Carbon Sequestration and Greenhouse Gas Fluxes in Agriculture: Challenges and Opportunities

CAST

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Foreword

Recognizing the need for an update to CAST's 2004 landmark report on climate change and agriculture, the CAST Board of Directors authorized preparation of a new report on the challenges and opportunities for agriculture in dealing with carbon sequestration and greenhouse gas fluxes.

Four eminent scientists agreed to share the role of cochair: Dr. Ronald Follett, USDA–ARS–NPA, Soil Plant Nutrient Research, Ft. Collins, Colorado; Dr. Sian Mooney, Department of Economics, Boise State University, Idaho; Dr. Jack Morgan, USDA–ARS–NPA, Rangeland Resources Research Unit, Ft. Collins, Colorado; and Dr. Keith Paustian, Department of Soil and Crop Sciences, Colorado State University, Ft. Collins. A highly qualified group of scientists served as Task Force members. The group included individuals with expertise in agricultural economics, agroforestry, agronomy, animal sciences, crops and soils, environmental science, food and natural resources, resource ecology, and soil and plant nutrient research.

The Task Force prepared an initial draft of the report and reviewed and revised all subsequent drafts. A member of the CAST Board of Representatives served as the project liaison. The CAST Board of Directors reviewed the final draft, and the authors reviewed the proofs. The CAST staff provided editorial and structural suggestions and published the report. The Task Force authors are responsible for the report's scientific content.

On behalf of CAST, we thank the cochairs and Task Force members who gave of their time and expertise to prepare this report as a contribution by the scientific community to public understanding of the issue. We also thank the employers of the scientists, who made the time of these individuals available at no cost to CAST. The members of CAST deserve special recognition because the unrestricted contributions they have made in support of CAST also have financed the preparation and publication of this report.

This report is being distributed widely; recipients include Members of Congress, the White House, the U.S. Department of Agriculture, the Congressional Research Service, the Food and Drug Administration, the Environmental Protection Agency, and the U.S. Agency for International Development. Additional recipients include media personnel and institutional members of CAST. The report may be reproduced in its entirety without permission. If copied in any manner, credit to the authors and to CAST would be appreciated.

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Interpretive Summary

This publication is a timely update of the landmark 2004 CAST Task Force Report, *Climate Change and Greenhouse Gas Mitigation: Challenges and Opportunities for Agriculture*. Modern-day environmental issues include the need to decrease concentrations of carbon dioxide (CO_2) and other greenhouse gases (GHGs) in Earth's atmosphere. Agriculture is in the middle of this, and the challenges include adapting management and land use to cope with the changing climate and adopting mitigation strategies to decrease agriculture's net contributions to GHG production. While agriculture deals with its key production roles, it also must consider conservation and the protection of natural resources. This report examines the current science to inform the public and policymakers about this crucial topic.

Agricultural Involvement

Globally, agriculture accounts for 13.5% of GHG emissions. In the United States, agriculture is a small but significant component of the country's and world's GHG emissions. We are moving into an uncertain and changing climate pattern that could affect agriculture production, sea levels, and human health. This report's primary focus is on agriculture's role in the land-atmosphere exchanges of GHGs as well as agriculture's ability to decrease GHG emissions or sequester additional carbon in agricultural soils while continuing to supply the necessary food, feed, and fiber required for the world's growing population.

Mitigation Options

Emissions of CO_2, CH_4 (methane), and N_2O (nitrous oxide) from agriculture are the result of both human-induced and natural processes in the ecosystem carbon (C) and nitrogen (N) cycles. Although these causes of GHG emissions cannot be completely eliminated, they can be lowered through modified land use and management.

In general, agricultural activities can mitigate emissions by

1. Decreasing emissions of GHGs due to agricultural causes;
2. Increasing sequestration of C in soil organic matter and plant biomass, resulting in a net removal of CO_2 from the atmosphere; and
3. Using sustainable agricultural biofuels with their capacity to offset CO_2 emissions from fossil fuels.

This report outlines a number of practices for which increased C sequestration and decreased emissions of GHGs have been established or, in some instances, are presently under investigation. The practices are evaluated and presented in separate sections that cover annual cropland, pasture and range, horticultural crops, agroforestry systems, wetlands and organic soils, confined livestock, and biofuel feedstock production.

There are two principal opportunities for C sequestration in agricultural ecosystems:

1. Improved management of permanent agricultural land through practices that enhance C storage
2. Conversion and/or restoration of marginal and degraded agricultural lands to alternative, C-sequestering uses

Emissions from N_2O can be decreased mainly through more efficient use of N additions to soils, and the main opportunities for CH_4 reductions in U.S. agriculture are through improved livestock and manure management practices.

Impacts on Society, Including on Agriculture

Economics govern the adoption of GHG emission-decreasing or sequestration-enhancing practices. The many possible ways to design adoption incentives or implement policy tools include the following:

1. Emission Taxes—Emitters of GHGs would face a tax on their emissions whereby emitters would be encouraged to implement emissions reduc-

tion technologies and thereby decrease their GHG emissions.

2. Market-based cap and trade—An overall limit (cap) on GHG emissions is set by a regulator and regulatory credits are issued equal in number to the level of the cap.

These systems have various options and issues that the authors examine in this report. They also point out several other important factors:

- Biophysical estimates of emissions reduction potentials are generally overestimates as they do not account for adoption costs or the possibility of higher economic returns from competing practices, and, in fact, different practices will likely dominate at different market prices.
- In addition to providing more reliable emission estimates under current land use, detailed multi-GHG models are needed to reliably assess mitigation potentials at regional and national scales within the United States.
- Agricultural management practices that sequester carbon or lower GHG emissions may have other environmental benefits (cobenefits) such as decreased soil erosion, decreased N and phosphorus surface runoff, and improved wildlife habitat.

Land owners may engage in GHG mitigation efforts for a variety of reasons, such as a desire to practice good environmental stewardship or a reaction to incentives for participating in private-sector offset markets or government-sponsored mitigation programs.

The rapid development of user-friendly tools that also can incorporate state-of-the-art models and fine-scale information on soil, climate, and management variables can help support science-based mitigation activities for U.S. agriculture. Comprehensive GHG legislation would also impact agricultural income in three ways:

1. Restrictions in GHG emissions would induce an increase in energy prices, which would raise agricultural production costs for energy.
2. Through economy-wide adjustments to increased energy prices, stronger incentives to produce alternative energy sources such as biofuels would likely increase.
3. Legislation that creates a market for GHG mitigation credits with offsets may generate new streams of income.

Executive Summary

To abate climate change is one of the most pressing modern-day environmental issues (IPCC 2007a). As a signatory country to the United Nations Framework Convention on Climate Change, the United States is actively engaged in a critical international effort to find solutions to the problems posed by climate change. The particular challenges for agriculture include adapting management and land use to cope with the changing climate and adopting mitigation strategies to decrease agriculture's net contributions to three of the greenhouse gases (GHGs)—carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). These challenges are additional to agriculture's pivotal roles—to produce food, feed, and fiber as well as bioenergy feedstocks and to provide for conservation and protection of natural resources.

The Science and Uncertainties Important to Climate Change

Greenhouse gases, often called *trace gases*, are present in the atmosphere in small concentrations and are crucial in controlling the energy balance and climate of the Earth. Fossil fuel combustion is the largest single source of human-caused GHG emissions. Agricultural emissions of the GHGs CO_2, CH_4, and N_2O are relatively small compared to emissions from fossil fuels, but as agricultural emissions can be manipulated by human activity, options for their management are potentially important. This report's primary focus is on agriculture's role in the land-atmosphere exchanges of GHGs as well as agriculture's ability to reduce GHG emissions or sequester additional carbon (C) in agricultural soils while continuing to supply the necessary food, feed, and fiber required for the world's growing population.

Greenhouse gases insulate the planet from extremes in temperature, but those effects are becoming intensified with human-caused emissions of GHGs into the atmosphere and are significantly altering the Earth's climate. A recent analysis of future temperature trends for the United States suggests, under continuing high emissions, an approximate 2.2–3.6°C (4–6.5°F) increase in temperature will occur across most of the United States by the period 2040–2059 compared to the baseline years 1961–1979, and that temperatures will show a further increase of up to 6.2°C (11°F) by the end of the century (USGCRP 2009). Even if emissions of CO_2 and other GHGs were to cease immediately, changes in the Earth's climate system due to the rise in atmospheric GHGs over the past two hundred years will likely sustain this warming for several centuries (Solomon et al. 2009).

We are moving into an uncertain and changing climate regime, with an increased incidence of extreme weather, warmer temperatures, retreating glaciers, thawing permafrost, and pest infestations, but with consequences that are likely to differ regionally in severity. In the United States, less frequent but heavier downpours are documented (IPCC 2007a) and, along with increased drought in some parts of the United States, are predicted to increase in the future (Seager and Vecchi 2010). Significant increases or decreases in rainfall quantity, timing, and intensity can have considerable economic impacts on agriculture. Problems from more intense rainfall include delayed planting, field flooding, more within-season water stress, and decreased crop quality. In the southwestern United States, less frequent rainfall combined with warmer temperatures is predicted to increase drought. Declining or rapid melting of snowpacks in the United States and elsewhere is threatening agricultural and municipal water supplies. Sea level rise and storm surges are a particular concern for coastal areas, with consequences for transportation and energy infrastructure. Human health concerns include increasing heat stress, insect and waterborne diseases, poor air quality, extreme weather events, and disease.

The Role of Agriculture

Terrestrial carbon absorption is the relatively small difference between the C exchanged between terrestrial ecosystems and the atmosphere. Each year, about 60 petagrams C is exchanged in each direction between the atmosphere and terrestrial ecosystems. The release of long-sequestered fossil fuels is accelerating with human population growth and development and challenging the capacity of terrestrial sinks to store the increased atmospheric C. Further, the C stored in the biological sinks and in the soil is vulnerable to return to the atmosphere as natural or man-made disturbances can cause soil organic carbon to oxidize. Thus, terrestrial sinks may best be viewed as mid-term reservoirs that may not be permanent offsets to the emissions from fossil fuels.

Atmospheric CO_2 concentrations have increased from about 280 ppmv (parts per million by volume) at the beginning of the industrial era to current levels of 390 ppmv. Approximately 80% of the current global CO_2 emissions is from fossil fuel burning and the remaining amount from deforestation, land use, and land use change. The increasing concentrations of the non-CO_2 GHGs, N_2O and CH_4, are primarily from agriculture (Del Grosso et al. 2005; Denman et al. 2007; Smith and Conen 2004; USEPA 2011; USGCRP 2009). Well over half of global CH_4 emissions are attributed to human activities, with agriculture (primarily CH_4 produced in digestive tracts of livestock, rice cultivation, and sewage) contributing about 50% followed by mining, transportation, fossil fuels, sewage, and landfills (Denman et al. 2007). Approximately half of N_2O emissions are anthropogenic, and agriculture is the biggest anthropogenic N_2O source. An estimated 40% of the anthropogenic N_2O emissions come from agriculture, primarily nitrogen fertilizer, legumes, manure, and soil and crop management (Del Grosso et al. 2005; Denman et al. 2007; Smith and Conen 2004). Globally, agriculture accounts for 13.5% of GHG emissions. In the United States, the agricultural sector emits over 6% of total national GHG emissions (in CO_2 equivalents). Thus, U.S. agriculture is a small but significant component of our nation's and the world's GHG emissions.

Plant physiologists are showing that continued increases in CO_2 will stimulate photosynthesis and water use efficiency and have potentially positive effects on plant growth, but that plants differ in their sensitivity to CO_2. For instance, photosynthesis in plants possessing the C_3 photosynthetic pathway (cool season grasses; most woody plants; important crop plants like soybean; and a number of weedy species in the United States) are not currently CO_2-saturated (Pearcy and Ehleringer 1984). Continued increases in CO_2 should lead to higher photosynthesis, likely translating to high plant productivity. Such direct photosynthetic benefits will decline, however, as photosynthesis approaches CO_2-saturation. In contrast, photosynthesis in plants with the C_4 photosynthetic pathway (warm-season grasses; some major crop species like corn; and some weeds) are already nearly CO_2-saturated at present-day CO_2 concentration; further increases in CO_2 are not expected to have much direct effect on photosynthesis (Ehleringer, Cerling, and Helliker 1997). The increased water use efficiency that results from stomatal closure in both C_3 and C_4 plants exposed to high levels of CO_2 suggests water relations of especially dryland or droughted plants may be enhanced in a future CO_2-enriched world (Leakey 2009; Morgan et al. 2004, 2011).

Temperature has many impacts on agriculture, but two of the most important ones are that it regulates the rates at which biochemically driven reactions proceed and that it affects the exchanges of energy and matter between agro-ecosystems and their environments. Both the biochemically driven and energy aspects of warming can elicit positive and negative effects on plant production. Plant species have different critical temperature ranges for their growth and development. A minimal base temperature is required for plant activity, and increases in temperature will eventually reach an optimal temperature that maximizes growth or yield; further increases in temperature will reduce activity, growth, or yield. A potentially negative effect of warming will be increased incidences of drought. All else being equal, warming induces greater evapotranspiration, which leads to desiccation, but increased water use efficiency from higher CO_2 may help to offset drier conditions.

Mitigation Options

Emissions of CO_2, CH_4, and N_2O from agriculture are the result of both human-induced and natural processes in the ecosystem C and N (nitrogen) cycles. Although these causes of GHG emissions cannot be completely eliminated, they can be lowered through modified land use and management. In general, agricultural activities can mitigate emissions by (1) decreasing emissions of GHGs due to agricultural causes, and (2) increasing sequestration of C in soil organic matter and plant biomass, resulting in a net removal of CO_2 from the atmosphere. Rates of C sequestration are limited in quantities and duration by the nature of the biological C cycle as well as by inherent capacities to store C in soils and biomass. Thus, after a change in management designed to increase C stocks (e.g., by increasing C inputs and/or decreasing decomposition rates), soil C stocks tend to approach a new equilibrium level where C inputs and outputs are again balanced, after which there is no more net increase in C storage. Sustainable agricultural biofuels present a third mitigation option with their capacity to offset CO_2 emissions from fossil fuels. The United States is now the world's leading producer of ethanol, and although the growth of cellulosic biofuel production has been slowed by unavailability of technology, the recent economic downturn, and the lower price of crude oil in 2008–2009, the country has a goal—to have biofuels represent 22% of total transportation fuel needs by 2022.

Activities that generate GHG emissions from agriculture include conversion of unmanaged ecosystems to agricultural uses and many common soil-, crop-, and livestock-management practices. Large losses of biomass and soil C stocks as well as substantial GHG emissions can result from land use conversions involving deforestation, biomass burning, wetland drainage, grassland conversion, plowing, and accelerated soil erosion. Additional CO_2 emissions are associated with energy used for the production and application of agricultural inputs such as fuel, fertilizers, lime, and pesticides plus that used in planting, harvesting, drying, irrigating, processing, and transporting commodities. Drained organic soils can sustain continuing oxidation of organic matter and increased CO_2 and N_2O emissions over several decades. Agricultural lands are subject to N_2O losses, whether from application of fertilizers, use of N-fixing crops, or manure; and during manure storage. Enteric fermentation in ruminant livestock, manure management, and rice cultivation are the predominant agricultural sources of CH_4. In addition, agricultural use generally decreases the natural CH_4-oxidizing capacity of nonflooded soils, usually by a factor of 8–10 or more, which contributes to CH_4 increase in the atmosphere.

This report outlines a number of practices for which increased C sequestration and decreased emissions of GHGs have been established or, in some instances, are presently under investigation. The practices are evaluated and presented in separate sections that cover annual cropland, pasture and range, horticultural crops (including turf), agroforestry systems, wetlands and organic soils, confined livestock, and biofuel feedstock production.

In croplands, a number of practices are identified that can increase soil C inputs (e.g., high-yielding residue crops, manure additions), lower soil organic matter decay rates (e.g., no-till or reduced tillage practices), or accomplish both (e.g., conversion of annual crops to perennials, cover crops). A particularly important strategy for decreasing emissions of N_2O is to improve the efficiency of production inputs (especially N fertilizer), thus decreasing associated fossil energy-derived CO_2, as well as N_2O emissions from inefficient use of N inputs. Means also exist to decrease CH_4 emissions and/or capture them for use as an energy source.

There are two principal opportunities for C sequestration in agricultural ecosystems: (1) improved management of permanent agricultural land through practices that enhance C storage, and (2) conversion and/or restoration of marginal and degraded agricultural lands to alternative, C-sequestering uses. Soil C stocks are governed by a balance between C additions (via both above- and belowground plant residues, manures, or other organic amendments) and losses, primarily as CO_2 through decomposition (i.e., heterotrophic soil respiration). Thus by increasing C inputs to soils and/or decreasing the rate of organic matter decomposition, the C content of the soil can be increased. There are many practices that can increase soil C sequestration. Conversion of degraded lands can also increase C sequestration when properly managed, including the conversion of cropland to pastures in long-lived roots of herbaceous perennial crops used for forage (e.g., hay fields) or fuel (e.g., biofuel feedstocks such as switchgrass); woodland or grassland conservation set-asides (e.g., the U.S. Conservation Reserve Program); wetland restoration; or restoration of land severely degraded by mining, salinization, or other activities such as industrial waste disposal. In addition to soils, C can be sequestered in woody biomass through agroforestry practices (e.g., windbreaks, riparian forest buffers, and other tree-based conservation buffers) and establishment of perennial crops for food (e.g., orchards) or biofuel (e.g., hybrid poplars). Finally, production of agricultural biofuels provides opportunities to offset fossil energy CO_2 emissions from agriculture and other sectors of the economy.

Impacts on Society, Including on Agriculture

Economics govern the adoption of GHG emission-decreasing or sequestration-enhancing practices. Although some producers in the United States have already adopted such practices, further adoption will occur only if the practices become profitable (in the absence of regulatory mandates). There are many ways to design adoption incentives or implement policy tools that could be used to lower GHG emissions. Two commonly discussed policies are emissions taxes and a market-based cap-and-trade system.

Under a policy of taxation, emitters of GHGs would face a tax on their emissions whereby emitters would be encouraged to implement emissions reduction technologies and thereby decrease their GHG emissions. Pollution reduction credits would not be traded between sectors under a taxation scheme. The real question regarding the tax is what it would it cover. For example, would sequestration be a recipient of a tax credit; how would fertilizer N_2O be included or range livestock CH_4? Presently, there are no standardized models or modeling techniques that are appropriate to answer the wide range of policy questions related to changes in production practices under alternative incentives that might be designed to mitigate agricultural GHG emissions. The USEPA (2011) reports that emissions from the agricultural sector account for over 6% of total U.S. GHG emissions. In contrast, the energy sector accounts for about 80% of U.S. GHG emissions. Consequently, emissions for the energy sector will likely be the first point of attention. So where does agriculture come in? The answer lies in the relative costs of emissions reduction. Namely, if agricultural offsets are to be economically competitive, they must be cheaper than emission reductions in the energy and other sectors. This includes the production cost and any transaction costs attendant to conveying the credits.

Under cap and trade, an overall limit (cap) on GHG emissions is set by a regulator and regulatory credits are issued equal in number to the level of the cap. Coverage would again be an issue, with questions arising such as whether or not added sequestration can expand the pool of permits and which emissions sources would be capped. In some cap-and-trade systems, emission reductions from nonregulated entities also generate credits, often referred to as offsets. The items that trade reflect reductions in GHG emissions and are converted to CO_2 equivalents (based on their global warming potential) and then referred to as carbon credits (C-credits). Although regulatory credits and offsets share a common definition in terms of GHG emissions and can be traded with each other in many markets, they are distinct products because they embody different kinds of risks and obligations.

Physical estimates of emissions reductions are generally overestimates as they do not account for adoption costs or the possibility of higher economic returns from competing practices, and, in fact, different practices will likely dominate at different market prices. In developing and selling credits, there are costs associated with assembling enough credits to fill a contract, monitoring compliance, and negotiating the contract. These are commonly called transaction costs and have been identified as one of the greatest hurdles for tradable permit systems. In addition, the net GHG reductions from individual land parcels are too small to sell in a GHG market and multiple parcels will need to be assembled to fill a single contract.

A combination of approaches is used to estimate GHG emissions from agriculture at the regional and national levels in the United States. Complex "process-based" models likely yield more reliable results than nationally uniform emissions factors for soil C and soil N_2O, but the models have not been parameterized to represent all crops and situations. In addition to providing more reliable emission estimates under current land use, complex multi-GHG models are needed to reliably assess mitigation potentials at regional and national scales within the United States because mitigation options designed to increase C storage are likely to impact N_2O emissions as well. Complex models have the ability to represent these interactions. Comprehensive simulations intended to address GHG mitigation potential at the national scale have been attempted only recently. A difficulty in the use of complex models is that viable mitigation options differ across the many production environments in the United States and the different land uses.

Agricultural management practices that sequester carbon or lower GHG emissions may have other environmental benefits, commonly referred to as cobenefits. The *cobenefits* are improvements in environmental factors such as decreased soil erosion, decreased nitrogen and phosphorus surface runoff, and improved wildlife habitat. Such cobenefits may need to be considered under a cap and trade, particularly if they represent substantive value for farmers, ranchers, and/or society.

Individual land owners may engage in GHG mitigation efforts for a variety of reasons, such as a desire to practice good environmental stewardship or a reaction to incentives for participating in private-sector offset markets or government-sponsored mitigation programs. Regardless of the specific motivation, adoption of improved management practices to decrease net agricultural GHGs will necessarily be implemented at the farm, ranch, or producer level, and thus estimates of emissions (and emission reductions) are also needed at that scale in addition to consideration of broader issues of baseline, additionality, *permanence*, uncertainty, and leakage. A variety of protocols, decision-support tools, and models has been developed (and more are under development) to support producer-level GHG estimation for potential use in GHG mitigation policies as well as voluntary emissions offset markets. The rapid development of user-friendly tools that also can incorporate state-of-the-art models and fine-scale information on soil, climate, and management variables can help support science-based mitigation activities for U.S. agriculture. Equally important is the continued expansion of field measurements and monitoring systems to improve the underlying models and provide solid estimates of uncertainty.

Finally, any comprehensive GHG legislation would impact agricultural income in three ways. First, restrictions in GHG emissions could induce an increase in energy prices, which could raise agricultural production costs for energy inputs such as fuel and electricity as well as for energy-intensive inputs such as fertilizer. Second, through economy-wide adjustments to any increased energy prices, stronger incentives to produce alternative energy sources such as biofuels, and diverted production caused by additional mitigation activity, the prices of many agricultural commodities would likely increase. The net effect of these two changes on farm incomes would depend on whether the cost increase or revenue increase is larger, although analyses have generally shown the price effect to be greater. Third, for legislation that creates a market for GHG mitigation credits with offsets, agricultural producers may have new streams of income from generating and selling these offsets. Although projected impacts vary regionally and by agricultural subsector, overall projections are for increased farm income when offset sales are accounted for.

1 Introduction: Climate Change and Agriculture

Climate change is one of the most important environmental problems of our day. Rarely does a week pass without some new information on the changing climate and implications for our planet. Slowing the growth of greenhouse gas (GHG) concentrations in the Earth's atmosphere is a critical challenge for society (IPCC 2007a) and will be the primary focus of this report. In this introductory chapter, some background on climate change science, the role of agriculture in GHG emissions, and the potential impacts of climate change on society and agriculture will be briefly discussed. Subsequent chapters will address GHG emissions from agriculture, mitigation options, biofuels, economics of GHG mitigation strategies, and, finally, implementation and policy issues. The report will provide readers with a background in climate change science and a strong grounding in the science, economics, and policy aspects of GHG mitigation and implications for U.S. agriculture.

Background

The literature about climate change largely began in the early 1980s (Ausubel 1983; Lemon 1983; Waggoner 1983), accompanied by a growing realization of the implications for U.S. agriculture (Adams et al. 1990; Rosenberg 1982, 1988) and the potential role of soil carbon (C) (Schlesinger 1986). Linkages among the soil C pool, the global C cycle, and soils as a source or sink of atmospheric carbon dioxide (CO_2) had stimulated early discussions regarding soil management as a possible strategy to reduce greenhouse gases (Dyson 1977; Jenny 1980). In 1990, then USDA Assistant Secretary Hess turned to the Council for Agricultural Science and Technology (CAST) to ask on behalf of U.S. farmers and foresters:

- What role does agriculture play in having adverse effects on the climate?
- What should agriculture do to adapt to possible climate change?
- What can agriculture do to decrease emissions of greenhouse gases?

A subsequent report to the U.S. Department of Agriculture (USDA), distributed also to Congress, the Environmental Protection Agency, the Food and Drug Administration, the Agency for International Development, the Organic Trade Association, and the Office of Management and Budget (CAST 1992), served as a resource for USDA attendees at the United Nations Framework Convention on Climate Change (UNFCCC) in Rio de Janeiro, Brazil, in 1992. The United States subsequently became a signatory country to the UNFCCC treaty. Much of the discussion in these early communications concerned the potential of agricultural systems to sequester C and thereby help remove some of the most abundant greenhouse gas, CO_2, from the Earth's atmosphere.

A second report (CAST 2004), *Climate Change and Greenhouse Gas Mitigation*, was released in 2004, with a synthesis of the latest research on GHG mitigation in agriculture, covering the aspects of GHG emissions and mitigation as well as economic and public policy issues critical for the adaptation of agriculture toward decreased GHG emissions. Since that report was released seven years ago, advances in global change science have significantly enhanced our knowledge regarding the nature of GHG emissions from agriculture, as well as the effectiveness of various management practices to control those emissions. A rapidly expanding interest among scientists and society regarding the potential dangers of unchecked climate change is stimulating movement among national and international bodies to address the underlying causes of climate change through treaties and legislation. For these reasons, the CAST Board of Directors authorized preparation of a new report on carbon sequestration and GHG fluxes in agriculture to provide the very latest information on this topic.

Evaluating net GHG emissions in food, feed, and fiber production systems requires quantification of the entire suite of greenhouse gases—CO_2, methane (CH_4), and nitrous oxide (N_2O). Carbon dioxide is an important GHG exchanged in consequential amounts between soils and the atmosphere. Much attention has been given to the storage of atmospheric CO_2 into stable organic fractions in the soil as a means to sequester C in agricultural systems and offset the emissions of CO_2. However, N_2O and CH_4 are also emitted in significant quantities from agricultural systems. Because of their increasing rates of emissions and large global warming potential (IPCC 2007a), both CH_4 and N_2O need to be considered along with CO_2 if agricultural production system effects on net GHG flux and GHG intensity are to be properly evaluated.

In considering how best to implement an effective strategy for decreasing agriculture's GHG emissions, economic and societal considerations must be included. Some of the latest insights on what is presently known about the fundamental biology relevant to the exchanges of these three GHGs between agricultural soils and the Earth's atmosphere, as well as information about promising areas of GHG mitigation management that can lessen agricultural contributions to the atmosphere's GHG inventory, will be provided. The effect of management on GHG emissions will be weighed against the impact of proposed management changes on system productivity, profitability, and the environment to determine which management options are most desirable.

Enhancing the rates and amounts of soil C sequestered requires that agricultural producers must choose to use improved management practices. Therefore, economic implications pertinent to farmer adoption and policy implementation will be presented. In the final chapter of this report, critical research needs that will be required to fill knowledge gaps so the U.S. agriculture sector will be well poised with appropriate tools for reducing its contribution to GHGs, while remaining economically competitive, will be identified. This holistic approach to GHG mitigation in agriculture will provide the reader with the necessary tools to better understand the science of GHG mitigation in the context of current-day agriculture and society.

Climate Change

Climate is dominated by the balance of energy derived from the sun. The Earth's energy balance can be altered in three basic ways: by changing the incoming radiation (e.g., through changes in solar activity or the Earth's orbit around the sun), by changing the amount of radiation that is reflected back into space (e.g., through changes in vegetation or cloud cover), and by changing the amount of radiation that is re-radiated back into space through long-wave radiation (e.g., through changes in atmospheric GHG concentrations) (Le Treut et al. 2007). The concern in this report is primarily with agriculture's role in the land-atmosphere exchanges of GHG.

Of the solar energy (short-wave) that reaches the top of the Earth's atmosphere, 30% is reflected back into space, mostly by aerosols (small suspended particles such as dust, sulfates, and salt) in the atmosphere, but also by light-colored areas on the planet's surface (Figure 1.1). The remaining 70% is absorbed by the Earth's atmosphere and its surface and re-emitted back into space as long-wave radiation. Greenhouse gases in the Earth's atmosphere capture some of the long-wave radiation, causing warming of the Earth's surface through the well-known greenhouse effect. Water vapor and CO_2 are the most important GHGs, but CH_4 and N_2O are becoming increasingly important in the Earth's energy balance. Naturally occurring GHGs have a mean warming effect of about 33°C (59°F) (IPCC 2007b) and thus make life on this planet possible. The rapidly increasing concentration of GHGs in the atmosphere due to human activities, however, are altering the Earth's energy balance and warming the planet (Figure 1.2).

Global surface temperatures have increased along with rising concentrations of CO_2 and the other GHGs since the late nineteenth century (Figure 1.3), and modeling exercises that attempt to separate human from natural influences on the climate strongly suggest human-induced changes of the atmosphere are the dominant causes for warming of the planet (USGCRP 2009). There are many feedbacks that can amplify or diminish these GHG-driven changes in climate. For instance, volcanic eruptions or changes in the Earth's solar activity can alter the Earth's energy balance, although activities in the past 30 years in these areas have likely cooled, not warmed, the planet. The reader is referred to recent reports from the Intergovernmental Panel on Climate Change (IPCC) (Denman et al. 2007; Forster et al. 2007) and the U.S. Global Change Research Program (USGCRP 2009) for thorough treatments of climate change science.

A recent analysis of future temperature trends suggests an approximate 2.2–3.6°C (4–6.5°F) increase in temperature across most of the United States by the period 2040–2059 for a high GHG emissions scenario compared to the baseline years of 1961–1979, and a further increase of up to 6.2°C (11°F) by the end of the century (Figure 1.4). Emissions scenarios are plausible storylines of future human activity, with high or low GHG emission scenarios corresponding to high or low levels of population growth, energy production technologies, mitigation efforts, and so on. Although these future trends, and hence GHG emission rates, are uncertain, it is worth noting that even if GHG emissions were now to completely cease, some temperature increases, sea level rise, and other im-

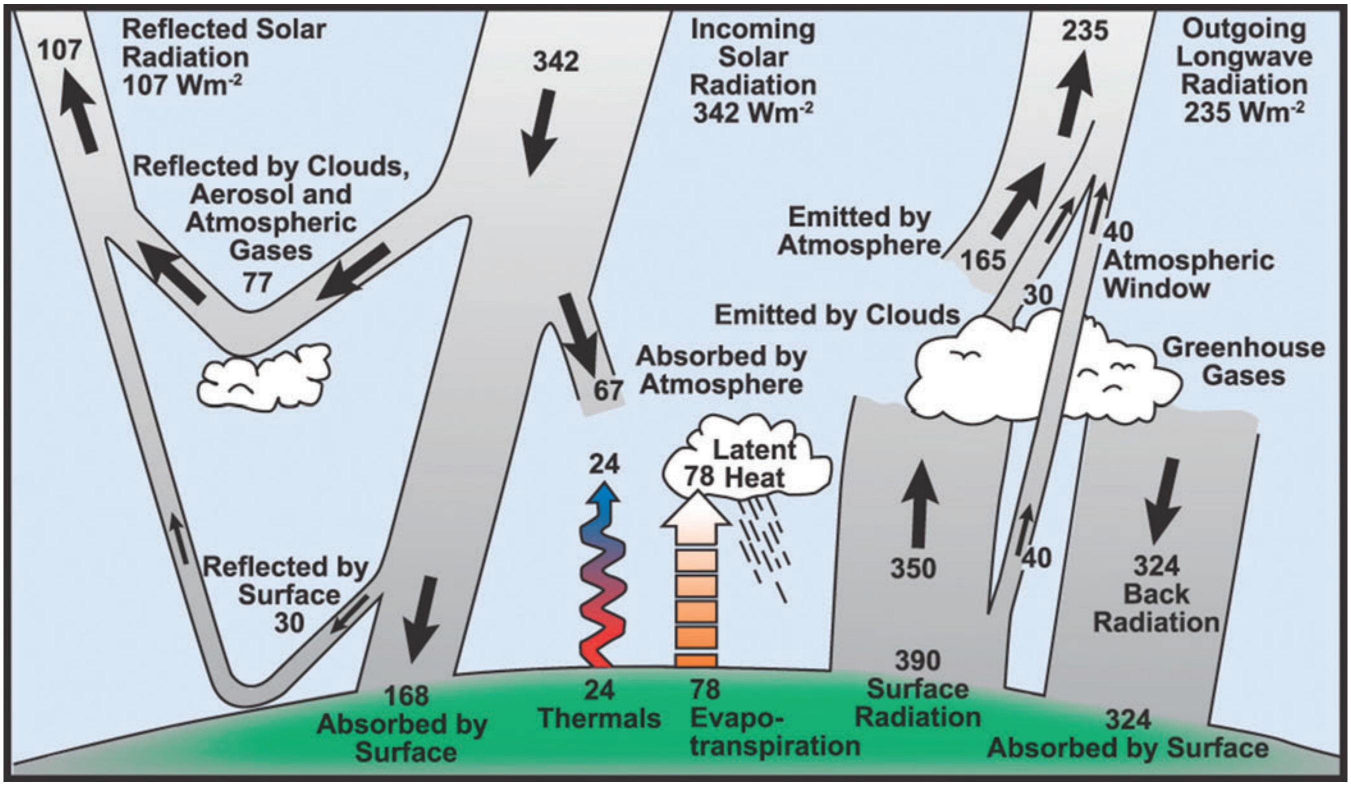

Figure 1.1. Estimate of the Earth's annual and global mean energy balance. Over the long term, the amount of incoming solar radiation absorbed by the Earth and atmosphere is balanced by the Earth and atmosphere releasing the same amount of outgoing longwave radiation to space. About half of the incoming solar radiation is absorbed by the Earth's surface. This energy is transferred to the atmosphere by warming the air in contact with the surface (thermals) through evapotranspiration and longwave radiation that is absorbed by clouds and greenhouse gases. The atmosphere in turn radiates longwave energy back to Earth as well as out to space (Kiehl and Trenberth 1997; Le Treut et al. 2007).

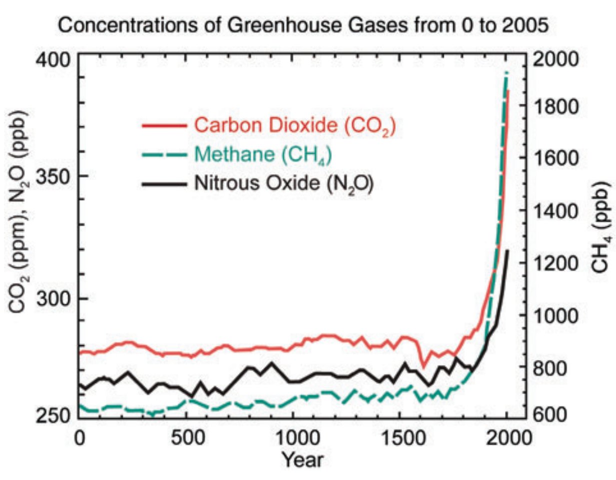

Figure 1.2. Increases in concentrations of these gases since 1750 are due to human activities in the industrial era. Concentration units are parts per million (ppm) or parts per billion (ppb), indicating the number of molecules of the greenhouse gas per million or billion molecules of air (USGCRP 2009).

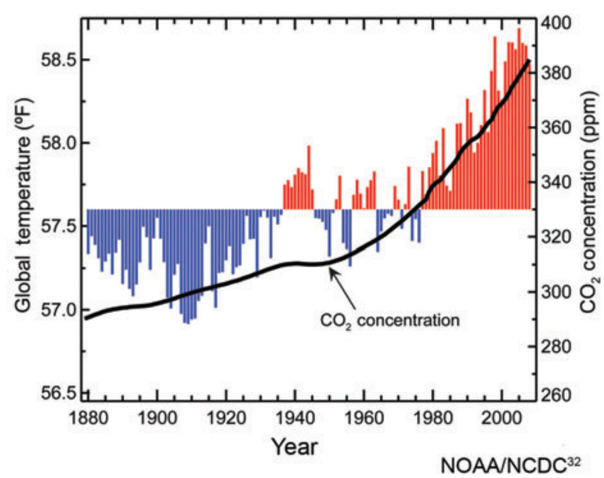

Figure 1.3. Increases in annual global surface temperature (over both oceans and land) since 1880. Red bars indicate temperatures above and blue bars represent temperatures below the average temperature period 1901–2000. The black line is atmospheric CO_2 concentration in parts per million (USGCRP 2009).

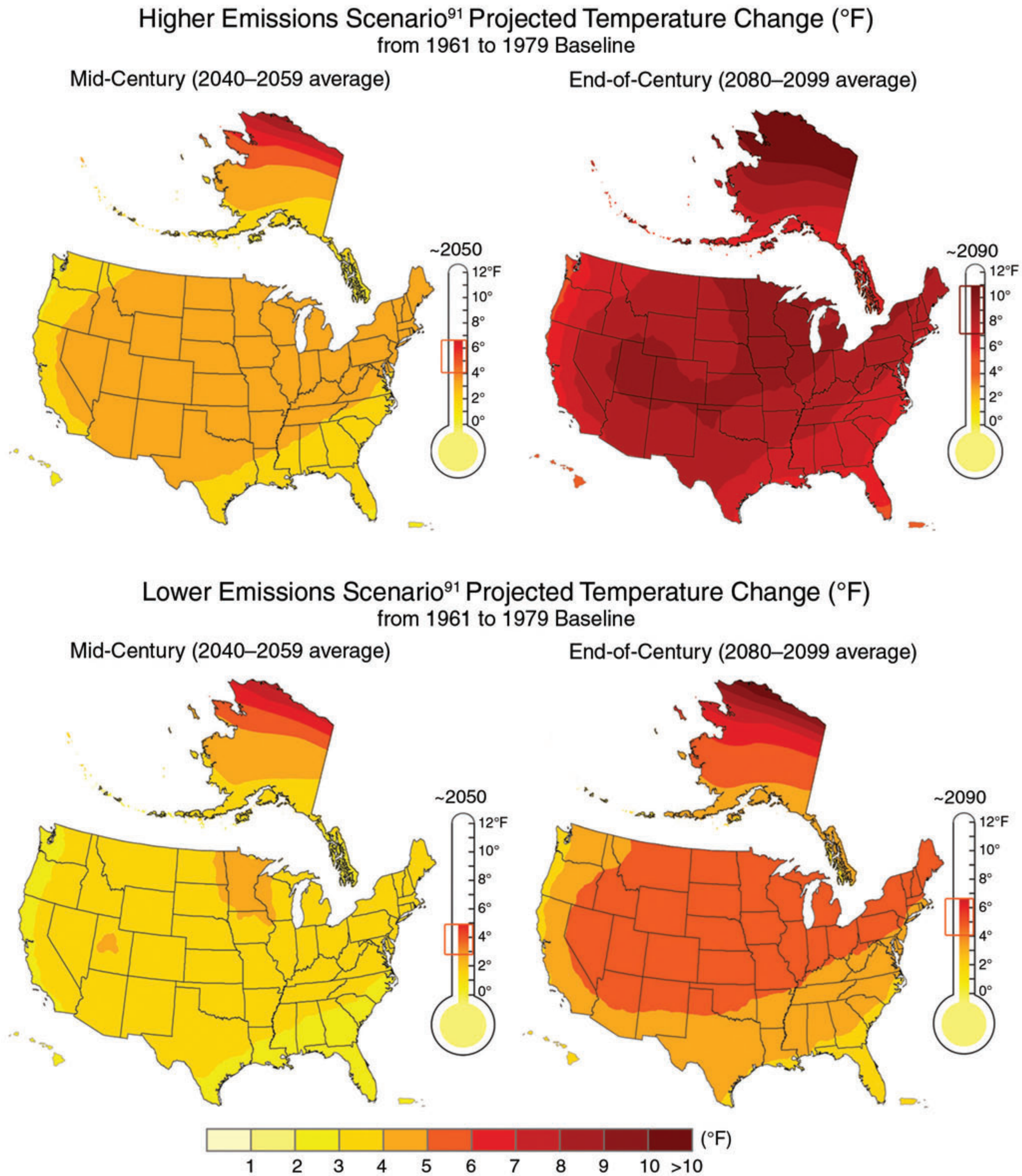

Figure 1.4. Projections of future temperature from 16 of the Coupled Model Intercomparison Project climate models. The maps feature a higher and lower greenhouse gas emission scenario. Brackets on the thermometers represent likely ranges of model predictions (USGCRP 2009).

pacts are expected to continue for the next thousand years (Solomon et al. 2009)—with the magnitude of the eventual changes depending on GHG emission rates and concentrations in the atmosphere.

In general, increases in global temperatures will intensify the Earth's hydrologic cycle by increasing the atmospheric humidity and altering atmospheric circulation patterns. The end result, confirmed by observations, is that the amount of precipitation as well as its intensity and frequency are all being altered, and these effects vary dramatically in different regions. Precipitation amounts have already increased 5% in the past 50 years across the United States, especially in the northeastern United States (USGCRP 2009), and the IPCC presents results showing this is occurring in many areas globally. In contrast, other regions have experienced little change in precipitation, while the Southwest has become drier. These patterns are expected to intensify in the future (Figure 1.5). Deeper incursions of warm, humid air from the south are expected to lead to increased precipitation further north than has occurred in the past. Continued greater drought is expected for the southwestern quadrant of North America due to warming and declining annual precipitation (Seeger and Vecchi 2010; Wang 2005).

Climate Change and Society

Although the vast majority of scientists agrees that climate change is underway and is being driven largely by human activities, how quickly it will happen and its detailed consequences at regional and local scales remain uncertain. Nevertheless, the people of the world are moving into an uncertain and changing climate regime, with consequences that are likely to differ regionally in severity (Somers 2010). Increased incidences of violent weather, warmer temperatures, retreating glaciers, thawing permafrost, and increased pest incidences are already having significant impacts on water, energy, transportation, agriculture, ecosystems, and health.

A particular concern is water resources, including increased precipitation intensity, altered seasonality of precipitation, increased drought, and diminished water quality, all of which will likely be manifested differently in different regions of the nation. Declines in snow pack and glaciers in the United States and around the world are altering water supply seasonality and challenging agricultural and municipal water activities. Sea level rise and storm surges are a particular concern for coastal areas, with consequences for coastal development as well as transportation and energy infrastructure. Human, plant, and livestock health issues are a concern, including those arising from increased heat stress, waterborne diseases, pest incidence, poor air quality, extreme weather events, and widening spread of diseases (Hatfield et al. 2008).

As a result of the mounting evidence, societal organizations from local communities to regions and countries are implementing legislation and adopting practices to mitigate net GHG emissions. The UNFCCC continues deliberations on international climate change protocols and agreements with many recent and scheduled meetings.

Agriculture's Greenhouse Gas Emissions

While increases in atmospheric GHG concentration have been dominated by fossil fuel burning, an important secondary source has been from deforestation and associated land use change as well as from many agricultural practices. As a result, atmospheric CO_2 concentrations have increased from approximately 280 ppmv (parts per million by volume) at the beginning of the industrial era to approximately 390 ppmv today. In recent decades, approximately 80% of the CO_2 emissions globally have been from fossil fuel burning and 20% from deforestation (Forster et al. 2007; USGCRP 2009). Agriculture contributes roughly half of the total anthropogenic emissions of the other two main GHGs, N_2O and CH_4 (Del Grosso et al. 2005; Denman et al. 2007; Smith and Conen 2004; USEPA 2011; USGCRP 2009). The largest sources of non-CO_2 agricultural GHGs are N_2O emissions from soils and CH_4 from enteric fermentation. Globally, agriculture accounts for 10–12% of the total GHG emissions (IPCC 2007a). In the United States, agricultural GHG emissions are over 6% of total U.S. emissions (USEPA 2011). Thus, agricultural emissions of GHG are a significant component of our nation's and the world's GHG emissions.

In addition to contributing to the GHG problem, agriculture is also directly affected by climate change, rising CO_2, rising temperature, altered precipitation, extreme events, and sea level rise. Although the focus of this document is on carbon sequestration and GHG fluxes in agriculture, any recommended management practices for mitigating GHG emissions must consider a host of other contingencies that determine the economics and sustainability of the practice. An important consideration of relevance to this report will be how agriculture responds and can adapt to climate change. The remainder of this chapter will thus briefly

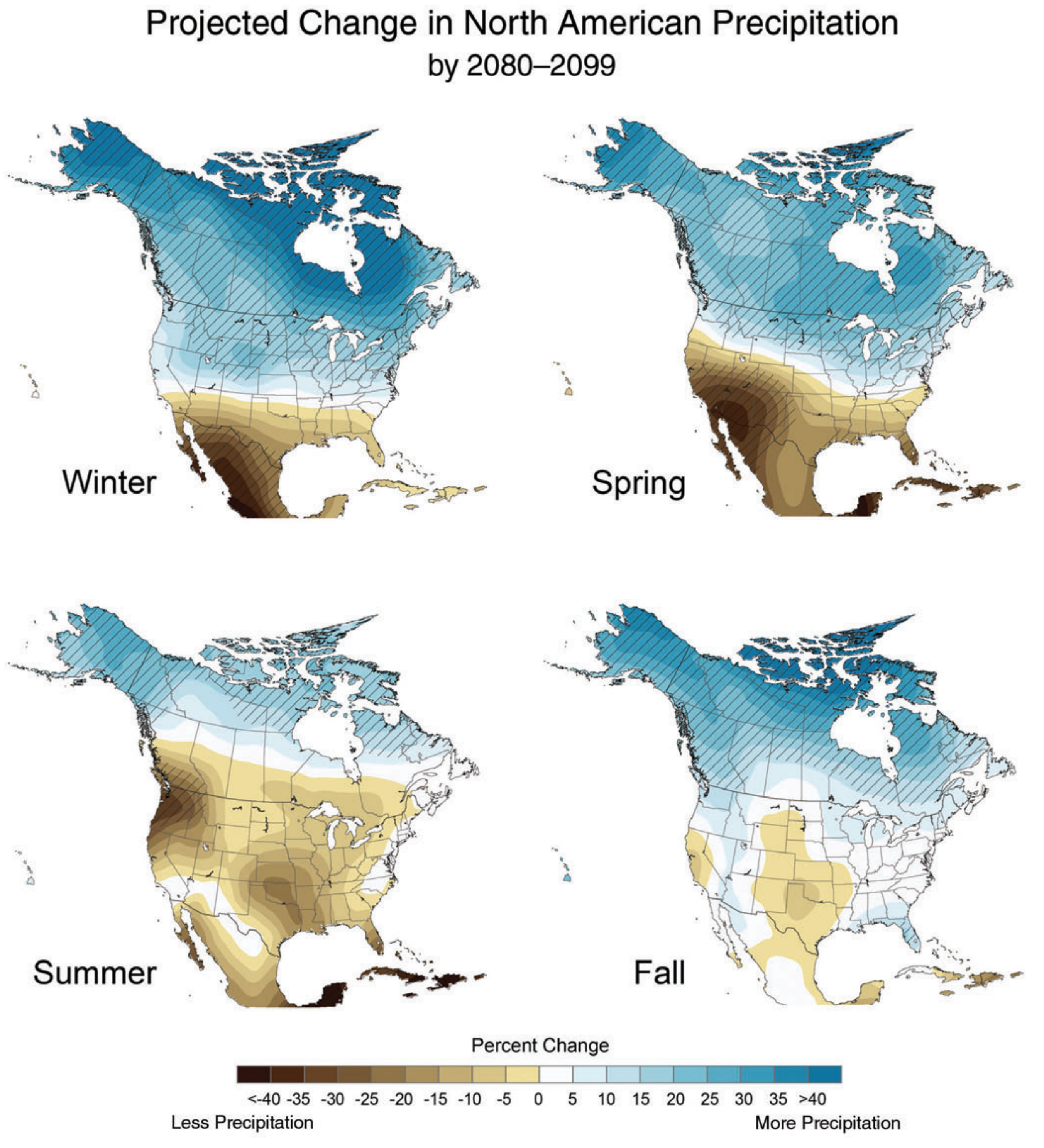

Figure 1.5. **Projected future changes in precipitation relative to the recent past as simulated by 15 climate models. Simulations are the late 21st century, under a higher emissions scenario. Confidence in the projected changes is highest in the hatched areas (USGCRP 2009).**

address the current understanding of how rising CO_2 and climate change are affecting agriculture.

Climate Change and Agriculture

Agriculture has always contended with a variable climate, but the changes presently underway driven by human-caused increases in GHG emissions are proceeding at a rate that will rapidly propel agriculture into new environmental circumstances (Williams and Jackson 2007). Warmer temperatures and changing precipitation patterns plus increasing incidence of extreme events will eventually alter the environment of many world regions to completely novel conditions. Concentrations of the GHG CO_2, which has a direct effect on plant physiology, have already increased to levels not experienced in well over 800,000 years (Jansen et al. 2007). In natural or semi-natural agroecosystems like rangelands and forests, these changes are likely already causing species shifts that will have profound impacts on biogeochemistry and land use (Morgan et al. 2008), affecting water and nutrient cycling, net primary production, and livestock/native fauna responses. In more intensive cropping and horticultural systems, climate change will alter hardiness zone classifications for plant species and will require both the adaptation of agricultural systems and the development of new germplasm to deal with these new environmental conditions.

Northward shifts of cropping patterns are being observed. Crops such as wheat, corn, and cotton are already being genetically adapted to grow under a wide variety of environments. Adaptability zones for warm-season, C_4 pasture grasses like coastal bermudagrass may move north in response to warming, although cool-season, C_3 grasses may benefit from the higher CO_2 concentrations. Changes in precipitation patterns and/or seasonality of water availability from irrigation stores in high mountain glaciers, snow packs, and reservoirs will affect local agricultural practices and the deployment of particular crops and cultivars. Ainsworth and colleagues (2008) argue that genetic modification of crops to optimize their direct responses to rising atmospheric CO_2 will be important in the success of future cropping systems.

Basic Plant Response to Carbon Dioxide and Climate Change

Carbon dioxide is a substrate for photosynthesis, and as such, increases in the concentrations of ambient CO_2 can stimulate photosynthesis (Pearcy and Ehleringer 1984). Atmospheric concentrations of CO_2 have been steadily rising since the industrial era, and plant physiologists have undertaken research to understand the implications of these changes for plants. They have learned that continued increases in CO_2 will stimulate photosynthesis and have potentially positive effects on plant growth, but that plants differ in their sensitivity to CO_2. For instance, photosynthesis in plants possessing the C_3 photosynthetic pathway (cool season grasses; most woody plants; most important crop plants in the United States) is not yet CO_2-saturated, and continued increases in CO_2 should lead to higher photosynthesis, likely translating to high plant productivity. The CO_2 response curve of photosynthesis for C_3 plants, however, indicates that such direct photosynthetic benefits will decline as CO_2 concentrations continue to rise and approach saturation for the carboxylating enzyme system that fixes CO_2 in green plants (Ehleringer, Cerling, and Helliker 1997).

In contrast, photosynthesis in plants with the C_4 photosynthetic pathway (warm-season grasses; some crop and weedy species; mostly herbaceous vegetation) is essentially CO_2-saturated at present-day CO_2 concentrations, so further increases in CO_2 are not expected to have much effect on photosynthesis. Other physiological differences in plants that can cause different sensitivities to CO_2 (e.g., N fixation, intrinsically fast- vs. slow-growing plants) have been useful, but certainly not perfect in predicting species-based differences in sensitivity to CO_2. Complicated combinations of plant and environmental traits sometimes obscure these functional group bases for predicting species responses to CO_2 (Polley, Morgan, and Fay 2011).

Carbon dioxide has another fundamental and important effect on plant physiology. Increases in CO_2 cause stomata to partially close (Hatfield et al. 2008; Wand et al. 1999). Stomata are the pores on plant leaves that allow the transfer of CO_2 and other gases between the leaf and atmosphere, and as such are critical regulators of plant water loss and photosynthesis. Stomatal sensitivity to CO_2 seems to operate in almost all herbaceous plants, with no apparent different sensitivities between C_3 and C_4 photosynthetic types (Wand et al. 1999). This closure can be a very positive plant response in that it reduces transpirational water losses and improves plant water use efficiency (Hatfield et al. 2008; Leakey 2009). In dry agro-ecosystems, this CO_2-induced stomatal closure may be more important than the direct photosynthetic response (Morgan et al. 2004, 2011).

Temperature has many impacts on agriculture, but two of the most important are that it regulates the rates at which biochemically driven reactions proceed and it affects the exchanges of energy and matter between agro-ecosystems and their environments. Both the biochemically driven and energy aspects of warming can elicit positive and negative effects on plant production.

Plant species have different *cardinal temperatures* (critical temperature range) that reflect the adaptation of their life cycle development and growth to the thermal environment (Hatfield et al. 2008). A minimum base temperature is required for plant activity; increases in temperature will eventually reach an optimal temperature that maximizes growth or yield, beyond which further increases in temperature reduce activity, growth, or yield (Table 1.1). The distribution of present-day recommended crop species and cultivars, as well as the natural species that make up native plant communities, reflect these cardinal temperatures. Global warming will continue shifting the zones for crop species and cultivars, and it will cause changes in native plant communities toward species better adapted to warmer temperatures. One concern with native systems is whether or not species and/or ecosystems will adapt quickly enough to move. Inability to do so is predicted to lead to massive species extinctions (Millennium Ecosystem Assessment 2005).

Warming also affects agriculture by extending the growing season, thereby potentially increasing annual productivity. The extent to which plants can take advantage of extended growing seasons will depend on how well their developmental life cycle meshes with the new thermal regime and the availability of essential resources to support continued plant growth. For instance, in a very dry climate, extending the length of the growing season may have little advantage if annual productivity is driven more by water supply than temperature. On the other hand, a warming environment with a longer growing season is likely to enhance productivity in regions where cold temperatures currently limit growth, like high-altitude or high-latitude regions. Developmental changes in crops may be required to optimize grain yield in warmer, longer growing seasons (Egli 2011).

A related issue for many fruit and nut crops is a winter chill requirement needed for optimal flowering and fruit set the following spring and summer (Westwood 1993). This aspect of plant adaptation to the thermal environment affects the optimal growing zones for many such woody perennial fruit and nut crops.

A potentially negative effect of warming will be increased incidence of drought. All else being equal, warming induces greater evapotranspiration, which leads to desiccation (Wang 2005). Increased water use efficiency from higher CO_2 may offset somewhat drier conditions (Hatfield et al. 2008), although there is still considerable question of how significant that CO_2-based offset may be (Frelich and Reich 2010; Seager and Vecchi 2010). Recent technological advances in simulating the combined effects of warming (Kimball et al. 2008) and CO_2 enrichment (Miglietta et al. 2001) in realistic field environments (Morgan et al. 2011) are

Table 1.1. Cardinal temperatures (°C) for economically significant crops[a]

Crop	Base Temp Veg	Opt Temp Veg	Base Temp Repro	Opt Temp Repro	Opt Temp Range Veg Prod	Opt Temp Range Reprod Yield	Failure Temp Reprod Yield
Maize	8	34	8	34		18–22	35
Soybean	7	30	6	26	25–37	22–24	39
Wheat	0	26	1	26	20–30	15	34
Rice	8	36	8	33	33	23–26	35–36
Sorghum	8	34	8	31	26–34	25	35
Cotton	14	37	14	28–30	34	25–26	35
Peanut	10	>30	11	29–33	31–35	20–26	39
Bean					23	23–24	32
Tomato	7	22	7	22		22–25	30

[a]Data include base and optimal temperatures for vegetative growth, reproductive development, optimal temperature range for vegetative biomass and maximum grain yield, and failure (ceiling) temperature at which grain yield fails to zero yield (Hatfield et al. 2008).

helping scientists explore such complex interactions between climate change and rising CO_2 (Figure 1.6).

The intensification of the hydraulic cycle that occurs with warming, leading to more intense precipitation dynamics with some regions receiving more, others less, annual rainfall, has already been discussed (USGCRP 2009). Problems from more intense, heavy downpours include increased periods of inter-event water stress, delayed planting, field flooding, and decreased crop quality, all of which have serious negative economic impacts on farming (USGCRP 2009). In other regions, like the Southwest, less frequent rainfall combined with warmer temperatures is predicted to increase drought (Seager and Vecchi 2010; Seager et al. 2007; Wang 2005). Finally, warming trends may have profound impacts on pests and diseases. For instance, warming in North America has been linked to the infestation of the mountain pine beetle in the Rocky Mountain region (Crozier and Dwyer 2006).

The outbreak of plant disease and pest incidence should be favored by increased incidences of climate extremes, and disease analyses suggest that recent warming trends are resulting in the movement of diseases from low to mid-latitudes (Easterling et al. 2007). For instance, computer simulation models suggest bluetongue, which affects mostly sheep, will eventually spread north to the mid-latitudes from the tropics (Anonymous 2006; Van Wuijckhuise et al. 2006). In addition to temperature, rising CO_2 and altered precipitation patterns may all affect disease, but so far investigations have focused solely on one global change factor (Easterling et al. 2007).

Figure 1.6. **USDA–Agricultural Research Service (USDA–ARS) scientists evaluate how combined CO_2 enrichment and infrared warming are affecting the microclimate and growth of prairie grasses and invasive weeds in a northern mixed-grass prairie at the High Plains Grasslands Research Station (latitude 41°11´N, longitude 104°54´W) near Cheyenne, Wyoming. This Prairie Heating and CO_2 Enrichment Experiment releases CO_2 from tubes surrounding the plots to maintain ambient CO_2 at 600 ppmv, and infrared heating above the plots warms them 1.5°C during the daytime and 3°C during the night. (Photo courtesy of Stephen Ausmus, USDA–ARS photographer.)**

2 Science of Greenhouse Gas Emissions from Agriculture

Processes, Sources, and Sinks of Carbon Dioxide, Nitrous Oxide, and Methane and the Drivers

Greenhouse gases, often called trace gases, are present in the atmosphere in small concentrations. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are the major greenhouse gases (GHGs) that are directly affected by human agricultural activities and are of substantial concern for global warming. Each of these gases, in addition to having emissions related directly to industrial activity, has significant components related to natural biogeochemical cycles. The biogeochemical cycles involving CO_2, N_2O, and CH_4 can be manipulated directly by human activity, thus providing options for influencing atmospheric concentrations of these major trace gases. The important details of their global dynamics are summarized in the next three sections.

Carbon Dioxide

Globally, there are five large global carbon (C) pools (Lal 2006): an oceanic pool estimated at 38,000 petagrams (Pg; 10^{15} g) C; a geologic pool estimated at 5,000 Pg C; a pedologic pool of soil C composed of 1,500 Pg of soil organic carbon (SOC) and 950 Pg of soil inorganic carbon (SIC); an atmospheric pool estimated at 800 Pg C and increasing at the current annual rate of ~4 Pg C (IPCC 2007a); and a plant pool of 550 Pg (Figure 2.1) with perhaps an additional 60 Pg of detritus material (Lal 2004). The increase in atmospheric CO_2, the main (GHG, is driven by the ~8 Pg per year (yr^{-1}) of C emission from fossil fuels and industrial activity. Also, there is an additional ~0.9 Pg yr^{-1} of C emissions from deforestation and land use change (Kelly 2008). The atmospheric C pool increased by 3.3 Pg yr^{-1} during the 1980s, 3.2 Pg yr^{-1} during the 1990s, and 4.1 Pg yr^{-1} between 2000 and 2005 (IPCC 2007a). Thus only about one-half of the CO_2 released from fossil fuels remains in the atmosphere (4 Pg out of 8 Pg) and the remainder is taken up by terrestrial ecosystems and the oceans.

Terrestrial absorption is the relatively small difference between large annual flows of C exchanged between terrestrial ecosystems and the atmosphere with around 120 Pg C taken up by plant photosynthesis each year and an approximately equal return flow to the atmosphere from plant and soil respiration. The average net difference in the terrestrial C pools is currently about 2 Pg C yr^{-1} with another ~2 Pg C yr^{-1} being taken up by the oceans (IPCC 2007a). Emissions from fossil fuel are essentially irreversible, with the terrestrial sink serving as part of an active

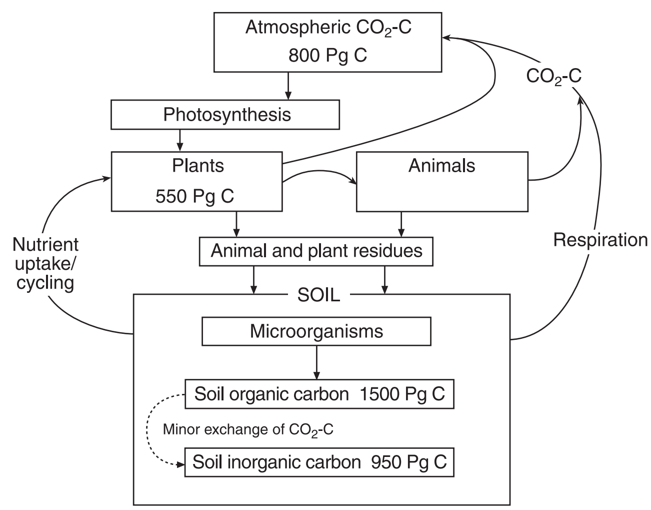

Figure 2.1. The Terrestrial Carbon Cycle. Inputs of carbon (C) into the soil organic carbon (SOC) pool originate from the fixation of atmospheric CO_2-C through photosynthesis by plants into simple sugars, and subsequently into the more complex materials (i.e., cellulose and lignin), eventually deposited in their leaves, stems, and roots. Plant material and its organic C can be consumed by animals or become humified into soil organic matter, which contains SOC, through the action of microorganisms. Carbon storage as SOC is controlled by the soil environment and the quality of the organic matter in which the carbon resides. Decomposition is the biological conversion of organic matter into more oxidized constituents, including CO_2, which is released back to the atmosphere. Decomposition rates are affected by soil structure and by soil temperature and moisture conditions (Morgan et al. 2010).

biological cycle that can potentially store some of the increased atmospheric C. During the next few decades, however, the C stored in the biological sinks is vulnerable to return to the atmosphere as natural or man-made disturbances can cause SOC to oxidize (Kunkel, Bromirski, and Lal 2004). Thus, terrestrial sinks are best viewed as mid-term reservoirs that may not be permanent offsets to the emissions from fossil fuels. Nevertheless they are important in that they can buy valuable time to reduce total emissions.

Plants are vital to capturing atmospheric CO_2-C, and terrestrial plants are estimated to contain 550 Pg C (Figure 2.1). The two major soil C stores are SOC and SIC, with estimates of both exceeding the total C in the atmosphere (Figure 2.1). Recognizing that the processes, although connected indirectly, are quite different, the changes in the SOC pool or in the SIC pool are related to the amount of C input minus the amount of C output (Equations 2.1, 2.2.).

$$\Delta \text{ SOC} = \text{C inputs} - \text{C outputs} \quad (2.1)$$

$$\Delta \text{ SIC} = \text{C inputs} - \text{C outputs} \quad (2.2)$$

Inputs of C into the SOC pool originate from organic forms of C that result from the fixation of atmospheric CO_2-C through photosynthesis by plants into simple sugars and subsequently into the more complex materials (i.e., cellulose and lignin) contained in their leaves, stems, and roots. Plant material and its organic C can be consumed by animals or become humified into soil organic matter (SOM) through the action of microorganisms. Some plant tissues persist for only a brief period before being shed and decomposed. For example, fine plant roots may last only a few weeks and deciduous leaves less than a year. Other tissues such as wood can, depending on forest type and disturbance frequency, persist for several decades or even centuries.

Carbon storage as SOC is controlled by the soil environment and the quality of the SOM in which the carbon resides. Decomposition is the biological conversion of SOM into more oxidized constituents, including CO_2. Soil organisms return most of the C in dead plant tissues to the atmosphere as CO_2. A smaller portion of this decomposing material is *humified*, or converted by soil organisms into more stable organic compounds, which are more difficult to decompose because of chemical resistance or physical protection by soil minerals. Humic compounds can remain in soils for hundreds to thousands of years before being converted into CO_2. As a result, most C in the terrestrial system is found in soils, not in living plant tissues.

Decomposition rates are affected by soil structure and by soil temperature and moisture conditions. Soil structure affects microbial access to oxygen, particularly in well-aggregated soils. Organic matter can be locked inside aggregates where oxygen limits microbial activity, so decomposition is slowed and carbon accumulates (Follett, Paul, and Pruessner 2007; Follett et al. 2009a). Respired CO_2 returns and again becomes part of the atmospheric pool (Figure 2.1). Carbon output (Equation 2.1) that decreases the amount of SOC results from losses caused by on-site soil erosion, leaching of organic C, and the decomposition of organic materials and respiration by plants, animals, and microorganisms.

Soils comprise the predominant C stock of agricultural ecosystems because plant biomass is either a relatively small component (as in perennial grasslands) or a seasonally transient component (as in annual croplands). Organic C contents of agricultural soils are typically on the order of 0.5 to 3% in the top 20 centimeters (cm), so for a typical soil bulk density of 1.3, this amounts to 13 to 78 tonnes C/hectare (ha). Organic C content tends to decline with soil depth (Follett 2009), and for most soils, 30 to 50% of the organic C to a 1-meter depth is contained in the top 20 cm. Soils with much higher C contents, including peat-derived (i.e., organic) soils, also are used for agricultural purposes, but to a limited extent. In the United States there are currently < 1 million ha of cultivated organic soils out of approximately 170 million ha of cropland (NRCS 2010).

Inorganic forms of C can also be an important component of the soil C cycle, although in most soils the net fluxes of inorganic C are small relative to the organic C. Carbon dioxide in the soil atmosphere equilibrates with dissolved CO_2, combining with water to form carbonic acid, bicarbonate, and carbonate (CO_3^{2-}) ions in the soil solution. Carbonate/bicarbonate ions can be leached from soils and enter surface and groundwater, and eventually the ocean. Inorganic C in the form of solid carbonate minerals (e.g., calcite) is present in soils derived from limestone and dolomite parent material. It also is formed as a secondary mineral in arid soils, through the reaction of calcium and magnesium with CO_3^{2-} ions. Carbonate minerals also are added to soils through certain types of agriculture liming and irrigation water. Depending on the balance between the dissolution and formation of CO_3^{2-} minerals, CO_3^{2-} (and hence CO_2) is either consumed or produced.

Nitrous Oxide

Nitrous oxide is a highly stable, long-lived trace gas found in the atmosphere at approximately 1/1,000th the concentration of CO_2. Anthropogenic activities have increased the Earth's annual emissions of N_2O by such that atmospheric N_2O levels have increased to 319 ± 0.12 ppb (parts per billion) in 2005, 18% higher than pre-industrial levels of 270 ± 7 ppb (Denman et al. 2007). Atmospheric concentrations of N_2O have been increasing approximately linearly in the past two decades at a rate of 0.26% yr^{-1} (Figure 2.2). Known sources of N_2O include bacteria in soils and sediments of both natural and managed ecosystems, industrial combustion, adipic and nitric acid manufacture, and biomass burning. Approximately half of N_2O emissions are natural in origin, the other half anthropogenic. Agriculture is the biggest anthropogenic N_2O source, with an estimated contribution of 2.8 teragrams (Tg) N yr^{-1} (Tg = 10^{12} g) of the anthropogenic sources (Del Grosso et al. 2005; Denman et al. 2007; Smith and Conen 2004). This compares with 6.6 Tg N_2O contributed annually by soils under natural vegetation, and 3.8 Tg N_2O contributed by oceans.

Nitrous oxide is produced in soils primarily by *denitrification* and *nitrification* (Figure 2.3). Both are microbial processes ubiquitous in most soils. Denitrification is the reduction of soil nitrate (NO_3^-) to N_2O and then to N_2 (nitrogen gas) by bacteria (Robertson 1999). Denitrification is an anaerobic process; thus, only if O_2 (oxygen in its molecular form) is unavailable will NO_3^- be denitrified to a nitrogenous gas. Because of this dependence on anaerobic conditions, it was once thought that denitrification was limited to saturated environments such as wetlands and lake sediments. Now it is known that substantial denitrification can take place in even well-structured upland soils; temporarily oxygen-depleted microsites are common inside soil aggregates (Sexstone, Parkin, and Tiedje 1986) and within SOM particles (Parkin 1987). If organic matter and NO_3^- also are present within these microsites, denitrifiers will produce N_2O and N_2.

The rate of N_2O production by denitrifiers thus depends on nitrate supply, the availability of oxidizable C (SOM), and the frequency and extent to which soil microsites are anaerobic. Because high soil moisture stimulates microbial respiration, restricts O_2 diffusion in soil, and increases NO_3^- diffusion to microsites, denitrification rates can be especially high after rainfall events and spring snowmelt. Although NO_3^- concentrations in some agricultural soils are low (e.g., in flooded rice), denitrification rates can be high because of coupled nitrification-denitrification. In these soils, nitrification in the oxidized rhizosphere creates NO_3^-, which quickly diffuses to adjacent anaerobic zones where it is denitrified to N_2O or N_2.

Denitrifiers are capable of reducing nitrate (through several intermediate compounds) to the principal gaseous end products of N_2O and N_2. As discussed earlier, N_2O is a powerful GHG found in

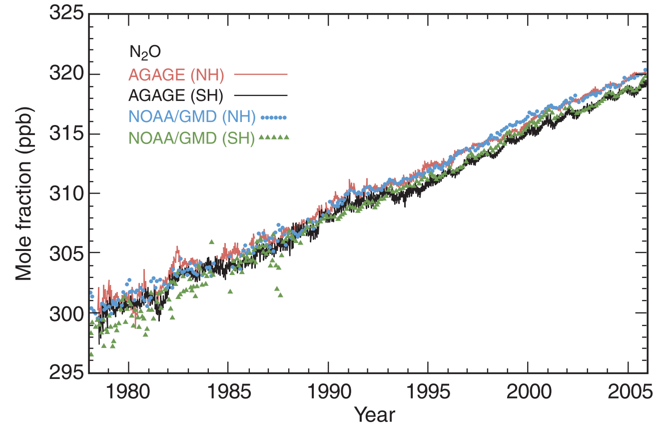

Figure 2.2. **Hemispheric monthly mean N_2O mole fractions (ppb) (crosses for the northern hemisphere, NH; triangles for the southern hemisphere, SH). Observations (in situ) of N_2O from the Atmospheric Lifetime Experiment (ALE) as well as the Global Atmospheric Gases Experiment (GAGE through the mid-1990s) and the Advanced GAGE (AGAGE since the mid-1990s) networks (Prinn et al. 2000, 2005) are shown with monthly standard deviations. Data from the National Oceanic and Atmospheric Administration (NOAA)/Global Monitoring Division (GMD) are shown without these standard deviations (Thompson et al. 2004). The general decrease in the variability of the measurements over time is due mainly to improved instrumental precision. The real signal emerges only in the last decade (Forster et al. 2007).**

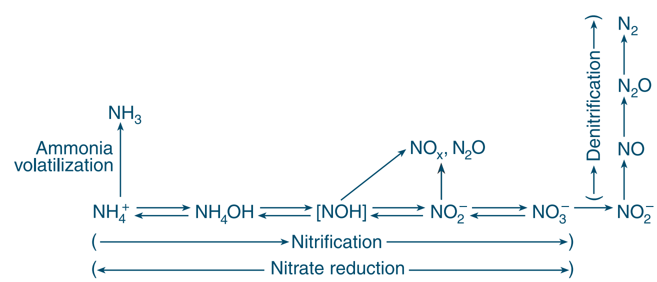

Figure 2.3. **Diagram of the major transformations of inorganic nitrogen (N) that can occur in soils, focusing on the major pathways of gaseous N losses, including N_2O. (Design courtesy of A. R. Mosier, USDA–ARS, Fort Collins, Colorado.)**

trace amounts in the atmosphere, whereas N_2 is the dominant gas present in the atmosphere and is not of concern with respect to the greenhouse effect. Thus, the relative amounts of N_2O versus N_2 produced by denitrification are of great interest. The proportion of end product emitted as N_2O is known as the *N_2O mole fraction* (N_2O:[N_2+N_2O]), which can range from 0 to 1 as a function of environmental conditions (Firestone and Davidson 1989) and microbial community composition (Cavigelli and Robertson 2000). Under highly anaerobic conditions, the N_2O produced tends to be further reduced to N_2, such that little N_2O is released to the atmosphere. In general, denitrification-derived N_2O formation will be favored during periods of low soil temperature, high but not saturating soil moisture (i.e., moderately anaerobic conditions), high NO_3^-, and low pH.

Nitrous oxide also can be formed during nitrification, the aerobic oxidation of soil ammonium (NH_4^+) to nitrite (NO_2^-) and then to NO_3^-. Intermediary compounds formed during NH_4^+ and NO_2^- oxidation can decompose chemically to gaseous N_2O, especially under acid conditions. Nitrifying bacteria also are known to use NO_2^- when O_2 is limiting, and such nitrifier denitrification (Poth and Focht 1985) may be the more common source of nitrifier N_2O (Firestone and Davidson 1989). Both nitrification and denitrification contribute to N_2O flux in soils of intermediate and low aeration (e.g., Panek et al. 2000; Stevens, Laughlin, and Hood 1997); in well-aerated soils with few anaerobic microsites, nitrifiers may be the dominant source of N_2O.

Spatial and temporal variability of N_2O flux can be extreme, making it difficult to quantify in most ecosystems. Spatially, N (nitrogen) gas fluxes are extremely heterogeneous on both field (e.g., Folorunso and Rolston 1984) and landscape (e.g., Groffman and Tiedje 1989) scales. To date, most field studies of N_2O have been based on small chamber-based methods with the chamber placed on the soil surface. It is not unusual for the coefficient of variation within individual plant communities to exceed 100% for chamber-based flux estimates, or for the specific types of plant communities or cropping systems to express different annual fluxes in different parts of a landscape. The use of tower-based micrometeorological methods, however, using laser techniques for measuring gas concentrations, is able to integrate over large areas, which can help reduce the impact of fine-scale spatial variability on field-scale flux estimates (Laville et al. 1999).

Temporal variability is no less important. Nitrous oxide fluxes can change quickly when environmental conditions change. Both natural events such as rainfall and human-induced events such as cultivation, fertilization, and other crop management practices can stimulate N_2O emissions markedly. In addition, Wagner-Riddle and colleagues (2010) observed N_2O emission events in Canadian cropland soil that occurred during winter and early spring when biological activity is typically low. The N_2O flux associated with the phase change during the main thaw event was an exponential function of the soil surface temperature increasing sharply when $T > 0°C$, but with smaller fluxes once T was $> 5°C$. The temperature response observed is consistent with the suggestion of a breakdown in the N_2O reduction process in the 0 to 5°C range, while the N_2O production enzymes are less affected by low temperature. When automated chambers have provided continual flux measurement (e.g., Ambus and Robertson 1998; Brumme and Beese 1992), order-of-magnitude flux changes occurred within a few hours.

Despite this variability, consistent differences among ecosystems have been documented. In both temperate and tropical regions, N_2O fluxes are greater from agricultural soils than from undisturbed soils under native vegetation (Keller et al. 1993; Mosier et al. 1991; Robertson, Paul, and Harwood 2000). Among all ecosystem types, fluxes tend to be smaller where soil NO_3^- availability is lower (Matson and Vitousek 1987; Robertson, Paul, and Harwood 2000; Smith et al. 1998). Differences in N_2O flux among different individual cropping practices are likely to be related to differential N availability.

Methane

Methane is a simple hydrocarbon compound that is most familiar as the main constituent of natural gas. Like the other trace gases, CH_4 is present naturally in the atmosphere in small amounts and is derived from a variety of natural and human-made sources. Currently, CH_4 accounts for approximately 18% of the radiative effects of increasing GHG concentrations (Forster et al. 2007).

Since the mid-1700s, the atmospheric concentration of CH_4 has increased by approximately 145%. Systematic observations of atmospheric CH_4 concentrations first were taken in the 1980s, but measurements of air trapped in ice cores now extend the record back in time (Figure 1.2). While atmospheric concentrations of the gas have increased by about 30% during the last 25 years, rates of increase have steadily declined from highs of around 1% yr^{-1} to lows near zero since the late 1990s (Figure 2.4). The global mean concentration in 2005 was 1,774 ppb (Forster et al. 2007). The reasons for the slower rate of increase in atmospheric CH_4 concentrations are still unclear, but they may involve both an increased rate of tropospheric destruction of CH_4, because of more hydroxyl radicals in the atmosphere, and a decreased growth rate of one or more of the sources of CH_4 emissions. Annual growth rates in CH_4 fluxes currently display tremendous variability, with rates as high as 14 ppb yr^{-1} in 1998 to less than zero in more recent years (Figure 2.4).

Total emissions of CH_4 to the atmosphere are approximately 500 to 600 Tg/yr, of which more than 60% are anthropogenic, or human-influenced, emissions (Table 2.1). Anthropogenic sources include energy (i.e., coal mining, well and pipeline leakage) and the remainder from biospheric sources. Agricultural activities, particularly rice cultivation and livestock, are major contributors to these biospheric sources, with annual emissions between the low 100s to 300 Tg CH_4 (Table 2.1). Other important sources include biomass burning, usually associated with deforestation and land conversion, as well as sewage-treatment facilities and landfills associated with urban populations. The recovery of CH_4 from waste streams (manure, sewage, or landfills) represents a potential energy source as well as a mitigation opportunity.

Methane is produced by microorganisms (Archaea) living in anaerobic environments. In agriculture, persistent anaerobic conditions mainly occur in flooded rice soils and in certain types of animal waste storage systems. Methane production also occurs within the digestive tracts of livestock, especially ruminants (e.g., cattle, sheep, buffalo, goats, and camels). These animals possess a large fore stomach, or rumen, in which plant materials are broken down through fermentation. Fermentation also generates CO_2 and hydrogen gas, the latter of which is used as an energy source by methanogenic microorganisms. Methanogens are strict anaerobes (i.e., functioning only in the absence of oxygen) whose substrates are limited to a few small molecules supplied as fermentation products released by other microbes. Most methanogens reduce CO_2 to CH_4 using acetate, formulate, or, sometimes, alcohol in their metabolism (Boone 1991).

Considerable CH_4 is emitted from the microbial decomposition of anaerobic livestock waste. The relative amount of CH_4 produced is determined by the waste-management system. When manure (some combination of urine and feces) is stored or treated in systems

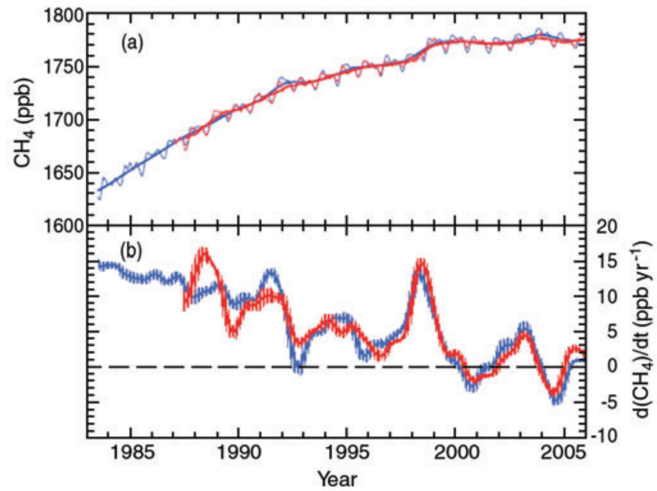

Figure 2.4. **Recent CH_4 concentrations and trends. (a) Time series of global CH_4 abundance mole fraction (in ppb) derived from surface sites operated by NOAA/GMD (blue lines) and AGAGE (red lines). The thinner lines show the CH_4 global averages, and the thicker lines are the deseasonalized global average trends from both networks. (b) Annual growth rate (ppb yr^{-1}) in global atmospheric CH_4 abundance from 1984 through the end of 2005 (NOAA/GMD, blue) and from 1988 to the end of 2005 (AGAGE, red). To derive the growth rates and their uncertainties for each month, a linear least squares method that takes account of the autocorrelation of residuals is used. This is applied to the deseasonalized global mean mole fractions from (a) for values six months before and after the current month. The vertical lines indicate ±2 standard-deviation uncertainties (95% confidence interval). One standard-deviation uncertainties lie between 0.1 and 1.4 ppb yr^{-1} for both AGAGE and NOAA/GMD data. Note that the differences between the AGAGE and NOAA/GMD calibration scales are determined through occasional intercomparisons (Forster et al. 2007).**

promoting anaerobic conditions (e.g., as a liquid in lagoons, ponds, tanks, or pits), CH_4 is produced from organic matter decomposition. When, on the other hand, manure is handled as a solid or deposited on grazinglands, it tends to decompose aerobically and produces little CH_4 (Safley et al. 1992; USEPA 1993, 2002). Usually, manures from animals on a high-quality diet have greater potential to generate CH_4 than manures from animals on a low-quality diet. The greatest emissions of CH_4 from animal manures tend to be associated with the most intensively managed animals, which also often use liquid manure storage facilities (e.g., dairy cattle).

In rice soils and in wastewater lagoons, metha-

Table 2.1. Sources of sinks and atmospheric budgets of CH_4 ($Tg/CH_4\ yr^{-1}$)[a] (Denman et al. 2007)

References	Indicative ^{13}C, 0/oo[b]	Hein et al., 1997[c]	Houweling et al., 2000[c]	Olivier et al., 2005	Wuebbles and Hayhoe, 2002	Scheehle et al., 2002	J. Wang et al., 2004[c]	Mikaloff Fletcher et al., 2004a[c]	Chen and Prinn, 2006[c]	TAR	AR4
Base Year		1983–1986		2000		1990	1994	1999	1996–2001	1998	2000–2004
Natural sources			**222**		**145**		**200**	**260**	**168**		
Wetlands	-58	231	163		100		176	231	145		
Termites	-70		20		20		20	29	23		
Ocean	-60		15		4						
Hydrates	-60				5		4				
Geological sources	-40		4		14						
Wild animals	-60		15								
Wildfires	-25		5		2						
Anthropogenic sources		**361**		**320**	**358**	**264**	**307**	**350**	**428**		
Energy						74	77				
Coal mining	-37	32		34	46			30	48[d]		
Gas, oil, industry	-44	68		64	60			52	36[e]		
Landfills and waste	-55	43		66	61	69	49	35			
Ruminants	-60	92		80	81	76	83	91	189[f]		
Rice agriculture	-63	83		39	60	31	57	54	112		
Biomass burning	-25	43			50	14	41	88	43[e]		
C_3 vegetation	-25			27							
C_4 vegetation	-12			9							
Total sources		**592**			**503**		**507**	**610**	**596**	**598**	**582**
Imbalance		+33								+22	+1
Sinks											
Soils	-18	26			30		34	30		30	30[g]
Tropospheric OH	-3.9	488			445		428	507		506	511[g]
Stratospheric loss		45			40		30	40		40	40[g]
Total sink		**559**			**515**		**492**	**577**		**576**	**581[g]**

[a] Table shows the best estimate values.

[b] Indicative ^{13}C values for sources are taken mainly from Mikaloff Fletcher et al. (2004). Entries for sinks are the fractionation (k_{13}/k_{12}-1), where k_n is the removal rate of $^{n}CH_4$; the fractionation for OH is taken from Saueressig et al. (2001) and that for the soil sink from Snover and Quay (2000) as the most recent determinations.

[c] Estimates from global inverse modeling (top-down method).

[d] Includes natural gas emissions.

[e] Biofuel emissions are included under Industry.

[f] Includes emissions from landfills and wastes.

[g] Numbers are increased by 1% from the TAR according to recalibration described in Chapter 2.

nogenesis occurs principally below the soil-water interface, where O_2 is depleted because of slow diffusion from surface waters and microbial respiration at the interface (Figure 2.5). Once formed, CH_4 can diffuse to the surface, rise to the surface entrained in bubbles, or, more importantly, be transported to the atmosphere through the rice plant via air channels (aerenchyma) within the plant that supply O_2 to the roots. This latter process is generally the most important emission mechanism and accounts for greater than 90% of total CH_4 emission from rice paddies (Cicerone, Shetter, and Delwiche 1983; Minami 1993; Nouchi, Mariko, and Aoki 1990; Seiler, Conrad, and Scharffe 1984).

Before entering the atmosphere, the CH_4 formed in soil can be oxidized by other *methanotrophic* bacteria, which use CH_4 as an energy source. But because methanotrophs require O_2, CH_4 oxidation occurs only in small bands at the soil-water interface and in the narrow zone around plant roots to which atmospheric O_2 is transported. During the course of the rice-growing season, a large portion of the CH_4 produced in flooded soil is oxidized before it can be released to the atmosphere (Sass et al. 1992; Schutz et al. 1989). Small amounts of CH_4 are dissolved in water and can be leached to groundwater. Thus, methane production in rice soils and other flooded environments is the net difference between CH_4 formation where O_2 is absent and CH_4 consumption where O_2 is available.

In nonflooded soil, CH_4 consumption dominates over whatever small amount of methanogenesis might be occurring in anaerobic microsites. We now know that the methanotrophs found in most aerobic soils can consume atmospheric CH_4 actively (Knowles 1993). Methane uptake is controlled by the diffusion rate and the potential biological demand. Diffusion is regulated by physical factors, and biological demand is regulated by physical and chemical environments. Both biotic and abiotic factors can limit CH_4 uptake.

Methane consumption is suppressed by restricted diffusion in wet soil. As soil dries and diffusion rate increases, CH_4 consumption increases to a maximum. When soil becomes very dry, consumption rate falls again as moisture stress decreases biological demand. In very cold soils, biological activity is quite restricted and the diffusion potential is more than adequate to meet the biological demand for CH_4. Methane consumption in aerobic soils does not cease in winter, however, as shown by studies of snow-covered mountain soils (Sommerfeld, Mosier, and Musselman 1993) and frozen prairie soils (Mosier et al. 1991). As temperatures rise in spring, biotic activity increases and consumption rates eventually plateau at a diffusion-controlled maximum.

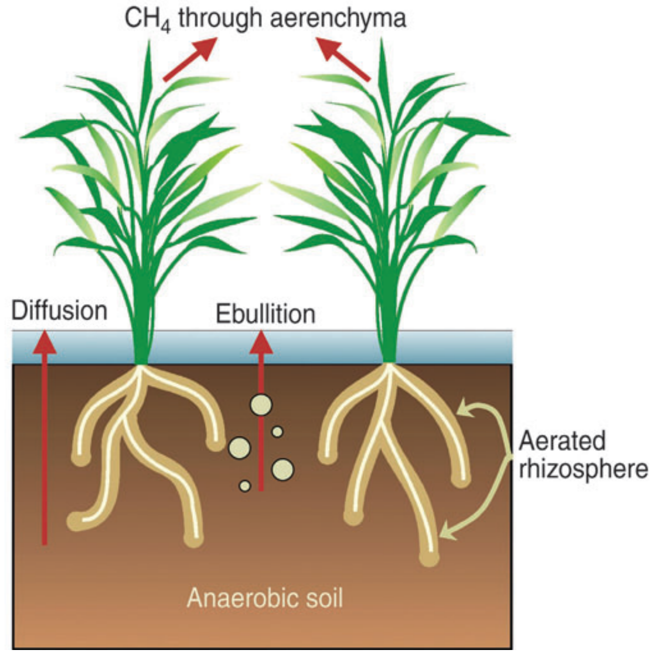

Figure 2.5. Pathways of CH_4 emissions from flooded soils such as under rice cultivation. (Drawing courtesy of A. Swan, Colorado State University.)

Methane consumption in aerobic soils has been recognized as a globally important sink for CH_4 only in the past decade or so, as measurable rates of CH_4 oxidation have been documented in a variety of upland environments, including agricultural soils (Keller et al. 1983; Megraw and Knowles 1987; Mosier et al. 1991; Robertson, Paul, and Harwood 2000; Steudler et al. 1989). Conversion of native soils to agriculture, however, has a major effect on the capacity of a soil to consume CH_4. Oxidation rates typically fall severalfold for reasons that are not well understood. Suggested mechanisms include the possibility that high NH_4^+ availability in agricultural soils competitively inhibits the intracellular enzymes oxidizing CH_4 (Steudler et al. 1989). Soil structure C in particular—its ability to impede or to promote diffusion of CH_4, O_2, and other gases between microsites and the atmosphere—also seems to play a role. Most likely, a combination of factors leads to suppression of CH_4 consumption in agricultural soils. Tillage, for example, destroys soil aggregates, which results in fewer aerobic/anaerobic interfaces in soil and impedes drainage as it diminishes soil porosity. Simultaneously, tillage increases SOM oxidation, which together with N fertilizers increases available NH_4^+ in the soil solution.

Measuring Carbon Dioxide, Nitrous Oxide, and Methane Fluxes

There are two main methods for measuring trace gas fluxes over crop and soil surfaces in agro-ecosystems: the static chamber and the micrometeorological methods. The chamber method involves the deployment of small chambers on the soil surface for a period of no more than 60 minutes (Parkin and Venterea 2010). This technique is often used for the concurrent measurement of CO_2, N_2O, and CH_4 fluxes, and it is particularly suitable for small plot work. The method involves installing permanent chamber anchors with minimum anchor or collar height to reduce micro-environment perturbations (Figure 2.6). During chamber deployment, samples of the chamber headspace gas are removed (Figure 2.7) at regular intervals and stored for later analysis by gas chromatography. Specific recommendations on chamber design, gas sampling and analysis, and flux calculations are provided by Parkin and Venterea (2010).

In addition to the static chamber method, it may be possible or desirable at some sites to use micrometeorological methods to characterize net ecosystem exchange (NEE) of CO_2 and evapotranspiration of water (H_2O). These measurements may be done with a Bowen Ratio or gradient system, but the most common approach currently used is eddy covariance (Figure 2.8) (Baker and Kimball 2010). In contrast to chamber methodology, micrometeorological methods require a large field for correct operation, and they measure the net fluxes of CO_2 and H_2O from soils and plants, i.e., the entire agro-ecosystem, not just soil fluxes as with the chamber methodology. Eddy covariance requires fast measurement of fluctuations in vertical wind speed and concentration of the scalar of interest. Both CO_2 and H_2O fluxes can be measured by open- or closed-path infrared gas analyzers. Closed-path instruments require a pump and some additional signal processing but are less affected by precipitation. To capture all eddies contributing

Figure 2.6. Example of a polyvinyl chloride (PVC) chamber used by GRACEnet research for measuring emissions/exchanges of greenhouse gases from the soil. The soil anchor (bottom left) is inserted permanently to a near surface level (right side). The chamber (top left) is attached to the anchor during the time that air samples are being collected using a syringe. Gas samples are subsequently analyzed using a gas chromatograph (Parkin and Venterea 2010).

Figure 2.7. Example of a rectangular chamber being used by ARS technicians to measure emissions of greenhouse gases from soil in a field of corn at the Ft. Collins, Colorado, research location. (Photo courtesy of Stephen Ausmus [D1535-8].)

Figure 2.8. Eddy covariance system measuring CO_2 and H_2O fluxes above an irrigated corn field. The net upward and downward components of wind are measured by sonic anemometers at the extreme right of the apparatus (Baker and Kimball 2010).

to the flux, signals should be sampled at 10 hertz or better. Eddy covariance data should be processed in 30- to 60-minute blocks. Smaller intervals may miss some low-frequency contributions to the flux, whereas larger intervals risk violation of stationarity considerations (Baldocchi, Hicks, and Meyers 1988). Recent advances in micrometeorological methods to measure N_2O and CH_4 now include improved laser devices (e.g., tunable diode lasers) with fast response time and high sensitivity, which can be deployed on towers or aircraft (Desjardins et al. 2010; Laville et al. 1999).

The concept of a global warming potential (GWP) has been developed to account for differences in the ability of each greenhouse gas to trap heat in the atmosphere relative to another gas. The GWP of a greenhouse gas, expressed in units of CO_2 equivalents (CO_2 Eq), is defined as the ratio of the time-integrated radiative forcing from the instantaneous release of 1 kg (kilogram) of a trace substance relative to that of 1 kg of CO_2 (IPCC 2007a). While any time period can be selected, the 100-year GWP is recommended by the IPCC and used by the United States for policymaking and reporting purposes where the GWP of CO_2 is considered to be 1 during a 100-year time horizon. The corresponding GWPs for N_2O and CH_4 as referenced to CO_2 were considered by IPCC (1996) to be 310 and 21 CO_2 Eq, respectively, and these values are still used for national GHG inventory reporting. More recently, Forster and colleagues (2007) report GWPs for N_2O and CH_4 during a 100-year time horizon to be 298 and 25 CO_2 Eq, respectively.

To report and interpret trace gas fluxes conducted at different sites and agro-ecosystems, standardized protocols are recommended not only in the measurement of fluxes, but also in the quantification of plant and soil attributes that ultimately determine those fluxes. The use of standardized protocols greatly improves the opportunities for cross-location, regional, and institutional research. Such uniformly collected data enhances our ability to test, improve, and validate research and models, all of which advances the science and its application by other researchers, producers, and policymakers. Liebig, Varvel, and Honeycutt (2010) recommend protocols to conduct soil sampling, processing, analyses, and archiving. Similarly, Johnson and Morgan (2010) provide plant sampling guidelines for both crop- and grazinglands for the interpretation of trace gas fluxes. Attributes like quality (includes C:N ratio), size of residue, and manner in which residue is incorporated into the soil are all important factors affecting soil conditions, microbial activity, and the resultant fluxes of trace gases.

Agricultural Greenhouse Gas Flux Estimates for the United States

The agriculture sector is reported to account for 503.9 Tg CO_2 Eq (USEPA 2011) and 501.2 Tg CO_2 Eq (USDA 2011) in 2008, or over 7% of total anthropogenic U.S. GHG emissions. In absolute terms, emissions from U.S. agriculture alone are equal to or greater than the total emissions (from all GHG sources) for many mid-sized European countries (e.g., Italy, France, Poland, Spain) (UNFCCC 2010a). Thus there is scope for significant emission reductions and C offsets through agricultural mitigation options in the United States, as detailed in subsequent chapters in this report.

United States agricultural emissions are dominated by a few main source categories (USDA 2011), namely, N_2O emissions (233 Tg CO_2 Eq) (largely associated with N management on crops and grazingland) and CH_4 emissions (185.8 Tg CO_2 Eq) (primarily from enteric fermentation by livestock and from manure management). In 2008, emissions of N_2O from agricultural soils were approximately 216 Tg CO_2 Eq and CH_4 emissions from enteric fermentation and manure management were 141 and 45 Tg CO_2 Eq, respectively (USDA 2011). Thus N_2O and CH_4 emissions make up about 80% of total agricultural emissions. Agriculture is the largest source of N_2O emissions and second largest source of CH_4 in the United States.

The CO_2 emissions from agriculture are largely confined to fossil fuel use for production (i.e., on-farm machinery, drying, irrigation) and CO_2 released from liming and urea amendments. Soil C on agricultural land (both cropland and grazinglands) constitutes a net sink, estimated at around 44 Tg CO_2 (for 2008), resulting from the net uptake of 79 Tg CO_2 on mineral soils and emissions of 35 Tg CO_2 from managed organic (e.g., peat) soils. Overall, this represents a net CO_2 removal from the atmosphere that offsets about 8% of total agricultural emissions. The main changes in land use and management since 1990 that have contributed to the soil C sink are cropland set aside in the *Conservation Reserve Program* (CRP), some increasing soil C stocks on grazingland, and some increasing soil C stocks due to long-term trends of reduced tillage intensity, crop residue production, and improved crop rotations (USEPA 2010).

Trends since 1990 show a slow growth of gross emissions from agriculture, equivalent to a rate of increase of approximately 0.2% on average since 2000 (Figure. 2.9). Considering total net emissions, however, the U.S. national GHG inventory also suggests

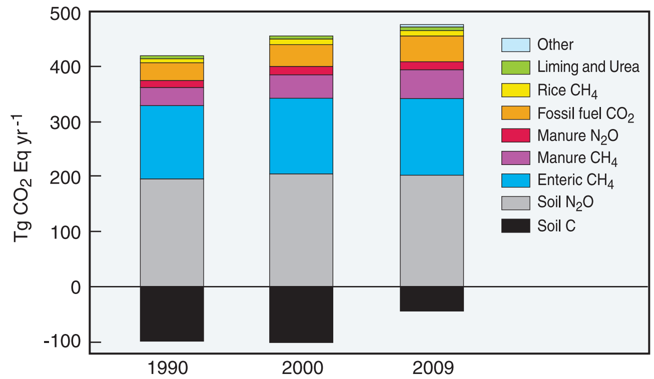

Figure 2.9. Greenhouse gas emissions, by source category, from U.S. agriculture (USEPA 2011).

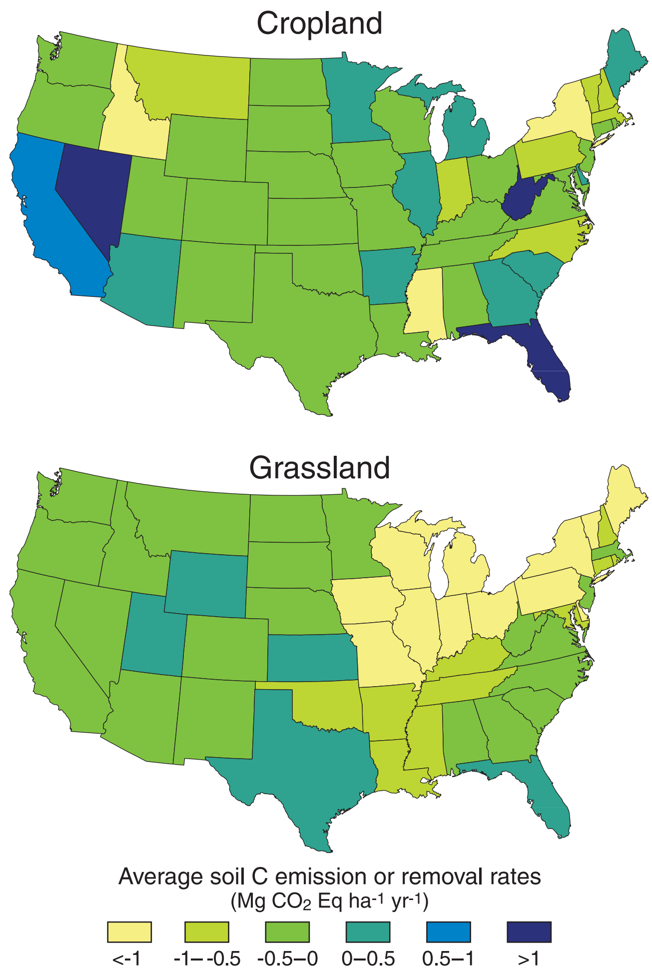

Figure 2.10. Soil C emissions or removals, displayed as the average per ha per year by state for total cropland area (top panel) and total grassland area (bottom panel), respectively. Note that emissions are given as positive values (as CO_2), while removals (i.e., increase in soil C stock) are given as negative values. Data are for the 2009 inventory year, derived from the U.S. national GHG inventory (USEPA 2011).

a reduction of approximately 0.3% per year in the rate of soil C sequestration since 1990. Recent factors that are likely contributing to the declining sink include leveling off of soil C increases in older CRP lands as well as conversion of CRP back to cropland and a leveling out (relative to post-1990) in the new adoption of no-till and reduced tillage[1].

For the two main land-based agricultural emissions sources, soil CO_2 and N_2O, rates vary geographically as a function of climate, soil properties (in particular, the occurrence of managed organic soils), and management system attributes (Figure 2.10). Mineral soils (i.e., nonorganic) are currently gaining carbon (i.e., acting as a sink) across most of the United States, with the highest average rates in the Midwest and Northern Plains regions (data not shown). For total cropland soils, however, high emission rates from managed organic (e.g., peat) soils—concentrated in Florida, California, and the Great Lakes region—are sufficient in some instances to offset gains on mineral soils, so that cropland soils overall are net emitters of CO_2 in several states (Figure 2.10). Grassland soils (i.e., under pasture and range) are also, on average, gaining soil C, with the highest rates in the more humid and subhumid parts of the country.

Current (2009) rates of C sequestration in pasture soils average >1 Mg (megagram) CO_2 ha^{-1} yr^{-1} (or a negative value, when expressed as an emission—see Figure 2.10) for much of the north-central and northeastern region, with somewhat lower rates of increase in states in the lower Mississippi basin. National inventory estimates show some states with low average net losses of soil C; in Florida, overall losses are largely due to emissions from managed grassland on organic soils. Much of the rangeland-dominated grassland areas of the western United States show either small average gains or losses in soil C. It should be noted that variability in interannual weather conditions in different parts of the country in a particular year (results shown in Figure 2.10 are for 2009, the most recent published results) affect the net soil C balance, particularly for rangeland (e.g., Derner and Schuman 2007). Therefore, some areas can shift from being a net emitter (during drought years) to being a net sink (during wetter years), largely as a result of climate variability.

[1] The impacts of recent changes (since 2007) in agricultural land use, however, are not fully reflected in the national GHG inventory because the most recent data on management practices are based on the 2007 National Resources Inventory data.

Nitrous oxide emissions also vary geographically, mainly driven by regional differences in climate and therefore crop productivity and N application rates. Based on computer modeling, the highest average rates of N_2O emissions are estimated to be mainly in the Corn Belt region, parts of the Northeast, and the intensively managed, high N input systems in California and Florida (Figure 2.11). These areas typically have the highest rates of N applied to cropland, and the more humid climates tend to create more favorable conditions for denitrification, one of the main pathways for soil N_2O loss. Average rates on cropland are lowest in regions dominated by semi-arid, low N input cropping systems in parts of the western United States (Figure 2.11). In addition, areas with substantial amounts of cropland set aside as CRP land, which is not fertilized, will have lower N_2O emissions compared to actively managed cropland.

Overall, grassland soils have lower rates of N_2O emissions, particularly in the western United States where rangeland, which is not fertilized, is the dominant managed grassland. Per hectare rates of N_2O are higher in the more mesic grassland area in the eastern United States, because of both higher N inputs and moister conditions. In parts of the mid-Atlantic region and Northeast, where a substantial part of the pastureland is more intensively managed with higher fertilizer N inputs, N_2O emission rates are as high as for cropland in the region, with several states showing average emissions of >1 Mg CO_2 Eq ha^{-1} yr^{-1}.

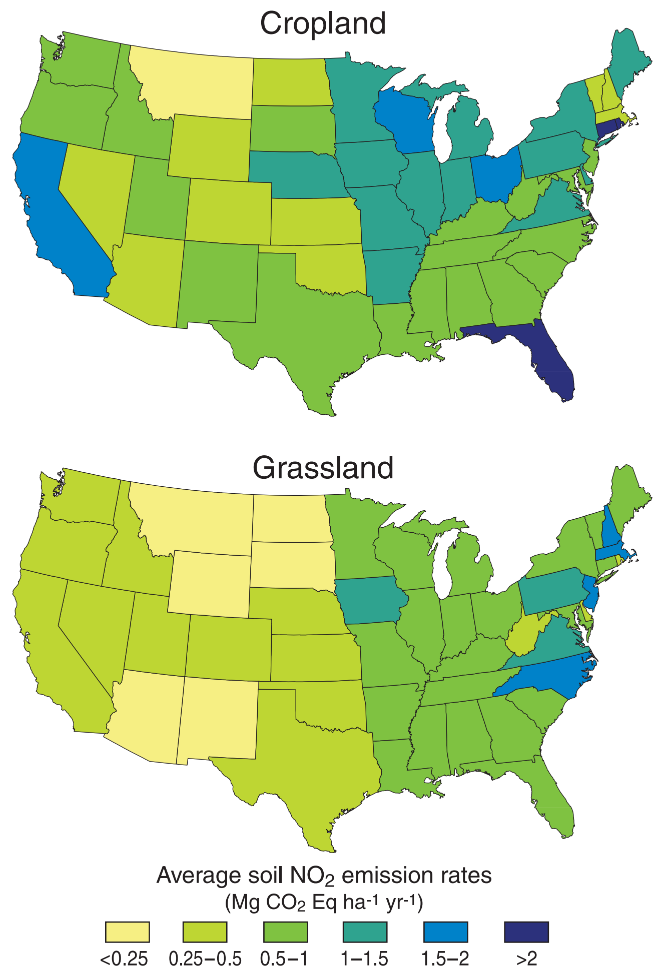

Figure 2.11. Soil N_2O emissions, displayed as the average per ha per year by state for total cropland area (top panel) and total grassland area (bottom panel), respectively. Values are expressed as CO_2 equivalents, assuming a GWP for N_2O of 310. Data are for the 2009 inventory year, derived from the U.S. national GHG inventory (USEPA 2011).

Feedbacks between Climate Change and Greenhouse Gas Emissions, and Implications for Future Emissions

Impacts of Rising Carbon Dioxide and Climate Change on Plant Productivity

Strategies to curb GHG emissions will be constrained by environmental conditions, including the impacts of climate change itself. Soil and plant responses to climate change will interact with management, affecting plant production and, ultimately, inputs of soil C. Rising atmospheric CO_2 enhances plant photosynthesis and water use efficiency, both of which potentially lead to increased production in cropping systems and on grazinglands (Brouder and Volenec 2008; Hatfield et al. 2008; Morgan et al. 2004; Runion et al. 2009). Predicted rising temperatures, and in some regions higher annual precipitation, may enhance agricultural production by extending the growing season and lowering the low-temperature limitations to growth in cool/cold environments at northern latitudes and high elevations (Brouder and Volenec 2008; Morgan et al. 2008). Assuming higher plant production leads to enhanced soil C inputs, these responses suggest that rising atmospheric CO_2 could enhance C sequestration (Runion et al. 2009). Continued or extreme warming, however, leads to desiccation and other secondary responses that will constrain CO_2-induced plant growth stimulation, and that may lead to decreased production in some regions of the United States (Arnone et al. 2008; Ciais et al. 2005; Saleska, Harte, and Torn 1999). This is likely to happen first in south-central and southwest portions of the United States where growing season temperatures are already high, especially in the Southwest where precipitation is expected to decline.

In more humid eastern states and the Pacific Northwest, plus Hawaii and Alaska, agricultural production may continue to be stimulated for a couple of decades. Declining availability of soil nutrients like N, however, may limit the long-term growth-enhancing benefit of CO_2 (Luo et al. 2004). In general, more intensive cropping and horticultural agriculture will respond by changing cultivars and species of plants, or by other management (irrigation, fertilization, cropping systems) that adapts the practices to the changing environment. Fewer options exist for extensively managed native systems like rangelands (Morgan et al. 2008), where climate change-induced species shifts are likely already underway.

Responses of Soil Carbon to Climate Change

The fate of soil C under the changing environment of climate change is uncertain and may not be linked simply to plant productivity (Heimann and Reichstein 2008). Soils do not respond directly to increases in atmospheric CO_2, but CO_2-induced changes in plant production potentially affect soil carbon through increased transfer of below-ground C into roots, rhizosphere exudates, and turnover of organic materials from the roots and from plant litter (Cheng 1999; Hoosbeek and Scarascia-Mugnozza 2009; Hungate et al. 1997b; Iversen, Ledford, and Norby 2008; Lichter et al. 2008). Although CO_2-enhanced growth can stimulate soil C sequestration via greater soil C input (Allen et al. 2006; Jastrow et al. 2005; Runion et al. 2009), CO_2-stimulated plant growth does not necessarily lead to higher rates of SOC sequestration in cropped (Peralta and Wander 2008) or native terrestrial ecosystems (Jasoni, Smith, and Arnone 2005; Niklaus et al. 2003; Parton et al. 2007; Stock et al. 2005). The inability of CO_2 to enhance soil C stores is more commonly observed in natural systems like grasslands and deserts where CO_2-induced reductions in critical soil nutrients like N may constrain the growth responses over time, leading to less CO_2-induced soil C inputs (Gill et al. 2006; Gruber and Galloway 2008: Luo et al. 2004; Pepper et al. 2005; van Groenigen et al. 2006).

Increases in CO_2 can affect the soil C cycle in ways that limit soil C sequestration, even in row crops like soybean where CO_2 enhances plant growth (Peralta and Wander 2008). Higher temperatures should lead to faster rates of SOM decomposition (Conant et al. 2008; Rustad et al. 2001) and decreased net C uptake. Where there are high amounts of organic C, relatively small changes in C emissions due to temperature-induced increased decomposition could substantially increase soil C emissions. Also, greater sensitivity of photosynthesis to water stress compared to decomposition processes suggests that increased drought and warming will further enhance soil C losses (Arnone et al. 2008; Derner and Schuman 2007). Thus, although climate change has likely stimulated C sequestration over the past 150 years, continued warming may lead to reductions in terrestrial C sequestration by the second half of this century (Heimann and Reichstein 2008; Pepper et al. 2005). Exact estimates of those losses are presently impossible to determine due to uncertainty about how climate change will unfold and impacts on the plant/soil C cycle (Conant et al. 2008; Davidson and Janssens 2006; Kirschbaum 2006).

Responses of Non-Carbon Dioxide Greenhouse Gases

Even less is known about the responses of non-CO_2 greenhouse gases, CH_4 and N_2O, to climate change. As discussed earlier, fluxes of CH_4 and N_2O are sensitive to levels of substrates, temperature, and soil moisture, and also to soil characteristics like texture, pH, salinity, and aeration (Dalal and Allen 2008). Limited trace gas research under various climate change scenarios confirms that land-atmosphere fluxes of CH_4 and N_2O will be driven in large part by how climate change affects the previously mentioned soil attributes. Elevated CO_2 generally lowers soil CH_4 consumption (i.e., lower atmosphere CH_4 removal) in dry systems (Kanerva et al. 2007; McClain, Kepler, and Ahmann 2002) and increases CH_4 emission into the atmosphere from wet systems such as wetlands and rice paddy fields (Allen et al. 2003; Luo et al. 2008; Whiting and Chanton 1993; Ziska et al. 1998). Thus, at global scale, a rise in the atmospheric CO_2 concentration will also cause an increase in the atmospheric CH_4 concentration.

Soil moisture is the most important soil characteristic influencing CH_4 fluxes, and thus soil moisture changes due to altered precipitation patterns can have a large effect on land-atmosphere exchange of CH_4 (Van den Pol-van Dasselaar, van Beusichem, and Oenema 1998). In areas where the annual precipitation is likely to increase, net exchange of CH_4 into the atmosphere could increase. Global warming stimulates microbial activity responsible for both soil consumption and production of CH_4, but these effects on CH_4 fluxes are also mediated by warming-induced changes in soil moisture (Knoblauch et al. 2008; Van den Pol-van Dasselaar, van Beusichem, and Oenema 1998).

Soil inorganic N is an important factor controlling N_2O emissions from the soil into the atmosphere. In systems where the N cycle is tight (i.e., the demand for inorganic N by plants is relatively high compared to the supply of inorganic N through decomposition) and soil inorganic N concentration is low, elevated CO_2 often has little effect on N_2O emission (Billings, Schaeffer, and Evans 2002; Hungate et al. 1997a; Mosier et al. 2002). In systems such as N-fertilized agricultural systems, however, where greater N is available, N_2O emissions into the atmosphere can be large (Mosier et al. 1991; Roelandt, van Wesemael, and Rounsevell 2005). Studies on the effects of elevated CO_2 on N_2O emissions in fertilized systems yield mixed results (Baggs et al. 2003; Cheng et al. 2006; Kettunen, Saarnio, and Silvola 2007), and more research is needed. Warming generally increases decomposition and release of soil inorganic N (net N mineralization) more than it increases plant N uptake, thereby increasing loss as N_2O (Rustad et al. 2001). Large emissions of N_2O into the atmosphere occur after large rain events, and thus changes in precipitation amount and frequency can alter annual rates of N_2O emission into the atmosphere.

3 Mitigation Options

Mitigation Principles

As outlined in Chapter 2, increased emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) from agriculture generally reflect human perturbations to natural processes in the ecosystem carbon (C) and nitrogen (N) cycles. Although the impact of these perturbations on greenhouse gas (GHG) emissions cannot be completely eliminated, it can be substantially lowered through improved land use and management. In general, agricultural activities can mitigate emissions by (1) decreasing emissions of GHGs, and (2) sequestering C in biomass and soils, resulting in a net removal of CO_2 from the atmosphere. Sustainable agricultural biofuels (Chapter 4) present a third mitigation category, with the potential to offset CO_2 emissions from fossil fuels.

Activities that cause GHG emissions from agriculture include the conversion of unmanaged ecosystems to agricultural use as well as many common management practices on long-established agricultural production systems (Figure 3.1). Land use conversion involving deforestation and biomass burning, wetland drainage, plowing, and accelerated soil erosion can result in large losses of biomass and on-site soil C stocks as well as substantial GHG emissions. There are also continuing sources of GHG emissions associated with already-established agricultural lands. Among these

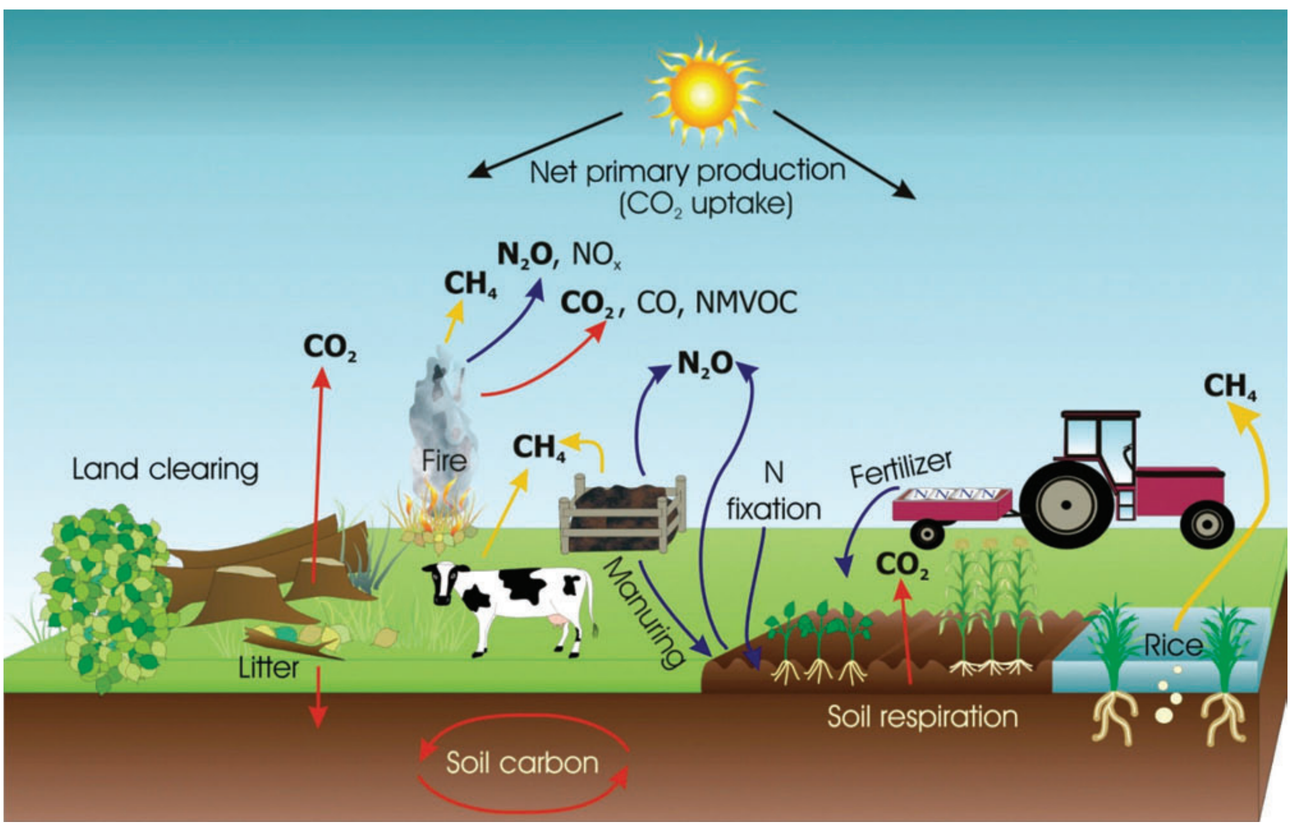

Figure 3.1. Overview of greenhouse gas emission sources associated with agricultural activities. (Source: Intergovernmental Panel on Climate Change; Design courtesy of Amy Swan, Colorado State University.)

are CO_2 emissions associated with energy used for the production and application of agricultural inputs such as fuel, fertilizers, lime, and pesticides. Drained organic soils can sustain continuing oxidation of organic matter, decreased soil C stocks, and increased CO_2 and N_2O emissions over several decades. Nitrogen additions—whether through application of fertilizers, N-fixing crops (e.g., legumes), or manure—are subject to N_2O losses. Livestock production, manure management, and rice cultivation are the predominant agricultural sources of CH_4 (N_2O is also released during manure storage). In addition, agriculture use generally lowers the natural CH_4-oxidizing capacity of nonflooded soils, usually by a factor of 8–10 or more, which contributes to CH_4 increase in the atmosphere.

Because of increasing human needs for food and fiber from agricultural systems, some increase in GHG emissions is an inevitable consequence of agricultural land use. Many opportunities exist, however, for decreasing emissions and, in some instances, even for making systems that are net sinks for GHGs. Decreasing the extent, and disturbance intensity, of land use conversions would dramatically decrease emissions, especially in the tropics where most deforestation is occurring. Reversing such practices can lead to sinks, i.e., through afforestation and conversion to grasslands. On established agricultural lands, an important strategy is to improve the use efficiency of production inputs (especially of N fertilizer), thus decreasing associated fossil energy-derived CO_2, as well as N_2O emissions from inefficient use of N inputs. Similarly, means exist to decrease CH_4 emissions and/or capture them for use as an energy source. Finally, production of agricultural biofuels (see Chapter 4) provides opportunities for offsetting fossil energy CO_2 emissions from agriculture and other sectors of the economy.

Agricultural carbon sequestration involves increasing the storage of C in biomass and soil, thus removing CO_2 from the atmosphere for some period of time. There are two principal opportunities for C sequestration in agricultural ecosystems: (1) improved management of permanent agricultural land through practices that enhance C storage, and (2) conversion and/or restoration of marginal and degraded agricultural lands to alternative uses like forest or grasslands. For most agricultural systems, soil organic matter (which is roughly 55% C) is the main persistent C stock (i.e., herbaceous biomass is largely ephemeral and/or removed annually by harvest). Soil C stocks are governed by a balance between C additions (via both above- and belowground plant residues, manures, or other organic amendments) and losses, primarily as CO_2 through decomposition (i.e., heterotrophic soil respiration). Thus by increasing C inputs to soils and/or decreasing the rate of organic matter decomposition, the C content of the soil can be increased; conversely, decreasing C additions or increasing decomposition rates will decrease soil C stocks.

There are many practices that can increase soil C sequestration. Conversion of degraded lands can also increase C sequestration when properly managed, including the conversion of cropland to pastures, woodland, or grassland conservation set-asides (e.g., the U.S. Conservation Reserve Program [CRP]); wetland restoration; or the restoration of land severely degraded by mining, salinization, or other activities such as industrial waste disposal. In addition to storing it in soils, C can be sequestered in woody biomass through agroforestry practices (e.g., windbreaks, forested conservation buffers) and establishment of perennial crops for food (e.g., orchards) or biofuel (e.g., hybrid poplars). Carbon can also be sequestered in long-lived roots of herbaceous perennial crops used for forage (e.g., hay fields) or fuel (e.g., biofuel feedstocks such as switchgrass).

Carbon sequestration rates are limited in quantities and duration by the nature of the biological carbon cycle as well as by inherent capacities to store C in soils and biomass. After a change in management designed to increase C stocks (e.g., by increasing C inputs and/or decreasing decomposition rates), soil C stocks tend to approach a new equilibrium level where C inputs and outputs are again balanced, after which there is no more net increase in C storage (Figure 3.2). The approach toward a new equilibrium state may take several decades, although the bulk of the changes in soil C storage under land use change tend to occur during the first 2–3 decades (West and Six 2007), with that from tillage change largely occurring in the first decade (West and Post 2002).

In addition to this equilibrium effect for cropland soils, some scientists have argued that there is an upper or *saturation* limit (Six et al. 2002; Stewart et al. 2007) above which additional organic matter cannot be effectively stabilized, regardless of the management intervention. Although this "ultimate limit" for soil C sequestration may be much higher than soil C levels in most cropland soil, the key point is that soils have a finite capacity to store C. Similarly, sequestration in woody biomass has an effective upper limit as the amount of vegetation that can be maintained. In addition, biomass as well as soil C stocks are not permanent and thus the stored C is subject to loss if the management conditions are not maintained.

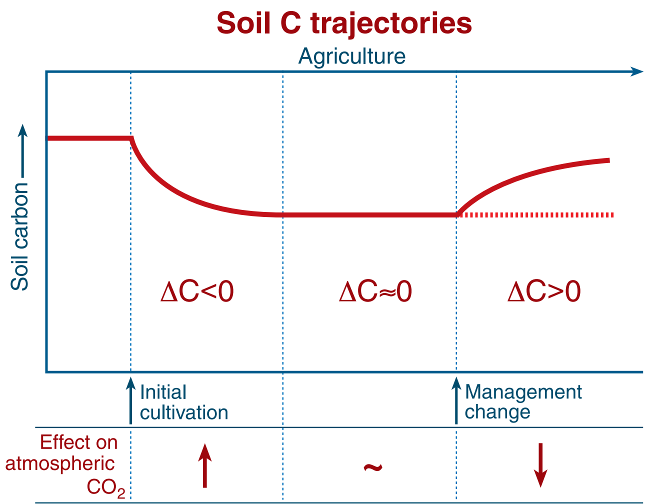

Figure 3.2. Conceptual diagram showing the initial decline in SOC that typically occurs following land conversions to agricultural use, resulting in a net loss of CO_2 to the atmosphere. Eventually, most cropland soils reach a new (lower) equilibrium and are neither a source nor sink of CO_2. With adoption of improved management practices that increase SOC, this acts as sink, thus removing CO_2 from the atmosphere.

Annual Cropland

Soil Carbon and Carbon Dioxide Mitigation

Currently, cropland soils in the United States are a net source of GHG emissions of around 150 teragrams (Tg) CO_2 equivalents per year (Eq yr^{-1}) almost entirely because of N_2O emissions (Table 3.1). With respect to soil C, however, U.S. croplands are roughly in balance. There is an estimated increase in soil C on most cropped soils of 46 Tg CO_2 because of a long-term trend of increasing crop residue production, reductions in the tillage intensity, conversion of annual cropland to pasture/hay, and set-aside of annual cropland in the CRP, but this is largely counterbalanced by emissions from cultivated cropland on organic (e.g., peat) soils of around 30 Tg CO_2 Eq yr^{-1} and net emissions (approximately 6 Tg CO_2 Eq yr^{-1}) from land recently converted to cropland (Ogle et al. 2010; USEPA 2011).

A number of practices, well-known from field experiments, can increase soil C stocks. The main classes of practices and their primary mode of action, i.e., increasing C inputs to soil (I) or reducing decay rates of soil organic matter (R), include the following:

- High residue-yielding crops and residue retention (I)
- Conversion of annual crops to perennial grasses and legumes (I, R)
- No-tillage and other conservation tillage (I, R)
- Cover crops (I, R)
- Manure additions (I)
- Reduced frequency of bare fallow (I, R)
- Other practices to increase C additions to soils (e.g., irrigation, improved fertility) (I)
- Rewetting (flooding) of organic (i.e., peat and muck) soils (R)
- Tree planting on annual cropland (I, R)

The changes in soil carbon stocks due to adoption of more "carbon-friendly" management practices depend on the *interaction* of several different management variables (e.g., crop type and rotation, tillage, nutrient and water management) as well as soil type, climate regime, and previous land use history (Paustian, Collins, and Paul 1997). Historically, few experiments were set up to specifically address combined best practices for promoting soil C sequestration—in fact, many older, long-term field experiments now used to assess management and soil C changes were not originally designed for soil organic matter studies (Paul et al. 1997).

Field experiments that typically involve changes in only one or two management components give an

Table 3.1. Greenhouse gas emissions from U.S. cropland, derived from the 2009 U.S. national greenhouse gas inventory (USEPA 2011)

	1990	2000	2009
Source Category		(Tg CO_2 Eq y^{-1})	
Soil N_2O[a]	140.4	153.3	149.8
Crop residue burning			
CH_4	0.3	0.3	0.2
N_2O	0.1	0.1	0.1
Rice cultivation	7.1	7.5	7.3
Soil C stock change[b]			
Mineral soils[c]	(57.1)	(58.2)	(41.8)
Organic soils	29.8	30.3	30.3
Liming[d]	4.7	4.3	4.2
Total gross emission	182.4	195.8	191.9
Total net emissions	125.3	137.6	150.1

[a]Includes both direct and indirect N_2O emissions.

[b]Values have been combined for the Cropland Remaining Cropland and Land Converted to Cropland categories.

[c]Values in parentheses denote negative fluxes to the atmosphere, i.e., net uptake of CO_2 by soils.

[d]Emissions from liming applications include applications to grassland and other land-use categories.

incomplete picture of the potential of best management practices to sequester carbon; hence models that can integrate the effect of multiple management, soil, and climate impacts are particularly useful when combined with empirical information from long-term field experiments. Moreover, a more integrated approach has been the objective of the recent GRACEnet program (Follett 2010), wherein consistent protocols for soil, trace gas, and plant sampling are used for a network of ~30 field-experimental locations across the United States to compare "business as usual" vs. improved management scenarios (USDA–ARS 2008).

Existing long-term field experiments and model-based assessments provide a relatively clear picture of management influences on soil C stocks for all the practices listed earlier, and a number of synthesis and meta-analyses for the United States has been published (e.g., CAST 2004; Follett 2001a; Franzluebbers 2010a; Lal et al. 1998; Ogle, Breidt, and Paustian 2005; Paustian, Collins, and Paul 1997; Paustian et al. 1997; West and Post 2002). These studies show that rates of soil carbon increase from adopting improved practices on conventional cropland typically range between 0 to 1 megagrams (Mg) C per hectare (ha^{-1}) yr^{-1} over a duration of 20–30 years, diminishing as soils approach a new equilibrium. The wide range is due to a variety of factors, including climate, soil physical factors, land use history, and which combination of practices are used. Because many of these factors are site specific, only a few broad generalizations can be stated with confidence. Typically C accumulation rates are lower in semi-arid regions compared to subhumid/humid regions (Ogle, Breidt, and Paustian 2005). Soils that have been substantially depleted in their original soil C stocks have the most capacity to increase them, provided adequate C inputs can be maintained. And synergistic practices that both increase C inputs (the I factor) and decrease soil disturbance (the R factor), such as with no till and cover crops or no till and hay crops in rotation with annual crops (Dick et al. 1998; Franzluebbers 2005), have the highest potential for increase on most soils.

Long-term experimental data compiled for the Intergovernmental Panel on Climate Change (IPCC) soil C estimation method, which integrates the effects of C additions to soil, tillage practices, and previous land use (Ogle, Breidt, and Paustian 2005), can be used to obtain a more integrated estimate for the potential of combined best practices (Morgan et al. 2010). This approach suggests that, on average, the carbon stocks in the top 30 cm could potentially be increased by 14 to 28% over a 20-year period. In absolute terms, considering a range in cropland soil C stocks of 20–80 Mg ha^{-1} to 30 cm, this would amount to rates of around 0.15 to 1.1 Mg C ha^{-1} yr^{-1}.

Recent discussions have raised some questions about the general efficacy of no-till adoption in increasing soil C (e.g., Baker et al. 2007; Manley et al. 2005; Powlson, Whitmore, and Goulding 2011). The benefits of no till for soil organic carbon (SOC) sequestration varied with soil, climate, cropping system, and depth of measurement (West and Post 2002), which may help explain apparent conflicts in the literature (Baker et al. 2007). It has been well recognized that no till alone does not increase soil C in all soils—in particular, no till in cool, moist environments and on heavy, textured soils often shows no increase or a small decrease in C stocks compared to conventional tillage (Angers and Eriksen-Hamel 2008). In a global analysis, Ogle, Breidt, and Paustian (2005) found that about 10% of the studies that compared full tillage with no till showed no difference or lower stocks of C under no till, when evaluated for the top 30 cm. On average and for the majority of cropland soils in the United States, however, several analyses show significant increases in C stocks when integrated to below the depth of tillage (Angers and Eriksen-Hamel 2008; Ogle, Breidt, and Paustian 2005; Syswerda, Corbin et al. 2011; West and Post 2002). Moreover, lower soil depths have lower and more variable carbon contents than surface horizons, making it more difficult to detect statistically valid changes because of management, which underscores the importance of observed changes in surface soils (Kravchenko and Robertson 2011).

There is also substantial support for mechanisms (i.e., greater SOM protection in stable aggregates, reduced aggregate turnover) by which no till can reduce the specific rate of decomposition of soil organic matter (SOM) (Follett et al. 2009a; Six, Elliott, and Paustian 2000). The lack of soil C increases in response to no till in some soils, however, is not fully understood. Decreased rates of residue C inputs under no till for corn and wheat, predominately in cooler, wetter regions (upper Midwest), may explain part of the lowered C sequestration performance of no till (Ogle et al. In Review). Also, soils in cool, wet environments often have already high surface organic matter contents and thus may be closer to a C "saturation level," and hence less of the residue added at the soil surface under no till can be stabilized (Gregorich et al. 2009; Stewart et al. 2007). In these situations, the tillage-enhanced movement of C to deeper soil layers, where SOM concentrations are lower and the SOM saturation deficit is greater, may allow for equal or perhaps greater overall C sequestration in tilled systems.

Amendments of organic material from external sources (i.e., outside the field boundaries) to soils generally increase SOC levels; however, assessing the net impacts on CO_2 mitigation requires including the C balance effects of the biomass removal at the location where the biomass was originally produced. In addition to conventional organic amendments such as livestock manure, there is growing interest in the potential of biochar applications to soil as a GHG mitigation option (Lehmann 2007). Biochar is the product of pyrolysis (heating in the absence of oxygen [O]) in which much of the O, N, and hydrogen (H) atoms from organic materials are driven out as gases, leaving behind a residue composed largely of aromatic (i.e., ring structure) C compounds that are largely resistant to microbial decay.

Biochar is ubiquitous in most agricultural soils in the United States (Skjemstad et al. 2002) as a result of past fires in the original prairie and forest ecosystems. Although the resistance to microbial decay (and hence its longevity in soils) of biochar varies depending on the temperature and duration of the pyrolysis process and the characteristics of the original biomass, a significant fraction of most chars can persist in soils for hundreds of years or longer (Kimetu and Lehmann 2010; Nguyen et al. 2008; Schmidt and Noack 2000) and thus can increase C stocks in soil, long term. In addition, initial studies in the United States suggest that biochar applications can improve soil fertility (Laird et al. 2010; Novak et al. 2009). Potential barriers to widespread biochar use in agricultural soils include cost and safety issues, its long-term impacts on crop production and soils, and even whether or not increased SOC from the application of biochar constitutes a net removal of atmospheric CO_2.

Mitigating Nitrous Oxide Emissions from Soil

By far the largest proportion of N from mineral fertilizers and manure in the United States is applied to annual cropland, and additional N is added from nitrogen gas (N_2) fixation by grain legumes (e.g., soybeans) and legume hay (e.g., alfalfa and clover) grown in rotation with annual crops. This added input of reactive N is the primary driver of increased N_2O emissions from cropland, with emission rates typically amounting to 0.3–3% of N input (IPCC 2006). And recent evidence suggests the percentage lost may be higher still at N input levels that exceed crop demand (Hoben et al. 2011; Ma et al. 2010). As discussed in Chapter 2, however, N_2O emissions are highly variable in space and time, and thus predicting emissions for a particular location and/or management system is difficult.

Because of the direct relationship between N_2O emissions and N input and the importance of N supply in determining crop yield, the main management challenge is to increase the efficiency of plant uptake of added N, including that added through symbiotic N-fixation (legumes) so that less is available for loss. Management practices to increase the nitrogen use efficiency (NUE) of high-productivity cropping systems fall into four main options (Robertson and Vitousek 2009): (1) adopting more diverse and continuous crop rotations that increase plant uptake of available N; (2) providing farmers with the decision support tools that allow better predictions of crop N requirements to avoid overfertilization; (3) managing the timing, placement, and formulation of fertilizer N to better ensure that N is available where and when plant demand is greatest; and (4) managing watersheds to mitigate indirect N_2O emissions—emissions from the downstream denitrification of nitrate leached from farm fields (Beaulieu et al. 2011).

Winter cover crops can capture N that would otherwise be available to soil microbes to transform to N_2O. Crops such as cereal rye (*Secale cereal*) planted in the fall after harvest of the summer crop—or overseeded into the summer crop during the growing season—will grow in the fall and following spring, effectively scavenging residual fertilizer nitrogen left by the summer crop as well as the newly mineralized nitrogen produced during decomposition of summer crop residue (e.g., Delgado et al. 2007; Strock, Porter, and Russelle 2004). Soil N_2O fluxes are typically highest in the weeks after fertilization and in the fall and spring when soils are warm enough to support microbial activity and moisture and nitrogen are readily available. A good cover crop can decrease soil mineral N and N_2O fluxes during fall and spring, while supplying N to the subsequent summer crop when plant N demands are highest. Additionally, some annual crops have lower N needs and are more efficient at removing N from soil than others, so including a mixture of crops in a rotation can increase the NUE of the rotation. For example, Syswerda, Basso, and colleagues (2011) found that less nitrate was lost during the wheat and soybean phases of a corn-soybean-winter wheat rotation than from the corn phase.

Until very recently most fertilizer N recommendations in the United States have been based on yield goals, which provide a fertilizer recommendation on the basis of expected maximum yield multiplied by an N yield factor. Soil N tests before fertilization can often improve yield-goal recommendations without loss of expected yield (Andraski and Bundy 2002) but are not universally effective. An alternative Mean Return to Nitrogen (MRTN) approach (Sawyer et al. 2006), now being adopted by most states in the Corn Belt, is based on the site-specific N rate at which the value of increased yield matches the cost of additional N fertilizer. In simple terms, N applied above an economically optimized rate is N that costs the farmer more than it provides in increased yield. Identifying the economically optimized rate involves constructing fertilizer-N response curves for different cropping systems on different soils, and several thousand of these curves have now been aggregated into state-specific recommendation for corn and corn-soybean rotations in the U.S. Midwest. The use of MRTN calculators and other decision support tools by a greater number of farmers will reduce the amount of fertilizer N applied to better match the N needs of the crops, thereby keeping N from the microbes that transform it to N_2O.

Fertilizer timing, placement, and formulation can also have an effect on N_2O fluxes. Applying N when, where, and in a form that plants can easily access maximizes crop N uptake, improving systemwide nutrient use efficiency. Ideally N should be added in multiple small doses as the crop matures, but because this is rarely practical, fertilizer should at least be applied close to the time of maximum uptake. Generally this means in the spring post-plant; best practice commonly calls for two applications to corn—a starter rate at planting and the remainder side-dressed once rapid plant growth has begun. Fall fertilization, widely practiced in some parts of the Midwest, and winter manure applications, common elsewhere, provides months of N freely available to N_2O-producing microbes.

Optimizing fertilizer placement is also an effective means for increasing fertilizer NUE. Banding fertilizer in the row keeps N closer to where the plants need it. And site-specific technologies for adding N at variable rates across a field, tailored to crop production potentials as they vary across the field, are another way to provide adequate N to plants without providing excess N to microbes (e.g., Scharf et al. 2005). On-the-go fertilization using spectral reflectance of the canopy to judge real-time crop N need (Scharf and Lory 2009) is a recent promising technology for better placing N fertilizer.

Enhanced efficiency N fertilizer can provide mechanisms for controlled or slower release of mineral N that can improve crop NUE and help avoid N_2O emissions. Enhanced efficiency N fertilizer can be considered in three basic categories: (1) stabilized materials wherein nitrification and/or urease inhibitors are added to delay conversion to N forms that would be more susceptible to loss, (2) physical coatings that place a barrier around a soluble N fertilizer to slow release, and (3) condensation products of urea and urea-aldehydes that are slowly soluble synthetic organic compounds (Akiyama et al. 2010). Slow-release fertilizers delay the dissolution of fertilizer N in soil, deferring its exposure to nitrifying bacteria that transform it to nitrate, and can, under some but not all circumstances, reduce N_2O emissions (e.g., Halvorson, Del Grosso, and Alluvione 2010). Nitrifying bacteria can also be inhibited by natural and manufactured compounds added to soil, though also with mixed success in practice (Parkin and Hatfield 2010).

Finally, watershed-level strategies can also be important for decreasing cropland N_2O fluxes. Fugitive nitrate that leaves farm fields can be kept from becoming N_2O downstream by planting downslope areas with conservation plantings to remove nitrate before it enters streams. Restoring headwater streams and small wetlands can promote the conversion of nitrate to inert N_2. Coupled with the in-field strategies described above, so that the amount of nitrate leaving fertilized fields is lowered, proper watershed management can effectively decrease overall cropland N_2O loss. Much of the nitrate present in surface waters within the Midwest Corn Belt is from subsurface field drainage, and thus additional methods are needed to decrease nitrate-N concentration and load.

Kasper and colleagues (2003) proposed to decrease nitrate-N concentrations in tile drainage water by modifying the design and management of tile drainage systems to enhance denitrification and to increase uptake of nitrate by plants. Two methods were proposed. First, drain tiles were installed a little deeper than current practice, while maintaining their outlets at the same depth as normal, based on the premise that deep drain tiles would be more efficient in drawing water from deeper saturated zones where denitrification occurs naturally. Second, Kasper and colleagues (2003) installed woodchip trenches on both sides of subsurface drains as denitrification biofilters that could provide soil microorganisms with wood chips as a C source to better enhance denitrification and decrease nitrate concentrations of water passing into the drainage tiles. The first treatment of placing the tile deeper had no significant effects on nitrate-N concentrations or loads. The research by Kasper and colleagues (2003) over a two-year period indicated that wood chip trenches and rye winter cover crops have the potential to decrease nitrate-N concentrations and loads in tile drainage water in a corn-soybean rotation. This process is shown in Figure 2.3 where denitrifying organisms can denitrify nitrate-N to harmless N_2, especially in the presence of an organic source of C.

Importantly, many of the practices that can reduce N_2O losses from cropped soils have other environmental benefits. All will reduce the amount of nitrate leached to groundwater and transported to coastal hypoxic zones, and several will also improve soil C retention. Capitalizing on such synergies is an important conservation opportunity.

Grazinglands

Background

The total land area in the United States is nearly 931 Mha (million hectares), with grazinglands (rangeland, pastureland, and forested grazingland) composing 35% of this total (316 Mha); this accounts for two-thirds of all agricultural use (USDA–ERS 2006). Grazing is the predominant use on the 237 Mha of pastureland and rangeland, grazing occurs on the 54 Mha of forested grazinglands, and fall and winter grazing of small grains and after-harvest grazing of hay lands occurs on 25 Mha (USDA–ERS 2006). The U.S. Department of Agriculture (USDA) Forest Service and U.S. Department of the Interior Bureau of Land Management manage grazing on 39 Mha and 64 Mha of rangelands, respectively.

This vast land area implies a high physical potential to sequester additional SOC, given appropriate management decisions. Lal, Follett, and Kimble (2003) suggested that grazingland SOC sequestration might represent about 15% (range of 13–70 Tg C yr^{-1}; 1 Tg = 10^{12} g) of the potential for U.S. soils to sequester SOC.

Soil organic C dominates the terrestrial C pool in grazinglands. Aboveground C is <5% of the total ecosystem C pool in nonwoody plant-dominated ecosystems (Derner, Boutton, and Briske 2006) and about 25% in pinyon-juniper-dominated ecosystems (Rau et al. 2010). Grazinglands are close to C equilibrium and thus may operate as C sinks or sources (Schlesinger 1997; USEPA 2011), with rates of soil organic C sequestration up to 0.5 Mg C ha^{-1} yr^{-1} for rangelands (Derner and Schuman 2007; Liebig, Hendrickson, and Birdahl 2010; Schuman et al. 1999) and 1.4 Mg C ha^{-1} yr^{-1} for pastures (Franzluebbers 2005, 2010a,b; Schnabel et al. 2001). Actual rates are often less than these apparent maximal rates of SOC sequestration because of management, weather, and other environmental constraints. Potentially high rates of SOC accumulation are predicted in newly established pastures and in degraded rangelands, while improper management and drought can result in significant C releases. Because of the large land area, the movement of C into and out of the reservoir of grazingland SOC can be an important feature of the global C cycle. In addition to SOC, a vast pool of SIC (soil inorganic carbon) occurs as carbonates in semi-arid and arid rangeland soils that can lead to either sequestration or release of CO_2 (Emmerich 2003), the direction and magnitude of which are currently poorly understood (Liebig et al. 2006; Svejcar et al. 2008).

Several key factors need to be understood in considering the potential of grazinglands for sequestering SOC: (1) The aboveground C pool is a minor component of total ecosystem C, and mean residence time of this C pool is only a few years; thus yearly variations in aboveground biomass only minimally affect C storage. (2) Most SOC is recalcitrant and well protected from minor natural disturbances. Microbial biomass and particulate or light-fraction organic C are most sensitive to management or land-use change, whereas resistant organic C and soil carbonates are least sensitive (Allen et al. 2010). (3) Major pathways of SOC input are through decomposition of belowground root biomass, surface deposition of animal feces, and decaying litter from aboveground vegetation (Follett 2001b; Pinerio et al. 2010). (4) Large perturbations in the SOC pool can occur with major soil disturbances, such as tillage, wind and water erosion, and surface denudation with overgrazing. These effects occur naturally with extreme weather conditions or through human-induced management decisions that cause poor vigor of plant communities (Follett, Kimble, and Lal 2001).

Finally, while there is high physical potential for rangelands, land management is often rather minimal and the land values low, so that large management manipulations are unlikely, particularly if they are costly. Thus the economic potential of a sequestration response is likely substantially lower than the physical potential.

Management Considerations

Two important management factors that control the fate of SOC in grazinglands are (1) long-term changes in production and quality of above- and belowground biomass that can alter the quantity of N availability and the C:N ratios of soil organic matter (Derner and Hart 2007; Pineiro et al. 2010); and (2) grazing-induced effects on vegetation composition, which can be equally important as the direct impact of grazing (e.g., grazing intensity) on SOC sequestration (Bagchi and Ritchie 2010; Derner and Schuman 2007). The rate of SOC sequestration decreases with longevity of a management practice (Derner and Schuman 2007), indicating that ecosystems reach a "steady-state" and changes in management or inputs may be required to sequester additional C (Conant, Paustian, and Elliot 2001; Conant, Six, and Paustian 2003; Swift 2001).

Stocking Rate/Grazing Intensity

The response of SOC to stocking rate/grazing intensity has been variable (Blackburn and Taylor 1986; Liebig et al. 2006; Liebig, Hendrickson, and Berdahl 2010; Schuman et al. 1999; Smoliak, Dormaar, and Johnston 1972; Warren, Biondini, Patton, and Nyren 1998; Wood and Blackburn 1984). Grazing has also been observed to either increase or have little effect on soil bulk density (Franzluebbers and Stuedemann 2010; Greenwood and McKenzie 2001). In northern mixed grass prairie, SOC is higher in grazed than in ungrazed areas (Frank et al. 1995; Ganjegunte et al. 2005; Liebig et al. 2006; Liebig, Hendrickson, and Berdahl 2010; Manley et al. 1995; Reeder and Schuman 2002; Schuman et al. 1999), partly from increasing dominance of the shallow rooted, grazing-resistant species blue grama (*Bouteloua gracilis*), which incorporates more root mass in the upper soil profile than do midgrass species that it replaces under grazing (Derner, Boutton, and Briske 2006). In managed pasture, SOC can be optimized with a moderate stocking rate compared with no grazing or heavy, continuous grazing (Franzluebbers 2010a,b).

Grazing Method

The response of SOC to grazing method has been sparsely investigated, at best. Two studies have suggested an increase in SOC with rotational grazing compared with continual season-long grazing (Conant, Six, and Paustian 2003; Teague et al. 2010), and another study found no difference between systems (Manley et al. 1995). Given that the preponderance of evidence suggests that rotational grazing does not influence vegetation production (Briske et al. 2008), changes in SOC with rotational grazing would be expected only if substantial vegetation change occurred independently of stocking rate. Much more research on grazing method is needed because of the strong adoption and promotion of rotational grazing by producers and agricultural advisors (Beetz and Rhinehart 2010; Budd and Thorpe 2009).

Prescribed Fire

Burning has the potential to alter SOC through effects on photosynthesis (Bremer and Ham 2010; Knapp 1985; Svejcar and Browning 1988), soil and canopy respiration (Bremmer and Ham 2010; Knapp et al. 1998), and species changes (Boutton et al. 2009; Pacala et al. 2007), in addition to increasing livestock gains, improving habitat diversity, and decreasing fuel loads (Anderson, Smith, and Owensby 1970; Rau et al. 2008; Toombs et al. 2010). Although C loss from burning grazinglands is a minor component of the annual C emissions (Owensby, Ham, and Auen 2006; Bremmer and Ham 2010), burning rangelands with a significant woody aboveground plant biomass can result in substantial immediate ecosystem C loss (Rau et al. 2010). Increases in near-surface SOC of rangeland soils in the western United States that have experienced woody plant encroachment over the past 100 years suggest that removal of woody plants by fire or other mechanisms may also significantly deplete these shallow, relatively susceptible SOC stores associated with encroachment (Boutton et al. 2009; Neff et al. 2009).

Improvement of Degraded Rangelands

Application of best management practices on poorly managed rangelands in the western United States could result in sequestration of 11 Tg C yr^{-1}, and continuation of sustainable management practices on the remaining rangelands would avoid losses of 43 Tg C yr^{-1} (Schuman, Herrick, and Janzen 2001). Many rangelands are N deficient, and N additions through interseeding of legumes can increase forage production and quality as well as C sequestration (Liebig, Hendrickson, and Berdahl 2010; Mortenson, Schuman, and Ingram 2004; Mortenson et al. 2005).

Pasture Management

Establishment of improved pastures on formerly degraded croplands in the eastern United States can sequester SOC at a rate >2 times that of no-tillage cropland (up to 1.4 Mg C ha^{-1} yr^{-1}) (Franzluebbers 2005) and comparable to those under forests. Livestock grazing with optimum stocking rates can sequester SOC at a rate exceeding that of unharvested (e.g., CRP land) or hay land (Franzluebbers and Stuedemann 2010). Vegetation composition of pastures can alter SOC, and changes may be due to species and microbial-plant-specific associations (Franzluebbers et al. 1999, 2000). Nitrogen fertilization of pastures improves forage production and can improve SOC content, but the C cost of N fertilization and its expected higher rate of N_2O emission may negate SOC benefits (Liebig, Varvel, and Honeycutt 2010; Schnabel et al. 2001). Soil organic C sequestration can also be affected by animal behavior (Franzluebbers and Stuedemann 2010) and soil sampling depth (Franzluebbers and Stuedemann 2009), although data have been derived from only a limited number of studies. Soil organic C storage under pastures is not only important for mitigating GHG emissions, but more importantly on the farm level for improving water relations, fertility, and soil quality.

Environmental Considerations

Environmental factors controlling the fate of SOC in grazing lands are (1) short-term weather conditions (e.g., droughts) on net ecosystem C exchange (Ciais et al. 2005; Ingram et al. 2008; Soussana and Lüscher 2007; Svejcar et al. 2008; Zhang et al. 2010), and (2) long-term changes in the global environment, such as rising temperature, altered precipitation patterns, and rising CO_2 concentration, affecting plant community composition and forage quality (Hatfield et al. 2008; Milchunas et al. 2005; Morgan et al. 2008; Soussana and Lüscher 2007). Some phenomena with important implications for C sequestration, such as woody plant encroachment (e.g., Boutton et al. 2009; Neff et al. 2009), likely emanate from several of these environmental factors (Morgan et al. 2008; Van Auken 2009).

Soil organic C sequestration in grazinglands is influenced by climate (Derner, Boutton, and Briske 2006), weather conditions (Franzluebbers and Stuedemann 2010; Ingram et al. 2008; Jones and Donnelly 2004; Svejcar et al. 2008), soil type or parent material (Burke et al. 1989; Causarano et al. 2008), plant community (Conant, Paustian, and Elliot 2001; Franzluebbers et al. 2000; Mortenson, Schuman, and Ingram 2004), plant growth characteristics (Derner, Boutton, and Briske 2006), site management (N addition, fire, harvest) (Bremmer and Ham 2010; Derner and Schuman 2007; Follett, Kimble, and Lal 2001; Rau et al. 2010), and livestock grazing (Derner and Schuman 2007; Derner, Boutton, and Briske 2006; Franzluebbers and Stuedemann 2010; Liebig, Hendrickson, and Berdahl 2010; Pineiro et al. 2010; Schuman et al. 1999). As a result of these interacting factors, SOC pools and sequestration are highly variable, both spatially and temporally (Allen et al. 2010). For example, grazinglands are typically characterized by short periods of high C uptake during the growing season (2–3 months) and long periods of C balance or small losses during the remainder of the year (Skinner 2008; Svejcar et al. 2008), which results in interannual variability in NEE (Zhang et al. 2010). Droughts in particular can limit periods of C fixation, thereby reducing C uptake and turning even productive grazinglands into C sources (Svejcar et al. 2008; Zhang et al. 2010).

In the past century, rising CO_2 combined with slightly warmer temperatures has likely enhanced net primary production (Hatfield et al. 2008), with possibly positive consequences for C sequestration. Such positive effects of climate change on productivity may continue for the near future at northern latitudes and at high altitudes where temperature is an important limiting factor on production and annual precipitation is not expected to decline. As warming accelerates, however, the CO_2 benefit to plant productivity and water use efficiency is likely to be offset by the negative effects of warming via desiccation, which will eventually constrain and possibly even decrease plant productivity, especially in the southwestern quadrant of North America (Seager and Vecchi 2010; Seager et al. 2007). Thus, on-going climate change is likely to alter the potential for rangeland C sequestration through changes in plant productivity, and it may decrease SOC sequestration in regions that are predicted to become more prone to drought.

Ecosystem Services Associated with Soil Organic Carbon

Soil organic C is a key indicator of soil quality (Doran et al. 1994). It improves the physical, chemical, and biological properties and processes in soil and, therefore, is a key contributor to many essential ecosystem services, such as biomass production, nutrient and water cycling, provision of habitat, climate regulation, and disease control (Franzluebbers 2010c; Havstad et al. 2007; Millennium Ecosystem Assessment 2005). Mineral soils with high organic C hold more water, provide more nutrients to plants, have better soil structure, and can help facilitate clean water and clean air. Estimates of SOC storage and rates of SOC sequestration in grazinglands are being developed by scientists for policymakers regarding the potential of grazinglands to mitigate rising atmospheric CO_2 concentration (Follett and Reed 2010; Lal, Follett, and Kimble 2003). Considerable interest is being generated in terrestrial C storage and marketing of stored SOC (Williams, Peterson, and Mooney 2004) as economic benefits from C sequestration programs have the potential to significantly contribute to household economies (Olsson and Ardö 2002). The potentially ephemeral nature of soil C (Ingram et al.

2008) and conflicts of management objectives between C sequestration and other services, however, may complicate management objectives in arid and semi-arid rangelands (Neff et al. 2009).

Knowledge Gaps

With the enormous diversity of grazingland environments and management considerations, there are many gaps in our knowledge concerning SOC sequestration in grazinglands (Derner and Schuman 2007). To address some of the more critical gaps, research is needed to (1) quantify SOC sequestration in arid shrub lands; (2) better understand the influence of soil order, soil texture, landscape position, history of land use, soil pH, and other edaphic factors on SOC sequestration; (3) account for and develop an improved understanding of the effects of management on SOC and the entire GHG budget (e.g., include CH_4 and N_2O emissions), as sometimes they may counteract the greenhouse impact of SOC sequestration (Liebig, Hendrickson, and Berdahl 2010); and (4) develop a robust modeling capacity to spatially and temporally scale estimates of SOC sequestration from site-specific findings to regional, national, and global levels. Furthermore, the development of research networks that foster collaboration among natural resource, ecology, and management disciplines will bring the knowledge and fiscal resources needed to address the complex problem of enhancing SOC sequestration in grazinglands.

Horticultural Crops

Vegetable Agriculture

Cover crops and crop rotation have long been used for improving soil fertility and productivity of row crops that are grown after these previous crops. Little attention has been given in the past, however, to the potential for C sequestration in vegetable production systems that occupied nearly 786,000 ha in the United States during 2007, along with 457,000 ha of potatoes (USDA–NASS 2008). In general, conservation tillage has not been practiced much in the arid, irrigated systems of California because of the diverse rotations and specialized field management practices needed for crop production in this region. Five years of winter cover crops (triticale, rye, and hairy vetch) and decreased tillage increased soil C in a cotton-tomato rotation in California (Veenstra, Horwath, and Mitchell 2007). Most of the SOC increase was found in the less stable light fraction form (particulate organic matter) rather than in the more stable mineral-associated carbon form. In a one-year study, a winter legume cover crop of hairy vetch (*Vicia villosa* Roth) and Australian winter pea (*Lathyrus hirstutus* L.) resulted in increased emissions of both CO_2 and N_2O from tomato plots (Kallenbach, Rolston, and Horwath 2010). No season-long or annual assessments, however, were made of the GHG emissions balance. The highest emissions of N_2O were associated with winter cover crop and furrow irrigation, and the lowest emissions were associated with no cover crop and subsurface drip tube irrigation. Although based only on point data and the associated limitations of such data, estimated annual N_2O emission was between 2 and 8 kilograms (kg) N_2O ha^{-1}. No attempt was made to correlate N_2O emissions with fertilizer application events.

In semi-arid south-central Colorado, Al-Sheikh and colleagues (2005) found that increasing the number of small grain crops in a grain/potato rotation increased the amount of C sequestered. Sainju, Sing, and Whitehead (2002) did not find any consistent long-term trends in SOC based on tillage, cover crops, or N fertilization treatments in sandy loam soils in Georgia, but bulk density was not used in these measurements. In addition to the potential for increasing SOM and soil C content, cover crops have the additional benefits of increasing vegetable crop yields, improving water quality and soil and water conservation, increasing cycling of macro and micro nutrients, and increasing nitrogen use efficiency (Delgado et al. 2007). Cover crops can cycle as much as 60 kg N ha^{-1} yr^{-1} fertilizer equivalent to the following potato crop (Delgado et al. 2004). These studies indicate that off-season cover crops could be a useful way to accumulate SOC in vegetable production systems. Further research is needed, however, to quantify SOC sequestration potentials in various vegetable production systems and to identify best management practices. Because cover crops have benefits beyond SOC sequestration, they also need to have an economic rationale for widespread use beyond carbon sequestration.

Orchards and Vineyards

The principal means for enhancing the capacity for sequestering soil carbon of vineyard and tree crops probably lies in the use of managed cover crops versus clean cultivation. The area of tree crops (fruit and nut) grown in the United States during 2007 is reported as 1.55 Mha (USDA–NASS 2008). The area of land in vineyards is uncertain; however, about 87% of all types are grown in California and the total U.S.

production of grapes in 2007 was 6.1 million metric tons on a fresh basis (USDA–NASS 2008). Cover crops, especially leguminous cover crops, have been used for many years in high-value cash crops to enhance soil fertility, including benefits of biological N fixation. Very little research, however, has directly addressed the issue of C sequestration in orchards and vineyards.

After five years of cover-cropping in a Monterey County, California, vineyard, Steenwerth and Belina (2008) found mean total soil C in the 0–15 cm depth was 1.32-fold and 1.53-fold greater with Merced rye and triticale cover crops, respectively, than in the vineyard with conventional cultivation. Measurements during the final year showed that microbial biomass C, potential microbial respiration, and dissolved organic C of the soil were consistently larger for the cover crops than for the cultivation treatments. Further, they found that although cover crops used more soil water during the spring than the cultivated treatment, this did not seem to seriously impact soil water by the time vineyard growth was underway. As with vegetable agriculture, there are many benefits of cover crops beyond C sequestration that should favor further adoption of cover crops. Nevertheless, further studies are needed to quantify soil C sequestration potentials in orchards and vineyards.

Turfgrass

During the past few decades, the total land under turfgrass, as part of the U.S. urban landscape, has increased while area of agricultural land has decreased. Construction is reported to have averaged 1.6 million new homes per year in the late 1990s (U.S. Census Bureau 1999), and although likely at a slower rate, the construction of new homes will continue to increase into the future. Turfgrasses are part of U.S. urban landscapes and are located in lawns and turf of residential, commercial, parks, athletic fields, road right-of-ways, and golf courses areas, often as monocultures, and in most climates in the United States (Jenkins 1994). Many land areas previously used for agriculture have now become part of the urban landscape, and where under turfgrass, such lands can continue to sequester C. Between 1990 and 2007, the acreage of land in farms decreased by 22.6 Mha (USDA–NASS 1997, 2008) with much of this decrease in acreage potentially being converted to turf and other urban or suburban use. Milesi and colleagues (2005) used high-resolution aerial photography to determine that 16.4 (±3.6) Mha in the continental United States was under turfgrass. This area, equivalent to 9% of the U.S. cropland, accounts for 1.9% of the surface of the continental United States and is likely the single largest irrigated crop (USDA–NASS 2008).

Rates of SOC sequestration under turfgrass have a fairly broad range. Within a semi-arid region of the United States (in Colorado), a study by Qian and Follett (2002) using ~45-year historical data from 15 golf courses indicated a rate of C sequestration about 1 Mg ha^{-1} yr^{-1} for a period of 25 to 30 years. A subsequent modeling study by Bandaranayake and colleagues (2003) estimated that 23 to 32 Mg SOC ha^{-1} could be sequestered in the top 20 cm of soil after about 30 years. Huh and colleagues (2008) studied putting greens and observed a significant and linear soil C sequestration rate that continued for 40 years and totaled 28 Mg ha^{-1} (i.e. ~0.7 Mg ha^{-1} yr^{-1}. Chronosequences in fairways constructed with disturbed sandy soils showed continuous but nonlinear soil C accumulation across 3-, 8-, 25-, and 97-year-old bermudagrass (Shi, Muruganandam, and Bowman 2006). Recently, Qian, Follett, and Kimble (2010) used stable isotope techniques to study SOC sequestration and SOC decomposition under fine fescue (rain fed and irrigated), Kentucky bluegrass (irrigated), and creeping bentgrass (irrigated) in a subhumid climate in eastern Nebraska. All the turfgrasses exhibited significant C sequestration (0.32–0.78 Mg ha^{-1} yr^{-1}) during the first four years after establishment. The net carbon sequestration rate, however, was higher for irrigated fine fescue and creeping bentgrass than for Kentucky bluegrass.

Using the lowest observed rate of sequestration of about 0.32 Mg C ha^{-1} yr^{-1} for the 16 Mha of turfgrass reported by Milesi and colleagues (2005), about 5 Tg C are sequestered by turfgrass systems across the continental United States each year. This, however, does not account for N_2O emissions. A critical need in this area will be incorporating the effects of urbanization, which removes agricultural land, on the expanding area in turfgrass and quantification of C-sequestering turfgrass systems.

Agroforestry

Introduction

Within the United States, agroforestry is broadly defined as the intentional integration of trees and/or shrubs into agricultural operations in support of many of the other options discussed in this report (i.e., annual cropping, grazing, and vegetable and horticultural crop production), as well as for providing other environmental, economic, and social services valued by landowners and society. A short list of these benefits includes clean air and water, soil conservation, wildlife habitat, protection/enhancement of crop and livestock production, economic diversification, conservation of energy, recreation, and bioenergy.

Agroforestry is also valued for its aesthetic contribution to rural landscapes (Grala, Tyndall, and Mize 2010) and for its role in preserving biodiversity both as a practical benefit and value in its own right (Leakey 1999). Five main categories of agroforestry are practiced in temperate North America: windbreaks, alley cropping, silvopasture, riparian forest buffers, and forest farming (Gold and Garrett 2009). These practices are highly diverse, comprising a wide range of planting densities, woody and herbaceous species compositions and configurations, and placements within landscapes. A brief description of the practices and some of their uses are presented in Table 3.2. Additional information for each of these practices can be found at the USDA National Agroforestry Center website (http://www.unl.edu/nac/) and in the book *North American Agroforestry: An Integrated Science and Practice* (Garrett 2009).

While providing these benefits to landowners and society, agroforestry also serves as a viable agricultural option for mitigating GHG emissions and adapting to shifting climate (IPCC 2000; Nair et al. 2010; Schoeneberger 2009; Verchot et al. 2007). In one of the most massive tree planting programs in the United States, the Prairie States Forestry Program (1935–1942), hundreds of miles of windbreaks were established on agricultural lands from the Dakotas down to Texas for the mitigation of the climate-induced soil erosion during the Dust Bowl years (Droze 1977). Regarding adaptation to future climate

Table 3.2. Categories of agroforestry practices commonly established in the United States

Practice	Description	Primary Use[a]
Riparian Forest Buffers	A combination of trees and other vegetative types established on the banks of streams, rivers, wetlands, and lakes.	• Reduce nonpoint source pollution from adjacent land uses • Stabilize streambanks • Protect aquatic and terrestrial habitats • Diversify income either through added plant production or recreation fees
Windbreaks (shelterbelts)	Linear plantings of trees and shrubs to form barriers to reduce wind speed. Depending on the primary use, the windbreak may be specifically referred to as crop or field windbreak, livestock windbreak, living snow fence, or farmstead windbreak.	• Control wind erosion • Protect wind-sensitive crops • Enhance crop yields • Reduce animal stress and mortality • Serve as a barrier to dust, odor, and pesticide drift • Modify climate around farmsteads • Manage snow dispersal
Alley Cropping	Rows of trees planted at wide spacings while growing food, forage, or feedstock in the alleys.	• Stratify/diversify crops in time and space for greater production • Diversify income streams • Protect soil quality and reduce nutrient loss
Silvopasture	Trees combined with pasture and livestock production.	• Stratify/diversify crops in time and space for greater production • Diversify income streams • Reduce nutrient loss
Forest Farming	Natural stands whose canopies have been manipulated to grow high-value crops in the understory, such as mushrooms, decorative florals, and medicinal herbs (i.e., ginseng).	• Stratify/diversify crops in time and space for greater production • Diversify income streams
Special Applications	Use of agroforestry technologies listed above to help solve special concerns such as disposal of animal wastes; filtering irrigation tailwater while producing a short- or long-rotation woody crop such as for biofeedstock.	• Treat municipal and agricultural wastes while generating additional products and income • Treat stormwater issues • Use of center pivot corners to generate additional habitat or income • Produce biofeedstock

[a] In addition to the targeted benefits listed above, agroforestry plantings can also be simultaneously managed to provide enhanced wildlife provisions for valued game and nongame species, including native pollinators, and, regardless of intent, will contribute to greenhouse gas mitigation through carbon sequestration and reduction in fuel emissions.

change in this same region, modeling efforts predict windbreaks in Nebraska should increase dryland maize yields compared to nonsheltered-grown yields for almost all levels of predicted climate change (Easterling et al. 1997). Because of the growing awareness of the many roles agroforestry can play in support of agriculture under uncertain climate change (see Verchot et al. 2007), it is gaining acceptance within the broader arena of climate change programs (e.g., Global Research Alliance on Agricultural Greenhouse Gases [http://www.globalresearchalliance.org]).

Greenhouse Gas Mitigation by Agroforestry

Agroforestry contributes to U.S. agricultural GHG mitigation activities by (1) sequestering C, (2) reducing GHG emissions, and (3) reducing reliance on fossil energy (Brandle, Wardle, and Bratton 1992; Dixon et al. 1994; Nair et al. 2010; Sampson 1992). Sequestration rates in agroforestry are estimated to be greater than for many other options (IPCC 2000), with large amounts of C being sequestered in the woody biomass. While afforestation-like in its C storage, agroforestry practices generally occupy only a small portion within farm operations (e.g., windbreaks established for crop and soil protection will generally occupy only 3–5% of the total cropland; riparian forest buffers for purposes of water quality services even less) and therefore do not meet the definition of an afforestation activity per se (see definitions in IPCC [2002] Appendix B). Agroforestry is best thought of as a tree-based suite of practices in support of agricultural land use that can produce sizable gains of new C per unit land area and has a longer duration before saturation occurs compared to other practices (USEPA 2010). For instance, the addition of windbreaks for purposes other than C to approximately 3–5% of a hypothetical farm operation under no till in east-central Nebraska was estimated to potentially increase total C sequestration on the farm by approximately 75% after 50 years (Schoeneberger 2009).

Data on the total area in and C sequestration for temperate agroforestry are lacking (Nair and Nair 2003; Schoeneberger 2009). Schroeder (1994) reported a median value of 63 Mg C ha^{-1} for aboveground C storage in temperate agroforestry systems. Dixon and colleagues (1994) estimated storage values from 12 to 228 (median value = 95) Mg C ha^{-1} (included belowground storage and standardized to a 50-year rotation) for systems from a broader range of ecoregions.

While demonstrating the C sequestering potential of agroforestry, these estimates are based on broad assumptions regarding the agroforestry practices and management as well as incomplete accounting of the C. There are, however, a growing number of agroforestry studies in North America with more comprehensive accounting data. A Canadian study found poplar and spruce alley cropping systems had 41% and 11% more total C, respectively, than the sole cropping (barley) system, with total C storage in the alley cropping systems being 75–96 Mg C ha^{-1} after 13 years (Peichl et al. 2006). Sequestration rates for the vegetation in the alley cropping systems ranged from 0.5 Mg C ha^{-1} yr^{-1} for the spruce alley crop to 1.2 Mg C ha^{-1} yr^{-1} for the poplar alley crop. Net C accumulation rates were estimated to be +13.2, +1.1, and -2.9 Mg C ha^{-1} yr^{-1} for the poplar and spruce alley cropping and sole cropping systems, respectively. It was noted that the values for the sole cropping system may have been on the low side as only one crop versus a rotation of crops was used in this treatment.

At 13 years of age, the trees had approximately 83% of the total tree C in the aboveground and 17% stored in the roots. In young afforestation plantings, tree biomass generally makes up the larger part of new C stocks (Nui and Duiker 2006; Vesterdal, Ritter, and Gunderson 2002). Nui and Duiker (2006), looking at the sequestration potential by afforestation of marginal lands in the Midwest, reported the majority of C occurring in the aboveground as compared to the belowground biomass, with aboveground comprising approximately two-thirds of the C sequestered within the four major carbon pools (roots, floor, soil organic C, and aboveground biomass). The woody component likely comprises a majority of new C contributions by agroforestry, especially in the early-to-mid years. King and colleagues (2007), however, found that roots comprised a significant biomass fraction, i.e., 17, 80, and 29% of the total biomass in red pine at ages 2–5, 8, and 55 years, respectively, suggesting that more information is needed on the partitioning of above- versus belowground biomass for many agroforestry species.

Soil Carbon in Agroforests

The contributions and significance of agroforestry in the United States to just soil C are difficult to ascertain. For instance, soil C sequestration rates in a number of afforestation plantings did not support the general thinking that the addition of woody plants on former arable land readily leads to increased storage. Plots from the Midwest were found to have soil C sequestration rates ranging from -0.07 to 0.58 Mg C ha^{-1} yr^{-1} and -0.85 to 0.56 Mg C ha^{-1} yr^{-1} (where a negative sign denotes a net C loss) in deciduous and coniferous afforested sites. Possible reasons for this range include (1) greater impact-slower recovery from tree planting, especially with conifers, and (2) the choice of present-day cropping fields as the C baseline for comparison (Paul et al. 2003). Additionally, as many of the agroforestry practices are purposefully designed to intercept and decrease soil erosion off-site, the soil C in agroforestry plantings may reflect varying contributions from direct capture of C via biomass and from interception of wind- and surface-soil eroded sources (see Figure 3.3).

In general, studies have found greater C stocks under agroforestry plantings as compared to treeless cropping or pasture systems, but these differences are highly variable, and as seen in afforestation studies, dependent on stand age, tree species, variability in practice design, impacts from prior management and establishment, and the differing soil and climate conditions in which the practices are placed (Haile, Nair, and Nair 2010; Nair et al. 2010; Peichl et al. 2006; Sharrow and Ismail 2004). Further, for many of these studies we can't determine the stability of this C and, perhaps more importantly, whether these differences are due to sequestration of new C in the agroforestry system or due to no or less loss of C than in the treeless operation it is compared with.

Recent studies are beginning to shed light on some of these issues. Haile, Nair, and Nair (2010) reported that silvopasture systems in Florida had greater C stocks than treeless pastures and were able to demonstrate that this increased C occurred deeper in the soil, were in more stable soil fractions, and were predominantly derived from tree components. Hernandez-Ramirez and colleagues (2010) found similar results in formerly cultivated Corn Belt soils that had been planted to coniferous trees for 35 years either as windbreaks or an afforested plantation, with increase in SOC being as much as 57% greater under the trees than in the adjacent conventionally tilled cropping systems.

Using a detailed sampling scheme for soil under this windbreak planting, Sauer, Cambardella, and Brandle (2007) estimated an annual accrual rate of 0.1 Mg ha^{-1} yr^{-1} over the 35-year period. They also found the patterns of litter mass, soil pH, and texture under these windbreaks, however, suggested the soil C inputs under the windbreak were from both internally and externally (deposition of windblown sediment) generated processes (see Figure 3.3). Sudmeyer and Scott (2002) found windblown soils contained greater levels of nutrients, including organic C, than the remaining topsoil of open croplands and attributed the higher levels of soil C under windbreaks in part to these windblown deposits. Forest edges have been demonstrated to serve as efficient traps for windblown materials,

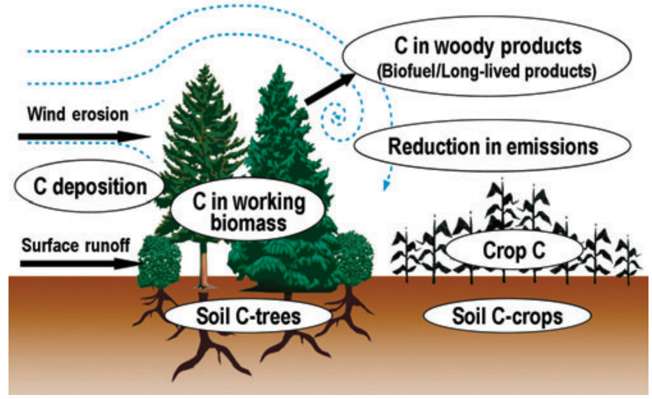

Figure 3.3. Major carbon sinks and sources in a field windbreak (Schoeneberger 2009).

creating higher concentrations of nutrients at the forest edge (Weathers, Cadenasso, and Pickett 2001), as would be expected with surface runoff from fields that concentrate at forest edges, especially those of riparian forest buffers. Because the majority of agroforestry practices in the United States are essentially "edge tree" plantings (e.g., windbreaks, alley cropping, and riparian forest buffers), it is important to distinguish how much of the soil C found in agroforestry systems is actually "new" C being sequestered and how much is from adjacent fields via wind or surface erosion, so numbers are not overestimated.

The high level of spatial variability in soil properties created by integrated agroforestry systems adds to the difficulty in determining C contributions from agroforestry. Sharrow and Ismail (2004) found soil spatial variability to be greatest in the silvopasture system, midrange for the tree plantation (forest), and least in the pasture, and they factored this variability into their sampling scheme for each of the treatments. In a recent review of soil C sequestration in agroforestry, Nair and colleagues (2010) acknowledge that the complexities of soil C "make measurement, estimation, and prediction of soil carbon sequestration potential a daunting task." Brown (2002) stated that although contributions to this pool may be significant, the costs to attain the needed levels of precision may be too high and sampling too intensive to be feasible for reporting in agroforestry C projects at this stage.

Other Greenhouse Gas Dynamics in Agroforests

Currently information is lacking to estimate net GHG impacts of agroforestry within farm operations because few studies have examined the impacts of temperate agroforestry on the other GHGs (N_2O and CH_4). Trees can potentially provide tighter nutrient cycling capabilities within agroforestry systems (Olson, Schoeneberger, and Aschmann 2000), which implies less N available for N_2O production. Allen and colleagues (2004) reported that trees reduced N leaching in a pecan/cotton alley cropping system. Reduced nitrate leaching attributed to the tree component was also found in silvopasture systems (Lopez-Diaz, Rolo, and Moreno 2011; Nair et al. 2007). Data from Allen and colleagues (2009) reported emissions of N_2O to be slightly higher for young afforested plantings compared to treeless pastures in Australia; however, this trend was reversed when the afforested plots were older (5–23 years old) with N_2O emissions being lower and CH_4 consumption higher under the trees than the treeless pasture, again emphasizing the need for better temporal information in these systems. Ryszkowski and Kedziora (2007) estimated N_2O fluxes from soils under windbreaks were less than from the adjacent cultivated fields. In the Canadian alley cropping system mentioned previously, Thevathasan and Gordon (2004) estimated a reduction of N_2O emissions from the alley-cropped fields of about 0.7 kg N_2O-N ha^{-1} yr^{-1} due to reduced fertilizer use and more efficient N cycling.

A study that examined N_2O and CH_4 fluxes in riparian forest buffers (7–17 years old) and adjacent crop fields in Iowa reported no differences in CH_4 emissions and significantly less N_2O emissions in the riparian forest buffer than adjacent crop field (Dong-Gill 2008). While the reduced N_2O emissions may be due in part to no N-fertilizer application within the riparian zone, Dong-Gill (2008) also found lower nitrate concentrations in the near-surface groundwater under riparian forest buffers, suggesting plant uptake rather than denitrification may also have been controlling N_2O emissions in these systems. Silvopasture systems, with the mix of management practices for the forage, tree, and livestock component, have potential for creating a more favorable net GHG grazing operation, but efforts are only beginning to look at these activities. More work is needed before we can understand agroforestry's impacts on the dynamics of these other GHGs.

Other Impacts of Agroforestry on Greenhouse Gas Dynamics and Emissions

The addition of an agroforestry component to a farm or ranch increases the complexity of and interactions between C storage pools and GHG fluxes within the agricultural landscape (Olson, Schoeneberger, and Aschmann 2000). For example, a simple windbreak system on cropland sequesters C in above- and below-ground woody biomass (Brandle, Wardle, and Bratton 1992; Schoeneberger 2009), but it also affects C dynamics out into the adjacent field (up to a distance of approximately 15 times tree height) (Figure 3.3).

Much less studied, but potentially providing an even greater contribution to the reduction of GHG emissions by agroforestry systems, are the energy savings and fuel and fertilizer reductions realized from decreased heating and cooling requirements for farmsteads; decreased fuel, fertilizer, and machinery costs due to removal of land from cultivation (either windbreaks that generally increase crop yields to compensate for the land put into windbreaks or on lands that are marginal and need the added conservation); decreased fuel use in snow removal; other off-site fuel use for mitigating off-site impacts of eroded soils; and potentially the substitution of fossil fuels by both the herbaceous and woody materials produced in agroforestry practices (Brandle, Wardle, and Bratton 1992; Kursten and Burschel 1993; Sampson 1992). Brandle, Wardle, and Bratton (1992) estimated that a minimum windbreak program for purposes other than GHG mitigation could potentially result in storage of 22 Tg C over a 50-year time span, but that the added GHG benefits of reduction in diesel fuel and fertilizer consumption, as well as energy savings from the protection of farmstead, could result in an additional 79 Tg over that time.

Agroforestry's Cobenefits

Agroforestry practices, as viable C sequestering and GHG mitigation options, need to be evaluated with regard to their broader impact on agricultural productivity and services to society. Their value lies in their strategic use within the agricultural landscape, especially on the more marginal and environmentally sensitive lands, to enhance the productivity and environmental services within agricultural production systems. These many cobenefits, along with C sequestration and other GHG mitigation, have been mentioned earlier (see Table 3.2), especially in regard to supporting greater diversity within agricultural lands. Higher bird abundance, richness, and diversity have been observed in systems where windbreaks (Schroeder 1986) and riparian buffers (Berges et al. 2010) are incorporated. Even recently established silvopasture systems were found to support increased numbers and species of certain invertebrate groups and bird species (Mcadam et al. 2007).

Concerned about the potential for some mitigation options to have adverse impacts on biodiversity, the IPCC (2002) listed agroforestry as an option that had the capacity to "sequester carbon and have beneficial effects on biodiversity." Agroforestry-enhanced diversity has many economic implications, from insect pest management through enhancement of arthropod and other natural enemy populations (Stamps and Linit 1998) and contributing to the state's economy through revenues generated from hunting, as well as from many other nonagricultural benefits ranging from "health values, transportation safety, aesthetics and property values" (Cable 1999; Kulshreshtha and Kort 2009).

Because of the multifunctionality in agroforestry created by this diversity, there is an emerging interest in the use of agroforestry plantings, especially those with herbaceous and woody components, for the production of biomass for biofuels and energy (Gruenewald et al. 2007; Schoeneberger et al. 2008). Agroforestry-enhanced diversity could be an option for providing the added resiliency and adaptability that will be required by these lands under future climate change (Verchot et al. 2007).

Inventory and Other Accounting Needs in Agroforestry

Looking at land area currently under or potentially under agroforestry in the United States, Nair and Nair (2003) estimated the C sequestration potential through agroforestry at approximately 90 Tg C yr^{-1}. Ascertaining any national-scale mitigation potential from agroforestry practices, however, especially given their off-site impacts, is currently not possible. Much work is still needed for providing the basis of C and other GHG dynamics and accounting for the many agroforestry practices and for the many conditions in which they can be established (Nair et al. 2010; Schoeneberger 2009). Unlike many of the cropping systems and forests, agroforestry in the United States is not explicitly included in either of the two national natural resource inventories (U.S. Forest Service Forest Inventory and Analysis and the NRCS NRI [Natural Resources Conservation Service National Resources Inventory]) (Perry, Woodall, and Schoeneberger 2005), which further limits our capability to account for agroforestry in reports like the 2010 U.S. Greenhouse Gas Inventory Report (USEPA 2010).

As with all the options presented in this publication, agroforestry requires standards for the quantification, monitoring, and verification of net GHG emissions that are accurate and economically feasible. Several 2008 Farm Bill conservation programs, such as the Environmental Quality Incentives Program (EQIP), that include many of the agroforestry practices will have an added focus on GHG mitigation. Such inclusion will make tools, like the Carbon Management Online Tool for Agriculture and Agroforestry Version 2.0 (USDA 2010), a valuable resource to farmers, ranchers and others in estimating C sequestration and net GHG emissions from soils and biomass based on local conditions and management decisions.

While many estimates to date have used readily available forest-derived equations and tools, they do not accurately reflect the conditions encountered in agroforestry. Unlike forests, the more open environment of agroforestry practices produces different light and climate conditions and results in woody plants with different growth forms (i.e., larger crowns, shorter boles or trunks, and different wood density), thus requiring development of new or modification of existing equations to more accurately reflect biomass and C stocks (Zhou et al. 2007; Zhou et al. 2011). Also lacking inventory and site index data for these "trees outside of forests," other approaches will be needed to assess current or future biomass (and therefore C stocks) in windbreaks and other agroforestry practices across the variety of soils and regions where they can be placed (Hou et al. 2011).

Further research in agroforestry is required before we can develop the understanding and reporting tools needed to accurately chronicle all the contributions of agroforestry to GHG mitigation within the whole-farm operations. Research that can assess agroforestry's impact on the productivity, as well as GHG emissions, of adjacent crop- and grazinglands will also be needed to fully assess agroforestry's value as a GHG mitigation tool for agricultural lands.

Wetlands Agriculture and Organic Soils

Wetlands Agriculture

Wetland agriculture implies mainly rice production in flooded fields. In 2009, the total area cropped in rice was 1.27 Mha in Arkansas, California, Louisiana, Mississippi, Missouri, and Texas (USDA 2010). These areas represent only slightly more than 1% of cropland in the United States, but they can be important sources of CH_4 and N_2O. Flooded rice soils are anaerobic with low redox potentials (Reddy, Feijtel, and Patrick 1986; Yu et al. 2001). Anaerobic methanogenic bacteria metabolize digestible organic matter and release CH_4 in the soil (Mayer and Conrad 1990). Methane is transported from soil to the atmosphere mainly by diffusion through air channels (aerenchyma) that can exist in plants (such as rice) that go from their roots through stems and leaf sheaths (Holzapfel-Pschorn and Seiler 1986; Mariko et al. 1991; Nouchi, Mariko, and Aoki 1990). Oxygen diffuses to the roots via the same air channels (Allen 1997), some of which diffuses into the soil and is used by methanotrophic bacteria to oxidize CH_4 (Schipper and Reddy 1996).

Only a fraction of CH_4 generated in flooded soil escapes to the atmosphere (Epp and Chanton 1993). Utilizing soil free of fresh organic matter, Allen and colleagues (2003) found that CH_4 emissions were minor for the first forty days of rice culture, although soil redox potential decreased drastically within a few days of flooding. In the absence of metabolizable SOM, methane emissions arise from root exudates or root sloughing as carbon sources (Minoda, Kimura, and Wada 1996; Watanabe and Kimura 1998). Methane emissions of wetlands are linearly related to photosynthetic rates (Whiting and Chanton 1993), and researchers have found that elevated CO_2 enhanced photosynthetic rates and increased CH_4 emissions of rice (Allen et al. 2003; Luo et al. 2008; Ziska et al. 1998).

Management practices that could decrease CH_4 and N_2O emissions from rice systems include (1) removal of rice straw to avoid incorporation of fresh plant residues; (2) timing a mid-season drainage to minimize CH_4 emissions so that a lower water table would not coincide with high soil nitrogen, leading to a burst of N_2O emissions; (3) managing N fertilization in multiple split applications and use of nitrification inhibitors to increase nitrogen use efficiency by plants; and (4) selection of rice cultivars with low root exudation or root sloughing and with low CH_4 transport capacity (assuming that this would not interfere with oxygen transport capacity) (Majumdar 2003).

Severe or prolonged drainage should not be implemented at panicle initiation (Baker et al. 1997a, 1997b) or flowering (Towprayoon, Smakgahn, and Poonkaew 2005) because it could decrease yields. Modeling could help predict the flow of events that govern GHG emissions and thereby guide mitigation options. Some authors indicate that CH_4 emissions could be predicted from organic matter additions, rice net productivity, cultivar characteristics, soil texture, and temperature (Huang, Sass, and Fisher 1998; Huang et al. 2004). Bossio and colleagues (1999) accurately predicted CH_4 emissions from a California rice field using a model by Nouchi and colleagues (1994) based on CH_4 concentration in soil water and temperature. Burning rice straw rather than incorporating straw in the soil can decrease CH_4 emissions, but it can also create air quality problems (Bossio et al. 1999). Harvesting and removing part of the rice straw for other purposes (e.g., as biofuel) might be an option to decrease CH_4 emissions. Because emissions from flooded fields include both CH_4 and N_2O, conditions that can decrease one emission might increase emissions of the other. Water table management to control emissions may be difficult to achieve (Jiao et al. 2006). Yu and Patrick (2003), however, found that redox potentials for minimal CH_4 and N_2O production were between 150 and +180 millivolts (at pH 7.0). This information may help lead to improved water table and irrigation management for rice to minimize emissions of both N_2O and CH_4.

Organic Soils

Organic soils or histosols (peat deposits) are formed as a result of flooded or water-logged conditions where plant decomposition is inhibited because diffusion of oxygen is impeded by water. Although organic soils cover about 10,000,000 ha in the contiguous 48 states (Joosten 2010; Lucas 1982; Stephens, Allen, and Chen 1984), only about 750,000 ha are drained for agriculture (USDA 2008). The largest drained areas in the United States are the 240,000–280,000 ha in the Florida Everglades Agricultural Area (EAA) (Allen 2007; Snyder 2005; Stephens and Johnson 1951) and the 100,000–150,000 ha in the Sacramento-San Joaquin Delta of California (Deverel and Rojstaczer 1996; Mount and Twiss 2005; Rojstaczer and Deverel 1993).

Minnesota and Michigan have larger total areas of organic soils (Lucas 1982) that are dispersed in many smaller units than the Florida EAA, but less total area is drained. Other drained organic soils are found mainly in Wisconsin, Indiana, Iowa, Ohio, Illinois, New York, North Carolina, and South Carolina. Although drained organic soils represent less than 1% of the U.S. cropland, they can contribute large CO_2 emissions per unit land area because of microbial oxidation, which results in soil subsidence. Based on early subsidence measurements (Shih et al. 1979) of about 1 in. (inch) per year (2.54 cm yr^{-1}), Allen (2007) estimated that EAA soils emitted annually 25.4 Mg C ha^{-1} (93.1 Mg CO_2 ha^{-1}). For 240,000 ha, this results in an annual emission of 22.4 Tg CO_2. Recent lower subsidence estimates of 0.57 in. yr^{-1} (1.45 cm yr^{-1}) (Shih, Glaz, and Barnes 1998) yield an estimate of 12.7 Tg

CO_2. This latter estimate is almost half of the USEPA (2011) estimate of total U.S. organic soil annual emissions of 27.7 Tg CO_2 and about 42% of the USDA (2008) estimate of 30.2 Tg CO_2 in 2005. Smaller, but wider-ranging, subsidence rates for recent years have also been reported for the Sacramento-San Joaquin Delta (Deverel and Rojstaczer 1996; Mount and Twiss 2005; Rojstaczer and Deverel 1993). Emissions from organic soils offset nearly half of gains in C sequestration by U.S. mineral soils (67 Tg CO_2) (USDA 2008).

Drained organic soils also emit N_2O. Organic soils of the Florida EAA had annual N_2O emissions ranging from 11 to 75 kg N_2O ha^{-1} for sugarcane, 25 to 152 kg N_2O ha^{-1} for St. Augustine grass, and 93 to 259 kg N_2O ha^{-1} for fallow conditions, with lower emissions during the low-rainfall year (Duxbury et al. 1982). Drained, cultivated organic soil annual emissions in New York were 113 to 133 kg N_2O ha^{-1} (Duxbury et al. 1982).

For Ohio drained organic soils, Elder and Lal (2008) found annual emissions of 152, 57, and 46 kg N_2O ha^{-1} for moldboard/disking (MB), no-till cropping (NT), and bare-herbicide (B) treatments, respectively. Elder and Lal (2008) also reported annual CO_2 emissions of 83, 69, and 76 Mg CO_2 ha^{-1} for MB, NT, and B treatments, respectively, which are comparable to the EAA (Allen 2007). Furthermore, Elder and Lal (2008) reported annual N_2O emissions of 47, 18, and 14 Mg CO_2 Eq ha^{-1} (based on global warming potential [GWP] conversion) for MB, NT, and B treatments, respectively. The CO_2 Eq of N_2O emissions ranged from 1/5 to 1/2 of the CO_2 emissions (56%, 26%, and 18% of the CO_2 emissions for MB, NT, and B treatments, respectively). Methane fluxes were negligible. Both CO_2 and N_2O emissions from drained organic soil cultivation are large inputs (per unit land area), but they are emitted from less than 1% of the U.S. croplands. If the large areas of fertilized and drained organic soils in other regions are also considered, then clearly N_2O emissions, as well as CO_2 emissions, from drained organic soils contribute appreciably to agricultural GHG emissions.

Within a given climatic (temperature) zone, annual subsidence rates are linearly related to depth to water table (Couwenberg, Dommain, and Joosten 2010; Stephens, Allen, and Chen 1984). Glaz (1995) and Snyder (2005) suggested that high water table management and crop selection might decrease microbial oxidation and ameliorate loss of organic soils. Research in Florida indicates that high water tables can be maintained and even short periods of flooding can be tolerated in sugarcane production without sacrificing productivity (Allen 2007; Gilbert et al. 2007, 2008; Glaz 2007; Glaz and Gilbert 2006; Glaz and Morris 2006; Glaz, Reed, and Albano 2008; Glaz et al. 2005; Morris 2005; Morris, Glaz, and Daroub 2004; Page, Rieley, and Banks 2011). Reddy, Feijtel, and Patrick (1986) and Reddy and colleagues (1993) found that cattail (*Typha* spp.) growing in nutrient-enriched shallow water of the Florida Everglades could accumulate organic deposits at the rate of 1.1 cm yr^{-1}. Such an accumulation implies that returning drained organic cropland soils to flooded conditions would not only eliminate emissions of CO_2 but would also lead to carbon sequestering. Whiting and Chanton (1993), however, showed that CH_4 efflux densities of wetland systems were linearly related to photosynthetic CO_2 uptake rates, which were generally higher in fertile, warm wetland systems such as *Typha*. Because methane emissions were calculated to represent about 2.7% of net ecosystem production of *Typha* (Chanton et al. 1993), the percentage GWP of methane emissions relative to carbon uptake would be about 21 x 2.7% = 57% of the carbon uptake.

Later analyses indicated that the annual molar ratio of CH_4 emissions/CO_2 uptake of subtropical (Florida) and temperate (Virginia) *Typha* wetlands ranged from 0.05 to 0.09 (Whiting and Chanton 2001). In the typical 20-year time horizon, the net GWP of *Typha* wetlands would be positive (21 x 0.05 = 1.05 or 21 x 0.09 = 1.89), whereas in a longer time horizon of 100 years, the GWP of *Typha* wetlands would be negative because of the shorter lifetime of methane molecules in the atmosphere. Although restoring flooding to drained organic cropland soils might decrease the GWP via the decrease of emissions of both CO_2 and N_2O and by sequestering C in floodwaters, there are other mitigation options. High water table management practices could be implemented (e.g., Gilbert et al. 2008; Glaz, Reed, and Albano 2008; Snyder 2005), which would maintain organic cropland soil uses while decreasing the GWP of GHG emissions. More measurements and assessment, however, would be required to accurately quantify the overall balance of GHG exchanges and concomitant GWP changes in management of organic cropland soils for crop production. Mitigation strategies intended to sequester C in organic soils and decrease or eliminate CO_2 and N_2O emissions will lead to CH_4 emissions. In the long term, some of the sensitive sites such as the Florida EAA might be partly taken out of production as Everglades restoration projects are implemented (Perry 2004). Part of the Sacramento-San Joaquin Delta might also be returned to nonagricultural wetlands, especially if the sea level rises (Mount and Twiss 2005).

Set-aside (Conservation Reserve Program) Programs

Legislation

The "*set-aside program*" in the United States required farmers to set aside a certain percentage of their total planted acreage and devote this land to approved conservation uses (such as grasses, legumes, and small grains that were not allowed to mature) to be eligible for nonrecourse loans and deficiency payments. The set-aside program per se, however, has not been used since the late 1970s, and its formal authority, called "set-aside," was eliminated by the 1996 farm bill (Public Law 104-127). During the early 1970s, concerns were increasing within the conservation community about U.S. soil and water resources because studies suggested that soil erosion was becoming as serious as it had been during the 1930s. The result was passage of the Soil and Water Conservation Act (RCA) of 1977 (Public Law 95-192). The RCA required the USDA to periodically prepare a national plan for soil and water conservation on private lands based on an inventory and appraisals of existing resource conditions and trends. The NRCS led this effort and completed appraisals in both the early and late 1980s, but a third appraisal effort, initiated in the early 1990s, was not completed. Although a national plan was adopted only in 1982, many of the activities envisioned when the RCA was enacted continued to be carried out to address conservation needs and priorities on U.S. private lands.

The predominant government set-aside program is the CRP that resulted from passage of the Food Security Act of 1985 (i.e., the 1985 Farm Bill) under Title XII (16 USC 3831). The 1985 Farm Bill authorized the federal government to enter into contracts with agricultural producers to remove highly erodible cropland from production in return for annual rental payments and established cost share. Active CRP contracts began in 1987. In addition to authorizing the CRP program, the 1985 Farm Bill contained provisions designed to discourage conversion of wetlands into nonwetland areas, collectively referred to as the "Swampbuster" provisions of the Food Security Act of 1985 (Title XII, Subtitle C). Swampbuster provisions denied federal farm program benefits to producers who converted wetlands after December 23, 1985. The *Wetlands Reserve Program* (16 USC 3837) authorized enrollment of wetlands for protection and restoration through permanent and temporary (30-year) easements.

The EQIP was approved in 1996 by amending the 1985 Farm Bill, reauthorized in the Farm Security and Rural Investment Act of 2002 (Public Law 107-171) and again reauthorizing it in the Food, Conservation, and Energy Act of 2008 (Public Law 110-234). Additionally, Conservation Innovation Grants (CIG) were authorized under the EQIP as a voluntary competitive program that was intended to stimulate development and adoption of innovative conservation approaches and technologies while leveraging the federal investment in environmental enhancement and protection in conjunction with agricultural production. Under CIG, EQIP funds are awarded competitively to nonfederal governmental or nongovernmental organizations, Tribes, or individuals with funding available for single- or multiyear projects.

The Food, Conservation, and Energy Act of 2008 (i.e., the 2008 Farm Bill) extended the CRP through fiscal year 2012. The CRP's general sign-up and continuous sign-up provisions remained unchanged, but starting on October 1, 2009, the program area was capped at 12.96 Mha (32 million acres), down from 15.87 Mha (39.2 million acres). Producers can offer land for *CRP general sign-up* enrollment only during designated sign-up periods, whereas *CRP continuous sign-up* is a nationwide, voluntary program designed to help farmers restore and protect environmentally sensitive land.

Besides updating previous provisions, the 2008 Farm Bill authorized an Environmental Services Market (ESM) within the USDA, updated and extended the 2002 Conservation Security Program, now the Conservation Stewardship Program, and made changes to the Grassland Reserve Program. The 2008 Farm Bill reauthorized funding for the Conservation Reserve Enhancement Program (CREP), a joint (federal-state) program to target specific agriculture-related environmental problems significant at the state or national level. The CREP is a highly targeted program, both geographically and to a specific resource concern, and offers additional financial incentives beyond the CRP to encourage farmers and ranchers to enroll in 10- to 15-year contracts to protect environmentally sensitive land, decrease erosion, restore wildlife habitat, and safeguard ground and surface water. An additional provision of the 2008 Farm Bill in Section 2709 directs the Secretary of Agriculture to establish technical guidelines that outline science-based methods to measure the environmental services benefits from conservation and land management activities to facilitate the participation of farmers, ranchers, and forest landowners in emerging ESMs.

Although various legislative approaches have been used over the years to establish set-aside programs, the CRP enjoys a relatively high degree of support for a set-aside program as it is not simply to provide supply control, but it is mandated to protect and enhance soil, water, wildlife, and other resources. In addition, it has a large potential for the sequestration of SOC and protection against wind and water erosion and various degrading processes to soil, water, and air quality.

Impact

Since its inception more than two decades ago, the CRP has had the participation of over 400,000 landowners, most of whom are farmers and ranchers. As of May 31, 2010, there were 12.67 Mha (31.3 million acres) of land enrolled in the CRP (Figure 3.4). A summary of many of the benefits of the CRP program during this time includes the following:

- 450 million tons of soil erosion decreased annually
- 0.81 Mha (2 million acres) of wetlands and buffers restored
- 48 million metric tons (53 million tons) of carbon dioxide removed
- 1,126 km (170,000 miles) of stream bank protected along rivers and streams
- An additional 2.3 million ducks produced each year from restored CRP wetlands
- Enhanced populations of pheasants, quail, and other wildlife species

There are currently 43 CREPs in 32 states in targeted watersheds, which has generated more than $1 billion in additional state and private funds for federal conservation efforts through the CRP. As of August 30, 2010, there were a total of 12.63 Mha (31.2 million acres) enrolled in the CRP.

Studies by the Food and Agricultural Policy Research Institute (2007), using the EPIC and APEX computer models, evaluated the control of wind and water erosion and estimated that both were lowered by 80–100% under the CRP when compared to the same land under crop production. Similarly, for water quality improvements, the models estimated N loss from fields under the CRP was lowered by 80–100% compared to cropland and phosphorus loss from fields was lowered by 60–100% by the CRP compared to cropland.

Experimental results on rates of SOC sequestration also continue to increase and, even though regional and local differences exist, rates of SOC sequestration are usually considered to be around

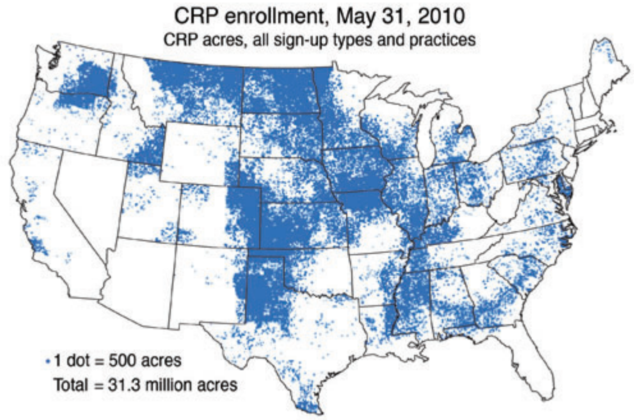

Figure 3.4. Land enrolled in Conservation Reserve Program.

0.5 Mg C ha^{-1} yr^{-1} in the top 20 cm of soil and more if considered to deeper depths (Follett et al. 2009b; USDA 2008). Assuming an average of 0.5 Mg SOC ha^{-1} yr^{-1} is sequestered across 12.6 Mha of CRP land in the United States, then as much as 6.3 Tg of SOC is sequestered annually in the United States. Cropped soils are estimated to have accumulated 18.1 Tg C (66.5 Tg CO_2 Eq) in 2005 (USDA 2008). Total net GHG emissions (N_2O, CH_4, and CO_2) from cropland soils, however, were estimated to be 41.7 Tg C (153 Tg CO_2 Eq) (USDA 2008). Therefore, the CRP accounted for about one-third of the net C sequestration on cropland soils and offset about 15% of all agricultural cropland GHG emissions.

As noted by Follett (2009), it is highly important to the future to recognize that current CRP contracts are expiring (two-thirds of the contracts that were in place in October of 2009 will expire by September of 2014 (USDA–FSA 2010). The multiple benefits obtained from the CRP, including sequestered SOC, resulted from a very large investment of taxpayer funds for more than 20 years. Accumulated benefits from the CRP have not been adequately measured. Even though conservation compliance provisions do exist for highly erodible land previously in the CRP program that returns to production, it is not fully understood how much of the land under expiring CRP contracts will be adequately treated if it is again tilled and planted to annual crops. The lessons from the 1970s, when studies suggested that soil erosion was becoming as serious as during the 1930s, need not be repeated.

Newer or better technologies are feasible to bring land back into production (Follett et al. 2009a), and it is important they be used if CRP land is converted back into cropland. Responsible soil-conserving government programs and policies at the national level need to continue to support conservation and to protect economic and societal benefits, including those accumulated under the CRP. It is reasonable to expect that future policy should recognize and allow sensitive acreages to qualify for continuous CRP sign-up, annual rental payments, and cost-sharing of approved practices. Such policies can provide heightened environmental benefits on selected areas such as waterways, filter strips, and habitat buffers. Such future activities, where eligible, also should be consistent with the emerging objectives of the USDA's Office of Environmental Markets.

Confined Livestock

Livestock, particularly ruminants (e.g., cattle, sheep, goats), produce CH_4 as a by-product of their digestion. Additionally, livestock manure can emit CH_4 and N_2O during storage and after field application. Storage conditions (aeration, temperature, pH) and manure composition influence the gases emitted and rates of emission. Methane emissions from manure that is stored can be lowered by cooling, covering, separating solids from slurry, or capturing the CH_4 emitted (Amon et al 2006; Clemens and Ahlgrimm 2001; Monteny, Groenestein, and Hilhorst 2001; Monteny, Bannink, and Chadwick 2006). Options for decreasing enteric CH_4 can be divided into three broad sets of practices: (1) improved diet digestibility, (2) additives, and (3) improved genetics of livestock. Options to mitigate GHGs from livestock manure are primarily limited to treatment, storage, or other improved management systems.

Improved digestibility of feed can increase the production/maintenance ratio of livestock and decrease the gross energy intake per unit of production, thereby lowering emissions per unit product. Diet composition can also affect the composition of volatile fatty acids produced, which affects CH_4 production—e.g., replacing sugars with starches in feed concentrates (Monteny, Bannink, and Chadwick 2006). Improvements in pasture forage quality, through improved pasture and grazing systems, have also been shown to improve animal production and lower CH_4 emissions (Leng 1991; McCrabb, Kurihara, and Hunter 1998; Wright et al. 2004).

Numerous potential dietary additives and agents have been associated with lowered enteric CH_4 emission. Common issues with several of these additives are inconsistent decreases in enteric CH_4, or only a temporary decrease in CH_4 production in the rumen as the rumen microbes adapt to the agent. The most common additives associated with a decrease in enteric CH_4 production are ionophores, such as lasolocid or monensin (Benz and Johnson 1982; McGinn et al. 2004; Van Nevel and Demeyer 1996), but the effect may be transitory (Guan et al. 2006; Johnson and Johnson 1995; Rumpler, Johnson, and Bates 1986). Certain oils and oilseeds have been shown to be effective in lowering enteric CH_4 emissions (Jordan et al, 2006a, 2006b, 2006c; Machmüller, Ossowski, and Kreuzer 2000), although this may be at the expense of decreased fiber digestion (McGinn et al. 2004). Halogenated compounds may also lower enteric CH_4 emissions by inhibiting the methanogenic bacteria populations, but they may also decrease feed intake and the effect may be short-lived (Van Nevel and Demeyer 1995; Wolin, Wolf, and Wolin 1964).

There is a growing body of literature evaluating the role of various compounds from plants such as saponins (Lila et al. 2003), essential oils, or tannins (Pinares-Patiño et al. 2003) on decreasing enteric CH_4. Probiotics, which may stimulate the growth of preferred populations, may also have a relatively small impact on enteric CH_4 (McGinn et al. 2004)—yet again there is limited literature regarding this effect. Fumaric or malic acid may serve as propionic acid precursors to decrease methane formation by serving as an alternative H sink, but typically only at high doses (Newbold et al. 2005), making them an unlikely mitigation factor.

In general, increasing the proportion of feed energy intake that is directed toward production (milk, meat) relative to energy used in maintenance will yield decreases in methane per unit of animal product. Activities include genetic improvements of the animal (for higher productivity), decreased time from birth to slaughter, and decreased dry periods for lactating cattle.

Methane capture from covered lagoons or in manure digesters (with either flaring of CH_4 or combustion for energy production) and the quantities of CH_4 captured can be relatively easily measured. Existing methods for estimating avoided CH_4 with these activities have been developed (e.g., methods developed for manure GHG abatement projects in the Clean Development Mechanisms of the Kyoto Protocol), although accurately estimating baseline emissions (without the storage and capture facility) is still challenging. Economic feasibility is the main constraint.

A variety of other manure treatments have been proposed including manure cooling ($<10°C$) to lower overall microbial activity (hence for both CH_4 and N_2O) (Sommer, Petersen, and Møller 2004), altering manure pH (Berg 2003; Berg and Pazsiczki 2003), and compaction of solid manure to lower O_2 (to denitrify all the way to N_2, lowering N_2O—but likely increasing CH_4). Frequent spreading of manure, if feasible, can lower emissions that occur under the storage period but may result in additional emissions occurring at application time (e.g., ammonia emissions that comprise a source for indirect N_2O emissions from manure) so that total GHGs are not decreased as much. Avoiding losses of gaseous N and leaching/runoff from stored manure will reduce off-site (indirect) N_2O emissions. Additionally, dietary N content can alter both urinary and fecal N content (Archibeque et al. 2007), which may contribute to the formation of N_2O by decreasing substrate available in the manure for N_2O production. There is, however, insufficient data at this time to properly estimate the potential impact of this form of management on subsequent N_2O emissions. Lastly, there are inherent differences associated with the various forms of manure management systems that a producer might use.

The IPCC (2006) indicates that the CH_4 conversion factors will range from 0 to 100% conversion of volatile solids. As the amount of CH_4 generated by a specific manure management system is affected by the extent of anaerobic conditions, temperature, and time that organic material is held within the system, there are obvious substantial differences in the amount of CH_4 produced from the various systems. There are typically, however, several logistic factors that may compel a producer to use a given manure management system. Additionally, the infrastructure and financial inputs required for many of the manure management systems may preclude the option of changing manure management systems as a mitigation strategy. This may, however, be the most important strategy producers constructing a new facility must consider if they hope to mitigate CH_4 from livestock manure. For example, in a system that uses aerobic treatment, essentially no solids are converted to CH_4, whereas systems that are highly anaerobic, such as deep bedding or anaerobic lagoons, may see as much as 80% conversion of volatile solids into CH_4. This variation in the rate of conversion of volatile solids into CH_4 illustrates how important the selection of manure management systems for various operations is for mitigating CH_4 production from manure management.

Scale

Although there are varying levels of effects derived from the various practices to alter enteric CH_4, the IPCC (Smith et al. 2007) provided region-level estimates for reduction potential of enteric CH_4 emissions. In North America, they assumed that improved feeding practices will lower enteric CH_4 emissions by 16% in dairy cattle, 11% in beef cattle, and 4% in sheep. Specific agents and dietary additives are indicated to decrease dairy cattle enteric emissions by 11%, beef cattle enteric emissions by 9%, and sheep enteric emissions by 0.4%. The final category of improving inherent animal performance is perceived to lower enteric CH_4 emissions by 3% in dairy cattle, 3% in beef cattle, and 0.3% in sheep. The IPCC assessment (Smith et al. 2007) did account for the fact that the emission reduction of subsequent practices is decreased by 20% for unknown nonadditivity effects. Additionally, it has been well established in the literature that each mitigation practice within each category of practices may have different levels of CH_4 reductions. As such, it is critical to evaluate each specific enteric mitigation strategy within the context of the particular production system (including dose and existing management practices) to achieve a more accurate estimate of the reduction in CH_4 emissions by each class of livestock.

In 2007, manure management was associated with 44 Tg CO_2 Eq from CH_4 emissions and 14 Tg CO_2 Eq from N_2O emissions (USEPA 2009). Although CH_4 emissions could be substantially lowered to negligible amounts with the construction of CH_4 digesters or systems that promote aeration of the manure, there are several logistical issues that will preclude this from happening. Decreases in N_2O may be more feasible, yet even a 10% reduction in N_2O emissions through improvements in manure management would equate to a relatively modest 1.4 Tg CO_2 Eq decrease. Within manure management, dairy cattle and swine produce the largest quantities of manure CH_4 (dairy cattle = 18.1 Tg CO_2 Eq; swine = 19.7 Tg CO_2 Eq), with beef cattle responsible for the third largest emission source at 2.4 Tg CO_2 Eq. This equates to 1.34 gigagrams (Gg) CO_2 Eq/1,000 dairy cattle, 0.303 Gg CO_2 Eq/1,000 swine, and 0.036 Gg CO_2 Eq/1,000 beef cattle.

With the variation in CH_4 production associated with these different practices, it is conceivable that almost all the 44 Tg CO_2 Eq from CH_4 emissions could be removed by adopting practices that either capture the CH_4 or prevent it from forming, although this is unlikely due to logistical and financial constraints and may be partly offset by increased emissions of N_2O or emissions due to the increased surface agitation and air flow through the system. Lastly, although there is a trade-off between decreases in CH_4 production and formation of N_2O, there tends to be much less variation in the rate of N_2O formation, with typically less than or equal to 1% of manure N being converted to N_2O, although some practices, especially actively mixed used bedding and intensive windrow composting, will see a high conversion rate of 7–10% of manure nitrogen into N_2O (IPCC 2006). There are substantial infrastructure costs associated with the construction of CH_4 digesters or any other manure management system. Additionally, to maintain the system, there typically needs to be a financial offset, either by using the electricity produced on the facility or selling the electricity into a local grid.

Other Impacts

Several of the techniques that may lower enteric CH_4 emissions may limit the accessibility of the final products from several markets. For example, although antibiotics and monensin may have the potential to lower CH_4, there is a growing push to decrease the use of these products due to concerns over the production of antibiotic-resistant bacteria. In addition, several markets (including the European Union [EU]) ban the use of many of these products in the production of livestock products sold in the EU. Additionally, there are limits to the quantity of concentrate feeds that can be included in a ruminant ration, and the leakage effects associated with the production of the grain may not yet be fully understood. Most mitigation options associated with decreased enteric CH_4, however, are also associated with improvements in the efficiency of feed use by the animals.

4 Bioenergy Feedstock Production

Introduction

Bioenergy offers a substantial opportunity to mitigate climate change by U.S. agriculture. Much of this opportunity stems from the ability of bioenergy feedstocks to offset contemporary fossil fuel use largely in the form of petroleum and coal. In particular, when energy forms based on recently photosynthetically captured carbon are substituted for energy based on fossil fuels with their associated carbon emissions, there is an opportunity to lower and ultimately stabilize atmospheric carbon dioxide (CO_2) levels rather than allow them to grow at rates commensurate with fossil fuel use. Carbon dioxide added to the atmosphere this year by biofeedstock or derivative fuel combustion was removed from the atmosphere by plants in the recent past—theoretically, biofuels simply recycle contemporary CO_2. Moreover, some bioenergy forms offer the additional opportunity to increase carbon (C) sequestration and decrease greenhouse gas (GHG) emissions in the ecosystems in which they are grown, providing an additional mitigation capacity. As currently practiced, however, bioenergy production provides only a fraction of the GHG emission reduction benefits possible.

What is bioenergy? Bioenergy is renewable energy derived from feedstocks grown by contemporary biological processes. In the United States today bioenergy is derived mainly from corn grain for ethanol production; a minor amount is derived from soybean and other oil seed crops for biodiesel plus some use of wood in electricity generation. Elsewhere, sugarcane and other oil crops are important feedstocks. Cellulosic biofuel production, not yet commercially viable, will use agricultural wastes and residues, perennial grasses, and woody vegetation for either cellulosic ethanol or other liquid fuels. Liquid fuel end products other than ethanol and biodiesel are also possible and likely to be part of a near-future biofuel economy; these include butanol, alkanes, and other so-called drop-in hydrocarbons. Bioelectricity uses agricultural wastes and residues, perennial grasses, and woody vegetation as firing or cofiring heat sources in generating electricity.

Corn-based ethanol production today consumes a substantial fraction of the U.S. corn crop. In 2009, about 107 million metric tons (MMT) (4.2 billion bushels) of corn were used to produce about 42 billion liters (11 billion gallons) of grain-based ethanol; this represents over 30% of the 2009 corn yield of 328 MMT (2.9 billion bushels). By 2015, legislative mandates now in place will have led to a maximum production of 57 billion liters (15 billion gallons) of grain-based ethanol annually, consuming about 50% of the 2009-equivalent crop. By 2022, the Energy Independence and Security Act of 2007 (EISA) mandates another 80 billion liters (21 billion gallons) of liquid biofuels, of which at least 61 billion liters (16 billion gallons) must be cellulosic and the remainder either cellulosic or another advanced feedstock with a lifecycle GHG emission reduction compared to fossil-derived fuels of 50% or higher as determined by the Environmental Protection Agency (EPA). Advanced biofuels include ethanol from cellulosic biomass and sugar cane and biodiesel from soybean, canola, and algae, as well as other fuels currently under development.

The United States is now the world's leading producer of ethanol, and although the growth of cellulosic biofuel production has been slowed by slowly developing technology, the recent economic downturn, and crude oil prices, the country maintains a goal to have biofuels represent 22% of total transportation fuel needs by 2022. For this to happen, however, a substantial amount of additional biomass feedstock must be produced, mostly from agriculture. To meet the EISA mandate of 80 billion liters (21 billion gallons) of ethanol with cellulosic feedstocks, for example, would require ~205 teragrams (Tg) of biomass per year; to meet expected future needs of 350 billion liters (92 billion gallons) by 2050 will require ~900 Tg.

Recent estimates of available biomass resources (NRC 2009) suggest that 109 Tg yr^{-1} (per year) might realistically be available from forest products, including logging residues, forest thinnings, and mill residues, and another 90 Tg yr^{-1} from municipal solid waste. This leaves the remaining 700 Tg yr^{-1} to come from agricultural production—from residues such as corn stover and purpose-grown cellulosic biofuel crops. This is a substantial amount of biomass; at today's average rates of on-farm cellulosic biomass production (Schmer et al. 2008) of 700 Tg yr^{-1}, as much as 90 million hectares (ha) (221 million acres) of additional cropland could be required—about half as much land as we use today for all annual crops.

In the future, there may be a major conversion of grasslands currently in the Conservation Reserve Program (CRP) to grain crops, particularly corn, because of the demand for grain crops or biomass for biofuels. Data by Follett and colleagues (2009a) have shown that with the use of no till, conversion from permanent grass to continuous no-till corn production did not result in any net change in soil organic carbon (SOC) during a six-year production period in the western Corn Belt of the United States. If CRP grasslands are converted to grain crop production, the data from this study strongly support the use of no-till farming practices as a method of conserving the SOC that was present in the soil while the land was in the CRP. Alternatively, it should be feasible to convert land in the CRP or other marginal lands to growing biofuel crops such as switchgrass.

Basis for Bioenergy Carbon Benefits

Most of the mitigation benefit of bioenergy comes from its capacity to replace an equivalent amount of fossil fuel, thereby avoiding the emission of additional fossil fuel carbon to the atmosphere. This is called the fossil fuel offset benefit, usually measured (at the national scale) in Tg of CO_2. There are fossil fuel costs, however, associated with growing biofuels such that the total biofuel carbon benefit must be debited by the amount of fossil fuel used in their production. In agriculture, this includes the fossil fuel used for tilling, planting, irrigating, harvesting, drying, and other agronomic activities; the fossil fuel energy and feedstocks used for fertilizer and pesticide manufacture, transport, and application; and the fossil fuel used for transporting feedstocks to biorefineries and for refining the feedstocks to liquid fuel. Moreover, should cultivation of the feedstock lead to a loss of stored soil carbon (C) to the atmosphere as CO_2, an increase in the emissions of the GHG nitrous oxide (N_2O) from soil, or a decreased capacity for the cropland to oxidize atmospheric methane (CH_4), this too must be debited from the total biofuel carbon benefit. Likewise, if conversion of land to biofuel leads to an increase in soil C storage, this would increase the total biofuel carbon benefit. Finally, if agricultural land producing food and fiber products is converted to biofuel production, the displacement of that food and fiber production and the associated GHG emissions need to be factored in. Thus the net carbon benefit of biofuels may be more or less than the total amount of fossil fuel carbon they offset—and in some instances the benefit can disappear altogether and biofuel production can become a net GHG source to the atmosphere. Thus it is crucial that lifecycle analysis be performed to document the full GHG benefits of specific biofuel production systems.

Though hotly debated, the EPA has determined that grain-based ethanol provides a lifecycle GHG benefit of 21%, meaning that for every 100 tonnes of ethanol carbon produced, 21 tonnes of fossil fuel carbon use is avoided. Most cellulosic feedstocks have a net benefit two to three times higher than this because they are either perennial crops, such as grasses that require little agronomic attention after their establishment year other than fertilizing and harvesting, or they are annual crop residues that would otherwise decompose in place, requiring additional fossil energy only for harvest and the cellulosic conversion process is less energy intensive. The net GHG benefit of biomass burned to produce electricity or heat is still higher, as more fossil fuel use can be offset because energy is not lost to biorefining. Table 4.1, updated from McCarl (2008), provides fuel comparisons of the use of biomass to power electric vehicles to internal combustion in vehicle engines.

Potential Feedstocks

Agricultural feedstocks for biofuel can be classified into three broad crop categories—annual grain crops and their residues, perennial herbaceous biomass, and perennial woody biomass plus manure. As noted earlier, in the United States today's biofuel industry is almost entirely grain based. Sugars and starches in grain require little processing and can be readily solubilized or hydrolyzed to sugars and then fermented directly to produce ethanol. Oil seeds can be pressed and the oil readily transesterified to biodiesel. The advantages of grain-based feedstocks are considerable: grain production is a mature technology that provides high yields with well-established efficiencies of scale, grain has a relatively high energy density that makes for efficient transport and storage, and biorefining to ethanol or diesel is technically straightforward with well-known economic costs.

Disadvantages, however, include the competing

Table 4.1. Percentage offset of net greenhouse gas emissions from the use of a biofeedstock (McCarl 2008)

	Form of Bioenergy Produced			
Feedstock Commodity	Crop Ethanol %	Cell Ethanol %	Biodiesel %	Electricity Fire with 100% Biomass %
Corn	30.9[a]			
Hard red winter wheat	13.2			
Sorghum	25.9			
Softwood residue		76.6		98.1
Hardwood residue		77.2		97.9
Softwood pulp		76.6		98.1
Hardwood pulp		77.2		97.9
Corn residue		73.5		94.6
Wheat residue		73.3		96.3
Softwood milling residue		81.9		
Hardwood milling residue		81.9		
Manure				97.7
Switchgrass		76.5		92.4
Miscanthus		87.7		97.1
Hybrid poplar		63.4		91.2
Willow		69.7		94.6
Energy sorghum		77.6		93.8
Soybean oil			70.4	
Refined sugar	64.8			
Corn oil			54.1	
Canola oil			91.4	
Bagasse		90.1		

[a] The percentage reduction in net GHG emissions when using corn-based ethanol is 30.9% relative to using gasoline. This means 69.1% of the potential emissions savings from replacing the gasoline are offset by the emissions from the use of fossil fuels in producing the corn, transporting it to the plant, and transforming it into ethanol. These percentages do not consider indirect land use.

need of grain for food and ultimately a supply capacity insufficient to meet future biofuel feedstock needs. Grain-based biofuels also contribute little to climate mitigation; with a lifecycle GHG benefit of only ~20%—and perhaps significantly less than this on consideration of additional indirect land use costs—they provide relatively little in the way of net C sequestration or GHG mitigation. Moreover, other environmental liabilities associated with intensive grain production as it is practiced today make them less attractive still. Although changes to cropping practices, such as the adoption of permanent no till and cover crops and better nitrogen (N) management, could improve their mitigation ability and overall environmental performance, even then supply capacity ultimately becomes limiting, especially in light of expected future demands for food by a larger and more affluent global population.

Cellulose-based biofuels are expected to meet much of the supply needs of a future biofuels economy, and they have a substantial GHG mitigation potential, but their production is presently constrained by the availability of affordable, large-scale technology. In contrast to grain-based ethanol, cellulosic biomass requires two extra processing steps when biorefined: an enzymatic step to convert cellulose and hemicellulose molecules to sugars, and a physical or chemical pretreatment step to break apart cell walls and expose the molecules to the added enzymes. Both enzymes and pretreatment add expense and complexity, although the expense is expected to come down quickly with technology advances. An additional advantage of cellulose-based feedstocks is that other end-products are also possible—drop-in fuels that substitute directly for contemporary diesel or petrol products and electricity and heat via direct biomass combustion.

Cellulosic material from agriculture includes three main biomass sources: annual crop residues such as corn cobs and stover; grasses and other herbaceous biomass such as switchgrass, hybrid Miscanthus, and mixtures of different species from old-fields and even restored prairie; and woody biomass from purpose-grown hybrid poplars and/or other wood energy crops. The enzymes and pretreatments responsible for deconstructing cellulosic biomass to sugars seem to be largely indifferent to the source of herbaceous biomass, which means that tomorrow's cellulosic biorefineries may find a broad diversity of feedstocks equally valuable. This has important advantages for providing biorefineries with a year-round source of feedstock and for the maintenance of diversity in agricultural landscapes.

Cellulosic biofuels face other technical hurdles as well. The relatively low energy density of cellulosic materials makes the efficient transport of material from field to refinery challenging. Advances in field-based densification will help to meet this challenge, as might decentralized pretreatment centers, perhaps county based, that could pretreat dry biomass to a more easily transported solution ready for fermentation or other refining. Seasonality of production and the need by refineries of a year-round annual supply raises issues about substantial seasonal storage and feedstock diversification.

The use of any particular feedstock, however—even cellulosic—does not guarantee net C or GHG benefits. Much depends on where and how feedstocks are grown. In the sections that follow, the considerations needed to evaluate alternative feedstocks and management scenarios are enumerated.

Electricity production is also possible and in instances far more technologically feasible, but it is generally more expensive than coal use, is not subject to subsidies as are liquid fuels, and is hampered in instances by a lack of large-scale firing technologies and the cost of hauling and storing bulky feedstocks.

Carbon Sequestration versus Carbon Loss and Debt

A crucial component of the overall GHG balance of bioenergy cropping systems, and thus of the overall capacity to mitigate climate change, is alterations to the amount of C stored in plants and soil. On converting an unmanaged system to cultivation, or on changing agronomic practices in a long-cultivated system, there is the potential to diminish established rates of C storage or to release stored C to the atmosphere. On the other hand, there is also the potential to increase rates of C storage—to enhance sequestration. The balance between diminishing C storage and enhancing it can make a huge difference to the system's overall GHG balance and thus to the biofuel's lifecycle GHG benefit.

Four items most influence changes to a biofuel system's C storage capacity and the creation of C debt: (1) land use conversion with its changes to the rates that C was previously accumulating; (2) tillage, with its capacity to promote the rapid oxidation of soil C to CO_2; (3) crop residue management, with its potential to promote or lower soil C accumulation; and (4) the use of perennial versus annual forms of feedstocks.

Land Use Conversion

Carbon debt is created on initial conversion of unmanaged land to biofuel production. The C stored in killed vegetation that is burned or left to decompose on site, including belowground C in roots, and the C in soil organic matter that is exposed to new environmental conditions that promote oxidation is C that would otherwise have stayed sequestered. Instead it is released to the atmosphere as CO_2 and must be debited against fossil fuel C offset credit produced by subsequent biofuel crops. Models suggest that this debt can be substantial, exceeding decades of C offset benefits when converting conservation set-aside lands and forests to annual grain crops for biofuel production (Fargione et al. 2008; Searchinger et al. 2008). Emerging empirical research is also suggesting, however, that C debt can be minimized with careful management (Follett et al. 2009a) and perhaps even avoided entirely with conversion that evades soil disturbance and replaces existing vegetation with higher-yielding perennial crops for cellulosic feedstock production.

Also included in C debt is foregone sequestration. This is the C that would have been sequestered had the converted land been left alone. If the replacement system sequesters C at a rate equivalent to that in the original system, then there is no net C benefit to sequestration in the replacement system—from the atmosphere's standpoint, sequestration must be greater than before to decrease atmospheric CO_2 loading. This is all the more reason to manage aggressively for soil C sequestration in subsequent cropping systems. In forested regions, it is also the reason it is important to plant cellulosic crops that grow faster than the trees they replace—if newly planted grasses or trees build biomass at the same rate as the original vegetation, then there is also no net C benefit. The rate at which the C debt gets repaid is thus a function of how much feedstock is produced and its GHG costs, including the potential reduction of prior rates of C sequestered in both soil and vegetation.

Tillage

For reasons noted in earlier sections, soil tillage can hobble the ability of a new biofuel cropping system to provide early GHG abatement. For land that is converted from an unmanaged state, particularly that inhabited by trees or from a no-till system, to one that uses even intermittent tillage, the GHG debit attributed to tillage can be considerable. Not only is substantial soil C oxidized immediately after the initial tillage, but the loss of prior sequestration if the land was previously still accumulating soil C—foregone sequestration—adds to the debt. Conversion without tillage, possible with careful residue management, is highly preferable—as is the maintenance of permanent no till after conversion (Follett et al. 2009a).

Residue Management

Together with tillage, residue management is key to maintaining soil C in annual crops. Early estimates (Graham et al. 2007) suggested that, on average, about 55% of the stover produced by the U.S. corn crop could be harvested without risk of erosion were no-till management widely adopted. Erosion, however, is not the sole arbiter of soil C levels—recent evidence (NRC 2009; Wilhelm et al. 2007) suggests that only about a third of this amount can be harvested if soil C stocks are to be maintained. Removing even this amount, however, is likely to be insufficient to sequester additional C, so the fossil fuel offset credit of harvested residue must be carefully compared to the lost soil sequestration benefit, particularly if the prior system was accumulating soil C via no-till or set-aside management. Furthermore, the need to replace nutrients removed in residues, through increasing fertilizer additions, is an additional consideration.

Biochar

Biochar, charcoal applied to soil, presents a specialized opportunity for sequestration that is related to bioenergy production (Lehman 2007). Biochar is created when wood or other plant biomass is burned under low-oxygen conditions, known as pyrolysis. An end-product of pyrolysis is charcoal, a C-enriched black solid that is resistant to microbial attack and so can persist in soil for long periods of time. The amount of charcoal produced during pyrolysis depends on a number of factors, including burn speed (residence time of the biomass in the combustion chamber), temperature, and pressure; other end-products include tar oils and synthetic gas ("syngas") that can be refined to a liquid fuel or used as a natural gas replacement. Likewise, the persistence of biochar in soil depends on a number of factors, including biomass source, pyrolysis methodology, and soil environmental conditions; evidence to date suggests that, on average, about 80% of biochar C is stable under humid temperate conditions, but this can vary significantly (see discussion under Annual Cropland in Chapter 3 of this report).

It is presently too early to say whether or not the net GHG benefit of biochar applied to soil is significantly greater than biomass or biochar burned to offset fossil fuel use. Lifecycle analyses (e.g., Roberts et al. 2010) suggest that much of the potential additional GHG benefit of soil-applied biochar is derived from purported decreases in N_2O production and greater fertilizer N-use efficiency in biochar-treated soils, but these effects have not yet been fully verified in field experiments. Absent these benefits, the net GHG benefits for direct biomass combustion vs. biomass-to-biochar-to-soil scenarios are similar, if not greater, for direct combustion. In addition, the value of the energy products generated is a major issue in practice profitability (McCarl et al. 2009).

The Importance of Nitrous Oxide and Methane in the Net Greenhouse Gas Benefit of Biofuel Systems

A significant portion of the potential GHG benefits of biofuels is related to their potential for lowering

N_2O emissions in cropping systems. Regarding grain-based food production, in grain-based biofuel systems N_2O loss represents the single greatest GHG source and, in the absence of soil C sequestration, can tip a system from providing a net benefit to being a net liability. For a conventionally managed corn-soybean-wheat rotation in the north central United States, for example, the impact of N_2O flux was twice that attributable to N-fertilizer production (0.52 vs. 0.27 Mg CO_2 Eq ha^{-1} yr^{-1}, respectively) and over three times that attributable to fuel use (0.16 Mg CO_2 Eq ha^{-1} yr^{-1}) (Table 4.2).

For cellulosic perennial systems, N_2O loss can be substantially lower—by as much as four to five times lower in systems like unfertilized hybrid poplar and mixed herbaceous feedstock systems (Table 4.2). These perennial-plant systems, even without a fossil fuel offset, can provide GHG benefits owing to a combination of low N_2O flux, little or no CO_2 costs of fertilizer production, fuel used only for harvest except during the establishment year, and soil C sequestration from root growth and persistence as well as the absence of tillage. The offset provides an additional benefit dependent directly on productivity—the greater the harvest, the greater the offset. Thus, maximizing the productivity of these stands—while minimizing their GHG liabilities—is the basic strategy for maximizing both energy and climate security.

In some instances, trade-offs must be weighed. Fertilizer use will increase biomass production in many (but not all) perennial biomass systems, and where this occurs the fossil fuel offset benefit from greater biomass production must be weighed against a potential increase in N_2O flux plus the CO_2 cost of fertilizer production. In some analyses the net GHG benefits of fertilization are nil at the field scale, but because the increased biomass in one location will avoid the need to convert land elsewhere to make up the biomass difference, the increased productivity—even in the face of no change in GHG intensity (CO_2 Eq MJ^{-1} [per megajoule] energy produced)—will be justified at the larger scale. Thus net GHG benefits must be assessed at a landscape or larger scale.

The potential for CH_4 oxidation to contribute to the net greenhouse benefits of biofuel cropping systems are also significant, if yet unrealized. Methane oxidation occurs at significant levels in undisturbed ecosystems, including the forests and grasslands that were replaced in the United States by annual cropland. For reasons not fully understood, methanotrophs become much less active in cropped soils, removing CH_4 from the atmosphere at only ~10% of rates pre-conversion, on average. There is evidence to suggest that soils under perennial vegetation recover some of this lost CH_4 oxidation capacity, though it occurs very slowly—on the order of several decades. Should this also be the case for perennial cellulosic biofuels, there may be another GHG benefit of these systems to include in the future.

Indirect Land Use Costs

Another consideration that affects the net GHG benefits of bioenergy cropping systems is indirect land use costs or leakage. These are GHG costs elsewhere that can be attributed to local bioenergy production. For example, if food production elsewhere is intensified to make up for the lost production due to conversion of existing local cropland to biofuel production, and if the food production elsewhere leads to GHG emissions that would not otherwise have occurred, then these emission increases must be debited against the net GHG benefit of the local biofuel system (Fargione et al. 2008; Murray, McCarl, and Lee 2004; Searchinger et al. 2008). Whether these effects occur

Table 4.2. Radiative forcing costs of field crop activities at a northern Corn Belt location

Cropping System	Soil Carbon Change	Fuel Use	N-fertilizer Production	Lime Dissolution	N_2O	CH_4	Net Balance
	(Mg CO_2 Eq ha^{-1} yr^{-1})						
Grain-based							
Corn-soybean-wheat	0	0.16	0.27	0.01	0.52	-0.04	1.02
Cellulosic							
Poplar (*Populus* sp.) trees	-1.17	0.02	0.05	0	0.10	-0.05	-1.05
Early successional vegetation	-2.20	0.02	0	0	0.15	-0.06	-2.11

All units are Mg CO_2 Eq ha^{-1} yr^{-1}. A negative net balance indicates greenhouse gas mitigation (more CO_2 Eq are sequestered than emitted). Note that the net balance does not include the fossil fuel offset credit (McSwiney et al. 2010; Robertson, Paul, and Harwood 2000).

internationally or in some other U.S. location is immaterial—from the standpoint of the atmosphere, GHG concentrations will not change if as many CO_2-equivalents are released from newly converted or intensified cropland as are saved by the conversion of existing cropland to biofuel production.

Calculating indirect land use costs is problematic and controversial, because of the difficulty of associating intensification elsewhere to a specific factor, the difficulty of knowing the extent to which local increases in food productivity alleviate the need to replace cropland now used for biofuels with intensified food production elsewhere, and our inability to measure the effect directly. We therefore rely on models with assumptions that are imperfect and feedbacks that are not transparent. The net effect is agreement that indirect land use costs are real but disagreement over their importance.

Two things are clear now, however. First, whatever is today's indirect land use cost, it will be greater tomorrow. As the world becomes more populous and affluent, the need for additional grain to feed both people and animals will increase proportionately. In the absence of agronomic productivity gains sufficient to keep up with these needs, more land will be needed to grow food crops. If cropland that is currently used to grow biofuels is not available for food production, then new land must be converted elsewhere—with the concomitant GHG costs of conversion. Those costs will create GHG debt that will, from the standpoint of the atmosphere, counteract many of the GHG benefits provided by the existing biofuel crops. Although the substitution of more productive cellulosic biofuel crops on land that is now used for grain-based biofuels (Somerville et al. 2010) could delay the acceleration of indirect costs, inexorably, in the face of increasing food needs and only slowly increasing productivity gains, additional cropland will be needed for food production.

Second, indirect land use costs could be avoided altogether by growing biofuels on land that is not currently used for food production. Marginal lands, long-abandoned cropland, and degraded forests could all be converted to biofuel production with minimal or no indirect land use costs. Although local C debt could be unnecessarily generated by careless conversion practices, the burden of C debt produced by indirect land use change would be avoided. In the language of C credit markets, there would be no "leakage." Cellulosic biofuel crops such as perennial grasses or hybrid poplars could be especially well suited for production on these lands.

Other Environmental Considerations

Other factors than GHG benefits must also be considered part of the biofuels equation. In particular are the biogeochemical and biodiversity impacts of biofuel expansion. Biogeochemical impacts include biofuel crop effects on reactive N in the environment and on water availability. From a biogeochemical standpoint, the expansion or intensification of annual crops to provide biofuel feedstocks have liabilities identical to the analogous food crops: greater nitrate loss to groundwater and coastal marine zones, elevated surface runoff and erosion, and more N_2O emission to the atmosphere (Robertson et al. 2011). Likewise, the expansion of annual biofuel crops can hamper the delivery of biodiversity services such as pollination and pest protection—services that depend on diverse habitats to provide refuge and year-round food sources for bees, predaceous beetles, and other arthropods and birds valuable to producers. One recent study (Landis et al. 2008) placed the lost biocontrol cost of expanded corn and decreased soybean acreage in four north central states at \$239M yr^{-1}.

In contrast to annual bioenergy crops, perennial cellulosic crops may have fewer environmental liabilities and more cobenefits. For example, perennial cellulosic crops can achieve high nutrient conservation by providing year-round cover, which reduces erosion and allows root uptake of N and other nutrients at times of the year when annual crops would be absent. Perennial biofuel crops also have a lower N demand because harvested tissue contains much less N than grain and perenniality allows nutrients to be retranslocated from leaves and stems to roots before harvest, allowing their re-use the following spring. And some crops may require little if any N fertilizer.

Likewise, planting a diversity of cellulosic crops in agricultural landscapes—either in multispecies assemblages or as single-species fields—would create a greater diversity of habitats for insects, birds, and other beneficials that pollinate and help to suppress crop pests and disease. On the other hand, planting perennial crops into existing landscapes that are more productive than the communities now present may increase overall water demand and could lead to greater evapotranspiration and less soil water drainage depending on crop water use efficiencies. Such changes could reduce local stream flow and groundwater recharge rates.

The potential for bioenergy crops to affect a number of environmental responses makes it important to consider their development in a systems context. This allows full consideration of environmental liabilities and benefits so that trade-offs can be identified and synergies promoted. In this way the full benefit of bioenergy systems can be realized: to simultaneously enhance climate, environmental, and energy security without jeopardizing food production.

Model Scenarios

Ecosystem models are often used to investigate the impacts of land use change on soil C levels and GHG emissions. In contrast to field experiments that are typically of relatively short duration and consider only a subset of available land management options, models are able to readily produce simulations that can be used to compare the short- and long-term impacts of a much greater variety of land use change scenarios. Models that account for how land management interacts with environmental conditions to control C and nutrient fluxes and have been validated by comparing model outputs with field measurements can make reliable projections. DayCent is a well-validated model that has been used extensively to quantify soil GHG fluxes under current land use and various land use change scenarios at scales ranging from the plot to the global. In this section, an overview of the model is presented and the results of biofuel production system simulations are summarized.

DayCent simulates key plant and biogeochemical processes, including plant growth and senescence, and the microbial processes decomposition, nitrification, and denitrification. Nutrient availability and environmental factors, particularly water and temperature, strongly influence these processes. Soil physical properties, daily weather, and land management information are required model inputs. Important outputs are crop yields, soil C change, nitrate leaching, and N gas emissions (N_2O, N oxide). The reliability of these outputs has been confirmed by comparing them with field data from experimental sites around the world. In 2005, the model was chosen by the EPA to calculate N_2O emissions from agricultural soils reported annually in the *Inventory of U.S. Greenhouse Gas Emissions and Sinks* (USEPA 2011) and submitted to the United Nations Framework Convention on Climate Change.

DayCent simulations suggest that both previous and current land use practices strongly influence soil GHG fluxes (Robertson et al. 2011). Corn-soybean cropping converted to long-phase corn cropping for ethanol production (four years of corn followed by one

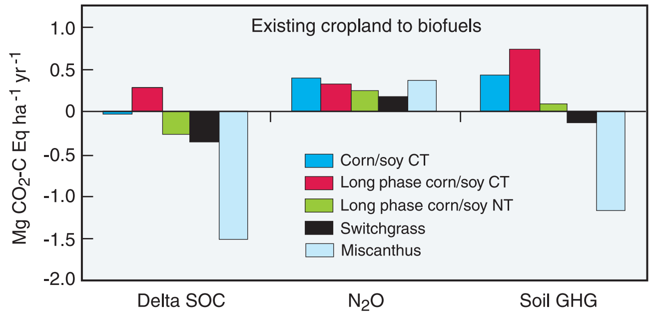

Figure 4.1. DayCent model predictions of soil carbon change, N_2O flux, and net greenhouse gas balance for existing corn-soybean conventionally tilled farmland in central Iowa converted to conventionally tilled (CT) long phase corn-soybean (four years of corn followed by one year of soybean), no-till (NT) long phase corn-soybean, or switchgrass biofuel production, and existing conventionally tilled corn-soybean farmland in east-central Illinois converted to a miscanthus biofuel production system. Net GHG change includes changes in soil organic carbon (SOC), direct and indirect N_2O emissions, and CO_2 emissions associated with production and application of N fertilizer. Negative values indicate net GHG mitigation. Not included are fossil fuel offset credits (Davis et al. 2010; Robertson et al. 2011).

year of soybean with 70% of corn residue harvested) increased net soil GHG emissions due to loss of soil C with conventional tillage but decreased net soil GHG emissions under no till, due to soil C storage and lower N_2O emissions (Figure 4.1). Conversion to perennial biofuel cropping systems resulted in a small net soil GHG sink for switchgrass and a large sink for Miscanthus (Figure 4.1). Both perennial grasses stored soil C, but Miscanthus stored more because production was much higher. Nitrous oxide emissions were lower than the long-phase corn system for switchgrass because N fertilizer inputs were lower (70 vs. 150 kilograms [kg] N ha^{-1}), and N_2O emissions were low for Miscanthus because no fertilizer N was applied. Nitrogen additions may not be required for Miscanthus production because it seems to facilitate biological molecular N fixation (Davis et al. 2010).

Conversion of native prairie to conventionally tilled annual grain production led to loss of soil C and substantially greater N_2O losses (Figure 4.2). Conversion to no-till corn, perennial switchgrass, or harvesting prairie biomass, however, can maintain or increase soil C without substantially increasing N losses (Figure 4.2). Management of N inputs has an important impact on net GHG balance. If prairie grasses are harvested without adding N, then production and soil C gradually decrease because N lost from harvesting

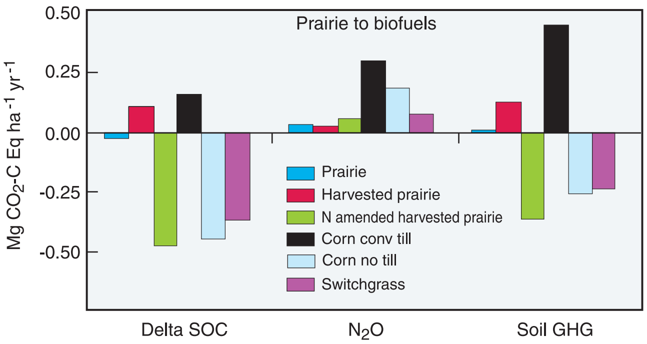

Figure 4.2. **DayCent model predictions of soil carbon change, N_2O flux, and net greenhouse gas balance for native prairie in eastern Kansas converted to harvested prairie, fertilized harvested prairie, fertilized switchgrass, conventionally tilled corn, and no-till corn biofuel production systems. Net GHG change includes changes in soil organic carbon (SOC), direct and indirect N_2O emissions, and CO_2 emissions associated with production and application of N fertilizer. Negative values indicate net GHG mitigation. Not included are fossil fuel offset credits (Robertson et al. 2011).**

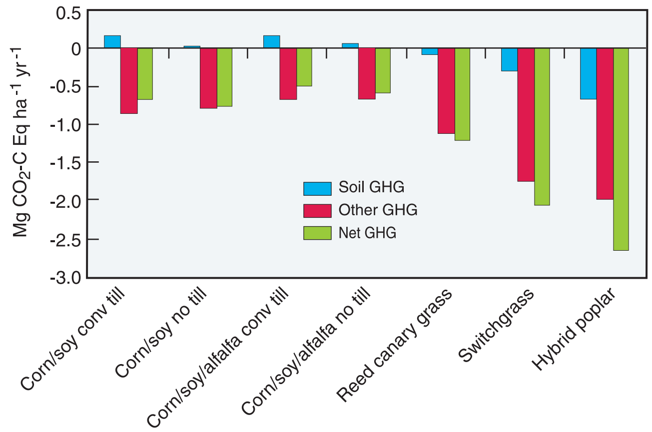

Figure 4.3. **Life cycle assessment of net greenhouse gas (GHG) balance for different grain and cellulosic biofuel cropping systems in Pennsylvania. Soil GHG includes changes in soil organic carbon and direct and indirect N_2O emissions; other GHG includes CO_2 emissions associated with production and application of farm inputs, operation of farm machinery, transport of biomass and conversion to fuel, and fossil fuel offset; net GHG is the sum of soil and other GHGs. Negative values indicate net GHG mitigation. Not included are fossil fuel offset credits (Adler, Del Grosso, and Parton 2007).**

is not replaced (Robertson et al. 2011). If a moderate amount of fertilizer N is added (e.g., 70 kg N ha^{-1} yr^{-1}), however, then production and soil C both increase without causing large emissions of N_2O. On balance, perennial systems harvested for biomass production can be a small soil GHG source for harvested prairie and a sink for fertilized prairie and switchgrass.

Conversion of native prairie or CRP lands to corn production does not necessarily decrease soil C if no-till cultivation is practiced (Figure 4.1). Increased production from added fertilizer increases C inputs to the soil while no-till management minimizes losses from decomposition so that soil C is maintained. Nitrous oxide emissions, however, are high with corn cropping systems that use conventional N fertilizers, and on balance these systems will be a net source of soil GHG emissions under conventional tillage (Figure 4.1), although these systems can still be a net GHG sink when the benefits of fossil fuel offsets and coproducts are considered in full lifecycle analysis (Adler, Del Grosso, and Parton 2007). Conventionally tilled corn-soybean and corn-soybean-alfalfa systems are a soil GHG source, but they are a net GHG sink when displaced fossil fuel and coproducts are included in a full GHG lifecycle analysis (Figure 4.3). In addition to the GHG of displaced fossil fuels and coproducts, the calculations represented in Figure 4.3 include the GHG costs of biomass transport and feedstock conversion to ethanol, production and transport of chemical inputs, and operation of farm machinery (Adler, Del Grosso, and Parton 2007).

Although perennial cellulosic biofuel systems have the largest GHG benefit, application of currently available technologies can improve the performance of annual grain-based systems used for ethanol production. Decreased tillage intensity can maintain or even build soil C, and using harvested stover for biorefinery heating (in the absence of cellulosic ethanol production) can improve the biorefinery's GHG balance. There are many N fertilizer technologies that can decrease N application rates and presumably N_2O emissions (Robertson and Vitousek 2009), including site-specific and on-the-go application methods as well as polymer-coated and other advanced fertilizer formulations (Halvorson, Del Grosso, and Alluvione 2010), although the efficacy of these methods has not been widely tested.

Conclusions and Future Considerations

Bioenergy is a major and growing part of the U.S. agricultural portfolio. Production is now dominated by grain-based feedstocks, primarily corn, which provides only modest GHG benefits owing to the amount of fossil fuel required to produce ethanol from corn and the large amounts of N_2O emitted from soils under annual crops. No-till management, residue retention, cover crops, and advanced fertilizer technology could improve the GHG performance of these systems, but they are not currently incentivized and therefore not widely practiced. Perennial cellulosic feedstocks offer a much greater climate benefit because of their lower dependence on fossil fuels (used only for harvest and perhaps N fertilizer after an establishment year), substantial belowground C storage, and low N_2O fluxes. Perennial cropping systems offer additional environmental benefits related to N conservation and biodiversity services such as pollination and pest protection.

Converting unmanaged land to bioenergy production can create significant C debt if sufficient care is not taken to avoid soil C loss and to establish a crop that captures more CO_2 than the plant community that is replaced. Indirect land use effects can also create C debt when existing food crops are converted to bioenergy crops and the difference in food production is made up elsewhere, with concomitant GHG costs. Indirect land use costs are difficult to calculate, imprecise, and contentious; they can be minimized or avoided altogether by planting bioenergy crops on marginal or other lands not currently used for food and fiber production. Models show the importance of treating bioenergy crops as systems, with the need to balance management options and productivity against the net GHG benefit of alternative management scenarios.

The commercialization of cellulosic biofuels, including developing end-products other than ethanol, promises to transform the agricultural energy sector. Genomic advances are likely to improve the productivity, energy yield, and diversity of crops suitable for energy production. Bioenergy crops done right offer substantial opportunities for providing GHG benefits.

5 Economics

Economic Fundamentals and Policy

Economics strongly influence adoption of greenhouse gas (GHG) emission lowering or sequestration enhancing practices. Some producers in the United States have already found it profitable to adopt the practices listed in Chapters 3 and 4. Broader adoption will occur only if the practices become profitable to a larger number of producers or in response to regulatory mandates. It is important to recognize that producers across the country face very different cost and production conditions. Thus, although specific production practices may be profitable for some producers in some locations, they are not likely to be profitable for all producers in all locations. For example, Chapter 3 suggests that adopting decreased tillage can often increase carbon (C) sequestration on annual cropland. For some producers, it is not economically superior to other production methods, based on cost and yield effects, increased yield risk, or some other barrier (e.g., the need for soil to warm up in the spring). Consequently, incentives may be needed to increase the number of producers that adopt this practice if it lowers GHGs. Incentives can take two main forms: a payment for lowering GHG emissions or a tax on emissions to encourage producers to switch to a technology that emits smaller quantities of GHGs. The size and types of incentives available to producers are primarily driven by the types of policies, rules, and eligibility criteria developed to lower GHGs. The "rules of the game" will drive the economic opportunities to adopt practices that increase C sequestration or decrease emissions of GHGs.

Policy Design—Taxes and Cap-and-Trade Policies

There are many ways to design adoption incentives (Keohane and Olmstead 2007; Tietenberg 2006) reflecting the many policy tools used to lower GHG emissions. Two common policies are emissions taxes and a market-based cap-and-trade system.

Under a policy of taxation, emitters of GHGs would face a tax on their emissions. It is expected that a tax on emissions will encourage emitters to adopt emissions reduction technologies and thereby lower their GHG emissions. Pollution reduction credits are not traded between sectors under a taxation scheme. The real question regarding the tax is what it will cover.

A tax only on fossil fuels would not fully encourage many agricultural practices that decrease GHGs. For example, several practices that enhance soil C sequestration would not be incentivized under a taxation system unless a tax credit was also provided for sequestration. A tax on inputs such as fertilizer and equipment (reflecting their GHG emissions) could provide agricultural producers with an incentive to change their land use mix. Bioenergy production described in Chapter 4 could be subject to taxes on inputs such as fertilizer and fossil fuels while receiving no recognition of fossil fuel emission offset or sequestration enhancement potential, which would decrease the incentive to adopt this and other practices that sequester C. In addition, under a taxation scheme, revenues from taxes accrue to the public sector (government). This pass-through of funds could be used for investment in energy-decreasing technologies but could also be diverted to support public goals other than decreases in GHGs.

A second policy is a cap-and-trade scheme. Under cap and trade, an overall limit (cap) on GHG emissions is set by a regulator and regulatory credits are issued equal in number to the level of the cap. In some cap-and-trade systems, emissions reductions from nonregulated entities also generate credits; these are referred to as offsets. The items that trade reflect reductions in GHG emissions and are commonly referred to as C credits (C-credits). They are, however, typically assumed to cover all GHGs with alternative GHGs converted to tonnes (megagrams [Mg]) of carbon dioxide (CO_2) equivalents (Eq) (based on their global warming potential). Regulated entities have several options available to comply with required emissions reductions; they can lower their own emissions and/or purchase regulatory credits from others as well as purchase offsets from unregulated entities if this is allowable under the policy design. In contrast to a system of emissions taxes, revenues from the sale of credits under a cap-and-trade system are likely to accrue to the private sector and thus encourage greater innovation.

Whereas regulatory credits and offsets share a common definition in terms of net GHG emissions and can be traded with each other in many markets, they are distinct products because they embody different kinds of risks and obligations (Williams, Peterson, and Mooney 2005).

Regulated entities that decrease GHG emissions by more than the number of credits they hold can sell the excess regulatory credits to other parties. This is likely to occur if the cost to them of lowering their GHG emissions (or sequestering additional C) is less than the market price of credits. Conversely, if it is less expensive for an emitter of GHGs to buy a regulatory or project-based credit rather than lower emissions, that emitter will buy these credits in the market. When regulated entities have emissions in excess of their permits, they must buy credits from others to cover all their emissions. Consequently, overall emissions are decreased to the cap level.

Trading of emissions credits between buyers and sellers establishes the market price. Globally, most GHG or C-credit trading currently takes place within a cap-and-trade regulatory framework. A cap-and-trade system is a policy option that provides an opportunity for agricultural producers to provide project-based emissions reductions to other sectors, but the real question involves the eligibility rules, i.e., what sectors are eligible to join a trading scheme and what practices within those sectors are considered eligible.

Yet another approach involves practice-based payments, in which incentive payments are offered to farmers who adopt particular GHG net emission-reducing practices. Such programs have been used extensively within the United States to encourage the adoption of soil and water conservation practices; e.g., the Wetlands Reserve Program, Conservation Reserve Program (CRP), and Environmental Quality Incentives Program. The effectiveness of these programs (and others) is discussed in Chapter 3. Similar incentives could be designed to lower GHG emissions or increase soil C sequestration. The government could generate the most cost-effective GHG offsets by focusing on the relative costs of practices in comparison to the amount of CO_2 Eq they remove from the atmosphere, i.e., provide incentives to adopt practices that result in the greatest reduction in CO_2 Eq dollar^{-1} of adoption cost. In principle, the funds for these purchases could be collected from fees levied on GHG emitters, setting up incentives similar to those where the emitters would be buying credits directly from farmers. These economic considerations will greatly affect the types of technologies adopted for mitigation and sequestration and the quantity of GHGs mitigated, including C sequestration.

Physical Potential versus Economic Potential

Often physical estimates of potential vastly overestimate what happens under implementation. The literature on the GHG reduction strategies is fraught with estimates that identify the physical "potential" of adopting a mitigation strategy, i.e., taking physical estimates of emissions reduction per hectare (ha), then multiplying that by total hectares and saying that is the potential. Such a procedure is a measure of physical potential and ignores the associated economics. Physical potentials overestimate likely GHG decreases because they do not account for the costs of adopting new technologies and practices or the possibility of higher economic returns from competing practices. Even at very high C-credit prices (up to $500 per tonne C Eq price), McCarl and Schneider (2001) show the economic potential to sequester C is less than half of the physical potential estimated by Lal and colleagues (1998). McCarl and Schneider (2001) also showed that at very high prices for GHG decreases, strategies like bioenergy and afforestation are more attractive than soil C sequestration and further decrease the economic potential for soil C sequestration to less than 1/10 of the physical potential suggested by Lal and colleagues (1998).

In fact, different practices will likely dominate at different market prices. At low prices for C-credits of under $20 per tonne, some researchers (Baker et al. 2010; McCarl and Schneider 2001; Murray et al. 2005) find that tillage and soil-based C sequestration practices are the dominant, low-cost technologies to lower GHGs. These are generally complementary to existing practices. At higher C-credit prices (i.e., greater incentives), however, decreased tillage is less dominant and is replaced by other land use change strategies that sequester C, offset fossil fuel use, or lower GHG emissions at a greater rate (generally forestry and bioenergy). These practices replace current land uses, achieving greater offset rates but forgoing current income sources.

Economic Aspects of Greenhouse Gas Policy Design

There are a number of economic aspects to consider related to the design of GHG policies. These design factors will influence costs and revenues as well as the potential for adopting different GHG mitigation technologies. A number of these factors are reviewed here.

Transaction Costs

The costs of developing sequestration or emissions reduction projects involve cost sources beyond the day-to-day costs of agricultural production activities; namely, in developing and selling credits, there are costs associated with assembling enough credits to fill a contract, monitoring compliance, and negotiating the contract, among other items (Mooney et al. 2004). These transaction costs are one of the greatest hurdles for tradable permit systems (Hahn and Hester 1989). Atkinson and Tietenberg (1991) review instances where the transactions costs were so high that they caused market participation to be substantially lower than expected. The existence of such costs means the prices received by farmers for their credits (or their net revenue) will be lower than the reported market prices, much like in the farm-retail price spread.

A few studies have estimated some of the transactions costs associated with measuring and monitoring C sequestered in agricultural soils under a C-credit trading system (Kurkalova, Kling, and Zhao 2004a; Mooney et al. 2004, 2007). Mooney and colleagues (2004) estimate that measurement costs range between 0.04% and 10.6% of credit price using a stratified sampling scheme with 5% error and 95% confidence. As the error bounds are expanded, or confidence levels are lowered, measurement costs decline because fewer samples are needed. Mooney and colleagues (2007) show that costs of measurement could be lowered further if more information was used to design a sampling scheme. Kurkalova, Kling, and Zhao (2004a) estimate that purchasers of soil C-credits could spend from 11.2% to 47.3% of their total C-credit expenditure on contract measurement and monitoring before the costs exceeded a payment scheme that provides a uniform payment per hectare.

These studies do not include the full range of transactions costs that would be present in implementing a project and thus may be underestimates. Additionally, policy design will influence these costs. Measurement technology (e.g., field sampling, remote sensing, or large-scale modeling), acceptable measurement error, the scale of projects, and other factors will affect measurement and other transactions costs. Generally, larger projects face measurement costs that are a small percentage of their total value. Also, as credit prices increase, measurement costs tend to form a smaller percentage of the total value of the project. Transactions costs may be project specific.

Typically the net GHG decreases from individual land parcels are too small to sell in a GHG market, and multiple parcels are assembled to fill one contract. For example, the Chicago Climate Exchange (CCX) required a minimum of 10,000 Mg of CO_2 Eq to form a single trade. At the CCX rate of ½ Mg CO_2 sequestered per acre (approximately 1.24 Mg ha^{-1}), 8,094 ha of no till are needed to support one contract, which, at a farm size of 200 ha, would require close to 40 farmers. As a result, many small land parcels were aggregated to enter into a CCX contract. The Iowa Farm Bureau charged a fee of 10% of the contract value to aggregate lands and producers into blocks large enough to sell on the CCX, whereas crop insurers charged 25%. The CCX was a pilot market and is no longer accepting additional C sequestration projects from agriculture.

Fundamental Economics of Carbon Offset Purchase

In 2008, agricultural emissions contributed more than 6% of total U.S. GHG emissions, while land use, land-use change, and forestry offset 14% of the total. In contrast, the energy sector accounted for 86% of U.S. GHG emissions (USEPA 2011). Consequently, to decrease emissions, the energy sector will likely be the first point of attention. So where does agriculture come in? The answer lies in the relative costs of emissions reduction. Namely, if agricultural offsets are to be economically competitive, they must be cost competitive with emission reductions in the energy and other sectors. This includes the production cost and any transactions costs attendant to conveying the credits.

Market Prices

Net GHG decreases in all sectors will be stimulated by high credit prices in the marketplace, but it is difficult to predict what the U.S. market price will be. The structure of any GHG policy will be one of the main drivers of credit price, because it will determine the demand and supply conditions faced in the market. Demand and supply are the factors determining market price. Williams, Mooney, and Peterson (2009) and Williams, Peterson, and Mooney (2005) summarized market conditions across current domestic and international markets, including the types of credits traded and recent prices. Prices seen in the European market are not a good predictor of U.S. prices because supply and demand conditions are likely to be different and the quality standards that credits are held to also differ (Williams, Mooney, and Peterson 2009; Williams, Peterson, and Mooney 2005).

Price Parity, Grading Standards, and Discounts

Several practices and technologies can be adopted to generate net GHG decreases in the agricultural sector, but these decreases and technologies will be evaluated against the full scope of possibilities from all industries by any credit purchaser. Many agricultural credits possess characteristics that may set them apart from credits generated by other industries and may differentiate their value, causing the need for grading standards.

A grading standard reflects differential use values on behalf of commodity consumers depending on commodity quality characteristics coupled with the production costs of achieving different commodity quality characteristics. For example, in the corn market there are differential prices per bushel depending on the moisture content, the incidence of foreign matter/broken kernels, and other factors. In a GHG market, the grading standards would reflect different credit characteristics important to the purchaser. Two examples of factors that may create differential prices for offset from different sources are (1) the extent to which the characteristics of the credit allow the purchaser to comply with the rules or regulations that define the GHG emission regulatory program, and (2) the amount of regulatory system credits that can be gained by registering the GHG decreases at hand. There are many other factors that could also create price differentials.

Characteristics that are important in some existing trading systems are global warming potential of a GHG, saturation and permanence (or lack thereof), additionality, leakage, and uncertainty. These are discussed in Chapter 6.

National and Regional Scale Analyses and Modeling

Several models have been used to examine the potential effects of a suite of policies on GHGs at the regional and national scale. For example, Antle and colleagues (2003); Kurkalova, Kling, and Zhao (2006); and Pautsch and colleagues (2001), used a range of modeling techniques and data sources to examine the regional adoption of alternative practices to enhance C sequestration under various policies. These models were constructed on a relatively aggregate level, e.g., state or major land resource area level, but examined a small suite of crop management decisions in detail. Very little regional modeling has been conducted to examine the economics of livestock and grazing practices to lower GHG emissions. Campbell and colleagues (2004) conducted a preliminary study for a location in Wyoming, but significant further research is required.

In contrast to the models above, Schneider (2000) and USEPA (2005) used the Forest and Agricultural Sector Optimization Model—Green House Gas (FASOMGHG), a national scale model with regional breakdowns (Adams et al. 2005), to examine the potential effects on the U.S. agricultural sector of policies lowering GHG emissions. In contrast to the region-specific studies, the FASOMGHG includes many agricultural sectors. Both regional and national economic models require significant economic and biophysical data collection. In the case of economic data, detailed descriptions of producer decisions are often required for in-depth modeling of a small number of practices. A significant challenge to collecting these data is that they are confidential and not universally available. As a result, many variables are obtained by conducting (costly) producer surveys, or less detailed secondary data are used from the Census of Agriculture or other federal statistics. Larger scale national models almost exclusively use secondary data sources because of the wide array of management decisions that need to be modeled. Information on prices, practices, and other relevant production data for different cropping or agricultural practices are then coupled with estimates of changes in GHGs as a result of adopting new management methods to estimate changes in production practices.

A combination of approaches is used to estimate GHG emissions from agricultural practices at the regional and national levels in the United States. These are discussed in more detail in Chapter 6. Although process-based models are likely to yield more reliable results than generalized emissions factors for soil C and soil nitrous oxide, the models have not been parameterized to represent all crops and all land uses. Economic analyses require this information to examine how crop production practices could change in response to policies that lower GHGs. Consequently, process-based simulation models are used to estimate GHG emissions for major crops (corn, soybean, hay, wheat, cotton, sorghum) and most grazed land on mineral soils at the subcounty level, and results are aggregated to state and national levels. Chapter 6 discusses the techniques available to measure GHGs, as well as their challenges.

Existing Markets

International and National Policy for Greenhouse Gas Reductions

Several countries have developed GHG markets to help meet their Kyoto Protocol obligations or other emission reduction needs. In the EU (European Union), mandatory GHG trading has occurred since 2005, within the EU-ETS (Emissions Trading System). Under the Kyoto Protocol and in the EU-ETS, there is participation by the agriculture and forestry sectors, including land afforestation and methane capture. No methodologies have been approved to certify tradable offsets from C sequestered through changed tillage practices (UNFCCC 2010b). The only methodology involving cropland practices, approved in 2008, allows for offsets to be generated by applying an inoculant instead of synthetic nitrogen (N) fertilizers on legume crops (UNFCCC 2010c).

In Canada, the government of Alberta has implemented mandatory GHG emissions performance standards for large industrial sectors. Since July 2007, facilities that emit more than 100,000 Mg of GHGs per year are required to lower their emissions intensity by 12% compared to their baseline. One of the options available to achieve this target is to purchase Alberta-based offset credits. Beef feeding and tillage management are eligible activities for credit generation (Alberta Environment 2011a; Alberta Environment 2011b).

At present, there is no U.S. federal policy that has given rise to a GHG market; however, a number of regional markets and some voluntary markets have emerged. Agriculture has a limited role in most of these schemes. The opportunities available to agricultural producers to sell C-credits may change in the future if new climate policies are adopted. Table 5.1 summarizes the main market types that exist at present.

Existing U.S. Policies and Markets

Mandatory State and Regional Policies

Although the United States has not ratified the Kyoto Protocol or passed domestic climate legislation at the federal level, there is significant interest in reducing GHG emissions at the state and regional levels. Some of the proposed initiatives are still in the planning phase, while others have been implemented. The agricultural and forestry sectors are not regulated under any of these initiatives; however, projects that lower GHG emissions or enhance C sequestration are eligible for payments in some instances. A summary of the main programs is provided below.

Oregon Carbon Dioxide Emissions Standards

In 1997, the Oregon legislature gave the Energy Sitting Council authority to set CO_2 emission standards from new energy facilities (State of Oregon 2007). Each plant must lower emissions and may implement a project themselves to decrease CO_2 or use what is referred to as the “monetary path” to lower emissions. Under the monetary path, the plant pays an organization called the Climate Trust $1.40/Mg CO_2 for excess CO_2 generated above the emission standard, plus transaction costs, to implement or participate in offset projects (Oregon Department of Energy 2010). Climate Trust projects for Oregon are currently limited to forest management. As of October 2010, there were no agricultural soil C sequestration projects undertaken by the Climate Trust, but they are exploring the possibility (Weisberg, P. 2010. Personal communication).

Regional Greenhouse Gas Initiative

The Regional Greenhouse Gas Initiative (RGGI) is an effort by ten Northeast and Mid-Atlantic states to develop a cap-and-trade program covering GHG emissions from power plants. The states included are Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont. It is the only mandatory cap-and-trade system in the United States (Kossoy and Ambrossi 2010). Some CO_2 decreases (up to 10% in some circumstances) can come from purchasing project offsets that lower CO_2 in other sectors and must be located in one of the participant states (RGGI 2010a). Eligible agricultural projects are limited but include avoided methane (CH_4) emissions from manure management and carbon sequestration due to afforestation of land previously in agricultural production (RGGI 2010a). Another potential opportunity is the biomass provision, because CO_2 emissions from

Table 5.1. Summary of main greenhouse gas markets and commodities

Market	Type	Credit Type	Uniform Commodity[a]
Regional Greenhouse Gas Initiative	Mandatory market—Cap and trade		
Western Climate Initiative	Allowances	Regulatory	Yes
	Rules for creation of offsets	Project based	Yes
Chicago Climate Exchange	Voluntary market (U.S.)—Cap and trade		
	Allowances (CFIs)	Regulatory	Yes
	Rules for creation of offsets (CFIs)	Project based	Yes
Contracts/tenders for projects that create offsets that are specific to a buyer's needs	Over the counter No mandatory rules for credit creation (some developed and being developed) No formal exchange Buying a service	Project based	Not necessarily[b]
Consumer offsets	Over the counter Variety of sources Buying a service Many not regulated	Project based	No
Kyoto European Union Emissions Trading Scheme	Mandatory markets—Cap and trade		
	AAUs	Regulatory	Yes
	ERUs	Project based	Yes
	CERs	Project based	Yes
	RMUs	Project based	Yes

[a]A uniform commodity is a commodity where each unit sold adheres to the same standards and specifications, i.e., each unit purchased will be approximately the same. Over-the-counter trades in specialized project and many consumer offsets do not conform to uniform guidelines.

[b]The Alberta Greenhouse Gas Reduction Program does have uniform standards for offsets.
AAUs—Assigned Amount Units
ERUs—Emission Reduction Units
CERs—Certified Emission Reduction Credits
CFIs—Carbon Financial Instruments
RMUs—Removal Units

the combustion of eligible biomass, including many produced by agricultural operations, can be deducted from the power plants' CO_2 compliance obligation. The prices within this market provide an indication of possible prices for agricultural biomass credits. Quarterly auctions began in September 2008, and allowance prices have ranged from a high of \$3.51 per tonne CO_2 Eq in March of 2009 to a low of \$1.86 per tonne CO_2 Eq in September 2010 (RGGI 2010b).

Western Climate Initiative

The Western Climate Initiative (WCI) includes seven states and four Canadian provinces: Arizona, British Columbia, California, Manitoba, Montana, New Mexico, Ontario, Oregon, Quebec, Utah, and Washington. Its goal is to decrease GHGs by 15% below 2005 levels by 2020 (WCI 2010a). A regionwide cap-and-trade system composed of individual partner jurisdictions' cap-and-trade programs is proposed to start trading January 1, 2012 (WCI 2010a). Regarding offsets, the WCI states, "To reduce compliance costs and encourage emissions reductions, offset certificates will reward emission reductions in sectors such as forestry and agriculture that are not covered by emission caps" (WCI 2010a). Offsets may be issued for projects located within and outside WCI jurisdiction. The role of agriculture offsets is uncertain at this time. Offset projects, which involve the risk of reversal or may be nonpermanent such as C stored in biomass or soil, must meet special rules. "If an emission reduction is reversed after credits are issued, the project developer must replace the reversed credits with other compliance units from within the system or return credits that were issued to the project" (WCI 2010b). The WCI criterion also requires that sequestered C be stored at least 100 years. The long-term persistence of soil C is difficult to assure over this period, and it is likely that developers would have to replace these credits over time as the soil C is released.

California Global Warming Solutions Act

The California Global Warming Solutions Act (CGWSA), more commonly referred to as AB32 (Assembly Bill 32), was signed into law on September 27, 2006. The act sets a statewide GHG emission cap that proposes to lower emissions to 1990 levels by the year 2020 with a further goal of 80% below 1990 levels by the year 2050 (CARB 2011). One of the key tools proposed to meet these goals is a cap-and-trade program. Although AB32 only applies to California, the cap-and-trade system was designed more broadly to be a regional program through the Western Governors' Initiative in conjunction with six other U.S. states and four Canadian provinces. Under the CGWSA, offsets can be used to cover up to 49% of emissions reductions (State of California 2006). Three of the four approved offset protocols provide opportunities to agriculture and forestry. Cap-and-trade regulations were adopted by the California Air Resources Board (CARB) in December 2010 with the intention of launching a cap-and-trade system by 2012. At present the cap-and-trade plan is subject to litigation within the state. More details on AB32, the cap-and-trade regulations, and associated analyses can be obtained from the CARB website (www.arb.ca.gov).

Voluntary U.S. Market

Perhaps the best-known cap-and-trade system for GHGs in the United States was the CCX, a voluntary pilot program launched in early 2003 (CCX 2010a) that ceased operations at the end of 2010. The CCX had approximately 450 members including industries from a number of sectors, municipalities, countries, state governments, offset aggregators, offset suppliers, liquidity providers, and others (CCX 2010a). Phase II of trading on the CCX began in 2007 and ended in 2010. No further trading is expected. Membership in the CCX was voluntary, but each member was obligated to lower GHG emissions by 6% below those in the 1998–2001 baseline period by 2010 (Capoor and Ambrossi 2007). Prices on the CCX since January 2008 have ranged from approximately \$0.10 to \$7.40/Mg CO_2 Eq.

More than 15,000 farmers, ranchers, and foresters (and more than 10.12 million ha of land) were enrolled in CCX projects to provide offsets (CCX 2010a). Several covered opportunities allowed one to generate credits by changing agricultural practices to encourage cropland and rangeland soil C sequestration as well as CH_4 reduction from livestock operations (CCX 2010b). Eligible projects for soil C-credits included continuous conservation tillage and grass planting. Under conservation tillage, estimates of C sequestration per hectare range between 0.5 and 1.5 Mg CO_2 ha^{-1} yr^{-1} (per hectare per year). For grassland, the assumed rate of C sequestration was between 1.0 and 2.5 Mg CO_2 ha^{-1} yr^{-1} (CCX 2009a). During 2004 through 2009, approximately 82,000,000 Mg CO_2 Eq of offsets were registered with CCX (CCX 2009b), of which approximately 33% were agricultural soil carbon offsets. The second and third largest categories were coal mine CH_4 collection and combustion (22%) and forestry C sequestration (17%). Offsets from agricultural CH_4 collection and combustion were only 1.8%. Kossoy and Ambrossi (2010) report that the volume of Mt (megatonne) CO_2 Eq, prices, and interest in the market declined after the Waxman-Markey bill did not explicitly include CCX credits for future use in the federal cap-and-trade program.

Voluntary Nonstandardized Trading

A variety of other voluntary trading is taking place. These trades are not always associated with a cap-and-trade system or other forms of regulation. Offsets within the voluntary market are commonly generated from projects that create emission reductions. Both nonprofit and for-profit organizations coordinate and manage these projects. They generally do not operate through formal trading markets or exchanges and are referred to as over-the-counter (OTC) trades. Examples of these OTC trades are in markets where individuals purchase GHG offsets to cover all or some of their annual CO_2 emissions from their auto, plane trips, and/or household energy consumption (among other emissions). The firm that issues the offsets invests a portion of the money paid by the purchaser in GHG reduction projects that may include, but are not limited to, renewable energy industry projects such as wind power and solar power. Other projects include CH_4 captured from landfills and coal mines. Agricultural projects may include CH_4 captured from manures and C sequestration in forests, cropland, and rangeland. The Stockholm Environmental Institute provides a list of voluntary C offset sources at http://www.co2offsetresearch.org/consumer/Providers.html.

Hamilton and colleagues (2009) report that suppliers of offsets include developers of GHG offset projects, aggregators that have ownership of a portfolio of credits from offsets, retailers selling directly to individuals or organizations that may have purchased offsets from aggregators, or project developers and brokers that facilitate transactions between buyers and sellers. Buyers could be individual consumers, groups, firms, other organizations, and government entities. Their motivation may be an interest in corporate responsibility improving the environment, marketing or public relation advantages, philanthropy, profitability from resale, and gaining experience for expected cap-and-trade policy. The worldwide voluntary market in 2009 was 93.7 Mt CO_2 Eq compared to 123.4 Mt CO_2 Eq in 2008, 66 Mt in 2007, and 23.7 Mt in 2006. Approximately 56% of the transactions in 2008 were through the CCX (Hamilton et al. 2007; Hamilton et al. 2009; Hamilton et al. 2010). The remainder of the trading was not through formal exchanges. Hamilton and colleagues (2009) report that of the approximately 14.5 Mt CO_2 Eq traded in OTC markets in 2008, 0.3 Mt CO_2 Eq was from agriculture soil sequestration and 1.2 Mt CO_2 Eq was from livestock CH_4 collection and combustion.

The most significant factors constraining demand in the U.S. C-credit market at present are the lack of binding emission reduction targets at the federal level and lack of a clear U.S.-wide GHG reduction policy (Young 2003). Opportunities for agriculture to participate in occasions to lower GHG emissions and sequester additional soil C will be affected by the type of policy instrument chosen for regulation (tax or cap-and-trade system). Most current bills in the U.S. House and Senate favor a cap-and-trade system, and many specifically include agriculture as a sector that could benefit from selling GHG mitigation and reduction projects in the marketplace. The actual market price for agricultural offset projects will be affected by the design of the regulatory policy that influences the demand for and supply of credits (Williams, Mooney, and Peterson 2009; Williams, Peterson, and Mooney 2005). Credit prices will also be affected by the rules for generating credits from projects that sequester C in managed ecosystems such as agriculture and forestry.

Cobenefits of Carbon Sequestration

Because of numerous interlinkages in natural ecosystems, the agricultural management practices that sequester C may simultaneously have other, mostly beneficial environmental co-effects commonly referred to as cobenefits. The cobenefits are improvements in environmental factors such as decreased soil erosion, decreased N and phosphorus surface runoff, and improved wildlife habitat. Many cobenefits arising from the adoption of different practices to lower GHG emissions are discussed in Chapters 3 and 4. The provision of these cobenefits often leads to the use of the term "win-win." These benefits from agricultural practices, however, need to be considered carefully against reductions in GHG emissions by the other sectors that could result in a different combination of cobenefits.

Recent studies suggest several reasons why it is important to examine and quantify cobenefits from C sequestration (Elbakidze and McCarl 2004, 2007; Feng et al. 2007). First, the magnitude of cobenefits determines whether or not GHG policy needs to address these cobenefits explicitly. The cobenefits represent the benefits that accrue primarily to society at large. If farmers are not compensated for these external benefits, they will not take the societal value of cobenefits into account when deciding whether or not to change farming practices to sequester additional carbon. In consequence, the cobenefits constitute externalities to a GHG offset market and too much or too little C may be traded in the market (or provided in response to a GHG mitigation policy) relative to what is optimal for society. If the cobenefits are relatively large, they justify government intervention to correct for the inefficiencies; if the cobenefits are relatively small, then they may not justify resources needed to alter the GHG mitigation policy or carbon market.

Second, although the generation of GHG offsets may significantly change the level of provision of other ecosystem services, the distribution of cobenefits could be very uneven across geographic areas and/or C sequestration practices (Feng et al. 2007; Kurkalova, Kling, and Zhao 2004b; Secchi et al. 2007). Depending on the activity that generates GHG offsets, the geographic areas that can sequester the most C may or may not be the same areas that provide most of the cobenefits such as soil erosion or nutrient loss reduction. The degree to which a policy that maximizes one environmental outcome will also maximize the total amount of the other environmental outcomes depends on the specifics of the farming practices, cropping patterns, and soils. For example, Kurkalova, Kling, and Zhao (2004b) estimated that a policy that targets C sequestration through an increased use of conservation tillage in Iowa could get high proportions of the total possible erosion and nutrient runoff reduction under realistic policy budgets. In contrast, Feng and colleagues (2007) estimated that targeting CRP enrollment based on C sequestration cobenefits would generate only small proportions of the total GHG offsets that could be potentially generated had the enrollment specifically targeted the fields with the highest C sequestration potential.

Third, the coexisting GHG offset markets and/or policies and the cobenefits-driven agri-environmental policies may affect the adoption of the same farming practices. Not accounting for the effects of the cobenefit targeting policies may lead to inaccurate assessment of the GHG baselines. For example, a large, continuing effort to improve water quality in agricultural watersheds has been affecting the use of many farming practices that impact GHG emissions, such as conservation tillage and cover crops. These changes on agricultural landscapes continue to alter the baseline in relation to which the GHG mitigation policy would need to be measured (Lawrence 2010; Osmond 2010). Likewise, policies that incentivize strategic removal and repositioning of riverside levees and creation of connected floodplains for decreasing flood risks often enhance numerous other ecosystem services, including C sequestration, in the affected floodplains (Opperman et al. 2009).

On the other hand, arguments can be made that cobenefits are misleading and should be neglected. Although the provision of these cobenefits often leads to the use of the term "win-win," they need to be considered carefully, as under a cap-and-trade system they may offset reductions in GHG emissions by the energy sector that could also result in a different combination of cobenefits. Elbakidze and McCarl (2007) examined the instance where use of agricultural sources allowed coal-fired emissions to continue and found the cocosts from allowing that were approximately equal to the water quality and other benefits from the agricultural activities. This means that it may be desirable to exclude such cobenefits in cost-benefit calculations, as argued in the context of water resources project appraisal in, for example, Stovener and Kraynick (1979).

Limited data are available on the magnitudes of the GHG mitigation cobenefits. As with the soil C sequestration potential, the magnitudes of the cobenefits depend on the specifics of soils, landscapes, present farming practices and cropping patterns, and climatic factors (Elbakidze and McCarl 2007; Mooney and Williams 2007). In addition to the challenge of estimating the physical measures of the cobenefits, researchers and policymakers are confronted by the need to convert the physical measures of, say, decreased erosion into economic (monetary) values. Consistent with the general law of demand for environmental goods and services, the monetary valuation of the cobenefits depends on the amount of presently existing environmental services. That is, other things being equal, the higher the quantity of a benefit the lower the willingness to pay for the next unit of environmental improvement. In consequence, the estimates obtained under the assumption of constant willingness to pay for cobenefits may be overstating the value of the potential cobenefits (Elbakidze and McCarl 2007).

Unlike the decreases in GHG that are of the same value to society no matter where they occur (Antle and Mooney 2002), the cobenefits related to water quality may be nonadditive: a decrease in N runoff close to a river may be more valuable to society than the decrease in the runoff on a field located further from water bodies. Consistent with the general theory of demand, other things being equal, the higher the number of people potentially affected by an environmental improvement, the greater the aggregate population's willingness to pay for the environmental improvement. This tendency results in the valuations of many cobenefits varying, depending on whether the affected regions are remote or close to populated areas (Elbakidze and McCarl 2007; Mooney and Williams 2007).

Elbakidze and McCarl (2007) and Mooney and Williams (2007) provide recent, comprehensive reviews of the magnitudes of the known cobenefits of agricultural GHG mitigation activities. Elbakidze and McCarl (2007) focus primarily on the cobenefits of afforestation activities and summarize the co-effects in monetary terms. Mooney and Williams (2007) review the estimates separately by the classes of cobenefits (water erosion, wind erosion, nutrient runoff and water quality, and wildlife and recreation) and provide the estimates both in physical terms and in combination with monetary valuation. Ongoing work on the improvement and expansion of the methods for transferring the economic valuations of environmental benefits to alternative geographic areas and/or populations constitutes a promising future source of models and data for connecting the physical measures of environmental improvement with the economic value of the cobenefits (Navrud and Ready 2007).

The present discussion omits the often considered pecuniary cobenefits in terms of economic support to farmers (Elbakidze and McCarl 2007; Feng et al. 2007; Mooney and Williams 2007). Feng and colleagues (2007) also note that the political support for GHG policy may be strongly linked to the expected changes in farmers' income that such policy may provide.

Finally, not all environmental co-effects of C sequestration activities are beneficial. For example, a switch from conventional to conservation tillage, considered beneficial for C sequestration on some soils, may also result in increased nitrate leaching (Meisinger and Delgado 2002). Moreover, lower tillage intensity has been commonly associated with increased use of pesticides (Fuglie 1999; Mooney and Williams 2007).

6 Implementation and Policy Issues

National Inventories

United Nations Framework Convention on Climate Change and Intergovernmental Panel on Climate Change Guidelines

In 1994, many countries joined the United Nations Framework Convention on Climate Change (UNFCCC) to address global warming and its consequences. Signatory nations agreed to report their national greenhouse gas (GHG) emissions annually to the UNFCC using agreed-on accounting methodologies developed by the Intergovernmental Panel on Climate Change (IPCC). Signatory nations can follow the revised 1996 guidelines (IPCC/UNEP/OECD/IEA 1997) or the 2006 guidelines (IPCC 2006). The 2006 guidelines define three methodological tiers and also address uncertainty assessments. Tier One methods are the easiest to use and use default emission factors and country-specific activity data to estimate emissions. Emission factors define GHG emissions per unit of agricultural activity; e.g., ten metric tons of CO_2-C (carbon dioxide-carbon) is emitted annually for every hectare of annual cropland on drained organic soils in warm temperate zones. Tier Two methods use the same approach as Tier One but apply country- or region-specific emission factors and typically require more disaggregated activity data. Tier Three methods use process-based simulation models and/or inventory monitoring systems (e.g., Del Grosso et al. 2010; Ogle et al. 2010). Models must be validated, and monitoring systems must have sufficiently detailed spatial and temporal resolution.

The IPCC guidelines recommend including estimates of uncertainty in activity data and emission factors. For Tier Three methods, the emission factor uncertainty involves an assessment of uncertainties in the model structure and parameterization. Methods to combine uncertainties are also prescribed, including simple error propagation and Monte Carlo approaches. Higher tier methods are expected to provide estimates of greater certainty than lower tiers; e.g., uncertainty in emission factors is greater for default (Tier One) than country-specific (Tier Two) factors. In theory, Tier Three methods should provide the most certain estimates because of greater specificity in the derivation of emission factors.

The IPCC guidelines also suggest accounting for how the following variables influence soil GHG emissions: current land use (e.g., cropland vs. grazed land), previous land use (e.g., land converted to cropland vs. land remaining cropland), climatic zone, soil type, and various land management practices. Emission factors for Tier One or Tier Two approaches are intended to account for the impacts of these different land management and environmental variables on emissions, but not necessarily their interactions. In contrast, process-based models used for the Tier Three approach can be used to estimate emission factors accounting for the influence of these variables as well as their interactions. Although Tier Three methods should yield more accurate and precise estimates, most nations use Tier One and Tier Two methods, mainly because Tier Three methods require extensive resources to develop and validate model outputs, acquire model input data, execute simulations, process model results, and verify quality control.

United States Soil Greenhouse Gas Inventory

The United States uses a Tier Three approach to estimate soil C stock changes and nitrous oxide (N_2O) emissions for major cropping systems (corn, soybean, wheat, hay, sorghum, cotton) and nonfederally managed grasslands used for livestock grazing. A Tier Two approach is used for soil C stock changes for minor crops and federal grasslands, while a Tier One approach is used to estimate N_2O emissions from minor crops, cropped and grazed organic soils, and federal grasslands, as well as methane (CH_4) emissions from flooded rice paddies (USEPA 2010).

Agricultural soils in the United States are responsible for about half of total agricultural emissions in the country, with the remaining due primarily to CH_4 emissions from enteric fermentation and CH_4 and N_2O emissions from managed manure systems. Nitrous oxide emissions account for the vast majority of soil emissions because other key soil emissions, such

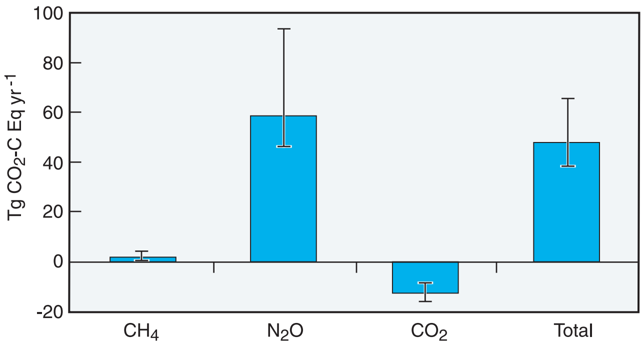

Figure 6.1. **Methane (CH_4), nitrous oxide (N_2O), carbon dioxide (CO_2), and total greenhouse gas fluxes for agricultural soils in the United States during 2008 with 95% confidence intervals. Negative values represent a greenhouse gas sink.**

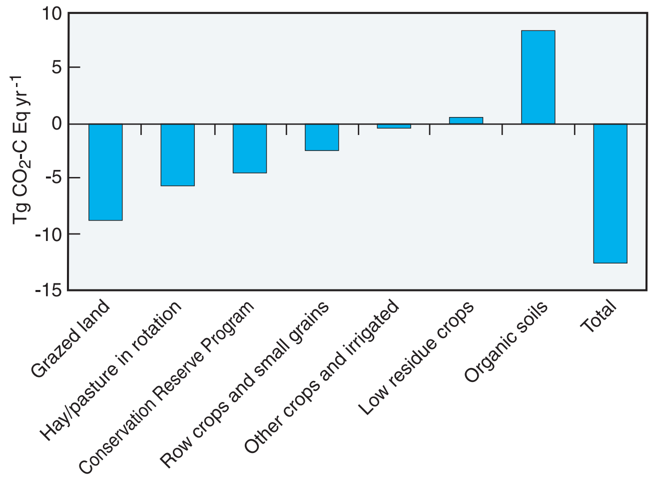

Figure 6.2. **Soil CO_2 fluxes for different agricultural land uses in the United States during 2008. Negative values represent a greenhouse gas sink.**

as CH_4 from rice paddies, generate a small portion of total agricultural land and agricultural soils are a net CO_2 sink (Figure 6.1). In aggregate, agricultural soils are estimated to be a GHG source of 48 teragrams CO_2-C Eq per year with a 95% confidence interval of -19 to +37%. Mineral soils are a fairly substantial C sink, but this is offset by drained organic soils used for crop production and grazing (Figure 6.2). Although cropland and grassland organic soils make up a very small portion of agricultural land (less than 1%), they are responsible for a disproportionately high share of CO_2 emissions because organic soils are carbon rich, and draining them greatly accelerates decomposition. The main land uses responsible for the CO_2 sink in mineral soils include grazed lands, Conservation Reserve Program lands (i.e., reserve cropland), and hay or pasture in rotations with annual crops. Decreases in tillage intensity have also contributed to C sequestration in soils used for row crop and small grain production. Over the last couple of decades, many farmers have converted to no-till or reduced tillage cultivation, leading to C sequestration in soils.

Nitrous oxide emissions are high in regions with intensive row cropping (Corn Belt states); in Texas, which has a large population of beef cattle as well as row and hay cropping; and in California, which has a large population of dairy cattle in addition to extensive row and specialty crop production (Figure 6.3). Soils in some states of the Corn Belt and Great Plains are CO_2 sinks, while Florida, Texas, and Minnesota are CO_2 sources (Figure 6.4). Florida and Minnesota are net CO_2 sources because the emissions from organic soils exceed any C sequestration in mineral soils.

Natural Resource Inventory Soil Carbon Monitoring Network

Verification of emissions and C sequestration is essential for reporting, particularly if there is a commitment to lower emissions. To verify future emission decreases due to C sequestration in agricultural soils, a national-scale soil monitoring network is under development that will track changes in soil C stocks and other soil variables for agricultural lands. The network is intended for long-term measurement of agricultural soils and is expected to continue monitoring soils for several decades or longer. Similar monitoring of soils currently occurs in other countries (e.g., Sleutel et al. 2003; van Wesemael et al. 2011) and is anticipated to provide a wealth of information for U.S. policymakers.

The measurement network will comprise a subset of survey points included in the U.S. Department of Agriculture National Resources Inventory (USDA–NRI), which is used for monitoring land use and management trends in the United States (Nusser and Goebel 1997). Several thousand sites will be sampled every five to ten years. These data can be directly associated with historical management, which is important because soil C pools change more slowly than other pools and a change in management can have an influence on trends for more than a decade.

The soil C measurements from the NRI network will be used to verify and ultimately assess uncertainty in the model-based estimates of soil C stock changes, which are produced using a Tier Three method (Ogle et al. 2010; USEPA 2010; see previous chapter). The model-based estimates will be adjusted

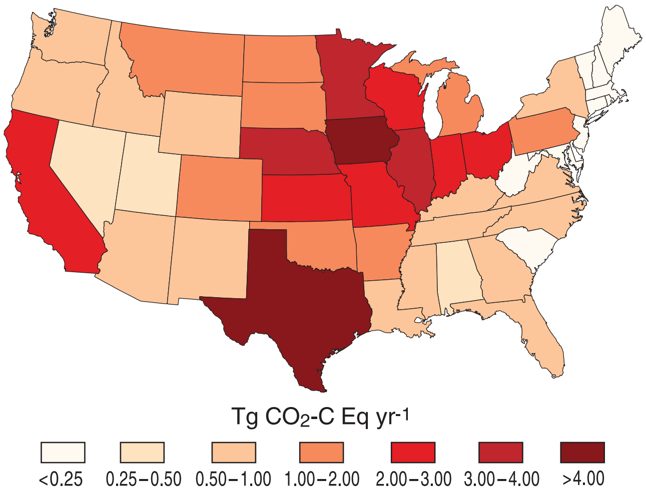

Figure 6.3. **State-level N_2O emissions from agricultural soils in the United States during 2008.**

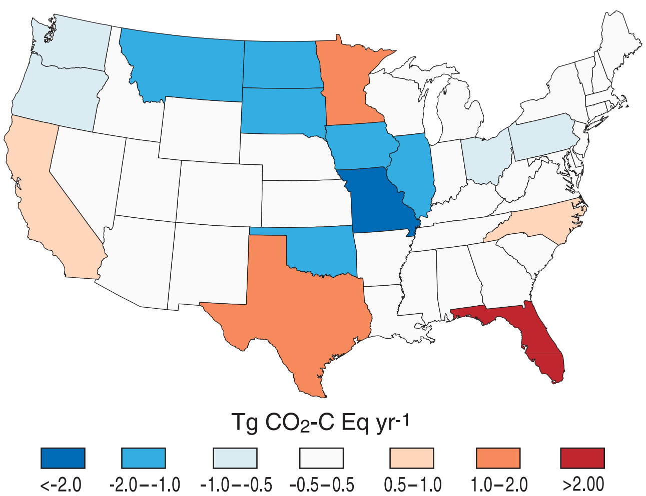

Figure 6.4. **State-level CO_2 fluxes for agricultural soils in the United States during 2008. Negative values represent a greenhouse gas sink.**

for bias and imprecision based on statistical analyses. This is an empirical approach that has already been tested and implemented in the U.S. inventory, but with a limited number of agricultural experiment sites (Ogle et al. 2007). Expanding to the larger network will allow for more generalization about the results and provide the verification of C sequestration activities that is needed for reporting purposes. Moreover, these results will provide a full analysis of the influence of all agricultural management on soil C stocks, which will account for leakage if and when it occurs within the United States. The results will provide additional insight into resource trends,

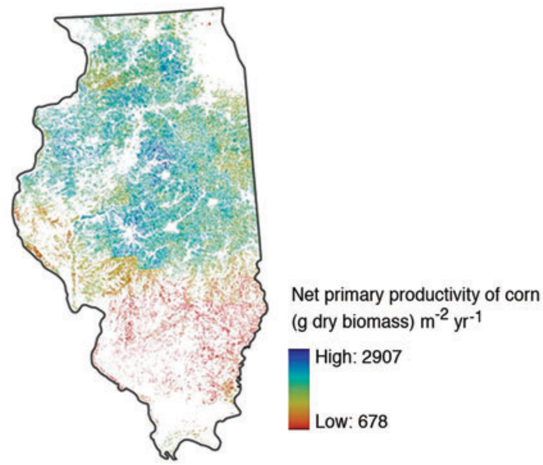

Figure 6.5. **Net primary productivity for corn in Illinois for 2007. Net primary productivity is the net production of plant biomass from photosynthetic assimilation of CO_2. It was estimated here using a light-use efficiency model, crop-specific parameters, and MODIS satellite data (MOD09Q1G) to estimate the amount of carbon dioxide taken in by plants minus the carbon dioxide emitted during respiration (Bandaru, V. [Joint Global Change Research Institute]. 2010. Unpublished data).**

informing USDA program initiatives by explicitly incorporating soil measures into farm policy analysis, which will include the development and assessment of national GHG mitigation programs.

Other Soil Carbon Stock and Flux Monitoring Networks

Additional methods of soil C monitoring are available for use independently or for use within existing monitoring frameworks. Satellite remote sensing, geospatial modeling, eddy covariance flux measurements, and atmospheric inversion modeling can each contribute to an improved monitoring network.

Satellite remote sensing can be used to estimate aboveground biomass (Figure 6.5), surface residue mass, leaf chlorophyll, leaf area index, and cellulosic absorption index (Brown et al. 2010). These components are key parameters in estimating annual changes in soil organic C. The temporal and spatial resolutions of remote sensing products have increased such that individual crop species can be identified without having multiple crop species or fields represented by a single pixel or reflectance value. Remote sensing products need to have a spatial resolution of <250 meters (m) to be useful for cropland delineation, with best results obtained from using a resolution of

50 m or less. Temporal resolutions of five days or less are needed to identify plant emergence and senescence, as well as to develop crop phenology curves that differentiate between major crops. Although the Moderate Resolution Imaging Spectroradiometer has high temporal resolution, Landsat and the Advanced Wide Field Sensor have high spatial resolution. At present, soil C cannot be monitored directly by remote sensing.

The use of geospatial modeling, including gridded spatially delineated data, improves modeled representation of the natural system. Improved representation of existing soil attributes, weather, land management, and crop rotations can all improve estimates of soil C. Gridded high-resolution data (<100 m) can now be handled for entire continents using appropriate computational resources and software (West et al. 2010). Integrating geospatial modeling with existing process-based biogeochemical modeling may likely contribute to development of the next generation of soil C models. Additionally, higher resolution gridded data will help monitor changes in annual land management and can act as verification and attribution of estimated soil C stocks and C fluxes.

Exchange of CO_2 between terrestrial ecosystems and the atmosphere can be estimated using eddy covariance flux methods. Carbon flux from soils can be estimated by subtracting harvested grain and fossil-fuel emissions from the measured total flux (Bernacchi, Hollinger, and Meyers 2005). Estimated C fluxes can be used to constrain biogeochemical models that estimate changes in soil C stocks. Networks of eddy flux towers can therefore provide regional constraints on national estimates of soil C change. A combination of process-based model output and eddy flux estimates can be used as input to atmospheric inversion models. Inversion models track the lateral flow of CO_2 across continental landscapes and can estimate regions of CO_2 sources and sinks (Schuh et al., 2010) (Figure 6.6).

Although changes in soil C stocks are difficult to ascertain from total atmospheric CO_2 movement, because of the small fraction of soil C flux compared to total plant growth and total fossil-fuel emissions, inversion models can serve as a validation for total modeled fluxes (Nisbet and Weiss 2010). Comparisons between flux tower data, process-based models, and atmospheric inversion estimates are currently being conducted within the North American Carbon Program. After comparison of these results, efforts may continue in the integration of these methods for a more comprehensive monitoring framework by utilizing the components of remote sensing and geospatial modeling.

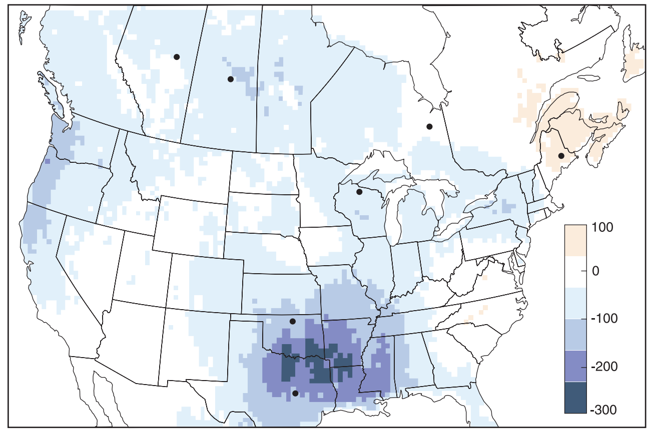

Figure 6.6. Atmospheric inversion modeling results illustrating a regional carbon sink in the southeastern/south-central United States in 2004. Units are g C m^{-2} yr^{-1} (Schuh, A. [Colorado State University]. 2010. Unpublished data).

Implementing Project and Farm-level Mitigation Activities

Individual land owners may engage in GHG mitigation efforts for a variety of reasons, such as a pursuit of income under credit trading systems, a desire to practice good environmental stewardship, or a desire to earn the benefits accrued by participating in government-sponsored mitigation programs. Regardless of the motivation, adoption of improved management practices to lower agricultural GHGs will be implemented at the farm level, and thus estimates of emissions (and emission reductions) are also needed at the farm scale.

Although the specific requirements for farm-level emission inventories will vary as a function of the policy design and objectives, some common attributes can be identified. First, estimates need to be unbiased and sufficiently accurate and precise. The degree of accuracy/precision required is, in part, a function of policy and decision-making requirements, so there is no single standard that can be specified for all instances. All estimates will have some error, and thus it is important that the uncertainty of estimates be known as well as the costs of achieving greater certainty. All other factors remaining the same, accuracy and precision are inversely related to measurement costs. For a policy to be practical, measurement costs should be a relatively small fraction of the value it produces. The quantification system also needs to address the large variety of practices that can be used for mitigation and their interactions with each gas—CO_2, CH_4, and N_2O—to do a full GHG accounting. The methods need to be applicable to the whole United States, but also locally specific, because the impacts of different mitigation practices are in many instances sensitive to soil and climate conditions at the local scale.

Three general approaches could be used for quantifying the GHG net emissions decrease from farm-level mitigation practices. In theory, direct measurement of all fluxes would provide the most accuracy and precision. Direct measurement of many fluxes, however—such as N_2O and CH_4 emissions from soils, livestock, and manure and biomass burning—is not feasible. For other emission categories such as soil C stock change, estimates based on on-farm measurements are feasible (Mooney et al. 2004), but for small mitigation projects they may be too expensive as a stand-alone approach.

At the other end of the spectrum are broad national or regional practice-based estimates such as developed for the Chicago Climate Exchange voluntary emissions trading. Although this type of approach is relatively easy to implement, it has a number of disadvantages. Because of the high degree of aggregation, there is a high probability of bias when applied to a specific area. Broad practice-based approaches can also be economically inefficient in terms of cost per tonne of net emissions reduction (Antle et al. 2003) in that they can disproportionately attract "under-performers" and inefficiencies increase with increasing spatial heterogeneity in climate and soil conditions. The inefficient aspects of this simple approach to measurement need to be weighed against the higher costs of implementing better measurement technologies. Finally, because many different components of an overall farming system (e.g., tillage, crop rotations, nutrient management) interact and can have contrasting effects on different GHGs, single practice standards cannot, in many instances, be rigorously defined. Thus, a third alternative approach to quantifying farm-level emissions is to use integrated models that can capture local-scale influences of soil, climate, and past land management, as well as deal with the interactive effects of different practices and multiple gases.

A variety of protocols, decision-support tools, and models have been developed (and more are under development) to support farm-level GHG estimation for potential use in GHG mitigation policies as well as voluntary emissions offset markets. Protocols for measurement of soil C stocks changes for projects engaged in C offset markets, using traditional field sampling methods, have been published (e.g., Pearson, Walker, and Brown 2005; Ravindranath and Ostwald 2009; Smith et al. 2007), although no "universal" measurement protocol is currently recognized. A somewhat different approach, using remeasurement of precisely located microplots (Ellert, Janzen, and Entz 2002; Spencer et al. 2012), may greatly lower sampling intensity required to detect soil C stock change over time (Lark 2009). A pilot effort to develop a U.S.-wide soil monitoring network using this approach is currently underway (Ogle, S. Personal communication; Spencer et al. 2012).

Various model-based farm-scale GHG calculators have been developed for providing estimates of GHG emissions and/or C stock changes for use by farmers, consultants, or other nonspecialists. These tools generally use an emission factor approach implemented within a spreadsheet environment most commonly based on IPCC emissions factors, although a few use factors derived from process-based models. In many instances a limited number of management systems and practices are represented, with limited specificity for soil and climate type, making them most applicable for use in broad, regional practice-based approaches.

Of the more comprehensive systems, the DNDC calculator (www.dndc.sr.unh.edu/?page_id=4) uses a process-based model to simulate CO_2, N_2O, and CH_4 emissions for U.S. cropland systems. HOLOS is a farm-level calculator (www4.agr.gc.ca/AAFC-AAC/display-afficher.do?id=1226606460726&lang=eng) for Canada that uses IPCC emission factors and a soil C stock change factor derived from the Century model to estimate soil C and N_2O emissions from cropland and pastures as well as CH_4 and N_2O emissions from livestock management. Cool-Farm (www.unilever.com/aboutus/supplier/sustainablesourcing/tools/) is another farm-level calculator, developed in the United Kingdom, which uses IPCC emission factors for C stocks and N_2O and CH_4 emissions from fields and livestock. It also includes capabilities for including up- and down-stream emissions for analyzing food supply chains. The COMET system is an online tool (www.cometvr.colostate.edu/; www.comet2.colostate.edu/) that performs on-the-fly simulations using Century and DayCent models for soil C, biomass C stock changes, and N_2O emissions from soil for annual and perennial crops (e.g., orchards, vineyards), pasture, range, agroforestry practices, and fossil fuel consumption. It also incorporates information on past land use history, computes baseline and mitigation scenarios, and estimates uncertainty for the fluxes. The COMET tool (Paustian et al. 2009) was developed originally for the U.S. voluntary emissions reporting system (1605B program). A new version of the system, COMET-Farm, uses a fully spatial interface at the field scale and also includes emissions from livestock management.

Although modeling approaches are initially time consuming and costly to develop, once developed, they are a relatively low-cost means for estimating emissions reductions and C sequestration under a wide range of conditions. Periodic and ongoing groundtruthing is beneficial to judge how well the models are performing.

The rapid development of user-friendly tools that also can incorporate state-of-the-art models and fine-scale information on soil, climate, and management variables can help support science-based mitigation activities for U.S. agriculture. Equally important is the continued expansion of field measurements and monitoring systems to improve the underlying models and provide solid estimates of uncertainty (Conant et al. 2010; Paustian, Ogle, and Conant 2010; Spencer et al. 2012; van Wesemael et al. 2011).

Greenhouse Gas Accounting Issues

Carbon Greenhouse Gas Market Design Issues

A number of major issues have emerged during international and domestic discussions of GHG market design. Some of the most contentious factors are the relative values of some classes of credits and the eligibility of some practices within potential markets. The major areas of discussion are summarized below.

Saturation and Permanence

Carbon accumulation does not continue indefinitely after a practice has been adopted to increase C sequestration. At some time after the adoption of a new practice, C accumulation will level off, i.e., saturation occurs, and a new equilibrium C content will take place. For example, accumulations of C after reductions in tillage intensity occur initially but may cease after as little as 15 or 20 years (West and Post 2002). A subsequent additional change in practice may result in additional C sequestration. Longer periods of C sequestration occur with grassland and forest establishment. With all practices, however, a reversal in the land use causes C to fall back to a different equilibrium. This raises questions regarding the permanence of any changes that have been adopted. Thus, there are two important characteristics to consider related to practices that increase soil C: the time over which the sequestration accumulates and the maintenance of the C after accumulation ceases. This means that payment schemes may need to consider maintenance and differential accumulation amounts. Kim, McCarl, and Murray (2008) consider this and show that, in some instances, willingness to pay may be less than 50% of that paid for a permanent GHG emission reduction.

Additionality

In the international GHG regulatory discussion, policymakers have reflected a desire to only credit GHG offsets that would not have occurred under the normal course of business (commonly called business as usual). Similarly, credit buyers would naturally desire to pay only for GHG credits that are recognized under regulatory schemes. Within the GHG credit discussion, this concern about whether or not GHG reductions occurred outside the scope of regular business is called *additionality*. The widely held stance is that the rulemaking and regulatory structure should only grant credits for GHG offsets that are additional to what would have occurred under business as usual. The main additionality issues, given a proposed project, are the following:

- How much of the potential GHG offsets created by a project would have occurred in the absence of the program?
- How much should the potential offsets created by the project be decreased to account for the activity that would have occurred in the absence of the program?

Many guidelines and implementation strategies have been suggested for programs to ensure that GHG mitigation or C sequestration meets the condition of additionality, i.e., whether or not additional is from a business-as-usual instance (Murray, McCarl, and Lee 2004; Smith et al. 2007).

Leakage

Market forces coupled with less than global coverage by a GHG regulatory program can cause net GHG emission reductions within one region to increase emissions in other regions. For example, promotion of corn ethanol generally is viewed to directly lower GHGs relative to gasoline use but, by increasing crop demand and prices, may increase emissions from enhanced crop production elsewhere. Such a phenomenon has been called *leakage* (Murray, McCarl, and Lee 2004; Murray, Sohngen, and Ross 2007; Searchinger et al. 2008).

Uncertainty

Land use-based production of GHG offsets will be subject to production and sampling *uncertainty*. Production uncertainty arises in much the same fashion as it does for any other agricultural or forestry commodity. Year-to-year weather variations along with the uncertain incidence of fire, diseases, and pest incidences coupled with many other factors will cause this uncertainty. Yields of crops commonly vary by 10% or more of their average value. Uncertainty also arises due to sampling issues. Measurement of GHGs across the landscape is not possible. Sequestered GHG concentrations and emission decreases are widely spread across every square inch of the landscape. From a practical viewpoint, one can never measure such geographically dispersed offsets. One must rely on sampling or modeling and both have associated errors causing uncertainty. Collectively the natural production and sampling, or modeling, uncertainty will exist, and thus the purchaser of potential GHG credits will be at risk of having the quantity of realized credits fall below the claimed number of credits, causing the purchaser to be out of compliance with regulatory limits. This, coupled with potential compliance penalties, leads to the uncertainty concern that has arisen in the consideration of GHG markets. Kim and McCarl (2009) and Kurkalova (2005) further discuss this issue and a discounting approach.

Early Adopters

An additional economic issue is that of early adopters. As mentioned earlier, some producers have already adopted practices that lower net GHGs, and a frequently discussed policy approach is only to pay for additional efforts, not previous commitments that fall under “business as usual.” If existing efforts are not eligible for incentives, however, it is possible that producers may reverse practices, releasing additional GHGs in an effort to become eligible for the incentives at a later date. This raises a major economic design issue of how to deal with payments to those using recommended practices begun before program implementation and has implications for income distribution, program cost/effectiveness, and implementation costs.

Policies under Consideration

As of July 2010 there were six market-based GHG bills introduced in the 111th Congress (Larson [H.R. 1337], Waxman-Markey [H.R. 2454], Kerry-Boxer [S. 1733], Stabenow [S. 2729], Cantwell-Collins [S. 2877], and Carper-Alexander [S. 2995]) (Pew Center on Global Climate Change 2010; RFF 2010). Another bill (Kerry-Graham-Lieberman) exists in draft form (Pew Center on Global Climate Change 2010; RFF 2010). Waxman-Markey, Kerry-Boxer, Stabenow, Cantwell-Collins, and Kerry-Graham-Lieberman propose an economy-wide cap on net GHG emissions. Emissions from the forestry and agricultural sectors are not specifically regulated under any of these bills. Under Waxman-Markey, which was passed by the House in 2009, agriculture and forestry are exempt from the cap, and enteric fermentation—the major source of CH_4 emissions from livestock—is exempt from future regulation as an uncapped sector (Raysor 2010). Four bills (Waxman-Markey, Stabenow, Cantwell-Collins, and Kerry-Graham-Lieberman) explicitly permit the forestry and agricultural sectors to sell project offsets that lower GHG emissions or sequester soil C. These bills explicitly provide a role for agriculture and forestry and would allow emissions decreases or sequestration offsets from agricultural and forestry sources to be sold within the GHG market.

A preliminary analysis of the effects of increased energy prices (thought to result from the Waxman-Markey bill) on agriculture was conducted by the Economic Research Service of the USDA–ERS (2009). Even under restrictive assumptions of no technological change or alteration of inputs, the agricultural sector was predicted to incur modest short-run costs and potentially significant net benefits in the long run in response to the offset market. Carper-Alexander (S. 2995) proposed an electricity sector cap alone and does not include a role for agriculture and forestry (Pew Center on Global Climate Change 2010; RFF 2010). Agriculture may have an indirect role in producing biomass for the electricity sector as this decreases emissions. In contrast to other bills, Larson (H.R. 1337) proposed an economy-wide tax on fossil fuels (RFF 2010), as mentioned earlier; a system of emissions taxes does not provide opportunities for forestry and agriculture to sell emissions decreases and sequestration to other sectors. It is possible that national policies could supersede the provisions of regional policies, in which instance the role given to agriculture in any national policy would be an important factor for the industry.

Any comprehensive GHG legislation would affect agricultural income in three ways. First, restrictions in GHG emissions would induce an increase in energy prices, which would raise agricultural production costs for energy inputs such as fuel and electricity as well as for energy-intensive inputs such as fertilizer. Second, through economy-wide adjustments to increased energy prices and stronger incentives to produce alternative energy sources such as bioenergy, the prices of many agricultural commodities would likely increase. The net effect of these two changes on farm incomes would depend on whether the cost increase or revenue increase is larger. Third, and finally, for legislation that creates a market for GHG mitigation credits with offsets, agricultural producers may have new streams of income from generating and selling these offsets. In a review of the available studies estimating the impacts of the Waxman-Markey bill on U.S. agriculture, Golden and colleagues (2009) reported that although the projected impacts vary regionally and by agricultural subsector, the existing studies project an overall increase in farm income from the bill when offset sales are taken into account.

7 Conclusions

Climate

Future temperature trends for the United States according to the high greenhouse gas (GHG) emissions scenario are expected to rise approximately 2.2–3.6°C (4–6.5°F) by the period 2040–2059 and to increase by up to 6.15°C (11°F) by the end of the century for a high GHG emissions scenario compared to the baseline years 1961–1979. Temperature changes are expected to alter the hydrological cycle. Precipitation amounts in the United States are reported to have increased 5% in the past 50 years, especially in the northeastern United States, while the Southwest has become drier. These patterns are expected to intensify in the future. Projections for temperature change are likely more reliable than are those for precipitation. Deeper incursions of warm, humid air from the south are expected to lead to increased precipitation further north than has occurred in the past, but greater drought is expected for the southwestern quadrant of North America because of warming and declining annual precipitation. Changes in the amount, duration, and intensity of rainfall will vary regionally, resulting in a heterogeneous response to climate change across regions.

Agricultural Greenhouse Gas Emissions

Globally, agriculture accounts for 13.5% of GHG emissions. In the United States, agricultural GHG emissions are over 6% of national emissions (USEPA 2011). Although a small percentage of U.S. emissions, the absolute quantity of emissions from U.S. agriculture is large enough to exceed the total emissions from some countries. This report presents how agricultural practices can be used to reduce GHG emissions, enhance soil carbon (C) sequestration, and provide national-scale mitigation potential. Many of the practices also have additional positive benefits (cobenefits) for the environment. Chapters 2, 4, and 5 discuss the possible cobenefits from practices that could be adopted to lower GHGs on annual cropland, pasture/range (including residue management and grazing livestock), horticultural crops (including vegetable crops, orchards, and turf), agroforestry, wetlands and organic soils, set-aside (CRP [Conservation Reserve Program]) programs, and confined livestock.

It is clear that agricultural practices within the United States have significant potential to contribute to efforts to lower GHGs; however, this potential must be considered together with their economic viability—an important factor driving business decisions. Although agricultural producers strive for good land and environmental stewardship, this must be accomplished within the confines of a business environment. Prices, input costs, and other costs associated with policy implementation, rules, and regulations will affect the adoption of the suite of potential practices described in this report and significantly affect the role that agriculture will play in GHG decreases. The economic profitability of a single agricultural practice changes from region to region, and as such we expect that there is a wide range of practices that could be adopted across the United States as a whole to mitigate climate change and lower GHG emissions. Several of the more critical research needs for developing and implementing U.S. agricultural C sequestration and non-CO_2 (carbon dioxide) GHG mitigation practices were previously described by Morgan and colleagues (2010) and are shown in Table 7.1.

Soil Carbon

Continued efforts to develop management practices that increase the uptake of CO_2 by plants during photosynthesis or increase the residence time of organic C in soils will benefit GHG mitigation efforts. Carbon sequestration within the soil's organic matter (SOM) fraction is among the best options for C storage in terrestrial ecosystems. Besides helping offset CO_2 emissions, C sequestration and increased amounts of SOM provide multiple cobenefits; including improved soil quality, structure, aggregate stability, water holding capacity, and capacity to decrease the effects of toxic substances. Across the United States, mineral soils provide a substantial C sink. These include grazed

Table 7.1. Critical research and development needs for developing and implementing U.S. agricultural carbon sequestration and non-CO_2 greenhouse gas mitigation practices (Morgan et al. 2010)

Topics	Critical Needs
Agricultural Sectors	
Cropping Systems	• Quantify above/belowground C contributions • Long-term trace gas flux networks • Accurate, easy-to-use C/SOC simulation models • Clarify tillage/environment interactions on C
Grazinglands	• C sequestration in arid shrublands • Total GHG accounting • Management/environment interactions • Robust modeling/scaling
Agroforestry	• Account for contribution in natural resource inventories • Incorporate potential into C sequestration tools • Quantify C sequestration and other GHG fluxes within each practice over time
Horticulture	• Quantify management effects on soil C • Determine best management practices • Benefits beyond C sequestration
Turfgrass	• Incorporate effects of urbanization in national estimates • Quantification of C sequestration (reverse order?)
Potential High Flux Areas	• Management for minimizing CH_4 and N_2O fluxes in rice and other major wetland crops
National/Regional Scale Analyses	• Improve understanding of land-atmosphere GHG fluxes for incorporation into models (e.g., DayCent) • More extensive/representative soil/GHG monitoring networks
Implementation	
Measurements/monitoring	• Develop low-cost C/trace gas monitoring to integrate soil sampling networks with modeling/ remote sensing
Databases	• GHG: NACP and GRACEnet • Climate/weather: PRISM, Daymet • Soil maps: SSURGO, STATSGO • Agricultural production: NASS, Census of Agriculture • Management practices: NRI, ERS, CTIC
Emerging Issues	
Biofuels	• Maintain soil resources, especially SOC • Quantify biofuel impacts on C and N cycling • Clarify relationships between C storage and non-CO_2 GHG fluxes • Marginal lands evaluation • Impacts on CRP, grasslands, forests
Climate Change	• Process-level research on multiple CC factors • Modeling to predict future impacts of CC on C and GHG fluxes • Observational/monitoring systems for tracking CC impacts on agro-ecosystem attributes that indicate C

lands, CRP lands (i.e., reserve cropland), and hay or pasture in rotations with annual crops.

Nitrous Oxide

Emissions of N_2O (nitrous oxide) are high in regions and states with extensive row cropping (Corn Belt states), intensive row- and specialty-crop production, and large beef cattle and/or dairy cattle populations. Emissions of N_2O contribute about 84% of the anthropogenic inputs of N_2O to the atmosphere due to nitrogen (N) fertilization, legumes, manure, and type of management. Warming generally increases decomposition and release of soil inorganic N (net N mineralization) and thus may increase N loss as N_2O. Additionally, N_2O emissions from soil often occur after rain events so that changes in precipitation amount and frequency can alter annual rates of N_2O emission. Management practices, particularly fertilizer use, can be made more efficient to decrease N fertilization and lower N_2O emissions.

Methane

Agriculture produces methane (CH_4) from several practices. Ruminant livestock (cattle, sheep, and goats) produce significant quantities of CH_4 during digestion. Livestock manure emits both CH_4 and N_2O during storage and field application. Manure storage conditions (aeration, temperature, and pH) and manure composition have a major influence on gases emitted and emission rates. Stored manure emissions of CH_4 are decreased by cooling, covering, solids separations, or the capture of emitted CH_4. Options to lower enteric CH_4 include three broad sets of practices: (1) improved diet digestibility, (2) additives, and (3) improved livestock genetics. Wetland agriculture is another CH_4 source, mainly from rice production in flooded fields. In 2009, total area cropped in rice was slightly more than 1% of U.S. cropland area, but it is an important source of CH_4 and N_2O.

Bioenergy

Bioenergy crops offer important opportunities for U.S. agriculture to mitigate climate change by offsetting fossil fuel use with photosynthetically captured carbon. Bioenergy use can help stabilize atmospheric CO_2 levels by recycling of contemporary CO_2. Growing bioenergy can also increase C sequestration to additionally mitigate atmospheric CO_2. Biofuel ethanol is now primarily made from corn grain, and minor amounts of biodiesel are derived from soybean and other oil seed crops. Elsewhere sugarcane is an important ethanol feedstock. Corn-based ethanol production consumed over 30% of the 2009 corn yield, and by 2015 the amount is estimated at nearly 50% of a 2009-equivalent crop. By 2022, it has been mandated that at least 61 billion liters (16 billion gallons) of U.S. ethanol production must be from cellulosic sources. As the technology develops, however, agricultural wastes, plant residues, and woody vegetation can be used for cellulosic biofuels. Commercialization of cellulosic biofuels holds promise to transform the agricultural energy sector. Genomic advances are expected to improve the suitability and diversity of energy crops.

Economic Considerations

Economic considerations are critical for adoption of any mitigation options to decrease GHG emissions. Although some U.S. producers already use important mitigation practices, more producers will adopt them only if they are profitable. Producers across the United States face very different cost and production conditions so that specific practices may be profitable for some producers in some locations but not profitable for everyone in all locations. For example, decreased tillage is well known but not universally used because it is not necessarily superior, economically, to other practices. Increased yield variability of risk, variable soil or climate conditions, or many other factors can serve as barriers that prevent the adoption of new practices. Climate change itself will exacerbate these risks as a consequence of different temperature and precipitation regimens.

Policies to Decrease Greenhouse Gas Emissions

Incentives may be needed, and can be used, to increase the number of producers that adopt practices to mitigate GHG emissions. There are many different ways to offer incentives that encourage producers to adopt practices that mitigate GHGs. The type of incentive will drive producers' selections of GHG mitigation technologies, the numbers of producers involved, and possibly the regional location of different management practices. For example, a payment to lower GHG emissions or, alternatively, an emissions tax might be used to encourage adoption of a technology that emits less. Many think only of taxes on fossil fuels. Under a policy of taxation, however, some emitters of GHGs would face an emissions tax that would create incentives for them to adopt and use GHG reduction technologies. Practices that enhance soil C sequestration might not be incentivized under a taxation system and could need additional subsidies, or possibly higher market prices, before the economic benefit of adoption is greater than their cost of adoption.

Another popular policy design is "cap and trade" wherein a limit (cap) on GHG emissions is set and emission allowances are issues equal in number to the level of the cap. These allowances can then be traded on the open market. The types of practices adopted to lower GHGs will depend largely on the suite of practices allowed by cap-and-trade regulation as well as their relative economic competitiveness with those that could be adopted by other industries. In contrast to a system of emissions taxes, revenues from the sale of credits under a cap-and-trade system are likely to accrue to the private sector and thus encourage innovations. Based on eligibility rules, a cap-and-trade system is an option that gives agricultural producers an opportunity to provide emissions decreases to other sectors.

Several countries have developed GHG markets to help meet Kyoto Protocol obligations or emission reduction needs. Under the Kyoto Protocol and in the European Union-Emissions Trading System, there is participation by agriculture and forestry sectors to include land afforestation and CH_4 capture, but there is no approved methodology to certify tradable offsets from C sequestered by changing tillage. The only methodology for cropland practices is to allow N_2O offsets for applying an inoculant instead of N fertilizer on legume crops. Even though regional and some voluntary markets have emerged, there is no federal policy giving rise to a U.S. GHG market. Nonetheless, if new climate policies are adopted, the opportunities available to agricultural producers to sell C-credits may change in the future.

Cobenefits

Because of numerous interlinkages in natural ecosystems, agricultural practices that sequester C or decrease GHG emissions may simultaneously have other, primarily beneficial environmental "cobenefits," possibly including lowering soil erosion, decreased N and phosphorus surface runoff, and better wildlife habitat. Consideration of these benefits needs to be done carefully where they may offset GHG emissions by the energy sector to result in other combinations of cobenefits. The magnitude of cobenefits determines whether or not GHG policy needs to address them explicitly. Cobenefits can accrue mostly to society at large, but uncompensated land managers may not account for them in their farming management decisions. Cobenefits can constitute externalities to a GHG offset market and too much or too little C may be traded (or provided in response to GHG mitigation policy) to be optimal for society.

Good arguments exist that cobenefits are misleading and should be neglected because estimating them is inherently limited by lack of data on the magnitudes of the associated GHG mitigation as well as the quantity and quality of cobenefit provision. As with the soil C sequestration potential, magnitudes of cobenefits depend on soils, landscapes, farming practices, and climate. Irrespective, unaccounted for cobenefit targeting policies may lead to inaccurate assessment of the GHG baselines. Changes on agricultural landscapes can alter baseline relationships that need measurement to address GHG mitigation policy. Overstating their potential value should not be based on the assumption of a constant willingness of society to pay for cobenefits.

Models

National-level analyses of soil CO_2 and N_2O emissions and removals are included in the national GHG inventories of most developed and a few developing countries. Greenhouse gas inventories can be produced using one of three tiers. Tiers One and Two use either empirically based emission factors or a C stock coefficient or "emission factor" generated from national- or global-level estimates. The Tier Three approach for GHG inventories used in the United States and some developed countries is based on process-based simulation models such as Century and DayCent. Complex models are needed to reliably assess mitigation potentials at regional and national scales because mitigation options designed to increase C storage are likely to impact N_2O emissions as well. It is difficult to address GHG mitigation potential at the national scale using complex models because viable mitigation options are different across the United States.

Use of geospatial modeling improves the representation of a natural system. Better representation of soil, weather, land management, and cropping can all improve soil C estimates. Integrating geospatial modeling with existing process-based biogeochemical models can contribute to development of next-generation soil C models. Additionally, higher resolution gridded data can help monitor changes in annual land management and verify and attribute estimated soil C stocks and fluxes. A combination of process-based model output and eddy flux estimates allows input to atmospheric inversion models for tracking lateral flow of CO_2 across continents and estimating regions of CO_2 sources and sinks. Although changes in soil C stocks are difficult to ascertain from total atmospheric CO_2 movement, because of the small fraction of soil C flux compared to total plant growth and total fossil-fuel emissions, inversion models serve to constrain total modeled fluxes. Comparisons between flux tower data, process-based models, and atmospheric inversion estimates are being conducted within the North American Carbon Program. Future efforts are needed to integrate various methods so that a comprehensive monitoring framework can include components of both remote sensing and geospatial modeling. User-friendly tools that incorporate state-of-the-art models are expected to help support science-based mitigation activities for U.S. agriculture.

Policy Considerations

Although requirements for farm-level emission inventories vary as a function of the policy objectives and design, common attributes include that (1) estimates need to be unbiased and have sufficient accuracy and precision to serve policy and decision-making, (2) no single standard can be considered totally certain for all instances and estimates, and (3) the trade-off between accuracy/precision achieved and the cost of measurements needs to be known. For policy implementation to be practical, measurement costs should represent a small fraction of the value produced by the policy. The system of quantification needs to address the large variety of practices available for mitigation and their interactions with GHG for a full GHG accounting. Methods need to be applicable to the entire United States but also locally specific because the impact of different mitigation practices is often sensitive to local soil and climate.

Some major issues have emerged within international and domestic market design discussions that imply decreased value for some classes of credits and decision on eligibility. These are: (1) Saturation and permanence—Sequestration C accumulation does not continue indefinitely, and a new equilibrium C content will follow practice adoption. Thus the two important characteristics are time over which the sequestration accumulates and maintenance of C stock after accumulation ceases. (2) Additionality—Buyers do not wish to pay for potential offsets that would be disallowed by a regulatory body with the widely held stance that the rulemaking and regulatory structure should only grant credits for GHG offsets that are additional to those that would have occurred under business as usual. (3) Leakage—Market forces coupled with less than global coverage by a GHG regulatory program can cause net GHG emission reductions within one region to be offset by increased emissions in other regions. (4) Uncertainty—Land use-based production of GHG offsets will be subject to production and sampling uncertainty. Production uncertainty arises in a similar fashion as any other agricultural or forestry commodity. Annual weather variations, along with uncertain incidence of fire, diseases, and pests, and many other factors will cause this uncertainty. These, coupled with potential compliance penalties, lead to the uncertainty concern that has arisen in the consideration of GHG markets.

Appendix A: Abbreviations and Acronyms

AB32	Assembly Bill 32
B	Bare-herbicide
C	Carbon
CARB	California Air Resources Board
CCX	Chicago Climate Exchange
CGWSA	California Global Warming Solutions Act
CH_4	Methane
CIG	Conservation Innovation Grant
cm	Centimeter
CO_2	Carbon dioxide
CO_3^{2-}	Carbonate
CREP	Conservation Reserve Enhancement Program
CRP	Conservation Reserve Program
EAA	Everglades Agricultural Area
EISA	Energy Independence and Security Act
EPA	Environmental Protection Agency
Eq	Equivalents
EQIP	Environmental Quality Incentives Program
ESM	Environmental Services Market
ETS	Emissions Trading System
EU	European Union
FASOMGHG	Forest and Agricultural Sector Optimization Model—Greenhouse Gas
Gg	Gigagram
GHG	Greenhouse gas
GWP	Global warming potential
H	Hydrogen
H_2O	Water
ha	Hectare
in.	Inch
IPCC	Intergovernmental Panel on Climate Change
kg	Kilogram
m	Meter
MB	Moldboard/disking
Mg	Megagram
Mha	Million hectares
MMT	Million metric tons
MRTN	Mean return to nitrogen
Mt	Megatonne
N	Nitrogen
N_2	Nitrogen gas
N_2O	Nitrous oxide
NH_1^+	Ammonium
NO_2^-	Nitrite
NO_3^-	Soil nitrate
NRCS	Natural Resources Conservation Service
NRI	National Resources Inventory
NT	No-till cropping
NUE	Nitrogen use efficiency
O	Oxygen
O_2	Oxygen in its molecular form
OTC	Over the counter
Pg	Petagram
ppb	Parts per billion
ppmv	Parts per million by volume
RCA	Soil and Water Conservation Act
RGGI	Regional Greenhouse Gas Initiative
SIC	Soil inorganic carbon
SOC	Soil organic carbon
SOM	Soil organic matter
Tg	Teragram
UNFCCC	United Nations Framework Convention on Climate Change
USDA	U.S. Department of Agriculture
WCI	Western Climate Initiative
yr^{-1}	Per year

Appendix B: Glossary

Additionality. Refers to a criteria for GHG mitigation activities or projects in which the reductions achieved are beyond what would have occurred in the absence of the activity or project (i.e., the business-as-usual case). Emission reductions that would occur in the absence of regulation of a carbon dioxide equivalent credit payment or other regulation would not be considered additional.

Cardinal temperature. Critical temperature range of plant growth and development.

Cobenefits. The benefits of policies that are implemented for various reasons at the same time—including climate change mitigation—acknowledging that most policies designed to address greenhouse gas mitigation also have other, often at least equally important, rationales.

Conservation Reserve Program. The Conservation Reserve Program (CRP) is a voluntary program for agricultural landowners. Through CRP, landowners can receive annual rental payments and cost-share assistance to establish long-term, resource-conserving covers on eligible farmland. The Commodity Credit Corporation makes annual rental payments based on the agriculture rental value of the land, and it provides cost-share assistance for up to 50% of the participant's costs in establishing approved conservation practices. Participants enroll in CRP contracts for 10 to 15 years.

CRP continuous sign-up. Environmentally desirable land devoted to certain conservation practices may be enrolled at any time under CRP continuous sign-up. Certain eligibility requirements still apply, but offers are not subject to competitive bidding (http://www.fsa.usda.gov/FSA/webapp?area=home&subject=copr&topic=crp).

CRP general sign-up. Producers can offer land for CRP general sign-up enrollment only during designated sign-up periods.

Denitrification. The breakdown under anaerobic conditions (i.e., oxygen limited) of nitrates by bacteria into less oxidized compounds (including nitric oxide and N_2O [nitrous oxide] gas), with the final end product being dinitrogen gas.

Humified. Organic matter that has been highly processed and altered through decomposition processes in soil, resulting in a more condensed and chemically heterogeneous structure that is more resistant to further decomposition.

Interaction. Cause-and-effect relationships between (groups of) organisms, biogeochemical processes, and/or environmental factors (e.g., temperature, moisture) that impact the structure and function of ecosystems.

Leakage. Unintended increases in greenhouse gas emissions that result from an offset project designed to decrease emissions. A typical example is that cap-and-trade regulations in one region may cause greenhouse gas emitters to move to less regulated or unregulated regions.

Methanotrophic. Having the biological capacity to oxidize methane to carbon dioxide and water by metabolism under aerobic conditions.

Nitrification. The oxidation of ammonium (derived from decomposing plant material or added in fertilizer) to nitrite and nitrate ions by soil bacteria, which is the main pathway for nitrate production in soils.

Nitrous oxide mole fraction. The proportion of N_2O gas emitted relative to the total N_2O and N_2 (nitrogen gas) gas emitted, i.e., $N_2O/(N_2O + N_2)$, on a molar equivalent basis.

Permanence. The long-term maintenance of a carbon stock or sink. Biological sinks, e.g., vegetation and soils, lack inherent permanence since disturbances (e.g., fire, plowing) can result in a loss of sequestered carbon back to the atmosphere.

Saturation. A level of carbon storage (e.g., in soils) that cannot be further increased by more carbon additions.

Set-aside programs. Although these have taken various forms in agricultural policy, they have generally been land retirement programs used in the past in which farmers decrease their planted acreage to participate in commodity programs or other agricultural programs.

Trace gases. A gas that makes up less than 1% by volume of the Earth's atmosphere and includes all gases except nitrogen and oxygen.

Uncertainty. A statistical measure of how well a set of observations or estimates represents the true value.

Wetlands Reserve Program. A program offering landowners the opportunity to protect, restore, and enhance wetlands on their property.

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