kooten et al. 2002; Mooney et al. 2004). Costs associated with such contracts, known as transaction costs, include normal financial transaction costs (legal and broker fees, etc.), as well as costs associated with verifying contract compliance. Very few reliable data are available to estimate transaction costs for agricultural emission credits within a private-market trading program. Based on experience with two pilot programs, McCarl (2002) estimated transaction costs (costs of organizing and participating in the market) for emission credits could be in the range of $0.83 per acre, or about $2.00 per hectare which, at an average carbon sequestration rate of 0.3 metric tonnes per hectare, implies a transaction cost of over $6 per tonne. However, it is difficult to generalize this estimate to other cases. Moreover, the costs of implementing contracts are likely to decrease with experience and with competition among service providers.

Mooney et al. (2004) estimated the measurement and monitoring costs that are likely to be required to verify the amount of agricultural-soil carbon sequestration achieved and therefore tradable. In a case study of their prototype measurement scheme, the upper estimate of measurement costs was 3 percent of the value of a carbon credit. Although these estimates should be interpreted with caution—because they were not based on actual contract implementation—they suggest that measurement costs are not likely to be large enough to prevent farmers from participating in a market for tradable emission credits.

The Antle et al. (2005) study is the only model-based study that has incorporated transaction costs. Results from this analysis are presented in Figure 6 (page 30), where carbon supply curves are shown for zero and $5 per hectare transaction costs. Transaction costs have the effect of creating a threshold price—a price below which no carbon would be offered by economically rational farmers. When carbon sequestration rates are relatively low, as is the case with the dryland wheat system in the Great Plains, transaction costs create a relatively high price threshold (equal to about $30 per metric tonne of carbon). For the relatively higher carbon rates associated with the corn, soy, and hay system in the Midwest, the threshold is about $20 per tonne. Figure 6 also shows that the effect of the transaction costs diminishes at higher carbon prices. Thus, it can be concluded that transaction costs are likely to be particularly important when carbon prices are low and in regions where carbon storage rates are low.

3. Access to Financing and Other Socio-economic Factors Affecting Technology Adoption

While financial constraints are not likely to impede adoption of new practices for large-scale, commercial farms, they may be a significant factor for smaller-scale farms. Some sequestration practices involve substantial up-front investment costs, and farmers must either self-finance the investment or have access to credit. Access to credit has been a long-standing problem for farmers. Although farmers in the United States have access to financing through both private banks and the federally subsidized farm-credit system (Barry 2002), at the present time participation in environmental programs is not considered in making lending decisions. However, with suitable changes in lending rules, a farmer could be allowed to use a GHG emissions-reduction contract as collateral on a loan, for example, for new equipment needed to implement conservation tillage. Alternatively, loans could be based on the present value of future carbon credits and be repayable with those credits (Antle and Diagana 2003).

A number of empirical studies have examined socio-economic factors affecting adoption of conservation tillage practices, as summarized by Uri (1999). This literature shows that, in addition to economic returns, characteristics such as age, education, experience, and farm size affect farmer decisions, although how they do so appears to differ across regions (Gould et al. 1989; Luzar and Diagne 1999; Soule et al. 2000). Using a sample of U.S. farms, Fuglie and Kascak (2001) found that diffusion of conservation tillage technologies has been relatively slow, with long lags in adoption due to differences in land quality, farm size, farmer education, and regional factors. Converting to conservation tillage may involve significant investments in knowledge and other capital, and farmers who adopted conservation tillage tended to have more college education and above-average farming experience. They also managed relatively larger farms, were younger, and were more likely to participate in subsidy programs. One explanation for the effects of age and farm size is that the profitability of the investment is likely to be higher for farmers managing large amounts of land and for younger farmers who have a longer time period to depreciate needed equipment.

C. Summary and Policy Implications

Recent model-based studies of the economic potential for soil and forestry sequestration show that agriculture can sequester carbon at a cost competitive with other methods of reducing GHG emissions. For carbon prices less than \$50 per tonne carbon (\$13 per tonne CO_2), the quantities sequestered are likely to be substantially less than the technically feasible amount. At the national level, estimates of economic potential for soil carbon sequestration from changes in agricultural practices range from several million metric tons per year at carbon prices of \$10 per tonne to 10 to 70 MMT per year at carbon prices around \$50 per tonne. Regional studies of both soil sequestration and afforestation show that the economic potential for carbon sequestration varies substantially across the United States, due to differences in technical potential and costs of sequestration. Most of the economic studies conducted to date focus on carbon sequestration through afforestation and changes in crop management and have not incorporated the economic and global warming effects of reducing the other two important GHGs—nitrous oxide and methane—due to data limitations.

Changes in crop management would sequester modest amounts of carbon but could be implemented rapidly and would provide emissions offsets for 20 to 30 years. Participation in GHG mitigation could provide income to farmers and encourage adoption of conservation practices that also bring other environmental benefits in the form of improved air and water quality, preservation of open space, and increased wildlife habitat. However, farms are businesses and they will generally only adopt carbon-sequestering practices if payments are available that are at least as great as any income reductions resulting from management-practice or land-use changes.

A key challenge in implementing soil carbon sequestration will be to develop cost-effective methods to organize contracts with farmers and to verify compliance with contracts. National studies of afforestation show that the potential quantities of sequestered carbon are much larger than would be achieved by changes in agricultural practices, and at a price of $50 per tonne carbon ($13 per tonne CO_2) are estimated to be in the range of 50 to 270 MMT per year. Key challenges for afforestation are potentially significant impacts on crop prices; that afforestation in some areas would be partially or completely counterbalanced by conversion of forests to croplands elsewhere (referred to as leakage); and the willingness of farmers to convert millions of acres of cropland and grazing land to permanent tree plantations.

IV. Bioenergy from Agricultural Lands

Bioenergy derived from plants is a growing, though still relatively small, part of the U.S. energy supply. If aggressive research and development programs succeed in substantially increasing yields of bioenergy crops and in reducing costs of biomass-to-fuel technologies, biomass from agricultural sources could supply as much as 19 percent of the total current U.S. demand for energy. This total would include energy from dedicated energy crops and biomass from conservation reserve program (see Box 2, page 11) lands (most likely perennial grasses), dual-purpose crops (e.g., corn and soybeans), agricultural residues, and animal wastes. This amount of bioenergy represents 82 percent of year-2004 petroleum energy used for on-road transportation in the United States.

Chapters II and III discussed strategies for GHG mitigation through abatement of agricultural emissions and by sequestering carbon in soils. This chapter explores another major opportunity for agriculture to contribute to GHG mitigation: the reduction of CO_2 emissions by using biomass to substitute for fossil fuels. Many promising technologies for converting biomass into energy already exist at some stage of development today, including:

- Direct combustion of biomass for production of heat and power;
- Gasification of biomass to produce a synthetic gas, often referred to as "syngas," that can be combusted in high-efficiency combined heat and power systems;
- Biological conversion of animal waste to methane as a source of heat, power, and fuels;
- Biological release and conversion of sugars in biomass to produce ethanol;
- Thermochemical conversion of biomass-derived syngas to produce transportation fuels; and
- Chemical conversion of natural oils to fuels, including biodiesel.

Energy from biomass is not a "magic bullet" solution to our energy and climate change challenges. Biomass can, however, be an important element within a diversified energy supply portfolio while providing climate change mitigation, energy security, and economic benefits. In terms of energy, depending on the success of efforts to improve yields and biomass-to-fuel technologies, biomass could supply 13 to 19 percent of the total 2004 U.S. demand for energy, or 14 to 20 exajoules (EJ) per year.[22] Under these conditions, the avoided greenhouse gas emissions from the replacement of fossil energy with biomass energy represents a reduction in GHG emissions of 670 to 1,710 million metric tons (MMT) of CO_2 annually (9 to 24 percent of total U.S. GHG emissions in 2004), depending on the mix of fossil fuels displaced.[23]

Of equal importance is the role biomass can play in improving the energy security of the nation by dramatically reducing our dependence on petroleum as a source of liquid fuels for transportation and organic chemicals. Fourteen to 20 EJ per year of biomass energy supplies would represent as much as 57 to 82 percent of current U.S. petroleum energy used in on-road light- and heavy-duty vehicles. Finally, rural America has seen both economic and population declines, and the opportunity to turn American farms into a major source of energy and chemicals—in addition to providing food, feed, and fiber—could open new markets for U.S. farmers and revitalize the economy of rural America.

Fossil fuels offer many advantages over biofuels (fuels derived from biomass). Fossil fuels generally contain more energy per unit mass than does biomass. Also, fossil energy sources tend to be located in concentrated reservoirs, as opposed to biomass, which is more dispersed. Petroleum, as a high energy-density liquid, offers tremendous benefits in handling and processing. Indeed, the U.S. transportation sector now relies on petroleum for 97 percent of its energy supply (USDOE–EIA 2003).

Fossil energy does have drawbacks, however, especially with regard to climate change. When fossil fuels are burned for energy, organic carbon that has been stored in sedimentary rock for millions of years is released into the atmosphere. Bioenergy-derived CO_2, on the other hand, participates in a recycling process on the time scale of one growing season to the next. CO_2 released from biomass used for energy can be removed from the atmosphere and converted into new biomass via photosynthesis on an annual basis.

The yearly recycling of carbon between plant matter and the atmosphere that occurs when we use biomass means that, theoretically, biofuels are "carbon neutral;" that is, these fuels cause no net increase or decrease in atmospheric CO_2 levels. The reality is more complex. For example, the growing of biomass involves the use of fossil fuels to drive tractors and to make the fertilizers and other chemicals needed by the farmer. Fossil fuel is also used to transport biomass to an energy facility and in some conversion processes. In addition, depending on the specific crops, and harvesting and other management practices used to produce biomass, carbon may be sequestered or released from soil and trees. Thus the GHG reductions achieved on a life-cycle basis compared to use of fossil fuels varies. In short, to understand what role biomass can be expected to play in reducing GHG emissions, several issues must be considered. Section A of this Chapter addresses how much energy the United States can produce from biomass, considers factors that affect GHG emission reductions when biomass is used for energy, and reviews the costs of using biomass for energy. Section B reviews bioenergy's net (life cycle) GHG impacts.

A. Agricultural Bioenergy's Potential to Displace Fossil Energy

Substantial increases in yields of perennial grasses and improvements in conversion technology, including cost reductions, could be achieved through an aggressive research and development program. Under such a scenario, dedicated energy crops alone could supply 6 to 12 exajoules per year of primary energy (6 to 12 percent of the current U.S. energy demand), without significantly raising food prices. This biomass could be used to produce heat, power, and/or transportation fuels.

Currently the largest use of biomass for energy is cogeneration of steam and electricity by the forest products industry, while ethanol made from corn grain is the largest source of biomass-derived fuel in the transportation market. Emerging opportunities include the use of agricultural residues to produce energy and the production of dedicated energy crops.

Estimates of potentially available residues run from 110 MMT per year to 380 MMT per year, which would supply 1.7 to 5.8 EJ per year. The low estimate reflects current farm practices while the high estimate represents aggressive shifts in farming practices and improvements in crop and residue yield. In the longer term, crops could be grown specifically for use in energy production, with perennial grasses being the most likely candidates. If yields of such grasses can be doubled or quadrupled—from current averages of 10 tonnes per hectare to 20 to 40 tonnes per hectare—6 to 12 EJ per year of primary energy could be derived from energy crops using approximately 15 percent of U.S. agricultural lands.

The upper limit on cost-effective production of ethanol from corn starch is estimated to be around 10 billion gallons per year, or less than 4 percent of current (2004) U.S. transportation demand. This is in contrast to production of ethanol from agricultural residues and energy crops, which could potentially provide 57 to 84 billion gallons of transportation fuels annually. To make ethanol from these biomass sources cost-competitive, it will be necessary to cost-effectively utilize the complex sugars found in the cellulose and hemicellulose[24] of residues and perennial grasses. Biodiesel made from oilseed crops and recycled animal and vegetable fats is rapidly gaining popularity. Biodiesel from these U.S. sources could provide up to one billion gallons per year.

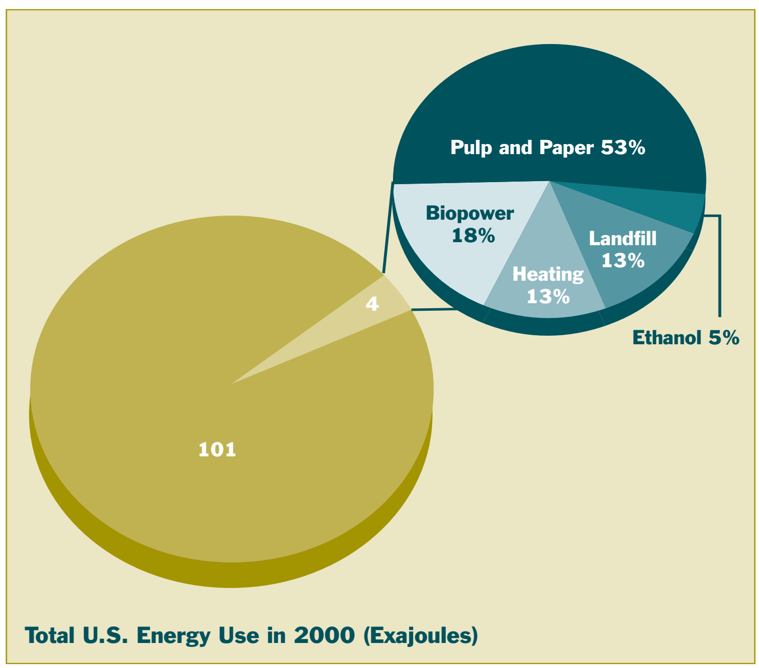
1. Current Situation

Bioenergy is already a growing, though relatively small, part of the U.S. energy supply. Of the total 101 EJ of energy consumed in the U.S. in 2000, 4.0 EJ were in the form of bioenergy generated from agriculture, forestry, and municipal solid waste (Figure 7). The pulp and paper industry used 2.1 EJ of bioenergy in 2000, representing 53 percent of the bioenergy consumed in the United States. The electric utility sector used 0.7 EJ of biomass in dedicated biomass-fired or coal/biomass co-fired power plants, 0.5 EJ of biomass was used for residential and commercial heating, and 0.5 EJ of methane-rich biogas from landfill operations was used for on-site heat and/or electricity production (Haq 2002; Chum and Overend 2001). Due to its focus on agricultural opportunities, this report does not further consider contributions of either the pulp and paper industry or municipal

Figure 7

Present Energy Supply from Biomass

in the United States



Sources: Clum and Overend. 2001.

landfills to U.S. biomass supply. These and other as-yet-unrealized biomass opportunities could result in a U.S. biomass supply exceeding 20 EJ.

Ethanol from cornstarch and sugarcane. The U.S. ethanol industry has grown from literally nothing in the late 1970s to a year-2003 production level of 2.8 billion gallons, contributing about 0.2 EJ, or approximately 2 percent of the total on-road transporation fuel used in 2003. Ethanol made from corn grain is the largest source of biomass-derived fuel in the U.S. transportation market. Grain ethanol is made by converting the starch in the corn kernel to sugar and then using yeast to ferment the sugars to ethanol. An upper limit on cost-effective ethanol production from grain is difficult to determine, although it is likely to be related to a maximum limit on sales of the co-product, distillers' dried grain, on which today's corn ethanol producers rely for profitability. USDA and U.S. Department of Energy (USDOE) researchers recently published a joint study of the potential magnitude of biomass supplies (Perlack 2005). They conservatively estimated the maximum economically feasible supply of ethanol from cornstarch at around 10 billion gallons per year, equivalent to 0.8 EJ of annual energy supply, three to four times current (2003) production.

The amount of fossil energy consumed in the production of the ethanol from cornstarch diminishes its effectiveness as a means of reducing fossil CO_2 emissions. The net energy balance of ethanol has long been a subject of controversy, with some experts claiming that it takes more energy to make ethanol than the fuel actually contains (Pimentel 1992); or that it is, at best, a break-even energy proposition (Chambers et al 1979). Certainly in the early days of the industry, this was true. But, over the course of the past 20 years, improvements in ethanol technology have turned ethanol into a net provider, rather than a net consumer, of energy. According to the latest estimates of the energy requirements for producing ethanol from corn grain, it takes 0.75 megajoules (MJ) (1 MJ = 1 x 10^6 joules) of fossil energy to produce one MJ of energy in ethanol (Shapouri et al. 2002a). For comparison, consider the life-cycle (or net) fossil energy balance for gasoline. Gasoline's net fossil energy balance includes two components: (1) the amount of fossil energy that goes into collecting and converting crude oil into gasoline (0.2 MJ), and (2) the amount of fossil energy contained in the crude itself (1.0 MJ). Corn ethanol's net fossil energy balance involves only one component, the 0.75 MJ for production and processing of the corn. Thus, considering only processing energy, it takes more fossil energy to produce one MJ of ethanol than one MJ of gasoline. However, if one considers the fossil energy used in production and consumption of the fuel, more fossil energy is consumed when using gasoline (1.2 MJ) than when using corn ethanol (0.75 MJ). The relatively high fossil energy demand in corn-grain ethanol production has several sources: fuel for farm machinery, significant use of nitrogen fertilizer, energy in drying operations, energy for transporting the grain, and energy in the distilling of the ethanol. Thus, while one MJ of ethanol directly saves only 0.25 MJ of fossil energy on a net energy basis, it avoids use of 0.45 MJ of fossil energy for every MJ of gasoline it displaces.

The cost of producing ethanol today is much lower than it was in the early days of the industry but is very sensitive to corn prices, which have varied from $1.94 to $3.24 per bushel in recent years. Using the midpoint corn price of around $2.60 per bushel, along with estimates of the operating costs for an ethanol plant producing 25 million gallons per year, ethanol is currently estimated to cost around $1.20 per gallon at the plant gate (McAloon et al. 2000).[26] This includes a credit for the sale of co-products worth $0.40 per gallon of ethanol and an annualized capital cost of $0.30 per gallon. Net operating costs[27] amount to $0.90 per gallon (McAloon et al. 2000).[28] This cost for ethanol is equivalent to an energy cost of $15 per gigajoule (GJ) (1 gigajoule = 1 x 10^9 joules) of fuel energy.[29] Annual average wholesale gasoline prices in the United States have risen steadily over the past 10 years and in the first 4 months of 2005 jumped to an unprecedented level of $13.17 per GJ ($1.59 per gallon) due to sharply increased prices for crude oil. The combined trends of increased gasoline prices and lower production costs for ethanol have significantly reduced, but not eliminated, the gap between corn ethanol and gasoline prices. USDOE projects wholesale gasoline prices of $11 per GJ in 2015, rising to around $13 per GJ by 2030 (USDOE–EIA 2006). This brings unsubsidized corn ethanol prices within range of, but not quite competitive with, gasoline as a bulk fuel in the long term.

In the United States, ethanol competes in the fuel market as an additive in gasoline at levels of up to 10 percent by volume. The current federal tax incentive of roughly $0.51 per gallon of ethanol reduces the plant gate cost of ethanol to around $8.34 per GJ.[30] At this price, ethanol looks attractive as a gasoline substitute both today and in the future, but only as long as the tax incentive remains in place. Ethanol has other competitive advantages over gasoline that raise its market value, including its ability to act as a fuel oxygenate and an octane booster.[31]

Ethanol for fuel is also made from sugarcane. The energy balance for sugarcane-derived ethanol is much better than the energy balance for corn ethanol. In Brazil only 0.1 MJ of fossil energy is used for each MJ of fuel ethanol produced (Macedo 1998). A major reason for cane-derived ethanol's significantly better energy balance—as compared to corn-starch-derived ethanol—is that bagasse (biomass that remains after the sugar has been extracted from the cane) rather than a fossil fuel is used as the energy source in the conversion process. A recent study by the International Energy Agency (IEA 2004) reported production costs of ethanol in Brazil of $1.07 per gallon ($13 per GJ) in 1990 and recent Brazilian ethanol prices as low as $0.57 per gallon ($7 per GJ), roughly half of U.S. ethanol plant gate prices. These lower production costs result largely from lower sugar-cane production costs. Due to the artificially high price for sugar in the U.S. market,[32] sugarcane production in the United States is highly profitable despite production costs that are four times higher than in Brazil (Shapouri 2002b). In the United States, sugarcane is produced in Florida, Louisiana, Hawaii, and Texas. Use of the roughly 2 MMT of sugarcane produced in the United States today (USDA–ERS 2005) would produce around 150 million gallons per year,[33] less than 5 percent of current U.S. ethanol production from corn. Imports of Brazilian ethanol are currently constrained by U.S. trade tariffs that add a 2.5 percent added-value tax and a secondary duty of $0.54 per gallon to the cost of fuel (RFA 2005).[34]

Biodiesel. Biodiesel today is where corn ethanol was more than two decades ago. In 2000 biodiesel production in the United States was negligible. By 2006, biodiesel production capacity in the United States reached almost 400 million gallons (NBB 2006). Biodiesel is a diesel fuel substitute made by chemically combining natural oils with methanol, which is currently made from natural gas. In the United States, the bulk of the biodiesel sold is made from soybean oil, with some production using waste greases and fats. Interest in biodiesel in the United States has increased dramatically in the past two years, mostly due to tax incentives in the form of commodity credit payments offered by USDA to soybean producers whose crops go to biodiesel production and federal tax breaks offered to biodiesel producers.

Like corn ethanol, biodiesel requires some fossil energy inputs; however, only 0.31 MJ of fossil energy is needed to produce one MJ of biodiesel fuel energy (Sheehan et al. 1998). There are several reasons for this. First, soybean farming requires no (or very little) nitrogen fertilizer; thus, unlike corn, it is not burdened with the energy inputs of producing and applying nitrogen fertilizer. In addition, the processing of soybeans and the conversion of soybean oil to biodiesel are not very energy-intensive.

As with ethanol made from corn, biodiesel is likely to enjoy significant market growth in the fuel additive and fuel-blend markets. Biodiesel can be used as an additive to improve the lubricity of diesel fuel,[35] and it is also a fuel oxygenate capable of reducing particulate matter and other regulated emissions. However, the overall impact of biodiesel from U.S. soybean supplies on the U.S. transportation fuel market is likely to be small in the long run because of the competition for soybean oil in the food market (NRC 1999). Waste fats and oils from restaurants and food processing could add almost one billion gallons of biodiesel capacity (0.1 EJ per year) in the United States, equivalent to 2 percent of the 6 EJ of petroleum currently used to meet on-road U.S. diesel demand.

Current plant gate costs for biodiesel range from $1.60 to $2.00 per gallon, with the low end representative of waste grease feedstocks and the high end representative of soybean oil. This corresponds to an energy cost of $13 to $16 per GJ. Diesel fuel in 2004 had a wholesale cost of around $1.19 per gallon (USDOE-EIA 2005a), corresponding to an energy cost of around $8.50 per GJ.[36] Thus, as an energy carrier, biodiesel is currently 50 to 90 percent more costly than diesel fuel.

Methane from animal wastes. Biological conversion of manure to methane is another current, viable, opportunity for energy production. However, even if all of the manure from all beef cattle, dairy cattle, poultry, and swine operations in the United States could be collected and converted to methane, the authors estimate that the amount of energy produced would be around 0.5 EJ per year or 0.5 percent of current U.S. energy consumption.[37] Moreover, it is important to recognize that not all manure can be practically collected for energy production. Nonetheless, opportunities for capturing and using animal wastes play an important role in near-term strategies for the development of agricultural bioenergy and have important co-benefits, including mitigating methane and nitrous oxide emissions from stored manure and addressing air- and water-quality issues associated with large-scale confined animal feed operations (CAFOs). (See Chapter II of this report.)

2. Emerging Bioenergy Opportunities—Agricultural Residues, Biomass from Conservation Reserve Program (CRP) Lands, and Biomass from Dedicated Energy Crops

Taken together, residues from existing crops, biomass from Conservation Reserve Program (CRP) lands, and biomass from dedicated energy crops, grown on 10 percent of U.S. prime agricultural land could, at present yields, supply around 5 EJ (5 percent) of current annual energy consumption in the United States. If 15 percent of U.S. cropland were dedicated to energy crops and a two- to four-fold increase in yields can be achieved, U.S. farmers could harvest another 6 to 12 EJ per year of primary energy (or 6 to 12 percent of the current U.S. energy demand) from energy crops alone. However, except for direct combustion, the technologies needed to use agricultural residues and energy crops such as perennial grasses for energy are not yet commercial in the United States.[38] Consequently, at present, use of these biomass sources is primarily limited to small demonstration projects. If current projects prove economically and technologically viable, increased use of residues, dedicated energy crops, and CRP-land biomass both in power plants and for transportation fuels can be expected. Significant use of agricultural residues, biomass from CRP lands (i.e., perennial grasses), and perennial grasses grown on prime agricultural cropland (i.e., energy crops) for transportation fuels will require cost reductions in conversion technologies (see Box 7).

Residues. Agricultural residues are the parts of the plants that farmers leave behind in the field after harvesting their crops. The use of agricultural residues for energy does not require changes in land use, and residues are a relatively low-cost feedstock compared with corn for ethanol or soybeans for biodiesel, making them an attractive mid-term option in strategies for bioenergy technology deployment. However, due to their multiple functions, residues are not "there for the taking." Residues contribute to the maintenance of soil organic matter, add nutrients, and protect soil from wind and rainfall erosion (Larson 1979). In 2000, Oak Ridge National Laboratory reported that, on average, 30 to 40 percent of the total corn and wheat residue produced in the United States could be collected, totaling approximately 150 MMT of residue per year in the United States, equivalent to 2.2 EJ of bioenergy per year[39] (Walsh et al. 2000). The recently published joint USDA–USDOE (Perlack et al. 2005) study of the potential for biomass as an energy supply

Box 7. Emerging Technologies for Converting Residues and Energy Crops to Ethanol

The key to use of biomass as a large and affordable source of energy is technology that can efficiently and cost-effectively transform the cellulose, hemicellulose and lignin in biomass into readily useable forms of energy, particularly liquid fuels for transportation. These three forms of biomass represent the single largest pool of renewable carbon in the biosphere. Two major technological pathways exist: (1) biological processing, and (2) thermochemical processing.

Biological routes involve the use of enzymes and microbes to release and ferment the sugars contained in cellulose and hemicellulose. Over two decades of research have dramatically reduced the cost of the enzymes needed to break down cellulose into simple sugars. DOE and its industrial partners have reduced the cost of these enzymes more than twenty-fold over the last four years. Meanwhile, researchers around the world have applied genetic engineering tools to develop microbes that can readily ferment the unusual mix of sugars found in cellulose and hemicellulose into ethanol.

Thermochemical routes involve applying heat to break apart biomass into chemical intermediates that can be used to make liquid fuel substitutes. Many of these thermal technologies have been around for over a century, used primarily in transforming coal (ancient biomass) into liquid fuels.

The key to cost-effective transformation of biomass is a smart combination of both biological and thermochemical technology. In such combined systems, it is possible to use biological systems that are best suited for handling sugars, while converting the non-sugar components of biomass with thermal processes. This can lead to maximum efficiency of the conversion of all of the components of biomass into useful forms of energy.

shows collectible residue ranging from 110 to 380 MMT per year (1.7 to 5.8 EJ per year). The low end of the range reflects no change from the status quo. The high end of the range represents aggressive shifts to sustainable farming practices and aggressive improvements in crop and residue yield. Such estimates of residue supply are based on the difference between the amount of residue that must be left on the field to meet erosion protection requirements and the total amount of residue produced.[40]

Ultimately the amount of residue that becomes available will depend on farmers. Farmers' choice of crop rotation and tilling practices is the single greatest determinant in how much residue can be taken without negative impacts on erosion and soil carbon. For example, the combination of no-till practices and continuous production of corn (rather than rotation of corn and soybean) is the scenario under which farmers in Iowa could collect the most residue (Sheehan et al. 2002 and 2004). This study demonstrated that there are opportunities to collect significant amounts of corn stover[41] without causing a net loss in soil carbon or excessive soil erosion, as long as farmers adopt no-till practices. However, particularly prior to widespread collection, more work is needed to understand the impacts of using agricultural residues as an energy source, in particular how they can be collected and used sustainably and economically.

Biomass from CRP Lands. Land currently held in the CRP offers another opportunity for mid-term biomass supplies. Farmers are currently paid to keep CRP lands out of traditional agricultural production because of the land's environmental sensitivity and often high erosion rates. In many cases, native grasses or trees are grown on this land. Sustainable harvesting of this biomass, in conjunction with some level of continued government payment for keeping the land out of traditional row crop production, could be a cost-effective source of biomass for energy production. A recently issued USDA–USDOE study (Perlack et al. 2005) looked at this possibility and concluded that around 25 MMT of biomass could be sustainably harvested from CRP land, equivalent to around 0.46 EJ per year of energy (0.44 percent of current energy demand).

Energy crops. Energy crops—plants grown for the purpose of producing energy—represent a paradigm shift from using land to provide for food needs to using it to provide for energy needs as well. Energy crops under development by researchers in the United States and around the world include short-rotation woody crops (fast-growing trees) and herbaceous crops (native grasses) such as switchgrass. Switchgrass is representative of a family of grasses that can be grown for energy production, with the specific choice of species dependent on regional conditions. Because of the high value of wood fiber, short-rotation woody crops in the United States are more likely to be harvested and sold to the pulp and paper industry than to be used as a source of energy. For that reason, the emphasis of energy crop research in the past few years has been on perennial grasses such as switchgrass. Yields for switchgrass across regions of the United States where data are available average around 10 tonnes per hectare per year (t/ha/yr), equivalent to a primary energy yield of around 200 GJ/ha/yr (Turhollow 1994; Wright 1994; McLaughlin et al. 2002; Walsh et al. 2003). These yields are significantly higher than the 2 to 3 t/ha/yr yields associated with residue collection. In addition to its high yields, native grasses such as switchgrass offer other important benefits in terms of building and maintaining soil organic matter, improving soil structure, and reducing erosion. (Chapter II, this report; McLaughlin et al. 2002).

One major problem in projecting the potential of switchgrass as a source of energy lies in evaluating the competition for valuable land resources dedicated to food production. Researchers at Oak Ridge National Laboratory have addressed this question by using agro-economic models that allow switchgrass to compete with food crops for prime agricultural and CRP land. At a farm gate price of $2.44 per GJ of primary energy in the switchgrass (around $44 per tonne of switchgrass), at current yields, energy crops take over only 10 percent of agricultural land (17 million ha), producing a total of around 3 EJ per year in primary energy. Traditional crop prices in this scenario rise by only about 9 percent (Walsh et al. 2003).[42]

To put the farm gate price of $2.44 per GJ in an energy context, consider that mine-mouth coal, oil, and at-the-source natural gas prices in 2004 averaged $0.93, $5.69, and $5.00 per GJ, respectively (USDOE-EIA 2006; USDOE-EIA 2005a). While this comparison makes biomass look reasonably competitive, it is important to keep in mind that the only direct comparison is between the prices of biomass and coal, because both biomass and coal can be burned directly for heat and power using the same processing equipment.[43] Technologies to convert biomass to transportation fuels are less mature than their fossil fuel counterparts. Consequently these technologies often have higher capital and operating costs than their fossil fuel counterparts, so the price disparity between energy in switchgrass and oil and gas does not provide an advantage as great as the above numbers may suggest.

Two additional pieces of information are needed to assess the full potential for primary energy production on agricultural lands: (1) the yields that could be possible in the future, and (2) the amount of land that the United States can afford to dedicate to energy production. A number of researchers suggest that a doubling of current yields from 10 to 20 t/ha/yr (400 GJ/ha/yr) is achievable through genetic manipulation of grasses (Turhollow 1994; Wright 1994; McLaughlin et al. 2002). A recent publication from the USDOE's Genomes-to-Life Program estimated that such an improvement, achievable within 25 years, would increase the production of primary energy from 3 EJ on 17 million hectares to around 8 EJ on 20 million hectares of agricultural land (around 12 percent of current agricultural land) with "little impact on food production" (USDOE 2002).

Experience with improvements in corn suggests that yield improvements could be even greater. Corn yield in Iowa, for example, has increased four-fold since the first introduction of industrial fertilizers and three-fold since the introduction of the first hybrid breeding programs. It is possible, therefore, that yields could reach levels of 40 t/ha/yr (800 GJ/ha/yr). Assuming an upper limit of 15 percent on the amount of U.S. cropland that can be dedicated to switchgrass production,[44] and a 2 to 4-fold increase in productivity, U.S. farmers could harvest 6 to 12 EJ per year of primary energy (or 6 to 12 percent of the current U.S. energy demand) in the form of switchgrass and other similar energy crops.

Meeting these aggressive targets would require a major scientific effort to genetically modify grasses, and no such effort is currently underway. Furthermore, even if the financial commitment to such a research program were made, there is no guarantee of success. Introduction of genetically modified crops also entails risks, including potential political resistance. Finally, use of residues and grasses from CRP and croplands will require infrastructure for delivering the biomass to energy facilities. At present there is no efficient infrastructure to serve this function. The authors estimate, for example, that if biomass is baled and trucked, a 2,000 tonne-per-day biomass facility would have to receive 24 trucks per hour—one every two-and-a-half minutes—for 12 hours daily. The inconvenience and impact of this kind of truck traffic is not insignificant, but the overall energy consumption and GHG emission impacts associated with this transport is not very significant compared to the fossil-fuel energy and GHG emission savings that can be achieved. The current emphasis on large (e.g., 2,000 tonnes per day or greater) facilities is driven by economies of scale. However, the low bulk density and dispersed nature of biomass may argue against such large centralized operations in favor of smaller, more widely distributed facilities. In power generation, for example, it may make more sense to build small, distributed facilities in remote rural areas where electricity transmission costs combined with local availability of biomass may give local power production a cost advantage.

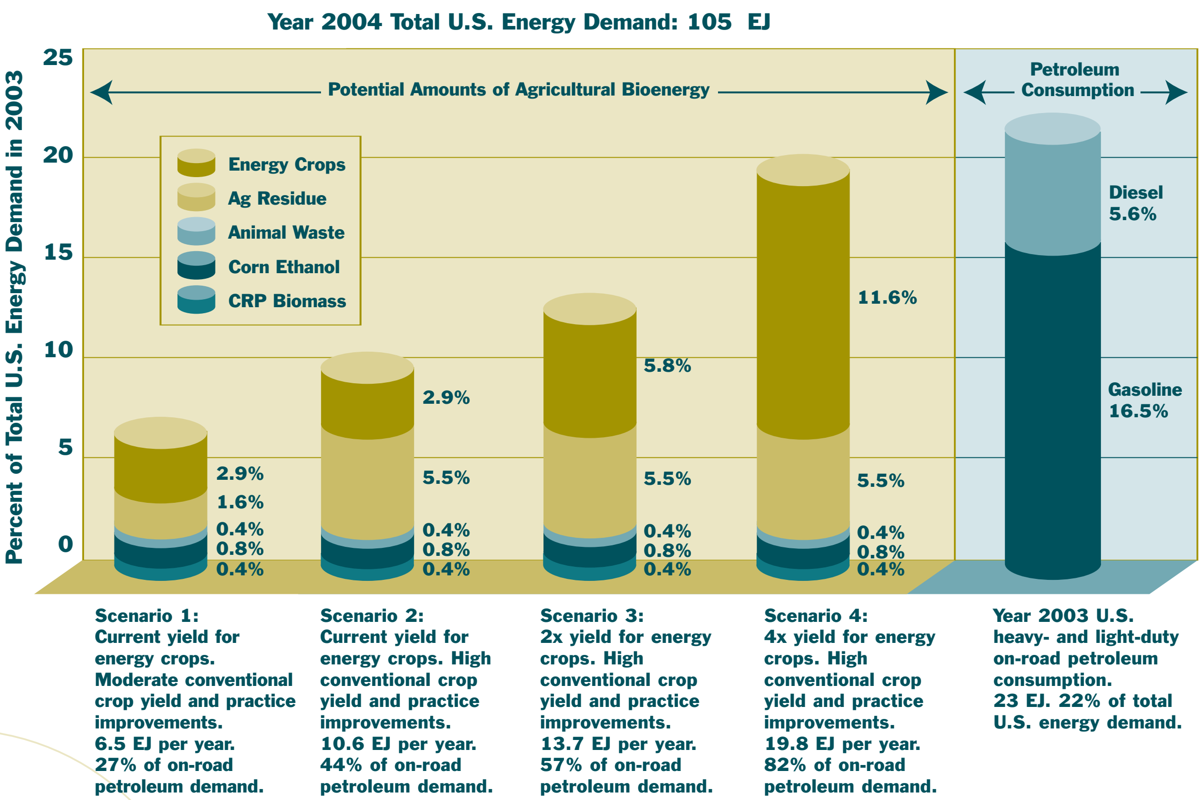
3. The Primary Energy Production Capacity and Cost Competitiveness of Biomass

The potential primary energy production from agricultural sources of biomass in the United States includes animal wastes, agricultural crops used for energy, agricultural residues, and energy crops from CRP and agricultural lands. Figure 8 shows the increasing potential of biomass to meet U.S. energy demand and transportation needs as energy crop yields increase and crop practices that favor high residues are adopted. The estimates shown in Figure 8 assume that ethanol from corn grain provides a maximum of 10 billion gallons per year.

In the best-case scenario of producing energy crops on 15 percent of our current agricultural land at four times current yield and higher collectible residues from conventional croplands, agricultural bioenergy could supply almost one-fifth of the total U.S. year-2004 demand for energy—a total of 20 EJ.

Figure 8

Potential Energy Supply from Biomass: Four Scenarios



Sources: USDOE-EIA 2003; Walsh 2000; Walsh 2002; Perlack 2005; USDA NASS Database.

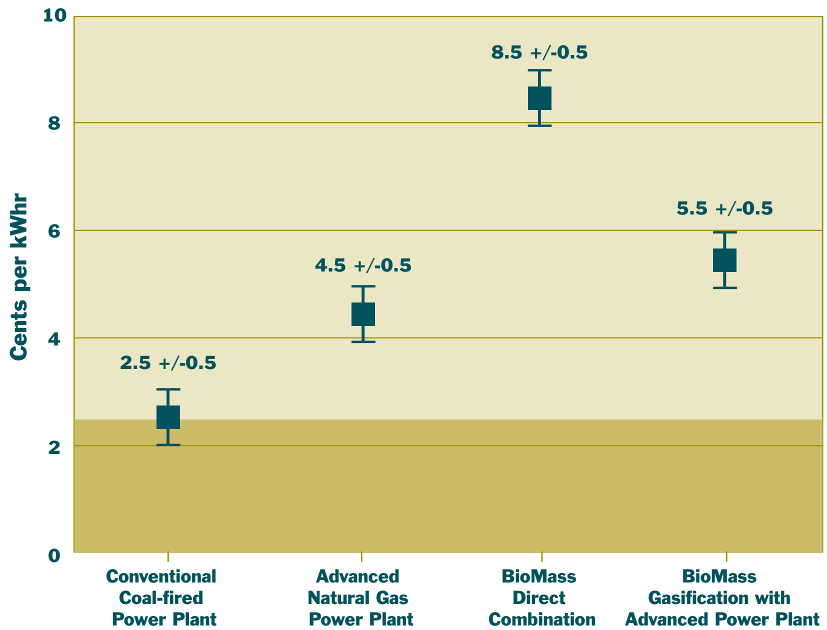
Realizing this potential will depend on biomass fuels becoming cost-competitive with other energy sources.

Figure 9 shows the range and midpoint of costs for power production from coal, natural gas, and several biomass technology options (Spath and Dayton 2003). As the numbers indicate, coal remains the lowest-cost option by far for producing electricity. Biomass technologies are from two to four times more costly. Predictions that coal prices will change little from now until 2025 (USDOE-EIA 2005b) makes the prospects for cost-competitive electricity generation from biomass a goal that will require advances in technology that have not, as yet, even been thought about.

Figure 10 provides a similar comparison for transportation fuels. None of

Figure 9

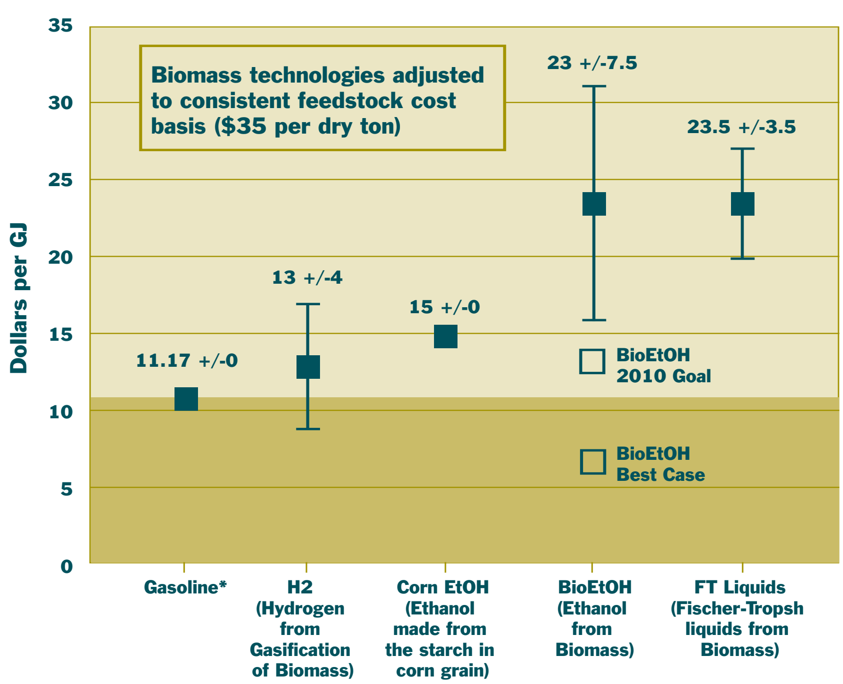
Comparison of Projected Costs for Biopower



Source: Spath and Dayton 2003.

Figure 10

Comparison of Projected Costs for Biofuels



*Gasoline price is USDOE projection for the year 2015.

Sources: USDOE-EIA 2006; Spath and Dayton 2003; Aden, et al. 2002; Wooley, et al. 1999.

the biomass technologies are ready to compete with gasoline today. In the case of ethanol from residues and perennial grasses—BioEtOH in Figure 10—USDOE has developed plans for reducing its cost to about $13 per GJ by 2010. This cost is in a range competitive with ethanol made from corn grain and with gasoline at costs seen during parts of 2005 and 2006, and is within reach of USDOE's long-term projected gasoline price of $13.17 per GJ. While analyses of long-term "mature technology" costs of ethanol from biomass suggest that it may be possible for bio-based ethanol to approach parity with its fossil fuel counterpart (Lynd et al. 1996), achieving such parity represents a significant stretch for the technology.

B. Bioenergy's Impact on Greenhouse Gas Emissions

With the exception of ethanol made from corn, use of biomass to produce heat, power, or transportation fuels essentially results in zero or close to zero CO_2 emissions. A number of studies have been conducted on the life cycle energy and GHG impacts of biomass for power production and transportation fuels. Direct combustion and gasification of biomass for power production are very effective means of offsetting fossil energy use. They displace conventional fuels on essentially a one-to-one basis,[45] reducing fossil energy use by almost 100 percent, with corresponding reductions in fossil fuel emissions. If combined with sequestration of exhaust CO_2, these options also offer significant opportunities for CO_2 removal from the atmosphere. However, use of biomass for heat and power does not offer the benefits of reduced oil dependence and improved energy security that are possible if the biomass is used to make transportation fuels. In addition, the market value of biomass for transportation fuels is significantly higher than its market value in power generation, where low-priced domestic supplies of coal offer stiff competition.

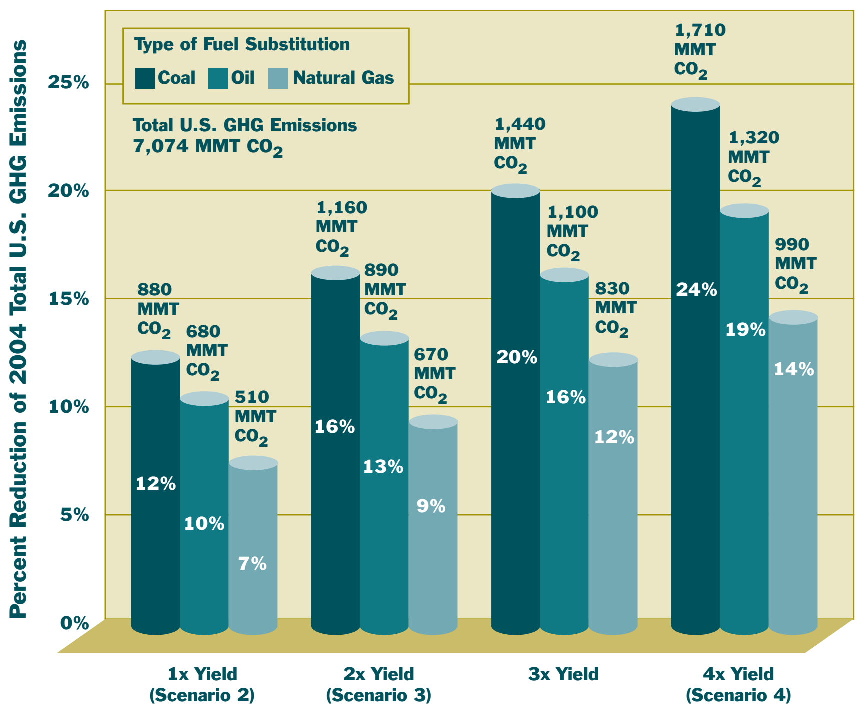
Fossil energy savings associated with the production of hydrogen, ethanol, and Fischer-Tropsch liquids from residues and perenniel grasses have been estimated to be 96, 102, and 91 percent, respectively (Spath and Dayton 2003; Sheehan et al. 2002, 2004). The fossil fuel savings in excess of 100 percent in the case of ethanol are due to the use of lignin—a substance found in residues and grasses—as the energy source for the conversion process. Use of the lignin results in excess energy which can be used to co-generate electricity, thus displacing additional fossil fuels. Ethanol and Fischer-Tropsch fuels are both liquids that can be used in existing internal combustion engine vehicles, while use of hydrogen would require significant changes in vehicle technology.

A number of estimates of the GHG implications of using ethanol in cars and light duty trucks are available (Sheehan et al. 2002; Wang et al. 1999; Macedo 1998). The choice of feedstock has a significant impact. The degree of GHG savings closely tracks the savings in fossil energy use. Replacing gasoline with ethanol made from corn grains reduces GHG emissions by 20 to 40 percent, compared with savings of around 100 percent for ethanol made from Brazilian sugar cane or lignocellulosic biomass such as corn stover or energy crops.

Figure 11 summarizes potential GHG savings associated with replacing fossil energy with biomass energy produced by U.S. farmers. The ranges shown here correspond to energy supply scenarios 2 through 4 shown in Figure 8 (page 52). The amount of GHG savings under each scenario varies, depending on the carbon-intensity of the fossil fuel that is displaced. Under the best-case scenario (four-fold energy crop yield improvements and conversion technology cost reductions), GHG savings range from 14 to 24 percent (990 to 1,710 MMT CO_2 per year). A two-fold improvement in energy crop yield represents modest, and very likely achievable, results for energy crop R&D. Two to three-fold yield improvements correspond to GHG savings of 9 to 20 percent of total year-2004 U.S. emissions (670 to 1,440 MMT CO_2 per year).

Figure 11

Potential GHG Impacts of Agricultural Bioenergy, Depending on Fossil Fuel Displaced



Scenario numbers correspond to scenarios in Figure 8. 3x yield is not shown in Figure 8.

Source: Sheehan, et al. 2002; Wang 1999.

C. Summary and Policy Implications

Bioenergy from agricultural sources offers significant promise as a means of reducing consumption of fossil energy, potentially providing energy equivalent to almost 20 percent of total 2003 U.S. consumption of fossil energy. This amount of bioenergy represents 82 percent of today's U.S. demand for petroleum energy used in on-road transportation. Commensurate with its ability to reduce fossil energy use is bioenergy's ability to dramatically reduce GHG emissions. The order-of-magnitude potential impact of bioenergy on GHGs ranges from 510 to 1,710 MMT CO_2 (140 to 470 MMT carbon-equivalent) annually (9 to 24 percent of 2004 U.S. GHG emissions), depending on which type of fossil fuel is replaced. However, in order to achieve this potential, significant reductions in the costs of converting biomass to transportation fuels will be required, or significant increases in use of biomass for heat and power, as well as significant increases in per-acre yields of crops grown for energy use.

An aggressive research and development effort could eliminate the current price gap between biomass and fossil fuels as sources of energy. In the long run, this means that the GHG savings from bioenergy would come at no additional cost to the nation. To achieve this, the United States would need to invest in biomass research at a scale hitherto unseen. Consistent support of long-term fundamental research for biomass production and conversion is a necessary component of any policy portfolio aimed at making bioenergy a significant part of our energy supply.

Another way to eliminate the price gap would be to recognize the benefits of biomass energy in the marketplace. In the near to mid term, a carbon tax could be used to equalize prices. For example, if USDOE meets its $1.07 per gallon target for ethanol made from biomass, ethanol would still not be cost-competitive with gasoline in 2015, assuming that gasoline's wholesale price is $1.37 per gallon, USDOE's projected price for that year. To compete as an energy carrier with $1.37 per gallon gasoline, ethanol's price would have to be $0.91 per gallon. Thus, a subsidy of $0.16 per gallon of ethanol would be needed to level the playing field. Since ethanol from corn stover saves 8 kg of CO_2 per gallon of ethanol used, a $0.16 per gallon subsidy corresponds to a carbon value of $20 per tonne of CO_2 ($73 per tonne of carbon). In the longer term, as gasoline prices rise in accordance with current projections, the need for a carbon tax or other market incentive will likely disappear. By 2030, for example, USDOE projects wholesale gasoline prices of $1.60 per gallon—equivalent to $1.07 per gallon of ethanol. Technology improvements for ethanol will also continue—further improving ethanol's competitive position in the marketplace.

V. Summary and Conclusions

Farmers' decisions about whether to adopt new management practices and whether to grow energy crops will ultimately determine the level of success of any agricultural sector GHG mitigation strategy. Farmers' decisions are motivated first and foremost by what they perceive to be most profitable. Thus, mitigation practices must be economically attractive to farmers. If farmers can be persuaded to adopt desired practices, the impacts on GHG emissions could be significant. It is technically feasible that 70 to 220 million metric tons (MMT) of carbon could be added to U.S. agricultural soils annually over two to three decades. This would remove 260 to 810 MMT of carbon dioxide (CO_2) from the atmosphere annually, offsetting 4 to 11 percent of current U.S. GHG emissions. Economic potential to store carbon varies substantially by region, and current studies suggest that at prices of $50 per tonne of carbon ($13 per tonne CO_2), soil carbon increases would be limited to 70 MMT per year. If an aggressive research and development (R&D) program succeeds in substantially improving per-acre yields of energy crops and reducing costs of conversion technologies, biomass from agricultural sources could supply up to 19 percent of total current U.S. energy consumption. This would yield GHG savings on the order of 180 to 470 MMT of carbon, which is equivalent to reducing CO_2 emissions by 670 to 1,710 MMT CO_2 per year (by substituting for fossil fuels) or 9 to 24 percent of total U.S. year-2004 GHG emissions.

Overall, studies so far indicate that agriculture is likely to be a competitive supplier of emission reductions if and when farmers are offered suitable payments. Among agricultural mitigation options, soil carbon sequestration will likely be most significant for lower carbon prices (less than $50 per tonne of carbon or $13 per tonne CO_2). At higher prices, afforestation and biofuel options become increasingly more competitive.

Agricultural activities have a broad and multi-faceted impact on all three of the main GHGs—carbon dioxide, methane, and nitrous oxide—and policies designed to mitigate GHGs must consider impacts on all three GHGs. Globally, land use (including agriculture) accounts for about one-third of all GHG emissions due to human activities. In the United States the proportional contribution is smaller, about 8 percent of net U.S. GHG emissions.[46] A variety of agricultural sources contribute to these emissions, including fossil fuel consumption in agricultural production; oxidation of soil organic matter and attendant CO_2 releases; nitrous oxide emissions from nitrogen fertilizer, manure, and plant residues; and methane emissions from ruminant animals, animal wastes, and flooded rice.

However, agriculture as a sector is unique in that it can function as a sink for both CO_2 and methane, helping to reduce their concentrations in the atmosphere. In addition, agricultural production of biofuels can provide a substitute for some of the fossil fuel currently used for energy. Thus, agricultural mitigation of GHGs includes utilization of agriculture's sink capacity, reduction of agricultural emissions, and bioenergy production. Utilization of agriculture's sink capacity is primarily accomplished through increasing soil carbon stocks. Soil carbon increases, which are typically in the 0.1 to 1 tonnes per hectare per year range, could be achieved through adoption of practices such as:

- Reducing the frequency and intensity of soil tillage;
- Including more hay crops in annual rotations;
- Production of high-residue-yielding crops and reduced fallow periods;
- Improved pasture and rangeland management; and
- Conservation set-asides and restoration of degraded lands.

Although soil emissions of nitrous oxide constitute the largest GHG emissions from U.S. agriculture in terms of global warming potential, both measuring emissions and achieving large reductions will be challenging. On average, nitrous oxide emissions are roughly proportional to the amount of nitrogen added to soils, through nitrogen fertilizer, manure, and nitrogen-fixing legume crops. Since nitrogen fertilizer use is an important component of modern, high-yield agriculture, more efficient use of nitrogen inputs is the key to reduction of nitrous oxide emissions through:

- Use of soil testing to determine fertilizer requirements;
- Better timing and placement of fertilizer; and
- Use of nitrification inhibitors and controlled-release fertilizer.

Agricultural methane emissions in the United States occur largely from livestock production through enteric fermentation and during manure storage. Methane capture and use to produce energy is an almost ideal way to address emissions from manure, as it reduces methane emissions, reduces GHG emissions from fossil fuels by providing a substitute energy source, and also provides air and water quality benefits. Strategies to address emissions from enteric fermentation include: improving animal health and genetics, feed additives, and more productive grazing systems.

Storing carbon in soils, reducing nitrous oxide and methane emissions, and producing energy from animal wastes all are potential sources of income or cost reductions for farmers. Relatively few studies of the economic feasibility of agricultural soil carbon sequestration have been done to date, and studies of the economics of nitrous oxide and methane reductions are even more limited. Initial conclusions from studies of the profitability of practices that sequester carbon include:

- Geographic differences in the technical potential and cost of carbon sequestration are substantial;
- Cost considerations are likely to limit agricultural mitigation to levels well below those suggested by technical potential; and
- Strategies based on contracts that pay per tonne of carbon stored or that take into account geographic variation in environmental and economic conditions are more economically efficient (less costly) than contracts based on average conditions.

In addition to profitability strictly defined, several other factors are likely to affect farmers' willingness to participate in mitigation programs:

- Risk, particularly given the likelihood of long-term contracts for carbon sequestration and the high likelihood of changes in economic and technological conditions that can result in unforeseen costs;
- Financial constraints and access to credit when adopting new practices;
- Uncertainty about the long-term effects on crop productivity of adopting carbon sequestering practices;

• Program implementation costs, including contract and transaction costs; and

• Sociological factors, such as age and education level of farmers, farm size, and access to information.

Production of biomass energy could provide a significant opportunity for agriculture to contribute to GHG mitigation. The overall impact of agricultural biomass on GHG mitigation depends on (1) how much energy can be produced from biomass, and (2) the net (life cycle) GHG impact of biomass use for energy. Biomass is particularly well-suited to providing liquid fuel substitutes for petroleum. However, further development of advanced technologies for conversion of biomass into transportation fuels is needed to make biomass more cost-competitive with petroleum.

Current U.S. agricultural bioenegy products for transportation fuels include ethanol made from corn grain, and biodiesel. Although the efficiency of grain-based ethanol production has improved over time, fossil energy use in its production is still high (three units of fossil energy required to produce four units of ethanol energy), limiting its value as a GHG offset. Moreover, there is likely to be an upper limit on the amount of corn-grain ethanol that can be produced economically, currently estimated at 10 billion gallons per year, less than one percent of current energy demand. Biodiesel, made from oil seed crops (e.g., soybean, sunflower) is more energy efficient—about 1 unit of fossil energy to produce 3 units of biodiesel energy, but biodiesel from oil seed crops is currently 50 to 90 percent more expensive than conventional diesel.

Responsible use of agricultural residues such as corn stover or wheat straw for biofuel production could supply 2 to 6 percent of current total U.S. energy demand or 7 to 24 percent of total U.S. petroleum energy demand in the on-road transportation sector. Addressing sustainability issues (soil conservation) is important in determining the amount of residues that could be utilized. Production of energy crops such as switchgrass at current yield rates could displace perhaps an additional 3 percent of current energy supply while utilizing about 10 percent of the total U.S. agricultural area. Improvements in grass genetics could potentially boost this amount to 6 to 12 percent of current energy supply, using up to 15 percent of prime cropland. Potential bioenergy supply from corn, animal manure, CRP lands, agricultural residues, and energy crops grown on prime agricultural land could represent almost one-fifth of total year-2004 U.S. energy demand and more than 80 percent of current U.S. petroleum energy demand in the on-road transportation sector.

Designing and implementing effective agricultural mitigation strategies depends on cost-effective and reliable methods to estimate GHG fluxes and carbon stock changes. Collecting information on management activities such as tillage practices, fertilizer use, and grazing practices at some or all of the NRI locations would improve GHG inventories and assessments. Establishment of a national soil-monitoring network along with additional long-term experiments that include measurements of nitrous oxide and methane fluxes are also needed to improve GHG estimation methods and reduce uncertainty.

A single “magic bullet” solution to the problem of reducing GHG emissions from fossil energy is unlikely, and biomass can play a useful role within a diverse portfolio of GHG reduction strategies. Practices that sequester carbon can maintain and increase soil organic matter, thereby improving soil quality and fertility, increasing water-holding capacity, and reducing erosion. More efficient use of nitrogen and other farm inputs is key to reducing GHG emissions and nutrient runoff, as well as to improving water quality in both surface and ground waters. Using digesters to capture methane from animal wastes can improve air quality and reduce undesirable odors. Consequently, policies should consider not only the GHG benefits but also associated co-benefits to arrive at the most effective solutions in a comprehensive framework. Further R&D is needed to improve the assessment of agriculture’s GHG contributions, to find better ways to manage lands to improve environmental quality, to design efficient policies to implement mitigation options, and to strengthen agriculture’s potential to contribute to producing renewable energy. Although challenges remain, agriculture has much to offer in helping to reduce GHGs in the atmosphere while at the same time improving the environment and the sustainability of agricultural resources.

Appendix A

Economic Studies of Carbon Sequestration Feasibility for U.S. Agriculture

Study	Activity	Region	Quantity (million metric tons/yr)	Average Cost ($/tonne)	Price or Implicit (Marginal) Cost ($/tonne)
Antle et al. 2001	• Change from grain crops to permanent grass with per hectare payment	Montana	0.12 – 0.34	50 – 300	90 – 500
Antle et al. 2003	• Reduced fallow or change from grain crops to permanent grass	Montana			
	— per hectare payments		0.2 – 0.9	10 – 65	10 – 160
	— per tonne payments		0.4 – 0.8	10 – 40	10 – 100
Antle et al. 2005	• Reduced fallow and conservation tillage	Central U.S.		n.a.	
	— zero transaction cost		1.2 – 7.9		10 – 200
	— $5/ha transaction cost		0.9 – 7.4		10 – 200
	(per tonne payments)				
Kurkalova, Kling and Zhao 2003	• Conservation tillage with per tonne payments	Iowa	0.2 – 1.1	n.a.	17 – 95
McCarl and Schneider 2001	• Reduced tillage with emissions tax, subsidy for CO_2, NO_2 and CH_4	U.S.	44 – 70	n.a.	10 – 500
	• Afforestation with per hectare payments	U.S.	4 – 183	n.a.	10 – 500
	• Biofuels with per hectare payments	U.S.	0 – 156	n.a.	10 – 500
Parks and Hardie 1995	• Afforestation of crop and pasture land	U.S.			
	— per hectare payments		45	10	n.a.
	— per ton payments		0 – 120	4 – 80	34 – 670
Pautsch et al. 2001	• Conservation tillage	Iowa			
	— per hectare payments		0 – 1.0	0 – 200	n.a.
	— per ton payments		0 – 1.0	0 – 300	n.a.
Plantinga, Maudlin and Miller 1999	• Afforestation with per hectare payments	Maine	n.a.	0 – 120	0 – 60
		S. Carolina	n.a.	0 – 90	0 – 45
		Wisconsin	n.a.	0 – 95	0 – 48
Stavins 1999	• Afforestation with per hectare payments	U.S. Delta states	7	< 38	< 66
		U.S.	0-518	70	<136
Lewandrowski et al., 2004	• Afforestation with per tonne payments	U.S.	8.5 – 133	n.a.	10 – 125
	• Conservation tillage with per tonne payments	U.S.	1 – 26.9	n.a.	10 – 125
	• Change from crops to permanent grass	U.S.	0	n.a.	10 – 125

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Endnotes

1. Agricultural lands, defined here as croplands and managed pastures, occupy about 25 percent of the vegetated area of the earth's surface and 27 percent of the lower 48 United States.

2. Organic soils, referred to scientifically as *Histosols*, are soils formed under water-logged conditions such as those found in bogs and wetlands, forming deep layers of mainly partially decomposed plant materials. When artificially drained, such soils can be used for agricultural production. However, the term "organic" does *not* denote any particular type of management system (i.e., organic agriculture).

3. Although see Ruddimann, W.F. 2005 for discussion of global pre-industrial agricultural GHG emissions (How did humans first alter global climate? *Scientific American* 292(3): 46–53).

4. Soybean typically produces about one-third the amount of crop residues as corn for the same location and growing conditions.

5. Irrigation in excess of crop needs can create anaerobic zones in soil that stimulate nitrous oxide (N_2O) emissions and/or increase the leaching of nitrate with subsequent emissions of N_2O from the drainage water.

6. High nitrogen balance conditions were classified as fields receiving nitrogen additions that exceed by 25 percent or more the nitrogen removed in harvested products.

7. Globally, methane emissions from rice are an extremely important source of agricultural GHG emissions, particularly from the large areas of rice cultivation in East Asia and the Indian subcontinent. Annual emissions amount to 100–250 MMT carbon-equivalents.

8. Specialized plant cells called *aerenchyma* transport oxygen to support root respiration in flooded soils. Methane produced in the root zone can diffuse through aerenchyma back to the atmosphere. Root morphology, tiller (i.e., stem) number, structure, and abundance of aerenchyma are among the plant characteristics that affect methane transport.

9. The latter is due to the fact that, under anaerobic conditions, nitrification is suppressed and any nitrate initially present will tend to be fully denitrified to nitrogen (N_2) gas. Thus the two main sources of nitrous-oxide emissions are absent.

10. These are broadly defined as facilities where manure is processed under anaerobic conditions, where the organic material breaks down through fermentation, and where methane is produced by methanogenic bacteria.

11. Stored forage that is maintained in a succulent, high-moisture status by being partially fermented.

12. The term *flux* refers to the production and/or consumption of a gas as well as its transport from one place to another (e.g., the flux of gas from the soil to the atmosphere).

13. In perennial vegetation such as forests, biomass stock changes similarly integrate the net carbon flux to the biomass, and measurements of biomass stocks over time are the standard method for estimating emissions or sinks involving forest biomass. Since above-ground vegetation in most agricultural systems is ephemeral (e.g., annual crops) or does not accumulate large standing stocks (e.g., grazed pastures), soil carbon stock changes are the primary focus for agricultural land.

14. To estimate carbon stock changes, the bulk density (soil mass per unit volume of dry soil) as well as the carbon concentration (mass of carbon per unit mass of dry soil) must be determined, and bulk density can vary spatially, seasonally, and in response to management (e.g., as a result of compaction and tillage). Conventional soil sampling involves extracting soil cores, which are then transported to a laboratory and prepared for analysis using wet or dry combustion methods. New analysis techniques, such as laser-induced breakdown spectroscopy (LIBS) (Cremers et al. 2001) and infra-red spectroscopy (McCarty et al. 2002), are being developed, but they are currently applied only in research environments.

15. For example, the 95 percent confidence level means that there is a 1 in 20 chance the true value lies outside a specified interval boundary. To achieve a given confidence level, the number of plots needed is a function of how variable the changes in carbon stocks are expected to be within the sampling area and what precision level is desired for the carbon measurements. For a given precision level (see endnote 16), the more variable the project area, the more plots are needed. For a given variability, more plots are needed to achieve higher precision.

16. Accuracy is the degree to which an estimated value matches its true value. Precision is the degree to which multiple estimations of the same value are alike.

17. NRI is currently pilot-testing the collection of such data at selected inventory points (J. Goebel, personal communication).

18. Analogous systems for forest biomass inventories have been established in most developed countries for decades.

19. More generally the payments can be made for carbon-equivalent units, by converting nitrous oxide and methane emissions into their carbon-equivalents using the concept of global warming potential discussed in Box 1.

20. The McCarl and Schneider model predicts price changes less than 30 percent; the Lewandrowski et al. study predicts changes less than 15 percent.

21. Studies of agricultural subsidy programs show that the costs of administering these programs have been as high as $0.50 for each dollar of subsidy given (Alston and Hurd 1990; McCann and Easter 2000; McCarl 2002).

22. 14 to 20 exajoules (EJ) of bioenergy out of 104 EJ of energy consumed in the United States in the year 2003. An EJ is 1×10^{18} joules. See Figure 8 for a summary of biomass energy potential. Replacing 14 EJ per year of fossil energy requires a doubling of energy crop yields over what is achievable today. Replacing 20 EJ per year of fossil energy requires a 4-fold increase in energy crop yields. Given the history for yield improvements in agriculture, a 4-fold improvement is doable, and a 2-fold improvement is very likely.

23. The greenhouse gas savings of replacing 20 EJ of fossil energy depend on the mix of fossil fuels actually displaced. USDOE reports that total carbon emissions from natural gas, petroleum, and coal are 14, 19, and 25 MMT of carbon per EJ, respectively.

24. Cellulose is the most abundant source of organic carbon in the biosphere. Both cellulose and hemicellulose are long-chain (i.e., polymer) sugar molecules. Hemicellulose contains a mixture of less commonly found, and less easily fermented, sugars than cellulose. Ethanol (C_2H_6O) derives its carbon from these sugars.

25. The two scenarios of two-fold and four-fold energy crop yield improvements shown in Figure 8 correspond to 700 million and one billion tons of agricultural residues, CRP biomass and energy crops. The authors project the ultimate achievable yield of ethanol from biomass to be 116 gallons of ethanol per dry tonne of biomass (or 76 gallons of gasoline equivalent per dry tonne).

26. *Plant gate cost* refers to the cost of the final fuel ethanol product before it is distributed for retail sale. Thus, this cost includes capital and operating cost but does not include distribution and marketing costs, nor does it include taxes.

27. *Net operating costs* are defined as the cost of all variable costs less the credit for co-product sales, and does not include any charge for capital.

28. These cost estimates are in good agreement with survey results reported by USDA for the average operating costs of ethanol plants in 1998 (Shapouri et al. 2001).

29. Ethanol contains 75,700 BTUs per gallon (lower heating value). Energy cost is $1.20 per gallon/75,700 Btus per gal x 948 BTUs/MJ x 1000 MJ per GJ. Gasoline contains 115,000 BTU per gallon (lower heating value).

30. Other state and local incentives also exist in many parts of the United States, which further improve ethanol's market competitiveness.

31. Until recently, EPA has required the use of oxygenated chemical compounds in gasoline in areas of the country that cannot meet national air quality standards for carbon monoxide. An oxygenated compound is simply a molecule that contains oxygen. The presence of molecular oxygen in the fuel in older vehicles reduced the amount of partial combustion of the fuel to carbon monoxide and encouraged complete combustion of fuel to CO_2. Ethanol has been one of the leading oxygenated chemicals used in gasoline. Ethanol also boosts octane number of a fuel. That is, it encourages smooth burning of fuel. Octane number of retail gasoline ranges from 85 for regular gasoline to over 90 for premium gasoline. By comparison, the octane number of pure ethanol is 120. Thus, ethanol is a valuable blend component for refiners interested in raising the overall octane of their gasoline. As an oxygenate or as an octane booster, ethanol has been able to command a higher price relative to its value as a simple carrier of energy (BTUs).

32. U.S. sugar prices have been well above world prices since 1982 because the U.S. Government supports domestic sugar prices through loans to sugar processors and, beginning in FY 2003, a marketing allotment program (USDA–ERS 2005).

33. Macedo 1998 reports a yield of 22 gallons per metric ton of biomass.

34. The purpose of the secondary duty is to prevent foreign ethanol producers from taking advantage of U.S. tax incentives on top of their respective government's incentives. Waivers under specific trade agreements do exist—most notably the Caribbean Basin Initiative.

35. Lubricity is a measure of a fuel's ability to coat engine surfaces and reduce mechanical wear. New lower sulfur diesel has been found to have lower lubricity that can be mitigated with the addition of small amounts of biodiesel.

36. Diesel fuel is slightly more energy-dense than gasoline, containing roughly 133,000 BTUs per gallon, and almost twice as energy dense as ethanol.

37. Based on estimates of methane production per animal per day (Balsam 2002).

38. Iogen (Canada) has a demonstration facility that has produced ethanol from wheat straw but not on a scale or at a cost that is commercially viable.

39. The amount of primary energy contained in corn and wheat straw is around 16.5 GJ per tonne. See Domalski et al. (1987). Thermodynamic Data for Biomass Materials and Waste Components. New York: ASME Press.

40. Results from a recent evaluation of residue removal on soil organic matter (Sheehan, J. et al. 2002. Is Ethanol from Corn Stover Sustainable? Draft. Golden, CO, National Renewable Energy Laboratory) have shown that residue requirements for maintaining soil organic matter may track well with requirements for controlling soil erosion, though much more study is needed to understand the interaction between soil organic matter and soil erosion constraints.

41. Stover is a common name for the residue left after corn harvest. Its name comes from its early use as a stove fuel on the farm.

42. There has never been a debate as to what level of impact on food supply and prices would be acceptable to the American public. That said, a nominal 10–15 percent increase in prices is well within the price fluctuations for commodity crops.

43. Biomass and coal share the same process equipment needs for handling solids, which are, by comparison, more expensive than the equipment-handling needs for liquids and gases like oil and natural gas. To date, experience with biomass gasification has been limited to heat and power generation. In the United States, USDOE has helped to test gasification of wood to provide electricity at a power plant in Vermont at the 200 tonnes per day (5 MW) scale (USDOE 2000). Because combustion and gasification technologies can use coal as well as biomass, the cost of the biomass used in these technologies must compete head-to-head on an energy basis with coal, unless subsidies or incentives for "green power" are available.

44. Assuming that current demand for food crops will not change substantially, we set the limit on agricultural land use for energy at 15 percent in order to avoid significant increases in food prices related to a reduction in supply versus demand for conventional crops.

45. Regardless of the technology used to produce electricity, each unit of biomass energy used in the power plant directly avoids the combustion of one unit of fossil energy.

46. The much higher fossil energy consumption in the United States compared to much of the world and the lack of emissions from deforestation that is occurring primarily in the tropics, accounts for much of the *proportional* difference between global and U.S. numbers.

This report describes opportunities for U.S. agriculture to contribute to reducing net greenhouse gas emissions. The Pew Center on Global Climate Change was established by the Pew Charitable Trusts to bring a new cooperative approach and critical scientific, economic, and technological expertise to the global climate change debate. The Pew Center continues to inform the debate through wide-ranging analyses that will add new facts and perspectives in four areas: policy (domestic and international), economics, environment, and solutions.

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