Climate Change ᵃⁿd Greenhouse Gas Mitigation:

Challenges and Opportunities for Agriculture

CAST

Climate Change and Greenhouse Gas Mitigation: Challenges and Opportunities for Agriculture

Task Force Report

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Foreword

Following a recommendation by the CAST National Concerns Committee, the CAST Board of Directors authorized preparation of a new comprehensive report on agriculture's response to the global climate change challenge.

Dr. Keith Paustian, Colorado State University, Fort Collins, and Dr. Bruce A. Babcock, Iowa State University, Ames, served as cochairs for the report. A highly qualified group of scientists served as task force members. The group included individuals with expertise in agricultural economics, agronomy, biogeochemistry, environmental science, global change research, natural resources, and resource ecology.

The task force prepared an initial draft document and reviewed and revised all subsequent drafts. A technical advisor assisted the task force during preparation of the report. The CAST Executive Committee and Editorial and Publications Committee 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, authors, and technical advisor who gave of their time and expertise to prepare this report as a contribution by the scientific community to public understanding of the issues. We also thank the employers of the scientists, who made the time of these individuals available at no cost to CAST. CAST thanks all members who made additional contributions to assist in the preparation of this document. 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 Agency for International Development. Additional recipients include media personnel and institutional members of CAST. Individual Members of CAST may receive a complimentary copy on request for a $3.00 postage and handling fee. 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

Agriculture is both a source and a sink for greenhouse gases (GHGs). As a source, agriculture can be burdened by regulations designed to curtail the growth in GHGs. As a sink (where carbon dioxide is removed from the atmosphere to increase storage of carbon [C] in soils), agriculture can benefit from those same regulations. This report synthesizes research on both mitigation of agricultural GHG emissions and enhancement of agriculture's ability to mitigate GHGs from nonagricultural sources. An improved understanding of the biophysical and public policy processes involved in translating a change in farming practices to a change in GHG emissions is essential to minimize the loss and maximize the gain from any future regulations.

Emissions and Mitigation of Agricultural Greenhouse Gases

Atmospheric concentrations of three gases—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)—have risen dramatically during the past century, accounting for more than 80% of human-induced global warming from atmospheric buildup of GHGs. Agriculture is linked directly to atmospheric concentrations of these gases through basic soil-plant-animal processes, and increased GHG concentrations in turn impact agriculture and other sectors of society because of their potential to promote rapid and undesirable changes in climate.

Agricultural mitigation of atmospheric GHG concentrations can be achieved by the following:

- Taking CO_2 from the atmosphere and sequestering it in biomass and soils;

- Decreasing the rate of land clearing for agriculture and taking marginal lands out of production;

- Changing agricultural practices on productive, established agricultural lands;

- Increasing efficiency of farm inputs such as fuel, fertilizers, and pesticides;

- Increasing production of agricultural biofuels (renewable biological-based energy fuels) to replace fossil energy emissions;

- Improving N-use efficiency as the primary means of decreasing N_2O emissions; and

- Decreasing CH_4 emissions by capturing or preventing emissions from animal manure storage and by increasing livestock production efficiency.

Carbon sequestration (causing removal of CO_2 from the atmosphere) can be achieved through management practices that increase C inputs to soils and/or decrease decay rates. Some options include crop rotations with high-residue yields, conservation tillage, decreased bare fallowing, conservation reserves on marginal cropland, grassland and grazing management improvements, and degraded-land rehabilitation.

Whereas it is important to account for the net changes in all GHG emissions when determining the advisability of a given management change, the potential for agricultural GHG mitigation is significant. Using existing technology and best management practices, U.S. agriculture could sequester 100–150 million tonnes of C per year, and current N_2O and CH_4 emissions could be decreased by 20–40%.

Policy Options and Design

Greenhouse gas mitigation in agriculture will not begin of its own accord. There are three scenarios within which mitigation programs might be established:

1. International agreements that allow agricultural GHG mitigation,

2. National policy goals of promoting agricultural GHG reductions, and

3. Nongovernmental mitigation efforts that capture consumer demand for "climate-friendly" products.

Current U.S. policy supports voluntary efforts aimed at decreasing GHG emissions, but current consumer demand for climate-friendly products is insufficient to induce development of private GHG programs.

If the United States moves toward participating in global efforts to decrease GHG emissions or if it moves to adopt firm national GHG goals, then acceptance and success of agricultural GHG mitigation programs will increase if those programs address the following concerns:

- Full Accounting. Effective and credible GHG mitigation programs will include full accounting across different GHGs and across locations. A particular management practice may decrease emissions of one GHG but increase emissions for another, and a specific land-use policy intended to mitigate GHG emissions may generate deleterious responses elsewhere.

- Observability. To be credible, control programs must be based on observable actions taken by land managers that can lead to predictable changes in net GHG emissions.

- Timing. Agricultural mitigation offers permanent and temporary decreases in atmospheric GHG levels. An effective program will provide mechanisms to discover the relative values of temporary and permanent mitigation and compensate accordingly.

- Adoption. Farmers and land managers will participate in GHG mitigation programs only if suffi-cient incentives are provided. Subsidies create positive incentives for adopting practices that mitigate GHG emissions; taxes create negative incentives.

Monitoring and Verification of Mitigation Success

Agricultural mitigation programs will be most likely to succeed if

- Protocols and procedures are developed that monitor and verify changes in C sequestration and GHG emissions;

- Both direct measurement of C stocks or gas fluxes throughout time as well as model-based predictions are used;

- Claimed changes are verified (because measurement of GHG emissions changes in agriculture is more difficult than measurement from large industrial sources); and, most importantly,

- A combination of statistical methods, advanced simulation models, and targeted sampling aimed at decreasing measurement uncertainties is used.

Executive Summary

Introduction

Agriculture is both a *source*[1] and a *sink* for *greenhouse gases* (GHGs)—part of the problem as well as part of the solution for GHG buildup in the atmosphere. Although the main source of GHG increase is fossil fuel combustion, three of the major GHGs—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)—are emitted to and/or removed from the atmosphere through agricultural activities. Increasing concern about climate change, driven in part by rising GHG concentrations, has focused attention on mitigation opportunities, including those within agriculture. This report summarizes and synthesizes research on mitigating GHG emissions through improving agricultural practices and elucidates key policy issues in implementing mitigation strategies. The intended audience includes policy- and decision makers in government and industry, agricultural producers, environmental and other nongovernmental organizations, and the general public.

Greenhouse Gases and Climate Change

Greenhouse Gases

The atmosphere contains a number of gases that affect the Earth's energy balance by absorbing and reradiating energy back to the Earth's surface. Naturally occurring GHGs include water vapor (H_2O), CO_2, CH_4, N_2O, and ozone (O_3); several man-made compounds (e.g., chlorofluorocarbons) also act as GHGs. The atmospheric amounts of three gases—CO_2, N_2O, and CH_4—have risen dramatically during the past century, however, and they account for >80% of the human-induced warming effect of all GHGs.

Of these gases, CO_2 is the most abundant in the atmosphere and has the greatest effect on warming. Since the mid-1800s, concentrations have increased

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[1] Italicized terms (other than scientific names) are defined in Appendix B: Glossary.

from approximately 280 ppm to 370 ppm. Large amounts of CO_2 are exchanged annually between the Earth's surface and the atmosphere through plant uptake (*photosynthesis*) and *soil respiration*. Although the gross annual fluxes of CO_2 between the atmosphere and terrestrial ecosystems are roughly equal, small differences in net C fluxes are changing carbon stocks significantly (most notably in the atmosphere). Of the 6–7 petagrams (Pg) C added annually to the atmosphere from fossil fuels, approximately 3 Pg remains in the atmosphere (causing the buildup in CO_2 concentrations), 2 Pg is taken up (net) by oceans, and 1–2 Pg is taken up (net) by terrestrial ecosystems and stored in biomass, soils, and sediment. The longevity and capacity of terrestrial and ocean sinks are issues of concern, particularly if fossil CO_2 emissions increase several-fold, as indicated in certain projections, throughout the next century.

Nitrous oxide is a highly stable gas occurring at lower concentrations (approximately 1/1,000[th]) than CO_2, but with a much stronger effect (on a molecule-per-molecule basis) on warming (approximately 300 times that of CO_2—a value referred to as *Global Warming Potential* [GWP]). Concentrations are increasing by 0.2–0.3% per year (yr), mainly because of human activities; approximately 40% of the increase is attributed to agriculture.

Methane also is present in the atmosphere at low concentrations, but it currently accounts for approximately 16% of the warming attributed to increasing GHG concentrations. It has a GWP approximately 20 times that of CO_2. Since the mid-1700s, CH_4 in the atmosphere has increased by 145%; current rates of increase are approximately 0.3% per yr. Agricultural activities, particularly rice cultivation, livestock, and manure, are major emission sources.

Impacts of Climate Change

Greenhouse gases are of concern because of their potential role in promoting rapid and undesirable change in climate. In this report, recent assessments of climate change impacts on agriculture are reviewed, providing context to the discussions of

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GHG mitigation.

The climate of the United States is warming, by an average of 0.6° Celsius (C) during the past century. Warming trends vary geographically across the country, tending to be more significant in the Northeast, the upper Midwest, the Southwest, and Alaska, and most warming has occurred in the winter. Throughout the same period precipitation has increased on average 5–10%, but with substantial regional variation.

For the *U.S. National Assessment: The Potential Consequences of Climate Variability and Change*, projections of possible future climate throughout the next century were analyzed by two global climate models, the Canadian Climate Centre (CCC) model and the Hadley Centre (HC) model, using an "intermediate" scenario for GHG emissions. Both models project warming for the United States of 2–5.5°C by 2100, varying by region and between models. Both models predict overall increases in precipitation, but they vary between small decreases in certain regions to large increases in others. Uncertainty in the precipitation projections, in both amount and distribution, is greater than for temperature.

Crops can be impacted by climate and GHG changes in a variety of ways. Increasing CO_2 concentrations can be beneficial for many plants by increasing photosynthesis and plant water use efficiency (Figure S.1). Whether benefits from "CO_2 fertilization" may be realized remains uncertain, however, depending on the effects of climate change on crop pests, water resources, climate variability, and extreme weather.

Assessments of global crop production suggest that warming temperatures and longer growing seasons may benefit higher-latitude regions, whereas warming at lower latitudes may have a negative influence by hastening maturity and shortening growing periods. Studies in the United States suggest that overall production of the major grain crops can be maintained. Climate change is likely to include changes in climate variability, however, with adverse effects on production. In particular, extremes of precipitation (i.e., droughts or floods) increase the risk of crop failure.

Potential impacts of climate change on water resources include decreased snow pack, changes in amount and timing of precipitation, increased crop water demand, and changes in water supply infrastructure. Most projections for the United States suggest that the relative abundance of water for agriculture can be maintained in the near term (e.g., 2030s). Uncertainty is high, however, and other factors, such as competition for urban water use, are likely to interact with climate change.

Spatial and temporal distributions and proliferation of insects, weeds, and pathogens are influenced significantly by climate. Under warming conditions pests overall are expected to expand their geographic ranges, and increased climate variability may increase the frequency and severity of pest outbreaks, as has been documented in the past. Increased costs for pest damages and pest control under the climate change scenarios used in the *U.S. National Assessment* have been estimated at $200 million per yr.

Climate change will affect livestock directly through heat stress and indirectly through impacts on forage productivity and quality, pests and diseases, water availability, and feed cost. As with crops,

Figure S.1.   To determine how elevated CO_2 may reduce water use by crops, a plant physiologist measures water vapor conductance of barley leaves grown at twice the current CO_2 concentration. Photo courtesy of S. Bauer, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

impacts will vary geographically: northern herds may benefit from warmer conditions, whereas southern herds may experience difficulties. There are several options for adaptation, including relocation of livestock production centers, but these options also will entail added costs.

Emissions and Mitigation of Agricultural Greenhouse Gases

Element Cycles

All three GHGs are involved in soil-plant-animal processes controlling carbon (C) and nitrogen (N) cycles in agricultural and nonagricultural ecosystems. These processes are linked intimately with soil organic matter (SOM), which plays a pivotal role in the fertility and sustainability of agricultural systems. Key attributes of SOM are its function as a repository for plant nutrients and its promotion of a soil structure favorable to plant growth and decreased soil erosion.

The ecosystem C cycle is governed by plant uptake of CO_2 through photosynthesis and incorporation into biomass, some of which is returned to soil as crop residues. The residues are decomposed by soil organisms, returning some of the C to the atmosphere as CO_2 (and as CH_4 in oxygen-poor environments) fairly quickly, but some partly decomposed material can remain in soil for decades or centuries. Throughout time, most soils accumulate organic C stocks that exceed the C stored in vegetation.

Nitrogen also cycles through agricultural ecosystems and is a critical nutrient for plants; its largest repository is as organic N in SOM. Fertilizer and N fixation by legumes constitutes the main input of N to agricultural systems, and large amounts of N cycle between SOM and inorganic mineral forms. Reactions involving these inorganic ions, largely mediated by soil bacteria, include the processes responsible for formation of N_2O. Agricultural management can alter significantly the C and N cycles and thus the emissions and sinks of the major GHGs.

Processes Controlling Emission of Greenhouse Gases from Soils

Carbon dioxide is emitted from soil by plant roots, microorganisms, and soil fauna—collectively referred to as soil respiration. Soil respiration is a large flux in the global C cycle, approximately 60 Pg/yr. This flux is balanced roughly by net C uptake by photosyn-thesis. Although large amounts of C are exchanged between the soil and atmosphere annually, the difference between fluxes and net change in global soil C stocks, on an annual basis, is small in comparison. Soil respiration is influenced by soil physical, chemical, and biological conditions and by soil disturbance. For a particular ecosystem, factors that increase C inputs and/or decrease respiration will favor the accumulation of C in soils, creating a sink for atmospheric CO_2. Conversely, decreasing C inputs relative to respiration will lead to a net efflux, or giving off, of CO_2 (a source) from soils. Conversion of native ecosystems to agriculture typically increases soil respiration; hence, agricultural soils historically have been a source of CO_2.

Nitrous oxide is produced in soils by *denitrification* and *nitrification*—ubiquitous microbial processes in most soils. Denitrification is an anaerobic process (occurring in the absence of oxygen [O_2]), whereby nitrate (NO_3^-) is chemically reduced to N_2O and N_2. Denitrification occurs in water-saturated environments (where saturation limits O_2 diffusion) such as wetlands and lake sediments, but also in well-drained upland soil, in anaerobic microsites. Other factors controlling denitrification rates include nitrate and labile organic matter availability. Nitrous oxide also is formed during nitrification, the aerobic oxidation of soil ammonium (NH_4^+) to NO_3^-, where N_2O is formed from intermediary compounds. Agricultural soils tend to emit more N_2O relative to native ecosystems because of higher N inputs and greater soil disturbance.

Methane is produced in soils under water-saturated conditions, such as flooded rice fields, and during animal waste storage (e.g., waste lagoons). Methane also is produced through enteric fermentation in livestock, especially ruminants (e.g., cattle, sheep, goats). In well-drained soils, CH_4 can be oxidized by certain bacteria. Methane oxidation, although occurring at low rates, occurs in most soils and constitutes a significant sink.

Basic Principles of Mitigation

Agricultural-based mitigation of GHG concentrations in the atmosphere can be achieved by (1) decreasing emissions of GHGs and/or (2) taking CO_2 from the atmosphere and sequestering it in biomass and soils. Effective mitigation strategies will seek measures that both decrease emissions and enhance C sequestration.

Greenhouse gas emissions derive from both land-

use conversions to agriculture as well as practices occurring on long-established agricultural lands. Decreasing the intensity and extent of land-use conversions could decrease emissions dramatically, especially in the tropics. On established agricultural lands, CO_2 emissions can be decreased by efficient use of fossil energy-based inputs (i.e., fuel, fertilizers, and pesticides). Improved N-use efficiency is key to decreasing N_2O emissions. Methane emissions can be decreased and/or captured for energy use. Production of agricultural *biofuels* provides opportunities for offsetting fossil energy CO_2 emissions from agriculture and other sectors of the economy.

Carbon sequestration strategies can involve (1) improved management of permanent agricultural land and (2) conversion of marginal agricultural lands to alternative uses. Historically, most agricultural soils have been depleted (by 20–50% or more) of organic matter compared with their preagricultural condition. Globally, net soil C losses from agricultural soils during the past 200–300 years are estimated to be 50–150 Pg. This range provides a rough benchmark for C sequestration (i.e., through partly rebuilding soil C stocks); several estimates yield global rates of 1–2 Pg /yr during a period of several decades.

In most instances, organic C levels exhibit an equilibrium behavior in which changes in C levels are proportional to changes in the rate of C inputs to soils and/or changes in the specific decay rate of organic matter already in the soil. After a change in these rates (e.g., through altered management practices), soil C levels tend toward a new equilibrium, during a period of many years. Thus, for a given management change there is a finite amount (and time span) of C sequestration. Inducing further changes in C stocks requires further changes of input rates or factors controlling SOM decomposition. This equilibrium behavior is distinct from the concept of soil *C saturation*, which implies an ultimate carrying capacity for soil C, independent of C input rates. Most soils, agricultural as well as native ecosystems, are well below any inherent C saturation level.

Mitigation of Carbon Dioxide Fluxes in North American Agricultural Soils

Carbon sequestration is favored under management systems that (1) minimize soil disturbance and erosion, (2) maximize amounts of crop residue return, and (3) maximize water- and nutrient-use efficiency of crop production.

Conservation tillage, particularly no-till, is appli-

cable widely for increasing soil C (Figure S.2). Many long-term experiments show increases of 0.1–0.7 tonnes/hectare (ha)/yr throughout several decades. Rates tend to be higher in moist climates with higher residue inputs and somewhat lower in semiarid regions. Soil responses to decreased tillage also depend on interaction with other practices such as crop rotation and fertilization. In semiarid regions, no-till can enable reduction in fallow frequency, thus providing a synergistic effect by increasing cropping frequency and residue inputs.

Crop rotations that include high-residue-yielding crops, eliminate fallow periods (e.g., winter cover crops), and/or include external organic matter additions (e.g., manure) promote soil C increases. Productivity increases since the 1950s in the United States

Figure S.2.   No-till farming of wheat in Whitman County, Washington. Photo courtesy of the U.S. Department of Agriculture.

likely have contributed to an overall stabilization and/ or increase in C in cropland soils. Productivity increases because of more N fertilizer and irrigation, however, also entail increases in N_2O and fossil-CO_2 emissions that offset gains in soil C. Thus practices that promote efficient water and nutrient use likely will have the greatest benefits in terms of decreased GHGs.

Perennial grasslands (for grazing, hay, or as conservation reserves) can maintain high C stocks, and thus conversion of annually cropped soils to grassland vegetation can yield high rates of C storage, up to 1 tonne/ha/yr or more. Several million tonnes of C per yr are being sequestered on Conservation Reserve Program (CRP) lands (Figure S.3). Carbon sequestration on grazing lands depends, in part, on previous management conditions; the highest potentials

are on previously poorly managed lands, through plant species improvements, better grazing management, and fertility and water management.

The restoration of highly degraded lands to perennial grass or forest vegetation, mine reclamation, and wetland restoration can rebuild soil C stocks at high rates, if adequate plant productivity can be established and maintained.

Using best management practices and existing technology, U.S. agricultural soils could sequester 100–150 million tonnes/yr during a few decades. Carbon sequestration carries additional benefits because of the contribution of SOM to soil fertility and quality. Higher organic matter content in soil tends to promote better physical structure for water infiltration and storage, plant rooting, and erosion resistance, and

Figure S.3. Conservation Reserve Program land that has not been cut. Photo courtesy of A. Eaglin, U.S. Department of Agriculture.

improved soil chemical properties for nutrient retention and buffer capacity.

Mitigation of Nitrous Oxide Fluxes in North American Agriculture

Global agricultural sources of N_2O are approximately 4–6 teragrams (Tg) N/yr, including both direct and indirect emissions. Emissions from agricultural soils are associated closely with N additions, averaging approximately 1.25% of N inputs, largely independent of the form of N added (i.e., as fertilizer, manure, crop residues). Emissions from stored animal waste can be significant, and rates depend on N content of the waste, whether the storage environment is aerobic or anaerobic, and length of storage. Indirect emis-

sions stem from N lost from agricultural soils; for example, through nitrate leaching or gaseous emissions (e.g., ammonia), which are added to neighboring ecosystems (such as riparian zones in the instance of NO_3^- in groundwater) causing N_2O emissions. Annual emissions from U.S. agriculture are slightly less than 1 Tg N_2O.

Because there is a direct relationship between soil N availability and crop yield, the main agronomic challenge for decreasing direct N_2O emission is to decrease N inputs without decreasing yield. Improving the efficiency of N use by crop plants is crucial for meeting this challenge. Practices to better synchronize N supply with plant needs include use of soil N tests, better timing and placement of fertilizers, cover crops during fallow periods, slow-release fertilizers, and nitrification inhibitors. Considerable opportunity for improvement in these practices exists. More efficient N use also acts to decrease indirect emissions, because less N is lost to nonagricultural ecosystems where it can stimulate N_2O emissions. Similarly, handling of livestock waste to maximize crop use of manure N can decrease emissions. Storing waste anaerobically decreases N_2O emissions, although under such conditions methane is formed, and ideally, methane-capture technology should be used. Using available technology and best management practices, current agricultural emissions of N_2O in the United States could be decreased 30–40%.

Mitigation of N_2O through improved N management has additional payoffs in decreasing fertilizer needs (and costs) as well as numerous environmental benefits, including less NO_3^- pollution and eutrophication of freshwater and coastal zones, decreased NO_3^- concentrations in groundwater (a human health issue), and decreased N deposition on sensitive natural ecosystems.

Mitigation of Methane Fluxes in North American Agriculture

Globally, agriculture is responsible for almost one-half of all CH_4 emission, with the major sources being enteric fermentation in ruminant livestock, livestock waste, and rice production. Agricultural practices also affect the sink strength of aerobic soils that oxidize CH_4. Emissions in the United States (for 1996) were estimated at approximately 8 million tonnes of CH_4, three-fourths of which were attributed to enteric fermentation.

Methane emissions from ruminant livestock (e.g., cattle, sheep, goats) account for 2 to 12% of gross feed-

energy intake, which represents a direct loss of energy to the animals. Manure from intensively managed livestock contributes 10–30 Tg CH_4/yr globally and approximately 1.6 Tg/yr in the United States, mostly from storage under poorly aerated conditions (e.g., waste lagoons). Flooded rice is a major source globally, but because of the limited area cultivated in the United States (<1.5 million ha) it is a comparatively minor source, accounting for approximately 0.3 Tg/yr. Methane oxidation in soils is a sink for atmospheric CH_4, globally estimated at 15–45 Tg/yr. Uptake rates are highest in native ecosystems, and conversion to managed pastures and cultivated crops generally decreases the CH_4 sink, by as much as 35–90%. Sink decreases result in an increase in net emissions.

Opportunities for decreasing CH_4 emissions from intensively managed cattle are limited somewhat in the United States because these operations currently are quite efficient. Improvements can be achieved through bettering feed efficiency, increasing feed digestibility, and inhibiting CH_4-producing bacteria.

Methane produced in manure stored under anaerobic conditions represents a wasted energy source that could be recovered. Large livestock operations lend themselves to use of covered lagoons and large-scale digesters to capture CH_4. Other waste storage practices, such a using solid manure handling and composting under aerobic conditions, decrease CH_4 but can increase N_2O production.

Options for decreasing CH_4 emissions from rice include water management, nutrient management, and *adoption* of new rice cultivars. Periodic drainage and reflooding (versus continuous flooding) has been found to decrease CH_4 loss; however, water availability may limit this option, and periodic drainage also may increase N_2O production. Straw and organic amendments stimulate CH_4 production, whereas certain fertilizers, such as ammonium sulfate and calcium carbide additives, can decrease CH_4 emissions.

Using available technology and best management practices, agricultural emissions from the major sources of CH_4 could be decreased by 20–40%. Many practices for decreasing emissions also improve crop and animal productivity. Less loss of feed energy intake because of CH_4 production means greater livestock weight gain. Methane capture from stored waste would allow farmers to purchase less energy and to decrease CO_2 emissions associated with fossil fuel use.

Full Greenhouse Gas Accounting

Sources and sinks of all GHGs need to be accounted for, and the use of GWP metrics provides a common currency for assessing net impacts. Many practices involve GHG trade-offs; for example, a particular type of manure handling may decrease N_2O emissions but increase CH_4 production. Similarly, increased crop production and residue inputs from higher N fertilizer use may increase soil C but also increase N_2O losses. Thus net effects of management practices, and how they may change throughout time, need to be considered.

Emission of fossil-derived CO_2 results not only from fuel consumption on the farm but also from the production and delivery of fuel, equipment, seed, fertilizer, lime, pesticides, and irrigation water, all of which require inclusion in a full GHG account. These estimates may vary substantially for different practices; for example, estimates of average fossil-derived CO_2 emissions for U.S. corn, soybean, and wheat production vary from 79 kg C/ha/yr for no-till soybean (lowest) to 268 kg C ha/yr for reduced-till corn (highest).

Net GHG accounting shows that N management can be as important as C management for GHG mitigation. In a study of Midwest cropping systems, five of the six systems studied had N_2O flux as the single greatest source of GWP. Ideally, net GHG accounting should be applied at the whole-farm or landscape scale to capture interactions between fields and pastures and confined livestock facilities.

Biofuels

Biofuels are renewable energy sources that include dedicated energy crops, agricultural residues, or CH_4 produced from agricultural wastes. Biofuels can substitute for fossil fuels and thus mitigate CO_2 and other GHGs released from fossil sources. The potential for agricultural biofuel production depends on (1) whether biofuel production can be cost competitive with traditional agricultural products, (2) whether energy produced from biofuels can be cost competitive with other energy sources, and (3) whether there are additional environmental or economic benefits that accrue from biofuel production.

Various energy crops and sources of biofuels and bioenergy, including crop residues, biodiesel, and bioethanol, are being examined. The best choice of a production system will vary geographically and with supply and demand for different bioenergy products. Ongoing technological improvements will affect the comparative advantage of bioenergy systems further. The extent to which bioenergy becomes a significant component of U.S. agriculture will depend on its

economic competitiveness with traditional agricultural products and with conventional petrochemical feedstocks, as well as social and political pressures for more renewable energy sources.

Policy Options and Design

Conceptual Framework

Although the economics of GHG mitigation from energy and industrial sources is understood relatively well, much less is known about the economics of soil C sequestration and other agricultural mitigation options. There are three primary scenarios under which agricultural mitigation options could be established: (1) international agreements that include the potential for agricultural GHG mitigation; (2) national (U.S.) policy promoting agricultural GHG decreases, even in the absence of international agreements; and (3) voluntary, nongovernmental mitigation efforts driven by consumer willingness to pay for "climate-friendly" products. Unless international or national policies generate official credits, C sequestration and GHG mitigation probably will not be a major determinant of farming practice or income. Current U.S. policy is a combination of scenarios 2 and 3, in that the United States officially is supporting agricultural C sequestration but on a voluntary basis.

Issues of Operation and Design

Acceptance and success of agricultural GHG mitigation programs will increase if programs address four key concerns: (1) accounting, (2) observability, (3) timing, and (4) adoption.

Effective GHG mitigation policies will include full accounting for GHG emissions in two respects: across GHGs and across locations (*full land accounting*). Because a given management practice can affect more than one GHG, the need for full GHG accounting is clear. In a national or global accounting system, partial land accounting will raise accountability issues, because areas not included in programs may be responsible for (unrecorded) emissions. But total land and GHG accounting may or may not be required in a domestic or voluntary policy setting.

Land-based emissions of GHGs are nonpoint-source pollutants because individual levels of emissions are difficult to observe, although aggregate emissions can be observable. For GHGs, direct observation may be more difficult than for other nonpoint pollutants (e.g., certain water pollutants where aggregate amounts can be measured in stream flow). An alternative to direct measurement of emissions is to base observations on the land-use and management practices that affect emissions. Thus, control programs could be based on knowledge of aggregate emissions and observable individual actions.

Duration and *permanence* influence the value of sinks, given that C stock gains for a particular change in practices have a limit and are reversible. Despite misconceptions to the contrary, temporary storage of C has value from the standpoint of mitigation, because perturbation of the climate system because of excess GHGs is related to the accumulated amount of GHGs and not (directly) to their annual emissions. On this basis, temporary sinks can be compared with permanent sinks (or emission decreases), where the value of a permanent sink is equal to the accumulated value of the sink during all time periods. Additionally, sinks may provide valuable temporary decreases in atmospheric GHG buildup if technological progress lowers the future costs of emission decreases—an argument often referred to as "buying time." The net value of agricultural sinks depends on their costs and benefits. The damages avoided by decreasing GHG emissions by one unit are the same, regardless of the source (e.g., agricultural sink, energy conservation, or decreased CH_4 emissions). Thus, the cost per unit of decreasing emissions may be quite different, which makes it desirable to choose the method that results in the least damage per unit of cost.

Farmers will participate in GHG mitigation programs only if sufficient incentives are provided. Generally, farmers respond to increased profit possibilities. Other factors, however, influence farmer adoption of alternative technologies—a process referred to as *diffusion* or the aggregated response of an individual farmer's decision to adopt or not to adopt. Diffusion often follows an *S*-shaped curve throughout time, the shape of which can be affected by the presence and imitation of "early adopters," education and outreach, perceived risk, suitability of technology, and thresholds in fixed capital requirements, all acting within a heterogeneous population.

Alternative Policy Designs

For an effective program to be implemented, policy details must be worked out, including (1) definition of the commodity to be targeted; (2) organizational structure; (3) identification and control of GHG "leakage"; (4) enumeration of payment rules, including

time frame and reversibility issues; and (5) monitoring and verification of contracts.

The definition of the commodity at the farm level likely will depend on GHG accounting in other economic sectors. As a rule, it is most useful to account for GHG changes where they actually occur (which would exclude emissions from off-farm production of inputs).

Organizational structures define the participants and their interactions, and, depending on the type of program, different organizational structures will be used. Possible scenarios include (1) programs based on binding international agreements; (2) a government-directed domestic program outside of international obligations; or (3) voluntary, consumer-driven programs.

With international agreements, the government will play a key role in designing and implementing sequestration programs. In a central bidding (tax and subsidy) system the government owns all permits, and GHG emitters must bid for the right to emit, or pay a tax. Farmers engaged in C sequestering practices might bid for the right to sell permits or receive a subsidy. Alternatively, in a tradable permit system the government could issue permits to emitters. In the United States, an example of such a system is the sulfur dioxide permit-trading market. Ensuring the integrity of either system requires (1) an independent monitoring organization that verifies actions taken by farmers and other market participants and (2) a centralized registry for recording transactions. Monitoring and registering organizations likely will need to be accredited by an international body. Because of the large number of farmers who could participate in a trading program, aggregators (e.g., existing farm organizations, cooperatives, or private companies) likely would emerge to represent farmers' interests.

In a domestic GHG mitigation program the basic organizational structure might be similar to the central bidding or tradable permit systems, in which aggregators representing farmers bid for the right to sell sequestered C. Demonstrating the efficacy of agricultural mitigation practices would require a high-quality monitoring and recording structure.

For a consumer-driven voluntary approach, there would be a somewhat different set of institutions and roles for government. The objective of an organizational structure would be to ensure consumer confidence in buying products contributing to mitigation of climate change. The role of government would be similar to that in other markets, including enforcement of property rights and transparent trading rules. Additionally, independent monitoring and registry

would be critical steps for maintaining consumer confidence.

The effectiveness of sequestration can be undermined by the occurrence of "leakage" (e.g., where actions to enhance C storage result in unintended increases in emissions elsewhere). For example, conversion of cropland to grasslands could increase C sequestration in one region but also cause new cropland (and soil C emissions) to be established elsewhere (through effects on commodity prices). Programs need to be evaluated under a broad accounting scheme so that leakage is estimated and the programs achieve cost-effective overall GHG decreases.

Payment rules for mitigation contracts should address (1) choice of levels and timing of benchmark emissions, (2) choice of long-term versus yearly contracts, and (3) impacts of reversibility. Benchmark emission soil C levels can be established based on past behavior. Without a benchmark, farmers might find it profitable to deplete levels today to enhance future sequestration. Long-term contracts are most attractive to offset purchasers (e.g., energy companies) to facilitate longer-term planning, but they entail greater risk to sellers (farmers) and thus generally require greater compensation. Short-term (e.g., annual) contracts provide more flexibility to farmers but typically generate a lower annual price. The impact of reversibility (e.g., reverting to intensive tillage on no-till fields) can be dealt with using a variety of contract structures, including pay-as-you-go, variable length contracts, and C annuity accounts. All three systems are efficient economically and their relative advantages in a particular circumstance will depend primarily on ease of enforcement and implementation.

Any successful GHG program must have effective monitoring and transaction processes at reasonable cost. Because contract values increase with reliability of monitoring, determining optimal monitoring is an economic problem—if monitoring costs are subtracted from contract price, then the optimal amount of monitoring occurs when the incremental benefits from additional monitoring are equal to the incremental cost. Low-cost monitoring systems likely will focus on observation of management practices that influence GHG emissions and/or direct measurement throughout larger aggregated areas, because of the prohibitive cost of direct measurement of GHG fluxes for individual contractors.

Ultimately, GHG mitigation policies in agriculture must be consistent with U.S. domestic agricultural policy and acceptable to the international agricultural policy community. Certain provisions for conservation in U.S. farm policy (e.g., the CRP and the Envi-

ronmental Quality Incentives Program) are broadly consistent with GHG mitigation goals; however, other policy components contain multiple objectives working at cross-purposes with regard to GHG mitigation. If GHG mitigation (and other conservation issues) becomes a more prominent policy objective, then legislative and administrative structures could be modified accordingly, for example, by tying crop payment and insurance subsidies more closely to environmental performance.

In the global context, agriculture presents many opportunities for C sequestration and multiple GHG emission decreases and for producing fossil fuel offsets (e.g., biofuels). Costs vary widely, but many mitigation options could be implemented at C costs below $30/tonne. A fundamental principle of economics states that increased flexibility decreases the cost of meeting a given objective. Applying this principle to the cost of decreasing GHG buildup means that a mitigation strategy that only focuses on using soil C sequestration would be more costly than one that allows a fuller range of offsets from multiple agricultural emissions sources.

Monitoring and Verification of Carbon Sequestration and Greenhouse Gas Fluxes

Quantifying C sequestration and emissions of CO_2, N_2O, and CH_4 presents a number of challenges, including their high spatial and temporal variability and low "signal-to-background" for short-term changes in soil C. These difficulties can be met with a combination of approaches involving (1) direct measurement of C stocks or gas fluxes through time, (2) extrapolation and interpolation using models, and (3) verification of estimated changes.

Direct Measurement

Analytical techniques for measuring soil C are well developed, and modern instruments are highly accurate. Conventional methods require field sampling (Figure S.4) and transport for laboratory analysis, although a variety of new in situ instruments are being developed.

The primary challenges for applying soil C measurements involve standardization of sample preparation and field sampling designs to deal with spatial variability. Required sample numbers increase with

spatial variability and the desired precision. Because much of the spatial variability in soil C is expressed at the field scale, the sampling density (i.e., as samples/ha) for a given level of precision decreases rapidly as the size of the area increases. Thus, sampling costs per unit area decrease as the measurement area increases. Relatively slow changes in total C stocks and high spatial variability dictate that multiyear intervals (e.g., 3–10 yr) should be used for repeated measurements.

Whereas net CO_2 flux can be inferred from soil C stock measurements, N_2O and CH_4 require gas flux measurements, which also can be used for CO_2. Chamber methods and micrometeorological techniques are the two main approaches.

Chamber techniques involve covering an area of soil and measuring the accumulation of gases (in

Figure S.4. Researchers from the University of Arizona download data from a meteorology/flux system that measures the amount of water and C exchanged among vegetation, soil, and atmosphere. Photo courtesy of P. Greb, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

closed-top chambers) throughout time or measuring flow rate and concentrations of gases flowing through open-top chambers. Manual chambers commonly are used, but automated chamber systems have been developed to increase the frequency of sampling. Micrometeorological techniques use frequent measurements of air movement, energy balance components, and gas concentrations to estimate net gas exchange between the land surface and the atmosphere. The techniques require highly accurate instruments with fast response time to make measurements several times per minute, which then need to be integrated over longer time periods. Approaches range in scale from tower-based facilities for field-scale determina-

tion to aircraft-borne sensors for estimating regional level fluxes.

Extrapolation and Interpolation Using Models

Regional estimation requires data gathered at local scales and uses statistical and/or simulation models to extrapolate to regional scales. The models integrate the multiple factors controlling soil C dynamics or GHG fluxes (e.g., climate, soils, topography, land-use, management) that themselves vary with time and space. Typically, regions are divided into many land units or patches (using geographic information systems) deemed relatively homogeneous for the controlling factors. Model calculations are carried out for each combination and then weighted by the area they represent and thus are expanded to the region. The United States has a relatively rich set of existing databases, including plant and crop productivity, soils information, long-term field experiments, land-use and land-cover data, land-management inventories and surveys, topographic data, and climate data that can be applied to regional analyses. Clear information gaps exist, however, and new and more complete national-level information on cropland management, in particular grazingland practices, is needed. New and planned data sources, particularly from remote sensing, will help in making regional analyses.

Net C stock changes and emissions from agricultural soils for the entire United States have been estimated using model-based methods for national reporting purposes. These estimates suggest that U.S. agricultural lands currently are a small sink for C (10–20 Tg/yr), although gains occurring on mineral soils are being offset partly by high C emissions from cultivated organic soils (5–10 Tg/yr).

Verification of Estimated Changes

Independent verification is a crucial component of GHG and C accounting and inventories. Ideally, verification procedures should include comprehensive uncertainty analyses of all components used in producing inventory estimates. These components include uncertainty in observations of management activity data as well as uncertainty in emission/sequestration rates for various practices. Statistical approaches, using Monte Carlo techniques, have been used to estimate uncertainty in the U.S. national soil C inventory. Estimates of the impact of tillage practices on soil C and emission rates from cultivated organic soils contributed the most to overall uncertainty in the inventory.

Targeted sampling, whether as part of a national network of permanent monitoring locations or for a specific C sequestration project, is an important component of an overall verification scheme. An efficient national soil C monitoring program will require a rigorous sampling scheme with well-understood limits preceding sample collection to maximize information gained while minimizing the number of samples, and cost, required.

Introduction

Climate always has been an issue of concern for agriculture, and because climate largely determines crop production potential as well as which crops are grown and where, the two are inexorably linked. Weather conditions in a particular year or part of a year have a major influence on crop yields and hence drive fluctuations in commodity prices and the economics of agriculture. Dealing with climate variability and its interactions with soils, water, and plant processes remains a difficult challenge for farmers and ranchers. How human-induced changes to the atmosphere and land surface affect climate and climate variability is, therefore, a topic of great concern to agriculture.

With adoption of the United Nations Framework Convention on Climate Change, signed by President George H. W. Bush in 1992 and ratified by the U.S. Senate, the United States and 187 other nations committed themselves to decreasing the buildup of greenhouse gases (GHGs) in the atmosphere to "prevent dangerous anthropogenic interference with the climate system." Along with various actions to decrease the atmospheric buildup of GHGs, interest has grown concerning agriculture's role in mitigating GHG increases. Three of the major GHGs—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)—are emitted to and/or removed from the atmosphere in significant amounts through agricultural activities. Thus the potential for agriculture to mitigate GHG emissions has been the subject of intensive scientific investigation the past several years.

This Council for Agricultural Science and Technology report summarizes and synthesizes recent research on the potential to mitigate GHG emissions through improvements in agricultural and land management practices. The report is designed to inform government and industry policy- and decision makers, agricultural producers, personnel in environmental and other nongovernmental organizations, and the general public. A major effort in preparation of the report was the blending of biophysical and ecological information with economic and policy findings and theories, yielding a clearer picture of the potential role of agriculture in GHG mitigation strategies. In addition, a primary aim was to address the three major GHGs that impact potential agricultural activities. Such a comprehensive approach facilitates examination of trade-offs and/or synergisms between carbon sequestration practices and emissions of N_2O and CH_4. It is hoped that this synthesis will enhance information on the potential role of agriculture in efforts, both nationally and internationally, to mitigate the increase in GHGs.

1   Greenhouse Gases and Climate Change

The underlying theme of this document is the general concern of human-induced climate change, largely because of the buildup of concentrations of certain gases in the atmosphere that result in an increase in heat trapped by the Earth. In this section the authors discuss the general nature of this effect and how it relates to agricultural activities. Further, to provide added context to the discussion of mitigation activities, there is a brief review of recent studies on the potential of climate change to impact agriculture.

Greenhouse Gases

Greenhouse gases (GHGs), often called *trace gases*, are present in the atmosphere in small concentrations. The most important gases causing the *greenhouse effect* are water vapor (H_2O), carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and ozone (O_3). Atmospheric levels of trace gases change naturally throughout time, but throughout the past several decades they have been increasing rapidly as a result of human activity. Moreover, new, man-made GHG compounds have been added to the atmosphere. These compounds include chlorofluorocarbons (CFCs), perfluorocarbons (PFCs), hydrochlorofluorocarbons (HCFCs), hydrofluorocarbons (HFCs), and sulfur hexafluoride (SF_6). The GHGs are important to consider because of their link to potential climate change and, as will be discussed later, the potential for altering activities to decrease the release of GHGs into the atmosphere.

The general physics involved in the greenhouse effect are well understood. Solar energy, predominantly as short-wave radiation, arrives from the sun at the top of the Earth's atmosphere. Approximately 30% of this energy is reflected back into space by clouds, aerosols, gases, and the Earth's surface. Of the remaining energy, 20% is absorbed by the atmosphere, including clouds, and 80% is absorbed by the Earth's surface. The Earth's surface is warmed and reemits energy as longer-wave (infrared [IR]) radiation. Part of this radiation is absorbed by GHGs, thus heating the atmosphere. Added heat is sent back to the surface, causing it to become warmer than if direct solar radiation were the only source of energy. This process is known as the greenhouse effect. If concentrations of these GHGs in the atmosphere are altered such that additional energy is absorbed, then global warming can occur (Figure 1.1).

Three characteristics of a gas determine its contribution to the greenhouse effect: its absorptivity for IR radiation, its atmospheric lifetime, and its concentration in the atmosphere. Depending on its chemical properties, a gas absorbs IR radiation in specific energy bands. Because there is overlap in the energy bands in which different gases absorb radiation, the net effect of increasing the concentration of any single gas depends on both the inherent characteristics of the gas and its interaction with the energy absorption bands of other GHGs. The second characteristic, lifetime, determines whether emissions in the atmosphere will contribute to global warming for many years or whether the gas will disappear relatively rapidly. These two properties, absorptivity and lifetime, often are combined into a single index called the Global Warming Potential (GWP) (Textbox 1.1). The third characteristic determining a gas's greenhouse effect is its concentration in the atmosphere. If a gas is not present at sufficiently high concentrations or does not have the potential to reach sufficiently high concentrations, then it is not an important GHG, regardless of its GWP.

The *global mass balance of gas* is the difference between the rate of emissions (E) into the atmosphere and the rate of losses from the atmosphere because of chemical breakdown and uptake by terrestrial and ocean sinks. The term *sink* refers to biological and chemical processes that remove a gas from the atmosphere—such as uptake of CO_2 through plant photosynthesis, dissolution of CO_2 and formation of carbonates in seawater, and oxidation of CH_4 by soil bacteria. The loss rate can be expressed by the atmospheric concentration (C) of the gas and its atmospheric lifetime (τ), where dC/dt denotes the change in concentration throughout time:

$$\frac{dC}{dt} = E - C/\tau$$

Figure 1.1. The greenhouse effect is caused by the absorption of long-wave radiation from the Earth's surface by trace gases, radiating the energy back to the Earth's surface (see also Table 1.1). (Source: National Assessment Synthesis Team, Washington, D.C.)
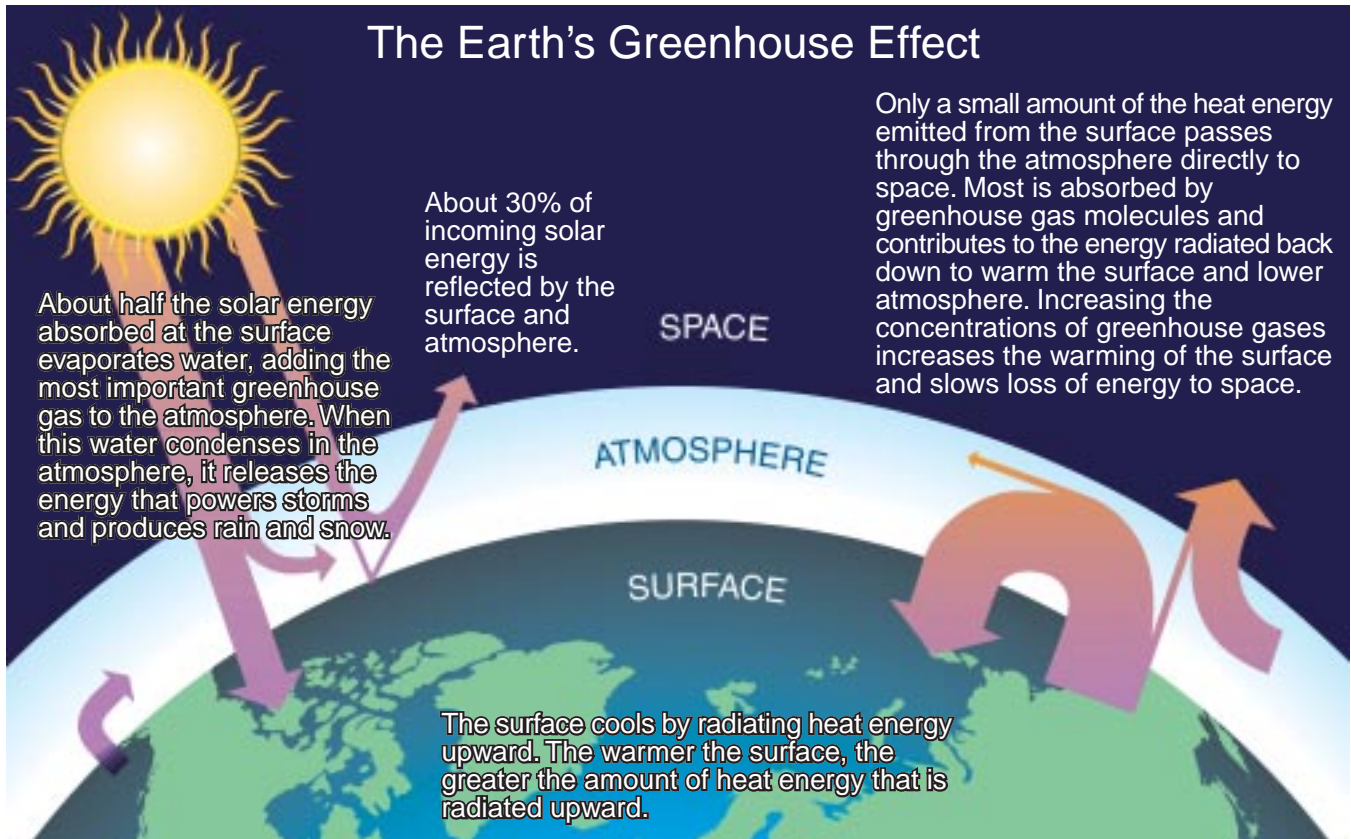
This equation indicates that there are three ways to change the atmospheric concentration of a gas: (1) if the emission rate, E, or the atmospheric lifetime, τ, is increased, then the equilibrium atmospheric concentration will increase proportionally; (2) even when emissions are constant, if the concentration of a gas is not at equilibrium, the concentration will change until the loss rate, C/τ, equals the emission rate, E; and (3) if a new gas is emitted at a constant rate to the atmosphere, time will be required for the concentration to increase to the point at which losses equal emissions.

Only a few of the many atmospheric trace gases have the combination of the three characteristics

$$GWP = \frac{\int_{0}^{T} a_i X_i(t)dt}{\int_{0}^{T} a_{CO_2} X_{CO_2}(t)dt}$$

Here, X represents the concentration of a gas (i), and a represents its specific radiative forcing (in watts [W] per square meter [m] per kilogram [kg] of gas). T is the time period of integration, or time horizon. Because gases have different average residence times in the atmosphere, GWP values vary depending on the time horizon chosen. The GWP for some of the GHGs is given here (from IPCC 2001b).

Gas	20-yr time horizon	100-yr time horizon
CO_2	1	1
CH_4	62	23
N_2O	275	296
SF_6	15,100	470
CFC-11	6,300	4,600

Textbox 1.1. Global Warming Potential (GWP) is a normalized measure of the effect of different gases on warming, where, by convention, CO_2 is used as a reference and is assigned a value of 1. The formula for calculating GWP is the per-unit radiation absorptivity (or radiative forcing) of a gas times its concentration, integrated throughout time, relative to CO_2.

required to contribute significantly to global warming (Figure 1.2). Chlorofluorocarbons have not achieved concentrations high enough to have a dominant effect on global warming, and the 1987 Montreal Protocol to protect stratospheric ozone has eliminated new production of CFCs. Unless something unexpected happens, CFCs will play a declining role in future global warming. Perfluorocarbons, HFCs, and SF_6, although having high GWPs, or very long lifetimes, are present in extremely low concentrations and have uncertain potentials for increasing to appreciable levels. Water vapor and ozone are governed primarily by climate feedbacks and indirect effects of human activity. Thus, CO_2, CH_4, and N_2O remain the major GHGs directly affected by human activities and are of the most concern for global warming. Each

the greatest amount of C (approximately 40,000 petagrams [Pg]; 1 Pg = 1 billion metric tonnes = 10^{15} grams [g]). The terrestrial reservoir contains a much smaller amount of C (2,000–2,500 Pg), of which 1,500–2,000 Pg C resides in the soil with approximately 600 Pg C in vegetation. The atmospheric reservoir contains the smallest amount of C (currently approximately 770 Pg), and nearly all C in the atmosphere is in the form of CO_2. By serving as a conduit between the other two reservoirs, the atmosphere plays an important role in the global C cycle, and the concentration of CO_2 in the atmosphere is governed largely by the dynamic exchanges among these three reservoirs.

The terrestrial system, including both vegetation and soil C pools, is the second largest of the three glo-

Figure 1.2. Changes in annual radiative forcing, measured as energy (watts, W) per square meter (m^2) of the Earth's surface, because of changes in the main GHGs (occurring during the period 1750–2000). Also shown is direct radiative forcing because of changes in aerosols (combined sulphate plus C aerosols from fossil fuel and biomass combustion) and from increased solar radiation. Forcing from mineral dust, albedo changes, and indirect aerosol effects not shown. (Adapted from IPCC 2001a.)

Figure 1.3. Overview of the global C cycle showing estimates of current terrestrial, oceanic, and atmospheric C stocks in petagrams (Pg) as well as annual anthropogenic C inputs. (Adapted from Schimel et al. 1995.)

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

Carbon (C) circulates among three distinct global reservoirs: the atmosphere, the oceans, and terrestrial systems (Figure 1.3). The oceans contain by far

bal reservoirs. The main flux between the atmosphere and terrestrial ecosystems results from the uptake of CO_2 by plants, through the process of photosynthesis, and the release of CO_2 to the atmosphere by plants, animals, and microorganisms, through the process of respiration. Some CO_2 also is released by fire.

Carbon dioxide uptake by plants, also called *gross primary production* (GPP), depends on the availability of light, water, nutrients, and temperatures suitable for metabolic function. Approximately one-half of the C fixed into organic compounds during photosynthesis is used directly by plants for energy and maintenance and is respired in a short time, returning CO_2 to the atmosphere. The remaining fraction, called *net primary production* (NPP), is the rate at which C is incorporated into plant tissues. Certain 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.

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.

The rate of C storage in terrestrial ecosystems, referred to as *net ecosystem production* (NEP), is the difference between NPP and the decomposition of plant litter and soil organic matter (SOM). At equilibrium, NPP is balanced by organic matter decomposition and NEP is zero. Of course, because of human or other disturbances and subsequent plant regeneration, no small area of terrestrial vegetation ever is in equilibrium for long. The notion that NEP may be approximately zero requires an averaging of sufficiently large areas for sufficiently long periods and making allowance for the effects of disturbances causing C losses to the atmosphere.

Over geologic timescales the concentration of CO_2 in the atmosphere fluctuates because of natural processes. During the last millennium, however, CO_2 concentrations had been relatively stable until the past 50–100 years (yr), during which time CO_2 concentrations increased at an accelerating rate (Figure 1.4). Since about 1850 the amount of atmospheric CO_2 released from the burning of fossil fuels, the making of cement, and the converting of native ecosystems to agricultural use has increased greatly. Before 1850, the atmosphere contained approximately 600 Pg C in the form of CO_2, equivalent to a concentration of 280 parts per million on a volume basis (ppm_v). In the mid-1990s the atmosphere contained approximately 770 Pg C, with a concentration of 365 ppm_v; as of 2003, atmospheric concentration exceeded 370 ppm_v.

Rates of change for the major global C stocks have been estimated for the past two decades (Table 1.1). Burning fossil fuel and producing cement contributed approximately 6.3 Pg C/yr to the atmosphere during the 1990s, almost a petagram more than in the preceding decade. The world's oceans have absorbed almost 2 Pg C/yr during the past two decades. Land-use conversions are estimated to have contributed an

additional 1.7 Pg C/yr of CO_2 emissions during the 1980s. The land-use emissions during the 1980s were roughly offset by enhanced uptake of CO_2 by the world's terrestrial ecosystems, likely because of regrowth of forests on land no longer cultivated, as well as stimulated growth because of CO_2 fertilization and nitrogen deposition. Several lines of evidence suggest that this absorption (commonly called
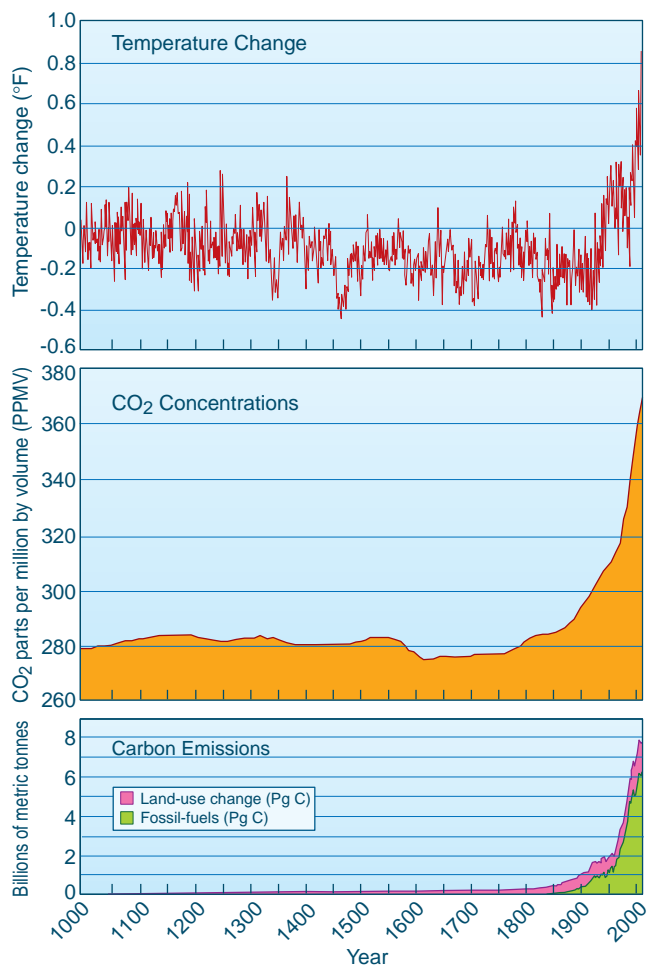
Figure 1.4. Mean Northern Hemisphere temperature (top panel), global concentration of CO_2 (middle panel), and annual C emissions from fossil fuel combustion and land-use change (bottom panel), during the past 1,000 years. (Source: National Assessment Synthesis Team, Washington, D.C.)

a sink) occurred largely in the northern midlatitude temperate zone, whereas emissions from land-use conversion occurred mainly in the tropics (McGuire et al. 2001; Tans, Fung, and Takahashi 1990). Overall, a small net uptake of CO_2 (0.2 Pg C/yr) was estimated for the 1980s, although the uncertainty around

Table 1.1. **Major net sources and sinks of C (in billion metric tonnes C/yr [Pg]) in the global budget for the 1980s and 1990s (adapted from IPCC 2001b[1])**

	1980–1989	1990–1999
	Pg C/yr	
Atmospheric increase in CO_2-C	3.3 ± 0.1	3.2 ± 0.1
Emission of C from fossil fuel and cement	5.4 ± 0.3	6.3 ± 0.4
Ocean-atmosphere flux	-1.9 ± 0.6	-1.7 ± 0.5
Land-atmosphere flux	-0.2 ± 0.7	-1.4 ± 0.7
(Net source from land-use change)	(1.7 [0.6–2.5])	NA
(Net sink from other terrestrial C uptake)	(-1.9 [-3.8–0.3])	NA

[1]Positive values = net increases in atmospheric CO_2 concentrations and net CO_2 flux to the atmosphere; negative values = net uptake of CO_2. NA = reliable estimates not available at present.

the terrestrial emissions and sinks is high. The net result of all these emissions and sinks has been an increase in atmospheric CO_2 at the rate of approximately 3.3 Pg C/yr.

Considerable scientific investigation has been directed at determining how long and at what rates terrestrial and ocean systems will continue to take up increasing concentrations of atmospheric CO_2. Future CO_2 emissions from fossil fuel burning undoubtedly will increase, but at what rate? The Intergovernmental Panel on Climate Change (IPCC) developed various emission scenarios using assumptions about future trends in key fossil-fuel-use determinants such as population, economic growth, technological change, land use, and emission control policy (IPCC 2001b). After the testing of various sets of plausible assumptions, scenarios were developed that ranged from projecting emissions of as little as 4.6 Pg C/yr to as much as 35 Pg C/yr in 2100; these figures compare with 6.3 Pg C/yr for the 1990s.

Nitrous Oxide

Nitrous oxide is a highly stable, long-lived trace gas found in the atmosphere at approximately 1/1,000[th] the concentration of CO_2. Atmospheric concentrations of N_2O are increasing at 0.2 to 0.3% per yr, most of which is attributable to human activities (Figure 1.5). Concentrations in the troposphere (314 parts per billion volume [ppb_v]) in 1998 were approximately 14% higher than those in preindustrial times (ca. 270 ppb_v) and are growing by approximately 0.8 ppb_v per yr (IPCC 1996a; Prather et al. 1995). Among the major GHGs, the global budget for N_2O (estimated here for the year 1990) is the least well quantified, especially with respect to terrestrial sources (Table 1.2). The atmospheric concentration increase means
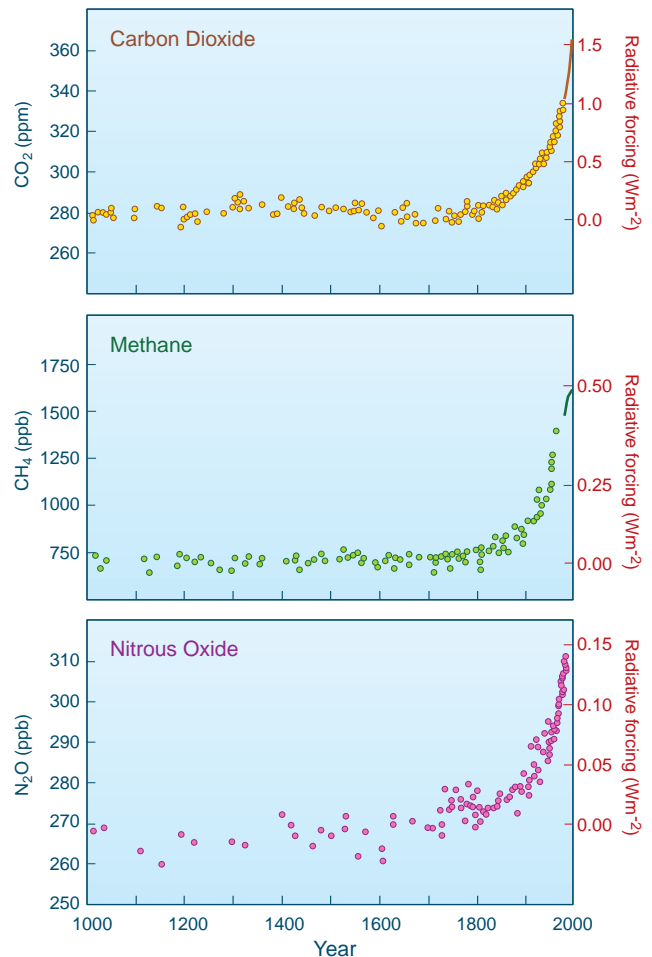
Figure 1.5. **Atmospheric concentrations of each of the biogenic GHGs: CO_2 (top panel), CH_4 (middle panel), and N_2O (bottom panel), during the past 1,000 years, based on ice core data and, in the past few decades, direct measurements in the atmosphere. (Adapted from IPCC 2001c.)**

Table 1.2. Estimated global sources and sinks of N_2O, for 1990. The difference between total sources and sinks reflects uncertainty in the estimates (based on data from Kroeze, Mosier, and Bouman 1999; Mosier et al. 1998a; Prather et al. 1995; adapted by A. R. Mosier)

Sources/Sinks	Tg N_2O-N/yr
Sources	
Industrial	
Combustion	0.9
Manufacturing	0.3
Biomass burning	0.6
Natural ecosystems	
Oceanic	3.0
Terrestrial	6.0
Agriculture (direct and indirect)	
Synthetic fertilizer	0.9
Grazing animals	0.6
Biological N_2 fixation	0.1
Crop residues	0.4
Cultivated peat soils	0.1
Animal waste handling	2.1
Nitrate leaching	1.6
Human sewage	0.2
N deposition on agricultural land	0.3
(Total agricultural)	(6.3)
Total sources	17.1
Sinks	
Photolysis and other reactions in atmosphere	12.3 (9.8–14.8)
Atmospheric increase	3.9 (3.1–4.7)
Total sinks	16.2 (12.9–19.5)

that a net amount of 3.9 (+/- 0.8) teragrams (Tg; 1 Tg = 10^{12} g) of N_2O-nitrogen (N) is added to the atmosphere annually. In the stratosphere, photolytic reactions annually remove approximately 12.3 (+/- 2.5) Tg N. There are no other known sinks for N_2O in the atmosphere, which implies that sources of N_2O must total approximately 16.2 (+/- 3.3) Tg N/yr in order to balance sinks.

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. Recent estimates (Kroeze, Mozier, and Bouwman 1999; Prather et al. 1995) suggest that manufacturing and industrial combustion together account for approximately 1.2 Tg N/yr and oceans for 3 Tg, which leaves the remainder—11.9 Tg N/yr—to nonindustrial terrestrial sources.

Biomass burning, both from natural and human-caused fires (including those associated with forest clearing and agricultural waste handling), seems to account for 0.6 Tg N/yr. Significant uncertainty ex-

ists with respect to how the remaining 11.3 Tg N/yr should be partitioned between natural ecosystems and agriculture, owing largely to the difficulty of directly measuring N_2O fluxes. Current estimates attribute approximately 6.3 Tg N/yr to agriculture; this is equivalent to 0.8 Pg CO_2-C/yr based on a 100-yr GWP *time horizon* (Robertson 2004).

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 16% of the radiative effects of increasing GHG concentrations (IPCC 2001b).

Since the mid-1700s, the atmospheric concentration of CH_4 has increased by approximately 145% (IPCC 1995). Systematic observations of atmospheric CH_4 concentrations were taken first in the 1980s, but measurements of air trapped in ice cores now extend the record back in time (Figure 1.5). From 1900 to 1950, atmospheric concentration of CH_4 increased from approximately 900 ppb_v to 1,100 ppb_v. After World War II the rate of increase accelerated until the early 1980s, when concentration reached 1,600 ppb_v. Subsequently, CH_4 has increased, but less rapidly; today, it is increasing at <5 ppb_v /yr (Dlugokencky 2000). The reasons for the slower rate of increase in atmospheric CH_4 concentrations still are unclear, but may involve both an increased rate of tropospheric destruction of CH_4, because of more hydroxyl (OH) radicals in the atmosphere, and a decreased growth rate of one or more of the sources of CH_4 emissions.

Total emissions of CH_4 to the atmosphere are approximately 450 to 600 Tg/yr (Table 1.3), of which approximately 30% is from natural sources and approximately 70% from anthropogenic, or human-influenced, sources (Khalil 1999; Watson et al. 1992). Anthropogenic sources are broken down further into approximately 100 Tg/yr from fossil fuel production (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. 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,

Table 1.3. Estimated global sources and sinks of CH_4. Main agricultural sources in bold (based on data from Watson et al. 1992; Matthews et al. 2000; adapted by A. R. Mosier)

Sources/Sinks	Tg CH_4 yr^{-1}
Natural sources	
Wetlands	100–200
Termites	10–50
Oceans	5–20
Freshwater	1–25
CH_4 hydrate	0–5
Anthropogenic sources	
Coal mining, natural gas and petroleum industry	70–120
Rice paddies	**9–25**
Enteric fermentation	**65–100**
Animal wastes	**10–30**
Domestic sewage treatment	25
Landfills	20–70
Biomass burning	20–80
Sinks	
Atmospheric removal	450–600
Oxidation in soils	15–45
Atmospheric increase	28–37

or landfills) represents a potential energy source as well as a mitigation opportunity.

Current estimates attribute approximately 180 Tg CH_4/yr to agriculture and biomass burning; this is equivalent to 1.1 Pg CO_2-C/yr based on a 100-yr GWP time horizon (Robertson 2004).

Impacts of Climate Change

Greenhouse gases are of concern primarily because of their potential role in promoting rapid and undesirable change in climate. Changes in the Earth's energy balance, and the resultant effects on climate, are caused by many factors, both natural and human-caused. A collective term used for perturbations to the earth's energy balance is *radiative forcing*. Of the factors that lead to global warming (i.e., a positive radiative forcing), increases in GHGs are estimated to account for approximately 70% of the warming effect (Hansen and Sato 2001). Black carbon (soot) and increased solar irradiance are thought to be responsible for most of the remaining 30% (Figure 1.2). Although other anthropogenic influences—in particular the enhanced production of sulfate and other aerosols—impart a negative (i.e., cooling) forcing, the change in the net radiative forcing compared with the preindustrial era has been toward a warming effect of approximately 1.6 Watts (W)/m^2 (Hansen and Sato

2001). Most climate scientists agree that GHGs have contributed to an observed warming of the Earth during the past 100 years, although uncertainty remains with respect to the exact amount of warming and the relative contribution of GHGs versus other sources (IPCC 2001b).

As for the Earth as a whole, climate during the last century in the United States has not remained constant. Observations from a network of more than 1,200 weather stations indicate that temperatures have increased, on average, by 0.6° Celsius (C) since 1900 (NAST 2000). This warming has not been uniform across the United States. The Northeast, upper Midwest, and Southwest have experienced an average increase approaching 2°C, and Alaska has recorded a near 3°C increase. The Southeast and the southern Great Plains, on the other hand, have been cooling on average during the past century, although temperatures have been rising in these regions since the 1970s. Most of this warming has occurred in winter. Average U.S. precipitation has increased by 5 to 10% during the last 100 yr, primarily through having more days with high rainfall. Increases have been greatest in the Midwest, southern Great Plains, and Pacific Northwest. Decreases have occurred in the northern Great Plains.

The array of possible implications for human activities and the ecology of the planet is significant. Consideration of the potential impacts of climate change and GHG increases has focused on the next several decades, during which time GHG concentrations are projected to continue to increase, giving rise to progressively greater climate perturbations. Global climate models (also known as general circulation models or GCMs) often are used to estimate what climate may be like in the future as GHGs increase in the atmosphere (IPCC 1996a, 2001b). Recent simulations of potential future world climate have been performed using the Canadian Climate Centre (CCC) model and the Hadley Centre (HC) model, as discussed and used in the *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000, 2001). Both models were initialized using current GHG concentrations and subsequently run from 1995 to 2100 using GHG forcings specified by the IPCC IS92a projections of atmospheric GHG concentrations (IPCC 1996a). These GHG projections were formulated based on projections of world population, economic growth, energy consumption, and policy initiatives that affect GHG emissions. The results of these GCM runs provide "intermediate" future scenarios for climate change and represent neither a "best-case" nor a "worst-case" scenario with

respect to future emissions, projecting a doubling of the 1990 CO_2 concentration (350 ppm$_v$) by the year 2100 (to 700 ppm$_v$).

In these future scenarios (Figure 1.6a, b), average warming in the United States will be somewhat greater than in much of the rest of the world throughout the twenty-first century. The CCC model projects a 5.5°C increase in the middle of the United States and a 2–3°C increase in coastal regions. Most of this increase is because of higher winter and nighttime temperatures. The HC model projects smaller increases in temperature: the eastern United States undergoes 2–3°C temperature increases, whereas the rest of the nation undergoes 4°C temperature increases by the year 2100.

The CCC model projects large percentage increases in precipitation of up to 100% for California and the Southwest. East of the Rocky Mountains, the southern half of the nation is projected to experience decreases in precipitation. A large region centered on the Oklahoma panhandle would experience the greatest declines, of up to 20%, with the Southeast next in

the severity of precipitation declines, extending, with more moderate temperature declines, northward into New England. Precipitation is projected to increase in the Northern Great Plains and the Great Lakes regions.

The HC model projects similar large increases in precipitation for California and the southwestern United States. For the eastern two-thirds of the United States, the HC model projects moderate increases in precipitation in most areas. It projects decreases along the Gulf Coast and in northern Washington and Idaho.

In both models, projected increases in precipitation are associated with an increase in heavy precipitation events associated with convective rain. In the Southwest, an increase in precipitation is associated with a warmer Pacific that would pump more moisture into the region through storms. In the Sierra Nevada and Rocky Mountains, much of this increased precipitation is likely to fall as rain rather than snow, thereby affecting the amount and timing of snowmelt available for irrigation waters.
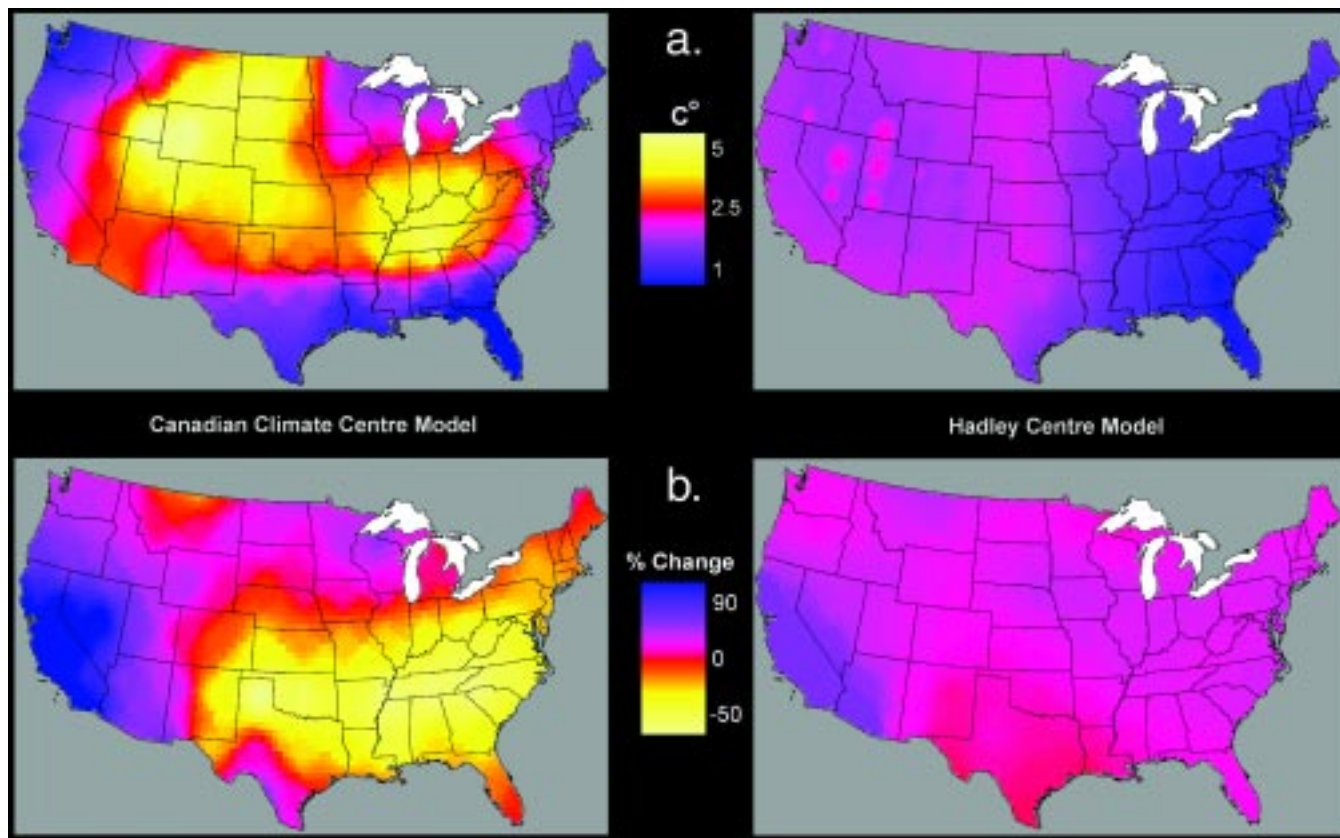
Figure 1.6. Projected changes in temperature (a) and precipitation (b) by 2050 for climate change scenarios using the Hadley Centre and Canadian Climate Centre global climate models, including projected changes in GHG concentrations and sulphate aerosols. (Source: Data from C. Rosenzweig.)

Effects of Climate Change on Agriculture

Climate change is a phenomenon that can affect natural as well as managed ecosystems, including agricultural systems, around the world. A number of studies have examined the potential effects of climate change on the Earth's ecosystems and on human society. The next sections review some of the recent work on potential impacts of climate change on agriculture in the United States, first examining physical effects on various agricultural components and then potential economic impacts.

Crop Growth and Production

Weather Effects

Ample scientific literature predicts that changes in climate will affect crop yields and agricultural productivity and the geographic distribution of crop species. Effects vary depending on the degree of changes in temperature, precipitation, cloudiness, etc., and the type of crop and management system involved. Yields will be affected by changes in temperature, precipitation, growing-season length, and extreme-weather events, particularly during critical phases of crop development (e.g., more frequent or prolonged heat waves at corn anthesis, or late frost at wheat flowering). Farmers readily understand that such changes in weather patterns have the potential to affect their crops.

Studies based on global projections of climatic changes still are fraught with uncertainty. Whereas most GCMs agree with respect to the regional distribution of temperature change—for instance, most agree that the high latitudes will warm more than the tropics—there is little agreement about how changes in precipitation will be distributed regionally. Thus it is necessary and responsible for modeling studies to use two or more scenarios of projected climate change to provide a range of projections of how certain crops will fare in different regions.

Effects of Increased Atmospheric Carbon Dioxide Concentration

It is well established that changes in atmospheric CO_2 concentrations can affect crop yields and agricultural productivity. Photosynthesis, respiration, and transpiration are the plant processes affected most directly by changing CO_2 levels. More CO_2 in the atmosphere tends to increase the diffusion of CO_2 into the leaves, increasing its fixation through photosynthesis into carbohydrates. Experiments in controlled environments have demonstrated that photosynthesis in single leaves and whole plants is increased in CO_2-enriched atmospheres (Acock and Allen 1985; Cure and Acock 1986; Kimball 1983; Poorter 1993). Free-air CO_2 enrichment (FACE) experiments (Hendrey, Lewin, and Nagy 1993) generally confirm the positive results obtained in controlled environments (Figure 1.7).

Plant species differ in their responses to CO_2 because of differences in their photosynthetic pathways. The two main types of photosynthetic pathways are referred to as C_3 and C_4 (so named because the precursor molecule in the C_3 pathway has 3 C atoms ver-

Figure 1.7.   Scientists measure the growth of wheat surrounded by elevated levels of atmospheric CO_2 near Phoenix, Arizona. The study, called the Free Air Carbon Dioxide Enrichment, is to measure CO_2's effect on plants. It is the largest experiment of this type ever undertaken. Photo courtesy of J. Dykinga, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

sus 4 C atoms in the C_4 pathway). The C_3 plants (small grains, legumes, cool-season grasses, and most trees) usually respond more positively than the C_4 plants (warm-season grasses, corn, sorghum, millet, and sugarcane). Responses also depend in part on environment (e.g., water and nutrient availability) and in part on genetics. There is some evidence, too, that CO_2 enrichment is relatively more effective in plants that endure high-temperature, moisture, and salinity stress, at least in the short term (Kimball 1983; Kimball and Idso 1983).

Another important physiological effect of CO_2 enrichment is the partial closure of *stomates*, the small openings in leaf surfaces through which CO_2 is absorbed and water vapor released. Accordingly, a rise in atmospheric CO_2 may decrease transpiration even

while stimulating photosynthesis. Thus, water-use efficiency (biomass accumulation per amount of water transpired) also is improved (Allen et al. 1987; Morison 1985). The influence of CO_2 enrichment on photosynthesis and transpiration often is termed the "*CO_2-fertilization effect.*" Whether the effects of CO_2 enrichment demonstrated in controlled environments and in the limited number of FACE studies will prevail in farmers' fields still remains uncertain, in part because of the unknown effect of CO_2 enrichment on potential interactions with weed and insect pests (Rosenzweig and Hillel 1998).

Projected Effects on Global Crop Production

Many modeling studies, summarized by the IPCC (IPCC 1996b, 2001a), the *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000, 2001), and others (e.g., Rosenzweig and Hillel 1998), lead to a number of generalizations with respect to the global distribution of climate change effects on crop production. At the higher latitudes, increased temperatures may benefit crops currently limited by cold temperature and a short growing season. In the midlatitudes, however, higher temperatures are likely to exert a negative influence on yields by hastening maturity and shortening the growing period. In the lower latitudes, higher temperatures also shorten growing periods. In addition, heat and water stresses are exacerbated in tropical environments, resulting in greater declines in crop yields than at middle and high latitudes, notwithstanding the potential benefits of atmospheric CO_2 enrichment. There is the possibility (although many variables make such predictions extremely difficult) that agriculture in certain parts of the world will benefit from global climate change, at least in the short term, especially in cold-limited, high-latitude and high-altitude regions and in dry regions where rainfall may increase. On the other hand, climate change may bring yield declines in other regions, especially in lower-latitude, semiarid regions that become drier.

Most studies addressing climate change and global agriculture conclude that production can be maintained to meet the needs of a growing population under conditions of a doubled CO_2 environment (IPCC 1996b; Rosenzweig and Parry 1994). Given current GHG emission rates, this doubling would occur sometime in the second half of the twenty-first century. Despite adequate global food production, however, projections consistently show regional disparities.

Generally, middle-to-high latitudes experience increases in productivity depending on crop type, growing season, temperature-regime changes, and precipitation seasonality. In the tropics and subtropics, however, where certain crops are near their maximum temperature tolerances and where dryland, nonirrigated agriculture predominates, yields are likely to decrease, increasing the risk of hunger for subsistence and low-income farmers.

Projected Effects on U.S. Crop Production

Studies focused on the United States suggest generally that overall production of the major grain crops will not decrease and may even increase, depending on the specific climate change modeling projection, region, crop, and time period. The recent *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000, 2001; Reilly et al. 2002, 2003) found a range of possible outcomes throughout the course of the twenty-first century. In certain scenarios, increases in precipitation and responses to CO_2 raised crop yields, thereby benefiting consumers but undermining farmers' profits. Simulations of wheat yields for projected climates during the 2030s (Figure 1.8) indicate that certain regions may have improved yields whereas others may have yield losses (Tubiello et al. 2002). Such effects could lead to shifts of agricultural production zones within the United States. Furthermore, crops will be affected
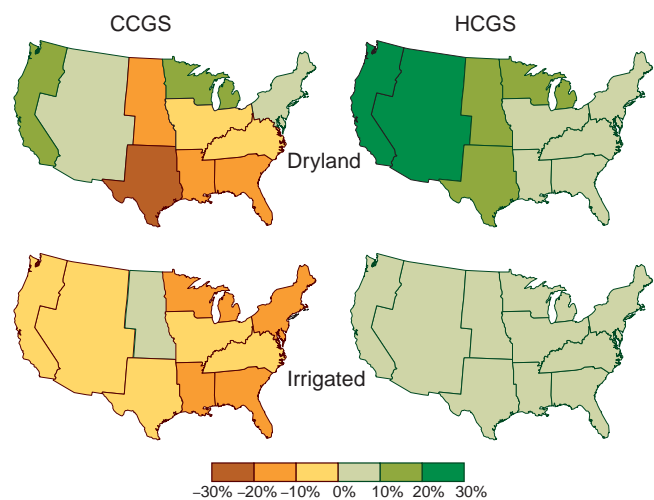
Figure 1.8. Projected changes in winter wheat yields under rainfed and irrigated conditions in the decade of the 2030s, based on climate change scenarios using the Canadian Climate Centre (CC) and the Hadley Centre (HC) global climate models. GS is greenhouse gases with sulfate aerosols; direct CO_2 effects are included. (Source: Tubiello et al. 2002.)

differently. Farmers, the industries supporting them, and commodities markets also may be required to adapt to changes in crop productivity and geography.

Brown and Rosenberg (1999) examined the effects of increasing climate change on yield of dryland corn in the major current U.S. production regions, using the HC climate model. The GCM climate change projection was evaluated for two points in time, 2030 and 2095. The HC model projected modest increases in temperature (1–2°C) by 2030 and precipitation increases of 25 to 125 millimeters (mm) per yr throughout most of the corn-growing region. Temperatures in 2095 were projected to increase by 2.0–3.5°C, and precipitation by more than 175 mm/yr throughout the entire region. The climate change scenario was applied with two levels of atmospheric CO_2 concentration: 365 ppm (near the current ambient) and 560 ppm to represent projected increases in CO_2 concentrations and the lack of, versus the occurrence of, a CO_2-fertilization effect, respectively. Yield results are summarized by region in Figure 1.9. Data in the figure are aggregated from simulations of crop yield on more than 200 representative farms.

Corn yield in the Erosion Productivity Impact Calculator (EPIC) model used for this analysis is directly proportional to biomass production, which is favored by a decrease in cold stress and a lengthening of the growing season in the Great Lakes region, the Corn Belt, and the Northeast. Under climate change, yields in these regions increase at current CO_2 concentrations and improve still more at higher concentrations. With no CO_2 fertilization (i.e., 365 ppm CO_2), yields decrease in the lower Mississippi Valley

(Delta), Appalachian, and Southeastern regions where higher temperatures shorten the growing season. Yield declines in these regions are overcome by CO_2 fertilization (i.e., 560 ppm CO_2) in all regions.

Climate Variability and Extreme Events

Climate change is likely to include changes in climate variability as well as in average conditions. If temperature variability increases, crops growing near the top or bottom of their optimal temperature ranges could be affected adversely, because diurnal and seasonal temperature fluctuations often exceed the optimal range for crop growth and development. If temperature variability diminishes, however, crops growing near their optimal ranges could benefit. Increases in daily temperature variability can decrease wheat yields through lack of cold hardening and resultant winterkill. Extremes of precipitation (i.e., droughts or floods) are detrimental to the productivity of dryland crops. Increased drought frequency would increase the need for irrigation, whereas increased flooding could lead to greater damage because of waterlogging of soils, crop lodging, and pest infestations (Rosenzweig et al. 2002).

Figure 1.10 shows modeled effects of increased

Figure 1.9. **Modeled dryland corn yields for major cropping regions in the United States, comparing current climate with climate in 2030 (H1) and 2095 (H2), from scenarios using the Hadley Centre model. Results are for current climate and climate change, with and without including increased CO_2 (365 ppm [ambient] versus 560 ppm). (Source: Brown and Rosenberg 1999.)**
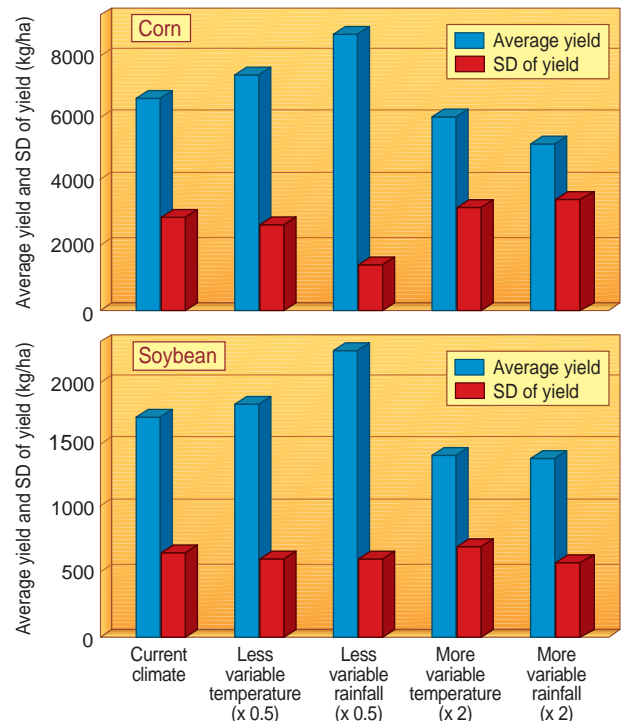
Figure 1.10. Effects of changes in variability of temperature and precipitation on average yields and yield variability of corn and soybean, from model simulations for Des Moines, Iowa. (Source: Rosenzweig et al. 2000.)

daily temperature and precipitation variability on corn and soybean at Des Moines, Iowa (Rosenzweig et al. 2000). The overall effects on corn and soybean are similar, but more severe for soybean. Increased climate variability increases crop yield variability. If variability in temperature or precipitation doubles, yields decrease and crop failures increase in number. Failures under doubled temperature variance occur because of a slowing of grain-filling that extends the growing period into the frost season. Doubling the variability of precipitation increases the frequency of crop failure because of drought in both corn and soybean. Halving precipitation variability results in an increase in mean yield and in a drop in the variability of yields year to year.

In addition, one marker for extreme events that has recently received considerable public attention is the El Niño-Southern Oscillation (ENSO) climatic phenomenon. Timmermann and colleagues (1999) present results from a modeling study implying that global climate change would alter ENSO characteristics causing (1) the mean climate in the tropical Pacific region to change toward a state corresponding to present-day El Niño conditions, and (2) stronger interannual variability with more extreme year-to-year climate variations and a more skewed interannual variability with strong cold events becoming more frequent. There could be implications for total agricultural production and economic value as investigated in Chen, McCarl, and Adams (2001).

Water Resources

Climate change will affect the availability of water supplies for irrigation and the requirements of crops and livestock for water. Future availability of water for agriculture depends, in part, on possible changes in precipitation, potential and actual evaporation, and runoff, at the scale of watersheds and river basins. Warmer winters will induce loss of natural storage in mountain snowpacks and subsequent shrinking of stream flows in late summer and fall (Gleick 1987). Crop water requirements will be affected by increased evaporative demand. These and other changes likely will affect the management of water resources, including reservoir operations, water allocations, and irrigation system development (Figure 1.11). Climate change also may increase the competition between industrial and domestic users, as well as the amount of water needed to sustain nonagricultural ecosystems in riparian areas and watersheds.

Studies have simulated the effects of climate

change on agricultural water supply and demand, both in the United States and globally. Although heavily dependent on the precipitation changes projected by climate models, most projections suggest that for the United States in the near term (to the 2030s), the relative abundance of water for agriculture can be maintained under climate change (NAST 2000). The *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000) emphasizes, however, that these findings should be viewed with caution, specifically as they relate to availability and quality of water for agriculture. Agriculture in water-scarce regions already is experiencing growing competition for water with municipal, industrial, urban, and environmental uses. Although reliable information on changes in storm frequency and intensity is not available yet, changes in these key processes can have important

Figure 1.11. Snowmelt runoff fills a reservoir in the Rocky Mountains near Dillon, Colorado. Photo courtesy of S. Bauer, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

effects on water availability and agricultural production. Furthermore, climate model projections indicate increasingly dramatic changes in the 2050s and beyond, with droughts and floods more likely to occur in several regions.

The seasonality of water supply also changes as a result of global climate change, and thus it is likely that the intraseasonal timing of water for crops will require increased attention. Global climate models project the potential for both increasing precipitation and intensifying hydrological variability; these projections cause concern regarding workability of soils in the spring and waterlogging damage to crops in the summer. Additional investment in drainage may be necessary if flooding of agricultural land occurs more frequently.

A study by Strzpek and colleagues (1999) linked climate change scenarios with hydrologic, agricultural, and water-planning models to evaluate future water availability for agriculture. Their analysis was applied to the U.S. Corn Belt using forecasts of climate change, agricultural production, population change, and gross domestic product growth. The study results suggest that, for the U.S. Corn Belt in the near term (2010s to 2020s), the relative abundance of water for agriculture can be maintained under climate-change conditions, but that greater changes are likely in the second half of the twenty-first century. Changes in the seasonality of water supply were projected, as well as increased decadal variation. Studies in more arid regions, such as the study by Chen, Gillig, and McCarl (2001) in the Edwards Aquifer region of Texas, suggest that effects of climate change on water resources may induce water-reallocation actions.

Grazing Lands

Grazing lands, predominantly composed of grasslands, occupy 47% of the Earth's land area (Williams et al. 1968) and 54% of the conterminous United States (Sobecki et al. 2001). A majority of U.S. grazing lands are in the central prairie region, with a transition from tallgrass to shortgrass prairie following the east-west moisture gradient. Grassland ecosystems are diverse and have evolved in response to large seasonal and annual fluctuations in moisture, temperature, and grazing intensity (Knapp et al. 1998; Schuman, Herrick, and Janzen 2000).

The magnitude and direction of potential climate change effects on U.S. grasslands are subjects of ongoing research (e.g., Ojima et al. 1993a, b; Parton, Morgan, and Kelly 2000) (Figure 1.12). Simulation studies projecting the effects of climate change on U.S. grasslands have produced conflicting results. Grassland productivity often increases with the interactive effects of temperature, moisture, and CO_2 enrichment found in most climate change scenarios (Parton, Morgan, and Kelly 2000; Thornley and Cannell 1997). As with crop responses to climate change, however, results depend on the GCM scenarios used to drive the ecosystem models. The *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000) found that grazing productivity on western grasslands decreased by 10% under the CCC model, whereas it increased by 5 to 10% with the HC model.

Figure 1.12. A researcher assesses the condition of grassland from an image produced by a satellite sensor. Photo courtesy of P. Greb, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

Given the already great variability and heterogeneity of climate in U.S. grasslands, major changes in grassland productivity are not expected with minor changes in climate (Seastedt et al. 1998). Grazing management practices (e.g., stocking rates, burning) are likely to remain more important in controlling grassland productivity, at least in the near term (Rice and Owensby 2001; Seastedt et al. 1998). Persistent drought and N deposition, however, could change species composition, diversity, and productivity. On the other hand, increased atmospheric CO_2 may be able to override, up to certain thresholds, the negative effects of drought (Owensby et al. 1999; Seastedt et al. 1998).

Few field studies have addressed climate change effects on intact grasslands. Owensby and colleagues (1996, 1999) studied the effects of elevated CO_2 on native tallgrass prairie for 8 yr. Both above- and belowground production was stimulated under elevated CO_2 when moisture was limiting (Owensby et al.

1996). Root production increased by an annual average of 41% under elevated CO_2 (Owensby et al. 1999). A key finding was that the C_4-dominated prairie showed substantial increases in water-use efficiency, resulting in greater production under moderate water stress. In a 5-yr study in the Colorado shortgrass steppe, Morgan and colleagues (2004) found that doubling atmospheric CO_2 concentration enhanced biomass production through improved water-use efficiency. The production response was driven by a single C_3 species.

Thus, grassland productivity changes because of decreased precipitation or increased temperature could be moderated by the increased water-use efficiency associated with increased atmospheric CO_2. One reason for the resiliency of grasslands to climate variations is that a large percentage of total net primary productivity in grassland ecosystems occurs belowground (Elliott, Hunt, and Walter 1988; Rice et al. 1998). Thus, the response of rangelands to land management and environmental change potentially is significant to the global C budget. Various modeling studies (e.g., Cole et al. 1993; Hunt et al. 1991; Ojima et al. 1990) indicate that in the future, grasslands could, depending on land management regimes, function as either sinks or sources of C.

Agricultural Pests

Climate affects not only agricultural crops but also their associated pests. Spatial and temporal distribution and proliferation of insects, weeds, and pathogens are determined to a great extent by climate, because their growth and development are driven by temperature, light, and water. Climate also affects the pesticides used to control or to prevent pest outbreaks. For instance, intensity and timing of rainfall influence pesticide persistence and efficiency, and temperature and light affect pesticide persistence through chemical alteration. Most analysts concur that in a changing climate pests will become better able to expand their geographic ranges. An expansion of pest populations may necessitate increased use of agricultural chemicals, implying health, ecological, and economic costs (Rosenzweig et al. 2000).

Major pest outbreaks occur during both favorable and unfavorable weather conditions. A 100-yr record of locust behavior in Kansas (1854–1954) indicates that the most severe damage was caused during dry years (Smith 1954). Insect damage to soybean increased during the severe drought of 1988 in the U.S. Midwest, when an estimated 3.2 million hectares (ha)

were sprayed with insecticides to control spotted spider mites across the region. Losses to Ohio farmers were estimated at $15 to $20 million in the same year (Stinner et al. 1989). Climate conditions during El Niño and La Niña years also seem related to pest damage in certain regions; for example, epidemics of wheat stem rust diseases in the U.S. Great Plains from 1921 to 1962 are correlated with El Niño episodes (Yang and Scherm 1997).

Several detrimental pests in the United States and Canada have expanded their ranges since the early 1970s (Figure 1.13), possibly indicating a response to rising temperatures. The soybean cyst nematode (*Heterodera glycines*) is the cause of great economic loss to U.S. soybean producers. In Iowa alone, this pest caused an estimated yield loss of 200 million bushels (worth approximately $1.2 billion) during the 1998 growing season. The pest has been expanding since the 1950s, but the increase has been more dramatic since the early 1970s (Niblack 1999), during which time soybean sudden death syndrome, a soilborne fungal disease caused by *Fusarium solani* f. sp. *glycines*, has expanded northward from Arkansas to Wisconsin (Roy et al. 1997).
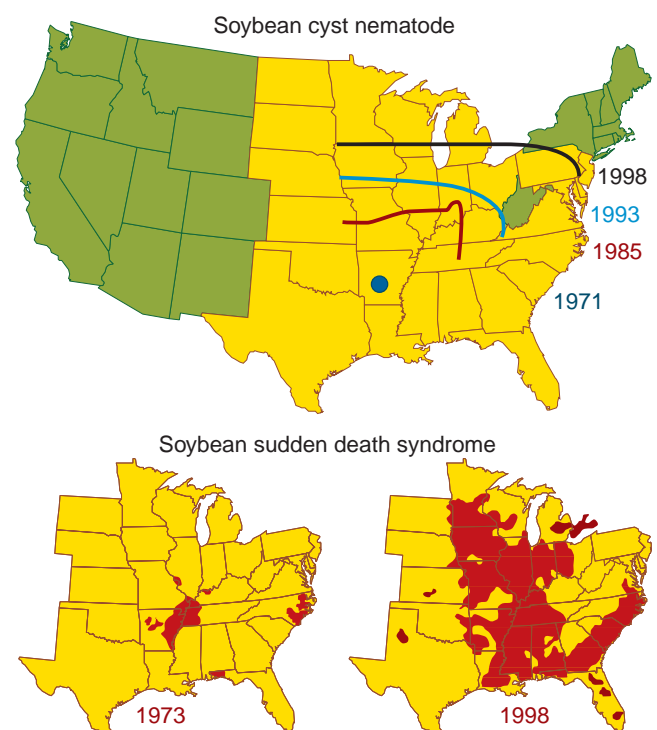
Figure 1.13. Range expansion of soybean cyst nematode (*Heterodera glycines*) from 1971 to 1998 and soybean sudden death syndrome (*Fusarium solani* f. sp. *glycines*) from 1973 to 1998. (Adapted from Niblack 1999; Rosenzweig et al. 2000.)

In a changing climate, pests may become even more active, thus posing the threat of increased economic losses to farmers (Coakley, Scherm, and Chakraborty 1999; IPCC 1996b). Whereas the majority of weeds are invasive species from temperate zones, certain weeds in temperate regions originated in tropical or subtropical regions, and in the current climate their distribution is limited by low temperatures. Such geographical constraints will be eased under warm conditions. Warmer temperature regimes have been shown to increase the maximum biomass of certain grass weed species significantly. Ranges of crop weeds, insects, and diseases are projected to expand to higher latitudes (Dahlsten and Garcia 1989; Sutherst 1990). (See Figure 1.14 for data regarding temperature effects on weeds.)
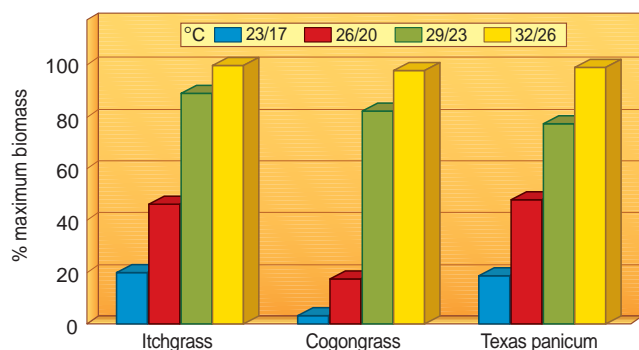
Figure 1.14. **Effects of varying day/night temperature regimes on biomass of three grassy weeds. (Adapted from Patterson 1993.)**

When temperatures are within their viable range, insects respond to higher temperature with increased rates of development and with decreased time between generations. (Very high temperatures decrease insect longevity.) Warmer winters will decrease winterkill, and consequently there may be increased insect populations in subsequent growing seasons. With warmer temperatures occurring earlier in the spring, insect populations may become established and thrive during earlier and more vulnerable crop growth stages. Additional insect generations and greater populations encouraged by higher temperatures and longer growing seasons likely will require enhanced pest management efforts.

The potato leafhopper (*Empoasca fabae*), a pest of soybean, alfalfa, and other crops, may expand its overwintering range—now limited to a narrow band along the Gulf of Mexico—and thus be positioned better to travel to the U.S. Midwest earlier and in greater numbers during the cropping season (Figure 1.15). Other insects predicted to become more damaging under warmer climate conditions include corn earworm (*Heliothis zea* [Hubner]); European corn borer (*Osrinia nubilalis*); and Mexican bean beetle (*Epilachna varivestis*) and bean leaf beetle (*Cerotoma trifurcata*), both of which are major pests of soybean (Stinner et al. 1989).

Increasing temperature, humidity, and precipitation favor the spread of plant diseases, because wet vegetation promotes germination of spores and proliferation of bacteria and fungi. Soil moisture also influences the life cycle of soil nematodes. In regions suffering from aridity, however, disease infestations may lessen, although certain diseases such as pow-
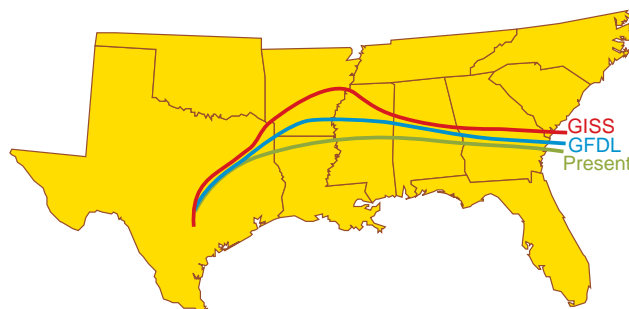
Figure 1.15. **Effects of climate change on overwintering range of potato leafhopper, based on climate change scenarios from two global climate models (GISS—Goddard Institute for Space Studies and GFDL—Global Fluid Dynamics Laboratory) under conditions of doubled CO_2 concentrations. (Adapted from Stinner et al. 1989.)**

dery mildews can thrive in hot, dry conditions as long as there is dew.

Interactions between crops and their associated pests in response to elevated levels of atmospheric CO_2 are not well understood and are difficult to predict. Because of the different growth responses of weeds and crops to elevated atmospheric CO_2, C_3 weeds are likely to become more aggressive (Patterson 1993). Carbon dioxide enrichment also may modify insect-crop relations. Changes in C- and N-partitioning in crops grown under elevated CO_2 conditions may affect nutritional quality and attractiveness of foliage to various insects. For example, experiments have shown that higher CO_2 tends to increase the C:N ratio in crop leaves, stimulating the feeding of and the damage caused by certain insects (Lincoln, Sionit, and Strain 1984; Salt, Brooks, and Whittaker 1995).

Chen and McCarl (2001) investigated how average

per-acre pesticide usage costs for corn, cotton, potatoes, soybean, and wheat were altered by changes in climate as observed across the United States in recent years. They found that climate had significant effects on pesticide use, cost, and economic returns. In the context of the 2090 scenarios used in the *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000), they projected that climate change would increase pesticide costs for corn, soybean, cotton, and potatoes and cause mixed results for wheat. In aggregate they computed that there would be an increase in net cost to society of approximately $200 million per yr.

Livestock

Livestock can be affected both directly and indirectly by climate change. Direct effects generally are associated with temperature effects on the animals' balance between heat dissipation and heat production. A change in this balance can alter animal mortality, appetite, feed conversion rate, weight gain, milk production volume, and conception rate (Adams, Hurd, and Reilly 1999; Hahn and Morgan 2000). These effects may be caused by long-term climate change or by short-term variation in extremes of day/night temperature patterns.

Indirect effects of climate change on livestock production arise through impacts on the productivity and quality of forage from grasslands, quality of other feedstuffs available (e.g., grain and hay nutrient content), incidences of disease and pests, availability of water in grazing areas, and market-influenced input and output prices.

The nature of individual direct and indirect effects depends on the climate in which livestock is located. Northern herds may benefit from warmer conditions, whereas southern herds may experience difficulties as a result of climate change.

Livestock producers can adapt to climate change in numerous ways. For example, the effects of increasing heat may be offset by provision of shading and sprinklers, improvement of airflow, lessening of crowding, and altering of diet, among other actions. Centers of livestock production also may be moved to more hospitable locations. Given these options, the IPCC's third assessment report (IPCC 2001a) concluded that intensively managed livestock systems are, especially because of their superior ability to adapt to extreme weather events, potentially more adaptable to climate change than are crop systems, although added costs are likely to be involved.

Summary of National and Global Assessments

Studies of the agricultural effects of climate change have been conducted for more than 15 yr. Several recent literature reviews and the newly completed *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* summarize the major findings (see Adams, Hurd, and Reilly 1999; Lewandrowski and Schimmelpfennig 1999; McCarl, Adams, and Hurd, in press; NAST 2000; Reilly et al. 2002, 2003; Rosenzweig and Hillel 1998). Among the major findings are the following:

1. During the next century, regional increases and decreases in crop and livestock production associated with climate change, as now foreseen, are not expected to result in large changes in global food production or any large global economic disaster in total food production. In part, this finding reflects the fact that the projected range of climatic alteration is less than the range of temperatures now experienced across productive areas of global agriculture (see Adams et al. 1998; Lewandrowski and Schimmelpfennig 1999; Reilly et al. 2002, 2003).

2. Impacts on regional and local food supplies in certain low-latitude (i.e., tropical) regions could cause large decreases in productive capacity and significant economic hardship (see Adams et al. 1998; Lewandrowski and Schimmelpfennig 1999; Rosenzweig and Parry 1994).

3. Climate-induced productivity changes that are harmful for consumers typically are beneficial to producers. In several studies of U.S. agriculture that include price effects, decreases in crop yields indicate that consumers would pay higher prices and receive smaller quantities of agricultural goods, thereby suffering economic losses (see Adams, Hurd, and Reilly 1999; Adams et al. 1990). On average, however, producers are projected to gain from revenue increases.

4. Climate change is likely to shift the comparative advantage of agricultural production regions in the United States and elsewhere. Such shifts are likely to alter the places in which specific crops are grown, both within countries and internationally, altering patterns of trade in agricultural commodities among regions and countries (see Darwin et al. 1995; Reilly, Hohmann, and Kane 1994; Rosenzweig and Parry 1994).

5. The economic consequences of yield changes will be influenced by adaptations made by farmers, consumers, government agencies, and other institutions. Farmers may adapt by changing planting dates, substituting cultivars or crops, changing irrigation practices, and changing land allocations among crop production, pasture, and other uses. Certain of these adaptations may be costly. Consumers may adapt by substituting relatively low-priced products for those that become relatively high priced as a result of climate change effects (see Adams et al. 1998 or Kaiser et al. 1993).

6. Pests currently are a major problem in U.S. agriculture, and climate change is likely to exacerbate that problem (Rosenzweig et al. 2000, 2002). A study by Chen and McCarl (2001) shows pesticide use would be expected to increase for most crops in most states under the climate scenarios used in the recent *U.S. National Assessment: The Potential Consequences of Climate Variability and Change* (NAST 2000). This assessment approach did not consider increased crop losses because of pests, implicitly assuming that all additional losses were eliminated through increased pest control measures. Further, in Chen and McCarl (2001) it was shown that this increase in pesticide use results in decreased overall national economic welfare. In addition, there could be substantial environmental consequences of increased pesticide use.

7. Changes in climate are expected to affect the productivity and aggregate demand for factors of production such as water, labor, energy, and equipment. Climate change is analogous to technological change in agriculture, which can increase or decrease total factor productivity and can increase or decrease the productivity of one factor relative to another. Most studies for the United States indicate that productivity changes triggered by climate change would generate changes in cultivated acreage by crop, total cultivated acreage, irrigation water consumption, farm employment, and other changes in factor demands. The consequences of changes in factor demands on regional or local economies are largely unexplored but potentially important.

2 Emissions and Mitigation of Agricultural Greenhouse Gases

Concern about the issues surrounding climate change, and the potential consequences for agriculture, has brought with it awareness that the rising concentrations of greenhouse gases (GHGs) in the atmosphere can have profound effects on the global environment and therefore on agriculture. As a sector of the economy that is itself responsible for significant GHG emissions, agriculture has a clear self-interest in pursuing opportunities for decreasing GHG fluxes. Because all major GHGs have significant sources and sinks associated with land use and management, agriculture can play a significant role in society's efforts to mitigate GHG increases and potential climate change.

Element Cycles

All three of the important GHGs—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4)—are produced through plant-soil-animal (i.e., ecosystem) processes that occur within both agricultural and nonagricultural ecosystems. The biogeochemical pathways involving these gases are part of the basic cycling of carbon (C) and nitrogen (N) that is fundamental to ecosystem function. In addition, C and N are the main elements making up soil organic matter (SOM), which plays a pivotal role in the fertility and sustainability of agricultural systems. Because the mitigation of atmospheric CO_2 through soil C sequestration is manifested by the buildup of SOM, special mention of the role of SOM in ecosystem function is made here.

Role of Soil Organic Matter in Ecosystem Function

Soil organic matter is a mixture of several components: plant and animal residues at various stages of decomposition, substances synthesized microbiologically and/or chemically from the breakdown products, and bodies of live microorganisms and small animals and their decomposition products (Schnitzer 1991). Soil organic matter is a minor component (often 1 to 5% by mass) of soil solids but greatly influences the soil's physical, chemical, and biological properties that are central to ecosystem function.

Soil organic matter functions as a main source of plant nutrient supply, particularly of N, phosphorus (P), and sulfur (S), through the mineralization of organic matter and the release of these elements by the soil biota. Soil organic matter and accompanying soil organisms also take up mineral nutrient elements. Thus, SOM functions as both a sink and a source of nutrients, depending on the nutrient concentration in the soil and the interactions of nutrient demand by plants and microorganisms (Paul and Clark 1989; Schlesinger 1997). The more humified components of SOM also can make up a significant proportion of soil ion-exchange capacity, and thus SOM influences soil pH and buffering capacity as well as the retention of base cations such as calcium (Ca) and magnesium (Mg).

Soil organic matter is a key attribute influencing soil structure and aggregation (Beare et al. 1994; Tisdall and Oades 1982). Soil organic matter accentuates the formation and stability of soil aggregates, increases total and macroporosity of the soil, and improves soil tilth (Karlen et al. 1997). Well-aggregated soils have good physical structure, which improves water infiltration and water storage, decreasing runoff and soil erosion.

In addition to playing a central role in plant production, SOM contributes a variety of other ecosystem services, including purification of water by filtering and immobilizing pollutants. Decline in SOM quality and quantity can set in motion degradative processes (e.g., soil crusting, compaction, diminished soil structure, or accelerated erosion) with adverse effects on plant growth and on water and nutrient cycling. These effects in turn contribute to the contamination and eutrophication of surface water, pollution of groundwater, and possible breakdown of ecosystem functions.

Table 2.1. Global C stocks in soil and vegetation, by major biome type (adapted from IPCC 2000)

Biome	Area (10^6 km^2)	Average soil C density (Mg ha^{-1})	Soil C stock (Pg)	Vegetation C stock (Pg)
Tropical forests	17.6	123	216	212
Temperate forests	10.4	96	100	59
Boreal forests	13.7	344	471	88
Tropical savannas	22.5	117	264	66
Temperate grasslands	12.5	236	295	9
Deserts/semideserts	45.5	42	191	8
Tundra	9.5	127	121	6
Wetlands	3.5	642	225	15
Croplands	16.0	80	128	3
Total	151.2	—	2,011	466

Carbon and Nitrogen Cycles of Agricultural Systems

The C cycle of agricultural systems describes the flows of C between the atmosphere and the plant-soil system (see Figure 2.1). Carbon, as CO_2, enters the system through the process of plant photosynthesis, creating biomass. In agricultural systems, a large portion of the biomass is exported as a harvested product; however, substantial biomass remains in the field as crop residues (including roots). The crop residues in or on the soil then are subject to decomposition, the process by which soil organisms, particularly bacteria and fungi, break down and use organic compounds for their growth and energy requirements. Thus, C is returned to the atmosphere, primarily as CO_2,

Figure 2.1. Overview of the C cycle in cropland ecosystems, showing the major fluxes of organic C. Drawing courtesy of A. Swan, Colorado State University.

through the respiration of the soil biota.[2] Most of the organic material making up crop residues is decomposed fairly quickly and is respired as CO_2, but certain portions are transformed further into more recalcitrant compounds that can become stabilized and remain in soils for many decades to centuries. Throughout time, soils accumulate C stocks that equal or exceed the amount of C stored in vegetation in most terrestrial ecosystems (Table 2.1).

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 typically are on the order of 0.5 to 3% in the top 20 centimeters (cm), amounting to 20 to 100 tonnes C /hectare (ha). Organic C content tends to decline with soil depth, and for most soils, 30 to 50% of the organic C to 1-m depth is contained in the top 20 cm. Soils with much higher C content, 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 (USDA–NRCS 1997).

Soils also contain C in inorganic forms that cycle between the atmosphere, soil, and hydrosphere. Carbon dioxide in the soil atmosphere equilibrates with dissolved CO_2, combining with water to form carbonic acid (H_2CO_3), bicarbonate (HCO_3^-), and carbonate ($CO_3^=$) ions in the soil solution. Carbonate/bicarbonate ions can be leached from soils and enter surface

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[2] Under conditions of low oxygen concentrations, such as in flooded rice fields, CH_4 can be formed through the anaerobic decomposition of crop residues and emitted to the atmosphere. From the standpoint of C cycling, per se, CH_4 fluxes are of minor importance but are significant as a GHG emission.

and groundwaters, and eventually the ocean. Inorganic C in the form of primary minerals (e.g., calcite [$CaCO_3$]) 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 Ca and Mg with $CO_3^=$ ions. Carbonate minerals also are added to soils through certain types of agriculture liming. Depending on the balance between the dissolution and formation of these minerals, $CO_3^=$ (and hence CO_2) is either consumed or produced. In most soils, however, the net fluxes of inorganic C are small relative to the organic C cycle.

Nitrogen also cycles through agricultural ecosystems where it is a critical element for plant growth (Figure 2.2). The largest amount of N in the ecosystem consists of the organically bound N in SOM—agricultural soils typically have a ratio of C to N of approximately 10 to 1. In most agricultural systems, the bulk of N inputs are through applications of fertilizer or manure. Some N is deposited from the atmosphere in rainfall and some also enters the soil through the activities of various N-fixing bacteria, which occur as free-living soil bacteria and as symbiotic bacteria associated with crop species such as soybean and other legumes. The bulk of the biologically fixed N in agricultural systems is through symbiotic (i.e., *Rhizobium*-legume) N fixation. Nitrogen cycling within the soil involves numerous other processes, including mineralization, nitrification, denitrification, and ammonia volatilization (see Figure 2.2). Organic N compounds are mineralized (i.e., released as

ammonium [NH_4^+]) by soil organisms and, under aerobic conditions, are converted further through nitrification to nitrate (NO_3^-), providing the sources for plant uptake of N. Nitrification, denitrification, and ammonia volatilization all are processes that lead to losses of N compounds to the atmosphere. Nitrification and denitrification, largely carried out by soil bacteria, are the primary sources of N_2O emissions. Ammonia is released to the atmosphere from fertilizer and manure and from senescing plant tissues.

Nitrogen and C cycling are linked within agricultural systems and at the global level. Carbon provides an energy source for the microbial decomposition of residues and SOM, which also drives most of the microbial transformations of N. Nitrogen in turn is a primary determinant of plant growth and hence C supply. Nitrogen also is a critical element for soil organisms; hence, N availability within the soil can influence rates of residue decomposition and microbial transformations involving inorganic N forms, such as nitrification and denitrification. Nitrogen fertilizer can stimulate plant growth rate and SOM accumulation in agricultural soils, but the industrial production of N fertilizer also releases CO_2, which contributes to CO_2 emissions from the agricultural sector (Schlesinger 2000).

Agricultural systems depend on the cycling of C and N through soil. Human agronomic activities attempt to maximize the capture of both C and N in plant material and to use these components as energy and nutritional sources in food and feed. Thus, management of cropping systems can alter the storage and release of C and N, with significant impacts on global C and N cycles and emissions and sinks of the major GHGs.

Processes Controlling the Emission of Greenhouse Gases from Soils

Carbon Dioxide Emissions

Carbon dioxide is emitted from soils in a process known as soil respiration. This CO_2 derives from the metabolic activity of plant roots and from soil microbes decomposing plant litter and SOM. Most of the plant debris entering the soil is respired relatively quickly as CO_2. Only a small fraction becomes humus, which accumulates in soils at slow rates during long periods (Schlesinger 1990). Soil respiration thus
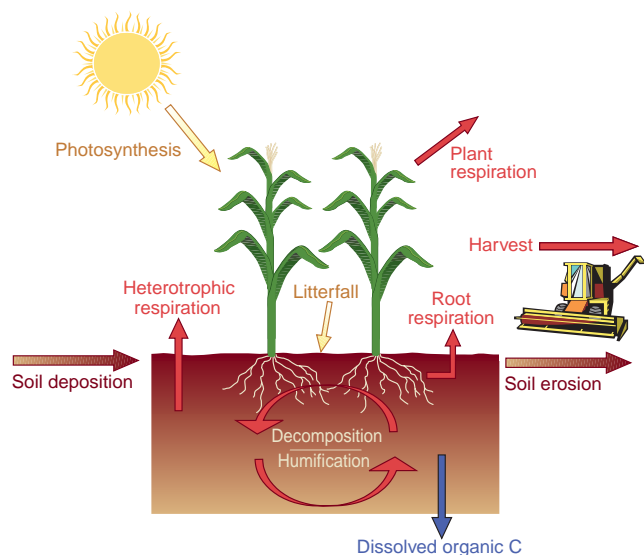

Figure 2.2. Overview of the N cycle in cropland ecosystems, showing the major fluxes of N. Drawing courtesy of A. Swan, Colorado State University.

is a major flux of the global C cycle. Each year, soil respiration returns nearly 10 times as much CO_2 to the atmosphere as emissions from fossil fuel combustion do. Soil respiration is roughly balanced by the net uptake of CO_2 through plant photosynthesis, however, and both net photosynthesis and respiration amount to approximately 60 petagrams (Pg) C per year (yr). Thus, although gross terrestrial C fluxes are very large, the net change in ecosystem C stocks, on an annual basis, is small in comparison (IPCC 2000).

Soil respiration increases immediately when native land is converted to agriculture (Schlesinger 1986). The increase in respiration stems from increased microbial activity in these soils, which often are warmer, wetter, and better aerated than native soils. But when the labile components of SOM are exhausted, soil respiration rate declines, eventually becoming lower in cultivated soils than in adjacent undisturbed soils. Indeed, Raich and Potter (1995) suggest that the global soil-respiration rate is currently 0.2 to 2.0 Pg C/yr lower than it would be if all the world's lands had remained in natural vegetation.

Net emissions of CO_2 from soils are dependent on the balance between C entering the soil and C losses through soil respiration. Factors that increase C inputs, and/or decrease respiration, will favor the accumulation of C in soils, creating a sink for atmospheric CO_2. Conversely, decreasing C inputs relative to respiration will lead to a net efflux of CO_2 (a source) from soils. Both sets of processes (i.e., C inputs and soil respiration) are under some degree of control through land-use practices, within limits imposed by climate and soil conditions.

Carbon inputs to soils are determined by the amount and distribution of primary productivity, the life cycle of the vegetation, and exogenous organic matter additions (e.g., composts or manures). In long-term experiments with known rates of organic matter additions (mainly done for agricultural systems), soils with low to intermediate organic matter levels often exhibit a linear relationship between C inputs and soil C levels (e.g., Paustian, Collins, and Paul 1997; Rasmussen et al. 1980), such that soil C levels increase in direct proportion to increased C inputs. Thus, land-use practices that increase net primary production (NPP) and/or return a greater portion of plant materials to the soil have the potential to increase soil C stocks. Soils with high initial C levels, however, often are less responsive to increased C inputs (e.g., Campbell et al. 1997), suggesting the possibility of a "saturation level" for C in mineral soils (Hassink and Whitmore 1997).

Organic matter decomposition and microbial respiration are influenced by numerous physical, chemical, and biological factors controlling the activity of microorganisms and soil fauna (Swift, Heal, and Anderson 1979). These factors include the abiotic environment (temperature, water, aeration, pH, mineral nutrients), plant residue quality, soil texture and mineralogy, and soil disturbance. Soil temperature and moisture are the most important environmental controls on decomposition, and their influence is fairly well understood. Generally, decomposition and soil respiration rates increase with increasing temperature up to an optimum range of 30 to 40° Celsius (C). Optimal moisture conditions for decomposition in most soils are around 55 to 60% water-filled pore space (Doran, Mielke, and Stamatiadis 1988), with decomposition decreasing as the soil dries. High water contents, near or at saturation, indirectly inhibit decomposition because of the decreased availability of oxygen (O_2). Optimal pH levels for decomposition generally are near neutral conditions (pH 6 to 8), but decomposition usually is not significantly repressed except in quite acid (pH < 5) conditions (Dyal, Smith, and Allison 1939).

Plant residues vary in their inherent decomposability because of differences in their physiochemical characteristics (Swift, Heal, and Anderson 1979). Lignin and certain polyphenolic substances tend to retard decomposition rates (Melillo, Aber, and Muratore 1982; Tian, Kang, and Brussard 1992). Long-term experiments with fixed amounts of organic additions of differing quality have shown higher soil C levels resulting from low-quality (e.g., high lignin) litter inputs (Paustian, Parton, and Persson 1992; Sowden and Atkinson 1968).

Soil texture and mineralogy are well recognized as influencing SOM levels. Other factors being equal, soil C levels tend to increase with higher clay content (Allison 1973; Burke et al. 1989). Humic material can bind to clay mineral surfaces to form highly stable organomineral complexes; 2:1 lattice (smectitic) clays, characteristic of lightly weathered soils, have a higher binding capacity than soils with 1:1 (kaolinitic) clay minerals, found in more highly weathered soils. The amount and type of clay minerals also influence the capacity of soils to form stable soil aggregates, which can partly protect organic matter from microbial attack (Adu and Oades 1978; Beare et al. 1994).

Changes in the global climate and in the concentration of atmospheric CO_2 are likely to change the global rates of decomposition and soil respiration in both natural and agricultural ecosystems. Higher temperatures are likely to increase the rate of soil res-

piration in all soils of the world, especially in cold regions (Kirschbaum 1995). On the other hand, higher concentrations of atmospheric CO_2 also will stimulate the growth of most plants, especially agricultural crops. Increased productivity can supply more plant residues to the soil, possibly increasing storage of SOM. But higher soil respiration from crops grown at high CO_2 levels, stemming from a greater release of CO_2 from soil microbes and increased root metabolism, often is seen when plants are grown under such conditions (Andrews and Schlesinger 2001). Thus the net effect of increasing both temperature and CO_2 may be positive or negative for agricultural soils, depending on how these two factors interact. The effects are likely to vary both geographically and throughout time and will interact strongly with management (Paustian et al. 1996).

Nitrous Oxide Emissions

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 NO_3^- to N_2O and then to N_2 by bacteria (Robertson 1999). Denitrification is an anaerobic process; thus, only if O_2 is unavailable will NO_3^- be denitrified to a nitrogenous gas. Because of this dependence on anaerobic conditions, at one time it was 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; 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 certain agricultural soils are low—for example, in flooded rice soils—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 N_2O and N_2. As discussed earlier, N_2O is a powerful GHG found in 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 *nitrous oxide 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 NH_4^+ to nitrite (NO_2^-) and then to NO_3^-. Intermediary compounds formed during nitrification 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 gas fluxes are extremely heterogeneous on both field scales (e.g., Folorunso and Rolston 1984) and landscape scales (e.g., Groffman
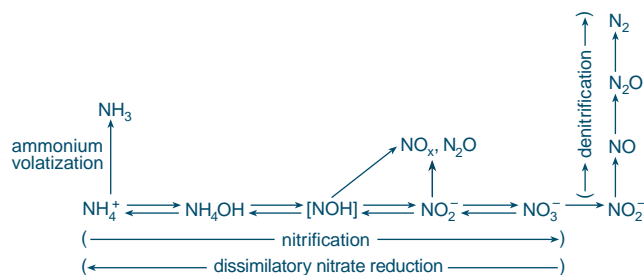
Figure 2.3. Diagram of the major transformations of inorganic N that can occur in soils, focusing on the major pathways for gaseous N losses, including N_2O. (Source: A. R. Mosier, Agricultural Research Service, U.S. Department of Agriculture, Fort Collins, Colorado.)

and Tiedje 1989). 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. There is, regrettably, no easy way to evaluate this heterogeneity; fluxes of N_2O from soils generally are small relative to the 319 parts per billion volume (ppb_v) concentration of today's atmosphere, a fact that limits sampling in most ecosystems to chamber-based approaches in which N_2O accumulation is measured in small chambers placed on the soil surface.

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 the few instances in which automated chambers have provided continuous 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 (see section on "Mitigation of Nitrous Oxide" in this chapter).

Methane Emissions

Methane is produced mainly in soils saturated with water for significant periods. In agriculture, this degree of saturation occurs mainly in flooded rice soils and during animal waste storage (see subsection on "Current Emissions and Sinks of Methane" in this chapter). 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 forestomach, or *rumen*, in which plant materials are broken down through fermentation. Fermentation also generates CO_2 and hydrogen gas (H_2), the latter of which is used as an energy source by *methanogenic* bacteria. Methanogenic bacteria 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 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 grazing lands, it tends to decompose aerobically and produces little CH_4 (Safley et al. 1992; USEPA 1993a, 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 are associated with the most intensively managed animals.

In rice soils and in wastewater lagoons, methanogenesis 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 2.4). Once formed, CH_4 can diffuse to the surface, rise to the surface entrained in bub-
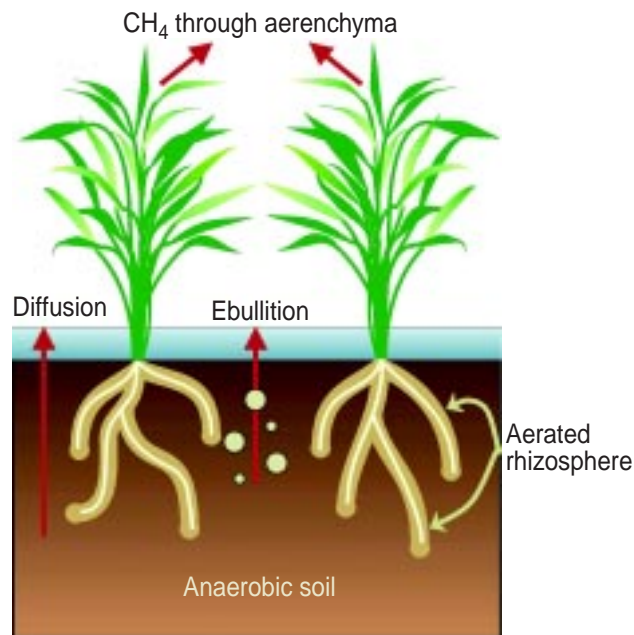
Figure 2.4. Pathways of CH_4 emissions from flooded soils, such as under rice cultivation. Drawing courtesy of A. Swan, Colorado State University.

bles, or, more important, when rice is present, be transported to the atmosphere through the rice plant itself, through air channels (called aerenchyma) within the plant that supply O_2 to the roots. This latter process is generally the most important emission mechanism and accounts for more 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. It is now known 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 by physical and chemical environments. Either biotic or 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.

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 up-

land 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 several-fold 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—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.

Enhanced understanding of the microbiological basis for CH_4 consumption in aerobic soils awaits further research, as does enhanced understanding of the spatial variability underlying both CH_4 and N_2O fluxes. In both instances, understanding should improve scientists' ability to model fluxes quantitatively. Existing field-scale models (see Frolking et al. 1998) are ineffective estimators for any but a few ecosystems, yet ultimately scientists will need to depend on modeling, coupled with field verification, to provide flux estimates for the myriad combinations of management practices potentially affecting fluxes.

Basic Principles of Mitigation

Contributions of soil and terrestrial ecosystems to atmospheric concentrations of GHGs can be decreased by (1) decreasing emissions of GHGs and (2) sequestering C, derived from atmospheric CO_2, within the ecosystem. Thus, an effective mitigation strategy involves adopting appropriate measures that decrease emissions while enhancing C storage in biomass and in soil (Table 2.2).

Decreasing Emissions

Anthropogenic activities leading to GHG emissions from agricultural ecosystems include the conversion

Table 2.2. Strategies for reducing emissions of GHGs and sequestering C and N

Reducing emissions of GHGs	Sequestering C and N
1. Decreasing deforestation	1. Increasing afforestation and reforestation
2. Decreasing conversion of natural to managed ecosystems	2. Restoring degraded soils and ecosystems
3. Reducing biomass burning	3. Adopting conservation tillage and growing of cover crops
4. Decreasing intensity and frequency of plowing, and eliminating bare fallowing	4. Soil application of biosolids (e.g., manure, compost, residue mulch)
5. Decreasing soil erosion rate	5. Conserving soil and water
6. Minimizing subsistence farming	6. Using integrated nutrient management strategies and agricultural intensification
7. Retiring marginal lands and reducing cultivation of organic soils	7. Adopting conservation reserve programs
8. Minimizing drainage of wetlands	8. Restoring wetlands and managing water tables in agricultural soils
9. Improving N-use efficiency	9. Increasing biological N fixation and strengthening nutrient cycling
10. Reducing use of agricultural chemicals	10. Adopting techniques of integrated pest management
11. Reducing use of monocultures	11. Growing deep-rooted plants and using agroforestry
12. Reducing flood irrigation	12. Using drip, furrow, or subirrigation
13. Enhancing energy efficiency and use of biofuel	

of native ecosystems to agricultural uses as well as many common management practices used on long-established agricultural lands. Land-use conversions can involve deforestation and biomass burning, wetland drainage, plowing, and accelerated soil erosion, resulting in large losses of biomass and soil C stocks and substantial GHG emissions (Figure 2.5a). On lands maintained for long-term agricultural uses there are continuing sources of GHG emissions. Among these are CO_2 emissions associated with energy used for the production and application of agricultural inputs such as fuel, fertilizers, lime, and pesticides. Intensively tilled soils, as well as drained organic soils, can sustain continuing oxidation of organic matter and decreases in soil C stocks throughout many decades. Nitrogen additions—a ubiquitous feature of all agricultural systems—are subject to N_2O losses, whether through fertilizers, N-fixing crops, or manure. Livestock production, manure management, and rice cultivation are the predominant agricultural sources of CH_4.

Hence, some level of GHG emissions from agricultural management is an inevitable consequence of agricultural land use. But many opportunities exist for decreasing emissions. Decreasing the extent of, and the disturbance intensity of, land-use conversions would decrease emissions dramatically, especially in the tropics where most deforestation is occurring. On established agricultural lands, an important strategy

is to improve the efficiency of production inputs (i.e., fuel, fertilizers, and pesticides), 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 to capture them for use as an energy source. Finally,
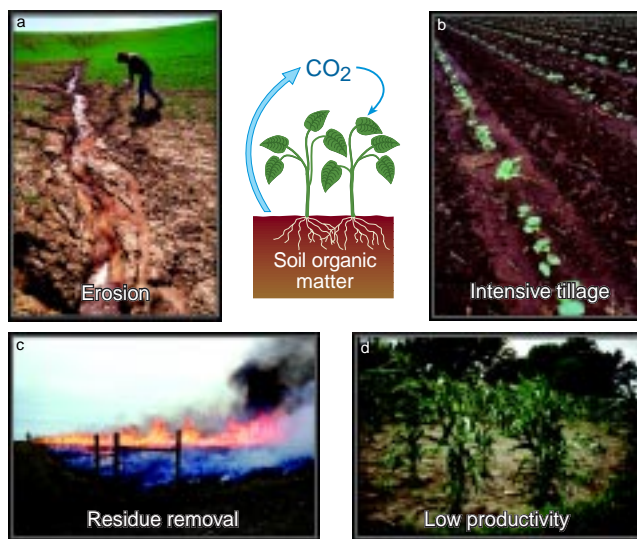
Figure 2.5a. Management practices and their effects that lead to degradation of soil organic matter stocks. Photos courtesy of (a) J. Dykinga, (b) B. Nichols, (c) R. Masters, USDA–ARS, and (d) K. Paustian, Colorado State University.

production of agricultural biofuels provides opportunities for offsetting fossil energy CO_2 emissions from agriculture and other sectors of the economy. These mitigation options are discussed in detail in the following subsections.

Carbon Sequestration

Soil C sequestration implies capture of atmospheric C and either its storage in biomass and soil within the terrestrial ecosystem or its leaching into deep groundwater as inorganic carbonates. There are two principal strategies for C sequestration in agricultural ecosystems: (1) improved management of permanent agricultural land and (2) conversion and/or restoration of marginal and degraded agricultural lands to alternative uses. The goals of a C sequestration strategy are to enhance net primary productivity and to increase organic matter return to soil while decreasing the rate of CO_2 loss resulting from decomposition. As will be described, a wide variety of technological options exists for enhancing C sequestration in agricultural soils (Figure 2.5b) including decreased tillage intensity, alternative crop rotations, and fertility and water management (Lal et al. 1998; Paustian et al. 1998). Restoration or conversion of degraded lands includes reforestation and afforestation (Brown and Lugo 1990; Gupta and Rao 1994; Li and Zhao 1998), conversion of cropland to pastures or to grassland set-asides (e.g., the U.S. Conservation Reserve Program [CRP]), restoration of salt-affected or chemically impaired soils and desertification control (Lal
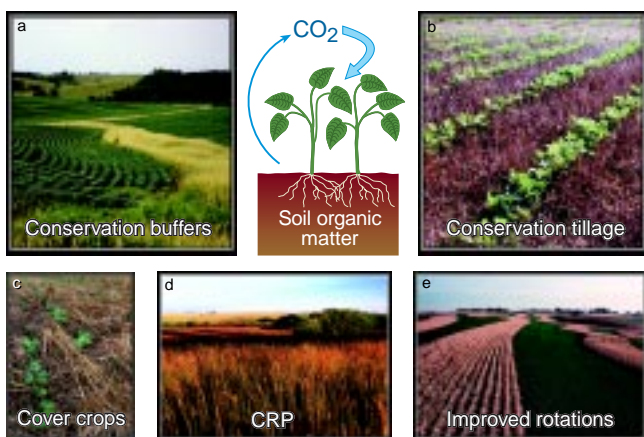
Figure 2.5b. Improved agricultural management practices can increase soil organic matter stocks. Photos courtesy of (a) K. Paustian, Colorado State University, (b) G. Alexander, (d) L. Betts, (e) T. McCabe, USDA–NRS, and (c) B. Nichols, USDA–ARS.

et al. 1999; Squires, Glenn, and Ayoub 1995), and wetland restoration.

Historic Carbon Losses Because of Land Use

The conversion of native ecosystems, such as forests, prairies, and wetlands, leads almost invariably to losses in vegetation and soil C stocks (Figure 2.6). For forests converted to agricultural use, the losses of C contained in the vegetation through wood harvest and burning are obvious. For conversion of nonforested ecosystems, biomass losses are of less importance, but soil C losses can be substantial in all types of ecosystems. Various factors are responsible for declines in SOM after initiation of agricultural land use, including decreased inputs of plant residue to soils, soil disturbance by tillage, more favorable environmental conditions for decomposition, and increased soil erosion.

The magnitude of losses after cultivation can vary substantially by location and soil type. For example, across 11 sites in the Great Plains, Haas, Evans, and Miles (1957) estimated decreases in soil C concentra-
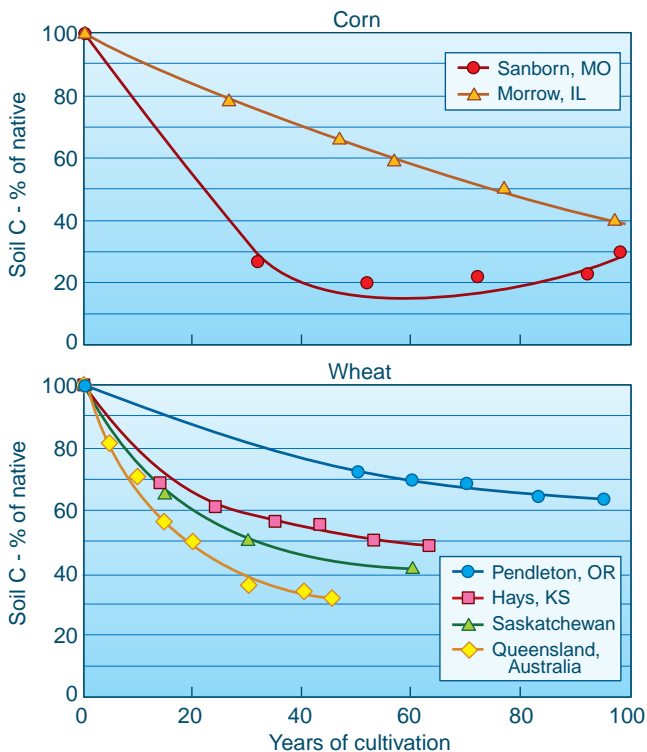
Figure 2.6. Examples of measured decreases in soil organic matter following conversion of native prairie soils to agriculture for sites in the Corn Belt (top panel) and in wheat-dominated croplands (bottom panel). (Source: Paustian et al. 1997a.)

tions of approximately 30 to 60%, after 30 to 40 yr of cultivation following plowing of the native prairie. In paired comparisons of native and cultivated soils across the United States, Mann (1986) calculated an average loss due to cultivation of 23% (in the top 30 cm) for all soils, but with a wide variation between soils. Losses on the order of 20 to 40% were estimated from a detailed analysis of soils from around the world (Davidson and Ackerman 1993). Losses typically are rapid the first few years and then decline over time.

A number of estimates of the historic C losses due to deforestation and conversion of forests, savannas, and prairies to agricultural ecosystems have been made. Current estimates, focusing primarily on changes in biomass stocks, are on the order of 125 to 150 Pg C during the past 150 yr (Bouwman and Sombroek 1990; Houghton 1999; IPCC 2000). The current C emission rate because of land-use change in the 1990s has been estimated at 0.6 to 2.6 Pg C/yr (Lashof and Hare 1999).

Estimates of the soil contribution to historic C losses because of agriculture are more uncertain, but the most recent values are on the order of 50 to 100 Pg C (Table 2.3), although certain authors have estimated much higher values. Lal (1999) considered historic losses of soil C stocks because of several processes including (1) mineralization losses of soil organic C (SOC) from increased soil disturbance and decomposition and lower amounts of biomass returned to the soil; (2) preferential removal of SOC with eroded sediments; (3) leaching of SOC as dissolved organic C (DOC); and (4) dissolution of soil inorganic C (SIC) by acidification. First-order estimates of these losses were 66 to 90 Pg C because of mineralization/oxidation and DOC leaching, 19 to 32 Pg C because of soil

Table 2.3. Estimates of the historic loss of soil organic C globally (adapted from Lal 1999)

Magnitude of the historic loss of SOC pool (Pg C)	Reference
30–60	Scharpenseel and Becker-Heidmann (1992)
40	Houghton (1995)
55	IPCC (1996b); Paustian et al. 1998
66–90	Lal (1999)
150	Bohn (1978)
230	Rozanov, Targulian, and Orlov (1990)
500	Wallace (1994)
537	Buringh (1984)

erosion, and 10 to 20 Pg C because of dissolution of SIC, yielding a total of 95 to 142 Pg C.

These estimates of historic C loss, tentative as they are, provide a reference point for soil C sequestration potential. Assuming that 60 to 80% of lost SOC could be recovered during a period of approximately 50 yr (IPCC 1996b), the global potential for SOC sequestration would be on the order of 60 to 110 Pg C at a rate of approximately 1.2 to 2.2 Pg C/yr. Similar estimates of global C sequestration potential for agriculture (including cropland, grazing land, and agroforestry) were made by the Intergovernmental Panel on Climate Change (IPCC) (2000) using a "bottom-up" approach. By considering C sequestration rates for improved management practices and sustainable land-use change as well as available area in developed and developing countries, they estimated potentials of 1.5 Pg C/yr by 2040.

Equilibrium, Capacity, and Residence Time of Carbon in Soil

Soil organic C is distributed across a multitude of organic compounds in soils, ranging from fresh plant residues to highly recalcitrant, amorphous humic substances, complexed with soil minerals. Typically, however, the distribution of organic C in soils is conceptualized in terms of broad classes or "pools" of organic matter that differ in their composition, formation, and rates of decomposition—most mathematical models of SOC dynamics consider three to five such organic matter pools (Paustian 1994). For purposes of illustrating the general behavior of organic C in soils, it is sufficient to further simplify this to a single pool, although the same principles can be derived from a more complex, multipool representation of SOC (Paustian, Collins, and Paul 1997).

The rate of change in soil C throughout time can be expressed as the difference between C additions and C losses, primarily through decomposition, i.e.,

$$dC/dt = A - kC \qquad (2.1)$$

where C is the SOC pool, k is the specific rate of decomposition, A is an accretion constant, and t is time. The accretion constant represents the amount of C added to soil through roots, aboveground crop residues, and biosolids applied as off-site input. Although these quantities vary from year to year, A can be thought of as the long-term average C input at a particular site. This model assumes that C losses because of decomposition are directly proportional to the size of the soil C pool. The value of k will vary from loca-

tion to location, depending on conditions that influence decomposition rates such as climate, soil physical properties, and soil management practices. Data from a number of long-term experiments suggest that for a given set of environmental conditions, k is constant and independent of C, over a fairly wide range of SOC contents (Paustian, Collins, and Paul 1997).

As shown in Eq. 2.1, SOC increases if the net value of the right-hand side of the equation is positive, and decreases if it is negative. For the soil to become a net sink, the rate of C input (A) must exceed decomposition losses (kC). When input balances losses, the rate of change of SOC (dC/dt) is zero and SOC remains constant (referred to as the *equilibrium state*). Where A and k are constants, then throughout time SOC will tend toward a unique equilibrium state, regardless of the initial amount of SOC. Under these conditions, the size of the equilibrium SOC pool ($^*C$) can be calculated as

$$^*C = A/k. \quad (2.2)$$

Under native conditions, most soils tend toward an approximate equilibrium state with respect to soil C. Conversion from natural to agricultural ecosystems, however, alters this dynamic equilibrium. In most instances, the accretion term, A, is considerably smaller than the depletion term, kC, and thus the SOC pool declines until a new equilibrium is reached (Figure 2.7).

Many agroecosystems, therefore, have the potential to sequester C through adoption of management practices that increase the accretion rate (A) and/or decrease the decomposition coefficient (k). After changes in agricultural land use/land management that impact the SOC balance, a fairly long period (e.g., 50 yr or more) is required before the system approximates a new equilibrium soil C content (Figure 2.7). Once soils reach a new equilibrium point, there is no net loss or gain of carbon. Thus, for a particular land use and management situation there is a finite limit to how much carbon can be sequestered. After a management change (such as increasing crop residue inputs) carbon sequestration rates will decline throughout time and eventually cease. At this point sequestering additional carbon would require further changes in management, which in turn would lead eventually to a stabilization at a new, higher equilibrium level. There is evidence that at very high carbon contents, mineral soils may approach a saturation limit (Hassink and Whitmore 1997); in essence the value of k in Eq. 2.1 increases as C increases—where soil C stocks no longer respond to further management changes to increase C.[3] This saturation phenomenon is still poorly understood, however, and most agricultural soils seem to be well below any inherent C saturation level.

If soils are to function for the long-term removal of C from the atmosphere, with storage as SOC, then the longevity or "residence" time of C in the soil is important. As discussed previously, soil organic C is maintained in a dynamic equilibrium between C entering the soil through plant residues and C leaving the soil
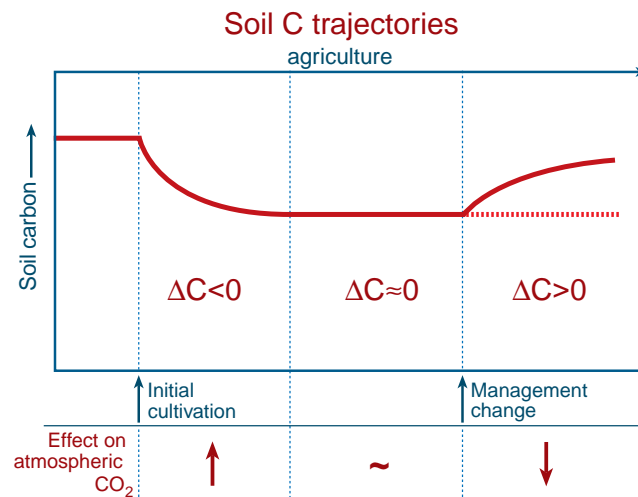
Figure 2.7. Conceptual diagram showing the initial decline in soil C that typically occurs when native ecosystems are converted to annual crop production, during which time there is a net loss of CO_2 to the atmosphere. Eventually, most cropland soils reach a new (lower) equilibrium content, and are neither a source nor a sink for CO_2. By instituting management changes that improve soil C status, C levels increase, acting as a sink for atmospheric CO_2. (Adapted from work by H. H. Janzen, Agriculture and Agrifood Canada, Lethbridge, Alberta.)

as respired CO_2, or as leached inorganic or organic C. Because SOC exists as a complex mixture of compounds, the residence time of these different components varies from a few weeks to several hundreds or thousands of years. The average residence time of C in soil, however, can be compared with that of, for example, C stored in vegetation where mean residence time is defined as:

Mean residence time (yr) = Total C stock/annual C flux.

Strictly speaking, mean residence time is defined only for C stocks at equilibrium (i.e., no net change

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[3] True saturation implies that the value of k in Eq 2.1 increases as C increases, instead of behaving as constant as is assumed normally.

throughout time), but for global comparisons, small departures from equilibrium can be ignored. Considering global organic C stocks of approximately 1,500 Pg for soils and 560 Pg for vegetation, and photosynthetic uptake of 120 Pg/yr by vegetation and soil respiration of 60 Pg/yr as approximate flux rates (Schlesinger 1995), then the average residence time of organic C entering soils is approximately 25 yr versus 5 yr for C in vegetation. The difference in residence times implies that changes in C stocks in soil, either as gains or losses, will occur more slowly than changes in C in vegetation. It should be emphasized, however, that terrestrial C sequestration, whether in vegetation or in soils, does not represent a permanent, irreversible "lock-up" of C, but rather that the build-up of terrestrial C stocks through changes in management practices is dependent on the long-term maintenance of those practices throughout time.

Mitigation of Carbon Dioxide Fluxes in Agricultural Soils

Current Emissions and Sinks of Carbon Dioxide

No comprehensive global estimates for current emissions and sinks of CO_2 from soils exist. Other components of the terrestrial C budget, including fossil-fuel emissions, atmospheric storage, and ocean uptake, imply that there was a net uptake of C by all terrestrial ecosystems of approximately 1.4 (±0.7) Pg/yr during the 1990s (see Table 1.1 in Chapter 1). This net amount includes C losses on the order of 1–2 Pg/yr through land-use change—principally through deforestation in the tropics—which have been more than offset by C gains in other terrestrial ecosystems. How much of this increase is associated with C stocks in vegetation versus soil is unknown.

For U.S. agricultural soils, it was estimated that C stocks increased by approximately 5–10 teragrams (Tg) C/yr for the period 1990 to 1996 (Eve et al. 2001; UNFCCC 2000; USEPA 2002), implying that agricultural soils in the United States presently are a small sink for C. Mineral soils, which make up the vast majority of cropland soils, gained approximately 10–15 Tg C/yr. Much of this increase is attributed to (1) recent trends in management of annual cropland, including the use of less-intensive tillage practices, decreases in the use of summer (bare soil) fallow, and changes in crop types and rotations; and (2) the conversion of annual cropland to perennial grassland and forest ecosystems, including CRP lands and conservation buffers. But C losses on cultivated organic land (i.e., histosols), which comprise less than 1 Mha of cropland in the United States, were estimated at 5–10 Tg C/yr, thus offsetting some of the gains on mineral soils. An additional factor to note is that the impact of greater crop residue production in recent decades (Allmaras et al. 2000) is not accounted for in the present U.S. inventory; thus, the current rate of soil C increase may be somewhat greater than the values cited (Eve et al. 2001). For comparison, current emissions of CO_2 associated with agricultural energy use on farms and for the production and distribution of fertilizers and pesticides used in agriculture are estimated at approximately 28 Tg C/yr (Lal et al. 1998).

Mitigation Practices and Rates

In general, C sequestration will be favored under management systems that (1) minimize soil disturbance and erosion, (2) maximize amounts of crop-residue return, and (3) maximize water- and nutrient-use efficiency of crop production. Although it may be impossible to achieve all these system attributes simultaneously, practices that effectively sequester C will share one or more of these traits.

Decreasing tillage intensity, especially by using no-tillage practices, is one of the most widely applicable techniques for increasing C sequestration. The physical disturbance accompanying tillage promotes the breakdown of soil aggregates, thereby making organic matter more accessible to microbial decomposition, as well as aerating and warming the soil (Reicosky and Lindstrom 1995). In long-term field experiments comparing no-till with intensively tilled annual crop systems, adoption of no-till typically results in increases in soil C of 0.1 to 0.7 metric tonnes or megagrams (Mg)/ha/yr (Dick et al. 1998; Janzen et al. 1998; Paustian et al. 1997a) during periods of 10 to 30 yr (Figure 2.8). An average U.S. rate of C sequestration under no-till of 0.34 Mg/ha/yr was estimated by West and Marland (2002). Rates tend to be higher in moist climates with high levels of crop residue inputs and somewhat lower in semiarid regions supporting less primary production. Cropping system responses to decreased tillage also will depend on other management practices such as crop rotation and fertilization (Dick et al. 1998). In semiarid regions, no-till adoption provides increased water storage, enabling more continuous crop rotations with elimination or decreased frequency of bare fallowing (Black and

Tanaka 1997; Havlin et al. 1990; Peterson et al. 1998). The effects of no-till systems under these conditions are synergistic, in that adoption of no-till leads to higher crop inputs through more continuous cropping, lower decomposition rates because of less (bare) summer-fallowing, greater water-use efficiency, and less soil disturbance (Peterson et al. 1998). Adoption of no-till alone, without decreasing the frequency of summer fallow, will have much less of a positive impact (or even a negative impact) on soil C sequestration (Jones, Steward, and Unger 1997; Peterson and Westfall 1997).
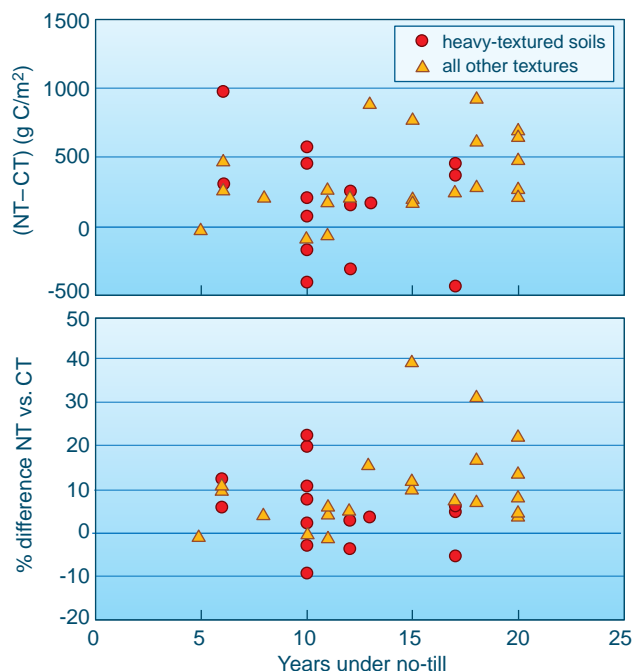
Figure 2.8. Differences in soil C between paired no-till (NT) and conventional till (CT) treatments from several temperate zone long-term experiments, showing absolute (top panel) and relative (bottom panel) differences. All values are calculated on a mass equivalent basis, to below depth of tillage, to account for tillage-induced changes in bulk density. Circles represent heavy-textured soils (clays, clay loams), and triangles represent all other soil textures. (Source: Paustian et al. 1997a.)

Increasing the amount of residue returned to soil can be managed through a variety of practices, including selection of high-residue-yielding crops (e.g., corn and sorghum), inclusion of hay crops in rotations, use of winter cover crops, application of manure and biosolids, improved management of fertilizer and water, and decrease of crop losses to pests. Most cropland soils show a clear response to increasing amounts of C return, such that SOC levels often are in direct pro-

portion to the amount of C added to soil under different management treatments (Huggins et al. 1998; Paustian, Collins, and Paul 1997; Rasmussen et al. 1980). The amount of soil C gain per unit of additional C input depends on the type of material added, the initial soil C content, and the time period over which the additions have occurred. Eventually, for any given level of input, soil C levels tend toward an equilibrium, limiting the amount and duration of additional C storage.

The average rate (per ha) of crop residue inputs (Figure 2.9) in the United States has increased dramatically since the 1950s, in part as a result of increased use of fertilizers, pesticides, and irrigation (Allmaras et al. 2000; Reilly and Fuglie 1998). Where
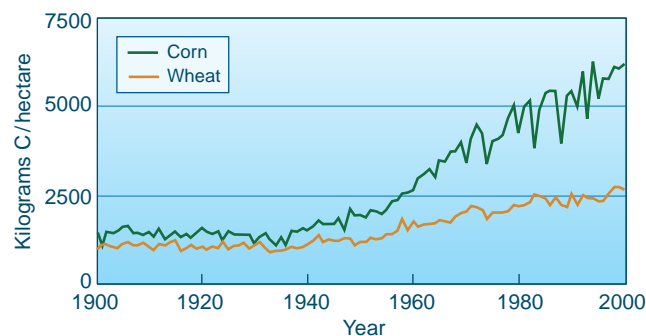
Figure 2.9. Average amounts of residue return (as kg C/ha) on corn- and wheat-growing acreage in the United States during the past 100 years. Data are based on annual crop yields compiled by the USDA's National Agricultural Statistics Service and conversion coefficients relating residue production to harvested products compiled from the literature. (Source: K. Paustian and S. Williams, Colorado State University.)

crop production is water- or nutrient-limited, irrigation and fertilization can contribute to C sequestration (Follett 2001). But CO_2 emissions associated with the manufacture and distribution of fertilizer, pumping of irrigation water, as well as emissions of N_2O and CH_4 from increased N fertilization and irrigation may offset part or all the gains in C storage. Fossil-fuel use and total GHG emissions from fertilizer and irrigation are highly dependent, however, on the type of input and its manner of use. For example, P fertilizers have no direct N_2O emissions associated with their use, yet they can increase production and C inputs in P-limited soils greatly (Russell 1960). Similarly, gravity-fed irrigation can increase production but requires little or no fossil fuel for its distribution (Follett 2001). Finally, it is worth considering that fertilization and irrigation are used to increase food production and not as a means to mitigate GHG emis-

sions. In general, practices promoting efficient water and nutrient use likely will have the greatest benefits in terms of decreased GHGs.

Various management practices on grazing lands (pasture and rangeland) can increase soil C. Compared with most annual crops, grassland vegetation effectively builds and maintains high soil C stocks. On perennial grasslands, tillage is absent or extremely limited. In addition, perennial grasses allocate much of their assimilated C to roots, resulting in high rates of C input to soil. Furthermore, the structure and rapid turnover of their fine roots favor the formation and incorporation of organic matter into stable soil-aggregate structures (Tisdall and Oades 1982). In comparison with short-season annual crops, for which conditions favorable to decomposition (e.g., warm, moist soils) may exist during spring and fall in non-vegetated soils, perennial grasses have a long growing season that enhances C inputs and acts to dry the soil and to suppress decomposition.

On poorly managed grazing lands depleted of SOM, practices that increase production and C inputs can build up soil C. Such practices include improving grazing management, using improved species, sowing legumes, fertilizing, and irrigating (Figure 2.10). In an analysis of more than 100 published studies, Conant, Paustian, and Elliott (2001) reported rates of soil C increase for different management improvements averaging between 0.1 to 1 Mg C/ha/yr, the highest rates occurring with conversion of cultivated land to perennial grasses (e.g., to pasture or CRP). Average rates of C increase for grassland management practices were approximately 0.3 Mg C/ha/yr for fertilization, approximately 0.2 Mg C/ha/yr for improved grazing or irrigation, and approximately 0.1 Mg C/ha/yr for introduction of legumes. Fisher and colleagues (1994) and Scholes and van der Merwe (1996) have reported some of the highest rates of C sequestration, on the order of 5 Mg C/ha/ yr or more, with the introduction of deep-rooted African grasses in tropical and subtropical savannas. Proper fire management also increases soil C. Studies in the Konza tallgrass prairie have reported a 5% increase in soil C as a result of annual burning (Rice et al. 1999)

Restoring degraded soils and ecosystems (Lal 1999), reforesting and afforesting degraded cropland and pastures (Brown and Lugo 1990; Gupta and Rao 1994; Li and Zhao 1998), retiring marginal lands through the CRP (Follett 1997), restoring wetlands through the Wetland Reserve Program (WRP), and controlling desertification (Lal et al.1999; Squires, Glenn, and Ayoub 1995) are important options for improving biomass productivity and sequestering C

in the soil and in the ecosystem. As an example, the global potential of desertification control for soil C sequestration has been estimated at 0.9 to 1.9 Pg C/ yr (Lal et al. 1999).

Converting marginal and highly erodible cropland to perennial vegetation, such as through the CRP and the WRP, has been of particular significance in the United States. A number of authors have documented substantial gains in soil C in these conversions (e.g., Follett, Kimble, and Lal 2001; Gebhart et al. 1994; Potter et al. 1999; Reeder, Schuman, and Bowman 1998; Robles and Burke 1998). At 14 sites across the Great Plains and western Corn Belt, Follett, Kim-
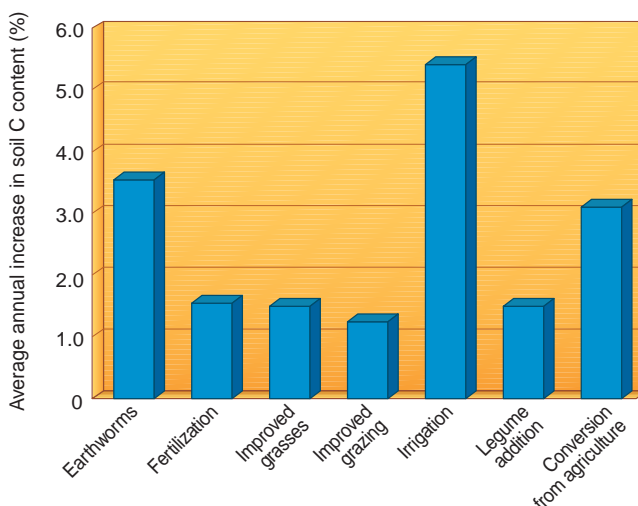
Figure 2.10. Average annual changes in grazing land C stocks for different management practices. (Source: Conant, Paustian, and Elliott 2001.)

ble, and Lal (2001) estimated average rates of C gain under CRP of 0.9 Mg/ha/yr. From an extensive review of the literature, Ogle and colleagues (2003) estimated that CRP increased soil C stocks by an average of 16% in the top 30 cm, equivalent to average gains of almost 7 Tg/yr on the 13.4 million ha of CRP land in the United States.

Additional Benefits of Carbon Sequestering Practices

The agronomic and environmental benefits associated with the buildup of SOM stocks are widely appreciated and have been documented in many studies. Soil organic matter is a primary indicator of soil quality (Larson and Pierce 1994), and the amount and quality of SOM impacts soil biological activity, soil structure and water dynamics, and nutrient cycling

and availability. Improvements in soil quality create favorable soil environments (Lal 1987), which can increase crop growth and yield, especially in situations where fertilizer use is low (Petchawee and Chaitep 1995). All other factors remaining the same, agricultural productivity increases with SOM (Ganzhara 1998).

Soil C is the main energy source for the soil biota, and hence increasing soil C promotes higher levels of microbial and faunal biomass and activity (Schnürer, Clarholm, and Rosswall 1985) and possibly greater soil biodiversity (Hooper et al. 2000). With no-till practices, the buildup of a residue layer on the soil surface acts to moderate extremes in temperature and moisture, providing a more stable environment for the soil biota. In certain areas, however, the higher surface residues associated with reduced- or no-tillage systems can result in decreased crop yields because of cooler soil temperatures at spring planting and higher levels of plant pathogens (Carter 1994; Kirkegaard 1995).

Soil organic matter serves as both a source and a sink for major plant nutrients, including N, P, and S, and thus is a fundamental component of soil fertility. Organic matter can make up a significant portion of the total soil ion exchange capacity, which influences the soil's buffering capacity and ability to retain nutrients and base cations. Pesticide immobilization and inactivation is enhanced by SOM (Tate 1987), which may serve to decrease off-site transport of pesticides to surface and groundwater.

One of the primary benefits of increasing SOM is associated with improving water dynamics within the soil profile. Organic matter, in general, tends to promote soil aggregation and the development of a stable, favorable soil structure with increased porosity and water-holding capacity (Buckman and Brady 1969). Higher organic matter contents and residue coverage of the soil surface decrease soil crusting and increase water infiltration rates, which enhance soil water storage and water available for plant growth. In semiarid environments, no-till practices increase the amount of water available for plant growth, allowing a decrease or elimination of the use of bare summer fallow, thereby increasing productivity during the cropping cycle and creating a positive feedback on SOM through increased C inputs (Peterson and Westfall 1997). Even in more humid regions, greater water availability under no-till compared with conventional tilled systems has been linked to higher productivity under no-till systems, especially during drought periods (Dick, Edwards, and McCoy 1997).

Coupled with increased water storage, higher SOC content and decreased soil disturbance promote a more stable soil structure, which leads to decreased potential soil erosion. Practices to increase SOC help decrease erosion in two major ways. First, residue retained on the soil decreases the impact of raindrops on the soil surface and prevents surface sealing, leading to a higher and more sustained infiltration rate and more rainfall absorbed into the soil. Second, well-aggregated, organic-matter-rich soils are more stable, and in such soils, although runoff may occur, sediment movement is decreased significantly. Several estimates suggest that soil erosion under no-till systems with residue retention is decreased by 4 to 5 times or more compared with rates under conventional tillage (Lindstrom et al. 1979; Nyakatawa, Reddy, and Lemunyon 2001). Erosion removes the upper soil layers, where the highest concentrations of organic matter tend to occur, such that erosion decreases have a disproportionate effect on preserving soil C stocks.

Improved soil structure also improves the "workability" or "tilth" of soil and decreases the energy required to move planting tools through the soil. In earlier times, soil tilth was assessed by the creaking of harnesses on horses or oxen, and the same principle applies today. The harder the soil, the more energy required to place anything—seed, fertilizer, and so forth—into it. No-tillage practices confer significant saving of fuel (and hence lower CO_2 emissions) compared with conventional practices, because of the need for fewer field passes and lower traction requirements. Frye (1984) estimated that fuel requirements for no-till were 55% of those for moldboard plowing. Considering all agricultural inputs (e.g., fuel, fertilizer, pesticides, and irrigation), West and Marland (2002) estimated C emissions of 168 kilograms (kg)/ha/yr under conventional till compared with 137 kg/ha/yr for no-till, or approximately a 20% decrease under no-till.

Mitigation of Nitrous Oxide Fluxes in North American Agriculture

Current Emissions of Nitrous Oxide

Independent estimates of global agricultural sources of N_2O range from 3.9 (Prather et al. 1995) to 6.3 Tg N/yr (IPCC 2001b; Kroeze, Mosier, and Bouwman 1999; Mosier et al. 1998a). The latter values are based on the new National Greenhouse Gas Inventories methodology (IPCC 1997; Mosier et al. 1998b), which includes separate estimates for N_2O emissions from

agricultural soils, N_2O emitted from animal waste-management systems, and indirect emissions caused by sources of agricultural N loss such as NO_3^- leaching and runoff. These three categories will be discussed in turn.

Current methodology for estimating N_2O emissions from agricultural soils is based on the observation that for most cropping systems N_2O flux is a nearly fixed proportion of total N inputs, typically close to 1.25 (+/– 1)%. Although not absolute, in surveys of published studies (e.g., Bouwman 1996; Mosier et al. 1996) more than 90% of reported N_2O fluxes from field studies fell within this range. Nitrogen inputs include synthetic fertilizers, manure and sewage sludge used as fertilizer, biological N_2 fixation, and crop residues. Fertilizer and manure inputs are discounted somewhat by N volatilization after application, and mineralization of SOM N is considered a significant source of N for peat soils. The methodology does not allow for differences in emission factors because of interactions among crop management practices, soils, climates, or input types—largely because insufficient information is available with which to parameterize for these different factors—but the method does provide a logical first-order estimator for soil-based N_2O emissions. Annual N_2O emission estimates for the United States and Canada are listed Table 2.4.

Nitrous oxide emissions from animal wastes can be significant (Tables 1.2 and 2.4). Confined animals excrete as dung and urine 80 to 95% of the N in their diet, and some proportion of this N is emitted as N_2O during collection, storage, and treatment. Proportions differ by type and handling of waste: in general, emissions increase with the N content of waste, the extent to which waste is allowed to become aerobic (allowing the initiation of nitrification-denitrification reactions), and the length of storage (Mosier et al. 1998a). For waste of a given N content, anaerobic lagoons and liquid systems will result in the least N_2O emissions, whereas solid storage and dry-lot handling will promote emissions. Mosier and colleagues (1998a) estimated that fluxes from liquid-based systems are on the order of 1 g N_2O–N/kg of excreted N, with fluxes from dry-storage systems 20 times greater. Nitrous oxide also is emitted from dung and urine patches in grazed pastures. Various studies have measured significant fluxes ranging from <1 to 50 g N/kg of excreted N on nonpeat soils; Mosier and colleagues (1998a) used a value of 20 g N/kg N excreted to estimate global pasture-flux.

Nitrogen is lost from cropping systems through means other than N_2O emission, and once lost will eventually become available again for emission as N_2O. Nitrate leached into groundwater will reemerge in riparian zones and if not denitrified or immobilized will continue into aquatic and coastal ecosystems, where it also may be emitted as N_2O. Some proportion of the organic N in crop yields either is consumed directly by humans or becomes meat or milk that is subsequently consumed. Most of this N then enters sewage treatment plants, where again it is available for conversion to N_2O or to NO_3^- that enters riverine systems and subsequently may be denitrified. Nitrogen volatilized as ammonia gas (NH_3) from crop fields, pasture, or confined animal facilities, or N oxidized from soil as NO_x, will reenter ecosystems downwind as inadvertent NO_3^- and NH_4^+ fertilizer. Such atmospheric deposition has been shown to stimulate N_2O emissions in forest soils at a rate similar to that of intentional fertilizers in cropped systems: approximately 0.2 to 1.6% of deposition-N inputs can be emitted as N_2O (Bowden, Melillo, and Steudler 1991; Brumme and Beese 1992).

The sum of emissions from cropped soils, animal waste handling and pasture systems, and indirect sources is approximately 6.3 (+/- 3.1) Tg N_2O-N/yr (see Table 1.2). Values for the major sources comprising global agricultural flux have a range of approximately twofold. Sensitivity analysis (van Aardenne, Kroeze, and Hordijk 1998) suggests that the overall agriculture flux is sensitive especially to two estimators: the proportion of N inputs estimated to be emitted as N_2O directly from soil, and the fraction of N inputs lost through leaching and runoff (which are an indirect source of N_2O).

Mitigation Practices and Rates

Examination of likely agricultural sources of N_2O suggests several practices that may mitigate the current increase in atmospheric concentrations of N_2O.

Table 2.4. N_2O emission from agricultural soil management in the United States and Canada for 1996 (based on AAFC 1999; USEPA 2002; adapted by A. R. Mosier)

Activity	United States	Canada
	N_2O emitted (Gg N_2O)	
Direct		
Agricultural soils	684	70
Grazing animals	140	25
Indirect	259	38
Total	1,083	133

Agricultural sources of N_2O seem to be split evenly among emissions from cropped soils, animal waste, and indirect sources affected by crop N loss. Each sector provides opportunities for decreasing current N emissions (Table 2.5). Many of these options have been detailed and reported in the literature (Cole et al. 1997; Kroeze and Mosier 2000; Mosier et al. 1998a; Oenema et al. 1998; Robertson, in press; Velthof, van Beusichem, and Oenema 1998).

Table 2.5. N_2O mitigation potentials for different agronomic practices (reduction estimates based on analyses in Cole et al. 1997; Kroeze and Mosier 2000; adapted by A. R. Mosier)

Practice	% Reduction from current N_2O flux
Soil emissions	
Soil N tests	10
Fertilizer timing	10
Cover crops	5
N fertilizer placement	5
Nitrification and urease inhibitors	5
Animal production	
Waste storage	25
Waste disposition	10
Improved pasture management	5
Indirect fluxes	
Crop N use efficiency	20
Riparian zone management	5
Ammonia management	5
Wastewater treatment	5

Mitigation of Nitrous Oxide from Agricultural Soils

Because most cropped soils emit N_2O at a rate of 0.2 to 3% of their N inputs, decreasing N inputs in cropping systems could decrease N_2O emissions directly, by approximately 1.25% of N inputs saved. Recent research suggests that synthetic fertilizers, manure, and biological N_2 fixation have equivalent effects on N_2O flux in most intensive cropping systems, and that tillage and other cultivation techniques have little effect on flux when soil N is readily available (Robertson, Paul, and Harwood 2000). Nitrogen is used inefficiently in most cropping systems: typically, only about half of N inputs are captured in crop biomass (Robertson 1997), and much of the remainder can be lost from the system through leaching and runoff (provided rainfall exceeds evapotranspiration) or through gaseous losses of N_2, N_2O, NO_x, or NH_3. Because there is a direct relationship between soil N availability and crop yield for crops in the United States, the main agronomic challenge for decreasing direct N_2O emission from agricultural soils is to de-crease N inputs without decreasing yield.

Kroeze and Mosier (2000) estimated that improved crop-N use efficiency could decrease soil-derived N_2O emissions from agriculture by as much as 35% globally, with even more savings in the input-intensive systems of North America, Europe, and the former Soviet Union. Such savings could be achieved by the application of existing technology, largely by better matching crop-N needs with soil N availability. For example, managing crops by banding fertilizer within the crop row or by applying N across a field at variable rates based on yield or soil test maps (i.e., *precision farming*) has significant potential to improve cropping system-N use efficiency. Such practices keep added N away from places in the field where it is less likely to be taken up quickly by plants; thus the N added is less available to N_2O-producing microbes and also less likely to be leached or volatilized.

Likewise, the timing of N availability in cropping systems can affect N use efficiency significantly. Timing fertilization and the decomposition of crop residues to coincide better with active crop growth can lead to improved synchrony between soil nutrient availability and crop demand. The longer N remains free in the soil solution before crop uptake, the more likely it is to be lost first to N_2O formation or some other fate. If N additions occur long before crop growth—either intentionally, as when fields are plowed or fertilized in the fall, or unintentionally, as when crop residues start decomposing months before the next crop—cover crops can be used to capture a portion of the N that otherwise would be available for N_2O formation. When killed the next spring, the cover crop will decompose, and its N will be released for potential crop uptake.

Any practice that tightens the coupling between soil N release and crop growth thus will lead to enhanced nutrient-use efficiency and to a diminished need for external N, thereby decreasing N_2O flux. And any practice capturing N within the system before its potential loss can help conserve available N for later use by the crop. Practices promoting efficient N use include the following (see Table 2.5):

1. **Soil N tests**. Nitrogen fertilizer is relatively inexpensive, and in the absence of specific information about soil N availability, many growers tend to overfertilize crops. Soil N tests—in particular, the pre-sidedress nitrate test (PSNT) performed at planting—can decrease overfertilization significantly. Soil N testing allows growers to adjust N application rates accurately to yield goals. Only approximately half of U.S. acreage planted to corn

in the mid-1990s was tested for soil N before planting; 80% of growers who tested applied the recommended rate (USDA 1997). Increasing the percentage of growers who soil test could decrease overfertilization significantly and thus decrease both direct and indirect fluxes of N_2O.

2. **Fertilizer timing**. Thirty percent of the U.S. corn acreage received fall-applied N in 1995 (USDA 1997). A significant number of producers fall-plowed. On only 30% of corn acreage was N applied after planting. Fall fertilization and fall plowing create a large pool of soil mineral-N available for overwinter loss through denitrification or leaching. Spring plowing and fertilizing as close as possible to the period of active crop growth keeps N away from competing sinks, including N_2O-producing microbes.

3. **Cover crops**. Winter or fallow cover crops can prevent the buildup of residual soil mineral-N and can decrease significantly nutrient losses during periods when the primary crop is not growing, which is most of the year in much of the United States (Figure 2.11). Yet cover crops were used on only 4% of major field crop acres in the United States in 1995. Using cover crops to catch the N that otherwise would be emitted as N_2O or lost by other means could significantly improve crop-N use efficiency. If the cover crop were a legume that also fixed atmospheric N, then some of the CO_2 cost of synthetic fertilizer also would be saved.

4. **Nitrogen placement**. Broadcasting is the most common method of applying N in the United States (USDA 1997). On only 40% of U.S. corn acreage in the mid-1990s were nutrients *banded*, or placed within the rows. Banding can provide significant N efficiencies: compared with broadcast applications, injecting anhydrous ammonia into soil near rows, for example, can decrease N leaching and volatilization by as much as 35% and increase yield by as much as 15% (Achorn and Broder 1984).

5. **Nitrification and urease inhibitors**. Nitrogen applied as NH_4^+ or mineralized from SOM must be nitrified to NO_3^- before it is available for denitrification. Urea must be hydrolyzed to NH_4^+ by soil urease. Nitrification inhibitors delay the transformation of NH_4^+ to NO_3^- to help match the timing of NO_3^- supply with peak demand (Figure 2.12). Urease inhibitors delay the hydrolysis of urea fertilizer to NH_4^+. Likewise, control-release forms of N fertilizer such as polyolefin-coated urea

Figure 2.11. Cover crops prevent erosion and add N to the soil. Here, technicians collect samples so they can estimate the amount of plant matter in a field near Thighman Lake, Sunflower County, Mississippi. Photo courtesy of S. Bauer, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.
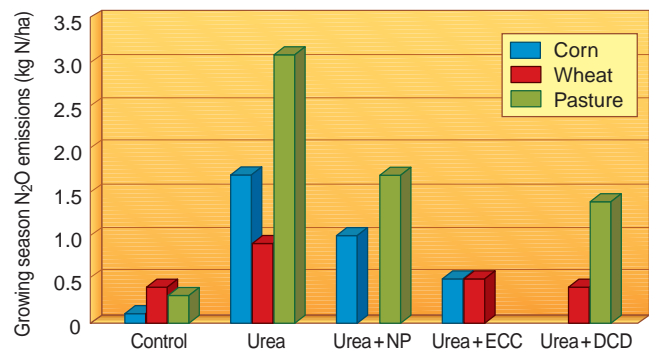
Figure 2.12. Effect of urease and nitrification inhibitors on N_2O emissions from corn, wheat, and pasture systems. Nitrification inhibitors include nitrapyrin (NP), encapsulated calcium carbide (ECC), and dicyandiamide (DCD). (Data for corn and wheat from Bronson and Mosier 1993, and for pasture from McTaggart et al. 1997.)

can delay enrichment of the soil mineral-N pool until crop growth occurs (Shoji et al. 2001). Nitrification inhibitors were used on less than 10% of U.S. corn acreage in 1995.

Mitigation of Nitrous Oxide from Animal Production

In 1992, approximately 28% of U.S. cropland was farmed by operators with confined animal units. The percentage is somewhat lower today, reflecting a long-term trend of increasing numbers of animals on fewer farms (USDA 1997). The increasing concentrations of high-density animal operations include dairy farms in Arizona and southern California, feedlots in the Southern Plains, hog operations in the Corn Belt, and operations in the Poultry Belt across the lower Mississippi Valley, Southeastern, and Appalachian states. For >90% of these operations, manure does not leave the farm, which means that it is stored and spread on-site—often at rates much higher than can be used by crops (USDA 1997). Approximately 14% of corn acreage in the United States received manure in 1995.

Most N_2O emitted in animal production occurs during waste storage, after waste spreading on the field, or after urine and dung deposition in grazed pastures. Management options available to decrease emissions significantly include the following (see Table 2.5):

1. **Waste storage**. Storing animal waste anaerobically can minimize N_2O losses. For N_2O to be produced, NH_4^+ from decomposing waste must be nitrified to NO_3^-. Nitrifying bacteria require O_2, so storing waste in liquid form or in lagoons can decrease N_2O emissions dramatically.

2. **Waste disposal**. How and when treated waste is disposed will affect the likelihood of subsequent N_2O emissions. If waste is used as a fertilizer, then mitigating poststorage emissions will require the same considerations as minimizing direct losses of N_2O from soil, as discussed in the preceding section. It is especially important to apply manure at rates based on crop needs (as determined by soil test), to avoid overfertilizing, and to maximize the likelihood that manure-derived N will be taken up by the crop rather than be made available to competing sinks such as N_2O production and leaching. Manure applied to fallow fields in fall and winter is likely to enhance both direct and indirect N_2O emissions. Manure applied to cover crops or just after planting will minimize N_2O emissions most effectively.

3. **Pasture management**. A larger proportion of the U.S. (lower 48 state) land base is used for grassland pasture and range (31%) than for cropland (24%) (USDA 1997), and mitigation of N_2O production on the grassland pasture and range land base could be significant. Most N_2O emissions from grazed animals occur from high-concentration urine and dung patches. Because the patchiness of these excretions cannot be controlled, the main management option is to ensure that pastures are not overfertilized with N added by the animals themselves. Emissions of N_2O will be promoted when animals are grazed in previously fertilized pastures or concentrated in small pastures, as more N will be added than can be taken up by pasture plants. Maximizing the likelihood for plant uptake of added N minimizes N_2O emissions.

Mitigation of Indirect Nitrous Oxide Fluxes

Indirect N_2O fluxes are those caused by agricultural N that escapes to the noncrop environment through NH_3 volatilization, runoff, and NO_3^- leaching. The most effective strategy for decreasing indirect N_2O fluxes is to minimize the export of excess N from the farm and to ensure that N exported in farm products is removed before being released back into the environment (see Table 2.4).

Specific mitigation potentials include the following:

1. **Maximizing crop-N use efficiency**. If the practices noted in the preceding section on "Mitigation of Nitrous Oxide from Agricultural Soils" are implemented, then N loss from crop fields can be minimized. Keeping excess N out of the noncrop environment is the most effective strategy for mitigating indirect N_2O fluxes.

2. **Managing riparian zones**. Nitrogen leached to groundwater and N in runoff eventually will enter lakes and streams. Planting filter strips near streams and encouraging the planting and maintenance of trees in riparian zones will help keep leached N from becoming N_2O at streamside or farther downstream. Nitrogen removed by streamside vegetation will be less available to the microbes that produce N_2O.

3. **Managing ammonia**. Ammonia gas volatilized from confined-animal facilities or from anhydrous ammonia fertilizers becomes rainwater NH_4^+, which reenters ecosystems as inadvertent fertilizer N. Animal waste can be handled to minimize

NH_3 emissions by the storage of waste in lagoons or other anaerobic systems in which pH is kept relatively low. Loss of anhydrous NH_3 applied as fertilizer can be minimized by the injection of NH_3 to an appropriate depth when soil moisture conditions are favorable (Nelson 1982).

4. **Treating wastewater**. Annually, U.S. sewage treatment plants release into the North Atlantic Ocean approximately 0.5 Tg N as dissolved N compounds, mainly NO_3-N. This volume represents approximately 12% of total N inputs to the North Atlantic (Howarth et al. 1996). Much of this N enters the waste stream as a former agronomic commodity consumed by humans. Removal of this N before it is released as effluent will prevent it from becoming N_2O in downstream rivers, estuaries, and pelagic zones (Kroeze and Seitzinger 1998).

Associated Benefits of Practices to Decrease Nitrous Oxide Emissions

Most management practices that promote N_2O mitigation are based on improved N management. Direct soil emissions are minimized by the creation of a more efficient crop-N economy. Emissions from animal production are decreased by appropriate storage and handling practices and by the timely and appropriate application of animal wastes to fields. Indirect emissions are minimized by decreasing the transfers of agricultural N to rainwater, streams, and rivers.

All these mitigation strategies have additional benefits for the environment. First, increasing farm-N use efficiency will help keep all forms of N out of the nonfarm environment. Groundwater NO_3^- loading will be decreased, as will subsequent loadings of surface and coastal waters. Eutrophication of U.S. coastal regions results in the formation of summertime hypoxic zones that can have a substantial negative impact on U.S. coastal water quality and fisheries (NAS 2000). High NO_3^- concentrations in well water degrade its quality for human consumption and can pose a serious health risk for young children.

Tighter farm-N cycles also will help decrease NH_3 and NO_x emissions to the atmosphere, subsequently decreasing deposition-N inputs to nonagricultural ecosystems (Holland et al. 1997). These inputs have been linked to forest decline in the northeastern United States (Fenn et al. 1998).

Making crop-N use more efficient also will decrease the need for synthetic N fertilizer, as will better man-

agement of manure N if the manure now applied in excess to certain crops is used instead as a fertilizer on crops now receiving synthetic N inputs. More efficient use of fertilizers and manure translates to lower fertilizer costs to farmers. Synthetic N fertilizer also produces CO_2 in its manufacture (Schlesinger 1999), so substituting excess manure for synthetic N will provide measurable CO_2 mitigation. Likewise, cover crops—if legumes—grown to mitigate N_2O production also could fix atmospheric N, thus also decreasing the need for synthetic N (Robertson, Paul, and Harwood 2000).

Certain N_2O mitigation practices also will result in direct CO_2 mitigation. Riparian forests that can mitigate against indirect N_2O fluxes will store C in growing vegetation for a number of decades, and both riparian forests and cropping systems with cover crops accumulate C in SOM.

Mitigation of Methane Fluxes in North American Agriculture

Current Emissions and Sinks of Methane

Approximately 70% of CH_4 production arises from anthropogenic sources and approximately 30% from natural sources. Agriculture is considered responsible for approximately two-thirds of the anthropogenic sources globally (see Table 1.3). Enteric fermentation in ruminant animals, livestock-waste management systems, and rice production are the main North American agricultural sources of CH_4 (Agriculture and Agri-Food Canada 1999; USEPA 2002). The main CH_4 sink affected by agriculture is the oxidation of CH_4 in aerobic soils (Mosier et al. 1997; Robertson, Paul, and Harwood 2000). General estimates of CH_4 emissions from U.S. and Canadian agricultural systems are provided in Table 2.6.

Emissions from Livestock

Methane emissions from livestock account for from 2 to 12% of gross feed-energy intake (Gibbs and Leng 1993; Johnson et al. 1993) or 5 to 20% of metabolized energy (Gadeken, Rath, and Sauerbeck 1990; Harper et al. 1999; Leng 1991; USEPA 1992). Because CH_4 production represents a loss of energy to animals, there is considerable interest in decreasing this loss in domestic livestock (Shibata 1994). Per-animal emission rates of CH_4 loss depend significantly on ruminant type, but for domestic ruminants, emission

Table 2.6. Estimates of CH_4 emissions from agricultural systems in Canada and the United States. Values are in Tg CH_4 /yr (based on data from AAFC 1999; USEPA 2002; adapted by A. R. Mosier)

	Canada	United States	Canada	United States
Source	1991		1996	
Livestock				
Enteric fermentation	0.8	6.1	0.9	6.2
Manure management	0.2	1.4	0.2	1.6
Rice production	0	0.4	0	0.3
Agricultural residue burning	<0.05	0.03	<0.05	0.03

rates are similar for a given feed type (Johnson et al. 1993; Shibata 1994) (Figure 2.13). Differences in CH_4 emissions among all ruminants are related to differences in levels of feed intake and extent of digestion, both of which depend on animal type and age. Generally, the fraction of feed converted to CH_4 (methane yield) decreases as feed intake increases (Branine and Johnson 1990; Johnson et al. 1993; Shibata et al.

Figure 2.13. Dryland grazing on the Great Plains in Colorado. Each cow on a pasture can emit about 350 liters (230 grams) of CH_4 per day. Photo courtesy of S. Bauer, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

1993). Methane yield also tends to decrease as feed quality increases (Gibbs and Leng 1993; Johnson et al. 1993); CH_4 emissions per unit of digestible energy generally are two to three times higher when animals are on a low-quality diet than when they are on a high-quality diet (Leng 1991; USEPA 1992). For example, a sixfold decrease in CH_4 emission was observed when grazed cattle were switched to a high-quality feedlot diet (Harper et al. 1999). For the United States, estimates of CH_4 production from enteric fermentation are based on livestock populations and emission factors multiplied by animal numbers

(USEPA 2002). Emission factors are based on animal diet and type (e.g., dairy or beef cow) and are derived from mechanistic models of rumen digestion and animal production (Baldwin, Thornley, and Beever 1987). In Canada, estimates have been made using similar average animal emission rates (Agriculture and Agri-Food Canada 1999).

Methane emissions from nonruminant livestock such as pigs and horses are lower than those from ruminants but also depend on diet quality. Emissions range from 0.6 to 1.2% of gross energy intake for a high-quality diet, and rates increase, by as much as twofold, for a low-quality diet (Crutzen, Aselmann, and Seiler 1986; Gibbs and Leng 1993; Johnson et al. 1993). Uncertainties in CH_4 emissions from these animals are high but of minor concern because approximately 95% of animal CH_4 emissions are from ruminants (Johnson et al. 1993). Cattle in North America and Europe are the most intensively managed in the world but account for only approximately 17% of total global emission from cattle and buffalo combined (FAO 2000).

Although almost all data on CH_4 emissions from cattle have been gathered from confined animals, a recent study of free-range cattle yielded results that compare well with emission estimates for confined animals (Agriculture and Agri-Food Canada 1999; Huyler et al. 1993). In a micrometeorology mass balance study of grazing cattle, Harper and colleagues (1999) found CH_4 emission values close to those predicted from the equations of Moe and Tyrell (1979), which were based on confined dairy cows.

Globally, CH_4 emissions from domestic animals are estimated at approximately 80 Tg/yr, with a range of 65 to 100 Tg/yr (Hogan 1993). In Canada and the United States, CH_4 production from enteric fermentation in livestock totals approximately 7 Tg CH_4 (6 from the United States and 0.8 from Canada) (Agriculture and Agri-Food Canada 1999; USEPA 2002) (Table 2.6)

Emissions from Livestock Waste

Globally, manure from intensively managed animals contributes 10 to 30 Tg CH_4/yr or approximately 18% of estimated direct emission from animals (USEPA 1993b, 1994). Seventy-five percent of global generation is estimated to come from cattle and swine manures, and 55% of that is from North America and Europe (Mosier et al. 1998c).

United States livestock-waste CH_4-emission values are obtained from the U. S. Environmental Protection Agency (USEPA 2002), which estimates emissions on the basis of manure production, the portion of manure that is volatile solids (by animal type), the CH_4-producing potential for volatile solids, the extent to which CH_4-producing potential is realized for each type of manure-management system, and the portion of manure managed in each system (by state and animal type). Canadian estimates are based on manure production and CH_4 emission rates (Agriculture and Agri-Food Canada 1999). According to these estimates, emissions from manure account for approximately 20% of the Canadian total (enteric fermentation + manure management) (Table 2.6). In the United States, approximately 33% of livestock CH_4 emissions are derived from waste management systems (Table 2.6).

Emissions from Flooded Rice

Rice production in North America is limited to seven U.S. states: Arkansas, Louisiana, Texas, California, Mississippi, Missouri, and Florida, which together accounted for a production area of 1.26 to 1.49 million ha between 1990 and 1997 (USEPA 2002). Because of the relatively small area involved, emissions from rice cultivation are small, approximately 4% of total methane emissions from U.S. agriculture.

Microbial Oxidation of Atmospheric Methane in Aerobic Soils

Soil- and land-use-related processes represent an important sink for atmospheric CH_4. Methane uptake rates determined during relatively long periods, for a variety of native ecosystems, show a surprisingly narrow range: from 23 micrograms (μg) CH_4 per square m per hr in taiga soils (Whalen, Reeburgh, and Kizer 1991), to 28 in desert soils (Striegl et al. 1992), to 20–50 in temperate grasslands (Mosier et al. 1997), and to 38–60 in temperate forests (Robertson, Paul, and Harwood 2000; Steudler et al. 1989). Reeburgh, Whalen, and Alperin (1993) estimated aerobic soil

CH_4 oxidation, globally, at 20 to 60 Tg/yr.

Conversion from native grasslands and forests to managed pastures and cultivated crops generally decreases the normal aerobic soil CH_4 sink (Keller, Mitre, and Stallard 1990; Mosier et al. 1991, 1997; Nesbit and Breitenbeck 1992; Robertson, Paul, and Harwood 2000). Management intensity seems to affect most the magnitude of decrease. Bronson and Mosier (1993) and Mosier and colleagues (1997) found in northeastern Colorado that fertilizing native grassland decreased CH_4 uptake rates by approximately 35% (Table 2.7). Cultivation decreased consumption an additional 15%. In irrigated maize and wheat, N fertilization did not decrease CH_4 consumption further, but rates were 85 to 90% lower than in native grasslands. Even if land is converted back to perennial vegetation (e.g., CRP land), recovery of CH_4 oxidation on former cropland likely will require several decades (Mosier et al. 1997).

Robertson, Paul, and Harwood (2000) found CH_4 oxidation rates in corn-soybean-wheat cropping systems in Michigan to be 80% lower than rates in adjacent native forests (Table 2.7). Rates were equally low in perennial crops (alfalfa and poplar trees) and recovered very slowly after abandonment from agricul-

Table 2.7. Effect of agricultural intensity on CH_4 oxidation (based on data from Bronson and Mosier 1993; Mosier et al. 1991; Robertson, Paul, and Harwood 2000; adapted by A. R. Mosier)

Location, Cropping system	CH_4 uptake μg C m^{-2} h^{-1}
Colorado	
Native pasture	26
N-fertilized pasture	17
Nonirrigated wheat	13
Maize	
Unfertilized	2.5
200 kg N/ha	2.5
Irrigated wheat	
Unfertilized	3.3
80 kg N/ha	3.8
Michigan	
Late successional forest	38
Corn-soybean wheat rotation	
Conventional tillage	6.8
No-till	7.2
Low input with cover crop	7.1
Organic with cover crop	8.0
Perennial crops	
Alfalfa	8.8
Poplar	7.4
Successional communities	
Early successional	9.8
Midsuccessional	24

ture. In 50-yr-old midsuccessional sites, oxidation rates were still 40% lower than in native forests. Among the cropped systems, rates were similar among conventional, no-till, and organic systems.

Emissions from Crop Residue Burning

Biomass burning emits CH_4 because of incomplete combustion. Globally, biomass burning (including forest and grassland) is estimated to emit 20 to 80 Tg CH_4/yr to the atmosphere (see Table 1.3). Levine, Cofer, and Pinto (1993) estimated CH_4 released from biomass burning to total 51.9 Tg/yr, whereas Delmas (1993) estimated a global source of approximately 35 Tg/yr. In the United States, agricultural residue burning is uncommon; CH_4 emissions from crop residue burning are therefore minor, on the order of 0.03 Tg CH_4/yr (Table 2.6) (EPA 2002).

Mitigation Practices and Rates

Successful development and implementation of mitigation strategies for agricultural sources of CH_4 require a comprehensive understanding of the effects of land-use change and agricultural practice on fluxes of these gases and on mechanisms for control. Current knowledge falls short of these criteria but is sufficient to identify key systems, practices, and geographic areas for targeting. To ensure that interactions and feedbacks are considered, proposed mitigation technologies should be evaluated within the context of whole-farm production systems. Trade-offs between CH_4 and N_2O especially are important because their production and consumption are controlled by different soil chemical and microbiological effects, and a strategy that minimizes one flux might promote another. Typically, CH_4 oxidation and N_2O emissions are inversely correlated (Mosier et al. 1996).

Mitigation of Methane from Ruminant Livestock

Opportunities for decreasing CH_4 emissions from intensively managed cattle are limited because these operations currently are quite efficient. For dairy cattle, potential decreases in current emission are approximately 10% through genetic improvement, 10% through bovine growth-hormone therapy, and 4% through improved feed formulation (CAST 1992; USEPA 1992). For beef cattle, pharmaceuticals being developed to promote protein-gain at the expense of fat could decrease CH_4 emissions by as much as 20% (CAST 1992). Longer-term opportunities include

the promotion of twins production to decrease the need for breeding animals, and biotechnological modifications to rumen fermentation to eliminate CH_4 production (CAST 1992; USEPA 1993b).

The U.S. Department of Energy (DOE 1999) recently reviewed the elements of U.S. GHG emissions and suggested that, as a result of expected improvements in milk production/cow, CH_4 emissions/unit milk produced will continue to decline. By 2010, a decrease in emissions of 30% is envisioned for the dairy industry, without a loss of profits. According to the DOE (1999) analysis, the main options for decreasing CH_4 emissions from the beef industry are refinements to the marketing system and improvements in cow-calf sector performance. Methane emissions from beef cattle in the United States could be cut 20% with no loss of profits. Specific practices to decrease CH_4 emissions from ruminants include the following:

1. **Feed rations to decrease digestion time**. Because most CH_4 is produced in the rumen, where feed is fermented anaerobically, the longer the feed remains in the rumen, the more organic matter is converted to CH_4. Practices to speed the passage of feed through the rumen include

   - harvesting forages at an earlier, more succulent growth stage;
   - chopping the feed to increase surface area;
   - minimizing the use of fibrous grasses and hays;
   - using easily digestible feed grains, legumes, and silage; and
   - feeding concentrated supplements.

2. **Edible oil additives**. Adding canola, coconut, or other edible oils to the livestock diet may decrease CH_4 production by inhibiting methanogenic activity.

3. **Ionophore additives**. Ionophores are feed additives that inhibit the formation of CH_4 by rumen methanogens. They are used widely in beef production. Because rumen microbes can adapt to specific ionophores, it may be necessary to use a rotating variety of ionophores.

4. **Specialized rumen bacteria**. Researchers are investigating the introduction of genetically modified bacteria into the rumen, resulting in less production of CH_4.

5. **Livestock production efficiency**. Improving the feed efficiency of livestock production will decrease CH_4 emissions because fewer animals will be needed to produce the same amount of product.

Many of these practices are in use now. If used by all North American producers, they have the potential to decrease energy lost through CH_4 release from approximately 5–8% of gross feed energy to 2–3% (Agriculture and Agri-Food Canada 1999).

Mitigation of Methane from Livestock Waste

Most CH_4 produced in anaerobic digestion constitutes a wasted energy source that can be recovered by adapting manure management and treatment practices to collect CH_4 (Hogan 1993). The by-products of anaerobic manure digestion can be used as animal feeds, aquaculture supplements, or crop fertilizers (Figure 2.14). Methods of decreasing CH_4 emissions include the following:

1. **Using covered lagoons**. This option is associated with the large-scale, intensive-farming operations common in North America, Europe, and regions of Asia and Australia.

2. **Using large-scale digesters**. Larger, more technically advanced CH_4 digesters can be integrated with management practices at large livestock operations. Recent DOE (1999) analyses have suggested that CH_4 recovery from large-scale, liquid manure-handling systems may be profitable for certain systems. Estimates of profitable emissions decreases from dairy and swine operations are 25% and 19% of 1990 emissions, respectively.

3. **Using other waste-storage practices**. Other practices that could mitigate CH_4 production from

Figure 2.14. Manure injection on a farm in Maryland. Photo courtesy of T. McCabe, U.S. Department of Agriculture.

livestock wastes include the following:

- using solid rather than liquid manure handling (although this practice may promote N_2O formation);

- applying manure to land as soon as possible (storing manure sometimes is necessary because the soil is frozen, too wet, or being cropped, and applying manure out of synchrony with crop demands for N will affect N_2O emissions);

- minimizing the amount of bedding in manure;

- keeping storage tanks cool; and

- aerating manure during composting (which may promote N_2O formation).

Mitigation of Methane from Rice Cultivation and Biomass Burning

Three major management options exist for limiting CH_4 emissions from rice: water management, nutrient management, and new rice cultivar adoption. It is unclear under which conditions these practices are agronomically profitable.

1. **Water management**. Methane emissions are influenced by flooding period and drainage schedule. Draining the rice field during the growing season seems to decrease CH_4 production by increasing the state of oxidation of the paddy soil (Sass et al. 1992). This strategy is best suited to rice fields with highly secure, controllable water supplies. Research is needed to determine whether water management to decrease CH_4 emissions also will increase N_2O production.

2. **Fertilizer management**. Application of rice straw increases CH_4 emission rates significantly, as does the addition of large amounts of compost (Schutz et al. 1989; Yagi and Minami 1990). The application of inorganic N fertilizers to rice seems to decrease CH_4 emissions relative to those from unfertilized fields or fields fertilized with organic materials. This decrease is believed to occur because the C:N ratio of the soil is decreased so that more organic C remains in the soil biomass. Fertilizing with NH_4^+ sulfate (Lindau et al. 1993) supplies N and sulfate, which maintain the soil-reduction potential above the level required to produce CH_4. Coated Ca carbide produces small quantities of acetylene (Banerjee and Mosier 1989; Bronson and Mosier 1991; Keerthisinghe, Freney, and Mosier 1993) that can limit ni-

trification and CH_4 production.

3. **New rice cultivars**. Methane emissions differ by rice varieties by a factor of two or more (Sass 1994). Thus, breeding for CH_4 emission potential in addition to productivity and taste may be a useful strategy for mitigating CH_4 (Lin, Dong, and Li 1994; Neue 1992).

4. **Biomass burning**. Frequency of burning croplands, grasslands, and forests may be decreased through sustained land-management programs and alternative land-use practices. The contribution of residue burning to the North American CH_4 flux is small, however, limiting the potential for mitigation.

In all, CH_4 emissions from agricultural sources could be decreased 30 to 40% using some combination of these practices (Table 2.8).

Table 2.8. CH_4 mitigation potentials for different agronomic practices worldwide (based on data from Mosier et al. 1998c and Matthews et al. 2000)

Source	Estimated amount emitted (Tg CH_4/yr)	Potential decrease Tg CH_4/yr	%
Ruminant animals	76 (65–100)[1]	24 (12–45)	32
Animal waste	14 (10–30)	3 (2–4)	21
Flooded rice	20 (9–25)	8 (4–10)	40
Biomass burning	22 (11–33)	6 (3–9)	27
Total	132 (95–188)	41 (21–68)	31

[1]Range of estimates.

Additional Benefits of Practices to Decrease Methane Emissions

Many of the management practices capable of mitigating CH_4 emissions in agricultural systems also can improve crop and animal productivity. Using feed additives to inhibit rumen CH_4 production increases the amount of feed available for livestock weight gain. Using covered lagoons to capture CH_4 from livestock waste and large-scale digesters to produce energy from the captured CH_4 would allow farmers to purchase less energy and will decrease CO_2 emissions associated with energy production. Properly spreading digested manure on crop fields would provide certain nutrients to crops, thereby decreasing reliance on synthetic fertilizer sources and saving the economic and CO_2 expense of fertilizer. Managing water and nutrients differently in flooded rice may provide both water and fertilizer savings.

Because certain practices decreasing CH_4 emissions may enhance N_2O emissions, their adoption must be considered in the context of whole-system Global Warming Potential (GWP) analysis (Robertson, Paul, and Harwood 2000). For example, spreading manure on crop fields could mitigate N_2O, but only if done in a manner optimizing rate and timing of application for maximum crop uptake. Substituting synthetic fertilizers for compost in flooded rice systems adds to the CO_2 cost associated with fertilizer manufacture. And whereas draining flooded rice fields during the growing season may decrease CH_4 emissions, it also may decrease soil C storage and enhance N_2O emissions.

Full Greenhouse Gas Accounting

To fully evaluate the effects of agriculture on GHG concentrations in the atmosphere requires consideration of all sources and sinks of GHGs in a given agricultural system. When sources exceed sinks, the system is a net source of GHG; when sinks exceed sources, the system is a net sink. Mitigation of GHG increases entails decreasing sources, increasing sinks, or both.

Full GHG accounting requires a common currency with which to measure all sinks and sources. The GWP provides this metric. Global Warming Potential is a measure of the cumulative radiative forcing, during a specified time horizon, of a particular GHG relative to some reference gas (IPCC 1996a); by convention, GWP is expressed as CO_2 or C equivalents.

As described earlier in this chapter, there are many different sources and sinks that contribute to the GWP of agricultural systems. In a modern row-crop system, potential sources include the use of fuel, N fertilizer, irrigation water, lime, and N_2O emissions. Methane emissions must be included for livestock operations. Potential sinks include soil C sequestration and CH_4 oxidation. All potential sources must be weighed against all potential sinks in order to calculate the net GWP of a particular cropping system or set of management practices (Robertson, Paul, and Harwood 2000; Schlesinger 1999). In this way full GHG accounting allows trade-offs to be evaluated properly. Irrigation, for example, can increase soil C sequestration by providing more crop residue, but if irrigation water is pumped from an aquifer it will have associated with it a CO_2 cost from fuel use and from the equilibration of carbonate-saturated groundwater

with surface soils. Similar trade-offs exist for many other management practices such as fertilization, crop choice, and manure management, to name a few.

Full Carbon Cycle Accounting

Over extensive regions of North America, soil C likely is accumulating as a result of increased adoption of conservation tillage methods, decreased use of bare fallow periods, increased efficiency of fertilizers, and increased use of high-yielding crop varieties (Almeras et al. 2000; Buyanovsky and Wagner 1998; Donigian et al. 1997; Eve et al. 2002). Certain of these management changes, however, imply increases in energy use that may, on balance, release more fossil fuel CO_2 to the atmosphere than is captured by soils. For the net effect of a change in practice to be determined, the total emissions of fossil fuel CO_2 associated with agricultural activities must be determined with a reasonable degree of precision. Preliminary estimates of CO_2 emissions from the production and use of agricultural inputs are approximately 28 Tg C/ yr in the United States (Lal et al. 1998).

Emission of CO_2 from the use of fossil fuels in agriculture includes not only fuel consumption on the farm but also emissions associated with production and delivery of fuel, equipment, seed, fertilizer, lime, pesticide, and irrigation water to the farm. For example, N fertilizer manufacture results in approximately 0.82 kg CO_2-C emission/kg of N produced (Follett 2001; West and Marland 2002). Additional CO_2 costs are associated with storage, transport, and application to the field. Izaurralde and colleagues (1998b) and Schlesinger (1999) estimated the total cost at 1.23 kg of CO_2-C released/kg of N applied to agricultural soil. Accounting for the field application costs is difficult, however, because fertilization is crop and field specific and fertilizers may be applied as part of another field operation or as a separate operation. Similar calculations for P and potassium (K) fertilizer, lime applications, irrigation water, pesticides, and seed production were analyzed by West and Marland (2002) and by Follett, Kimble, and Lal (2001). Because the extent to which each of these energy-intensive resources is applied varies by crop, soil condition, climate, and other factors, full C-cycle accounting is complicated. West and Marland (2002) suggested that analyses be differentiated at least by major crop type. Incorporating estimates of fuel use by agricultural machinery, West and Marland (2002) provided estimates of average U.S. CO_2 emissions by tillage practice for corn, soybean, and wheat produc-

tion. Estimates ranged from 79 kg C/ha/yr for no-till soybean (lowest) to 268 kg C/ha/yr for reduced-till corn (highest).

Irrigation is used on some of the most productive land and constitutes a significant, but highly variable, energy and CO_2 cost. Approximately 25% of the irrigated area in the United States (20 million ha) is gravity fed (i.e., through ditches and field furrows), which has little fossil energy cost, whereas approximately 75% of irrigated lands involve pumping. Follett, Kimble, and Lal (2001) estimated average CO_2-C emissions from energy for pumping irrigation to be 200 kg C/ha/yr (range 85 to 330 kg C/ha/yr) whereas West and Marland (2002) arrived at a somewhat higher value of 240 kg C/ha/yr (range 125 to 285 kg C/ha/yr).

Lime dissolution is another source of CO_2 in cropping systems: ground limestone (which is comprised of calcium and magnesium carbonates [$CaCO_3$ and $MgCO_3$]) is applied to many agricultural soils to counteract soil acidity, and as the lime dissolves between application events it will release CO_2 to the atmosphere and HCO_3^- to groundwater. Liming was the single greatest source of GWP in a midwestern alfalfa crop studied by Robertson, Paul, and Harwood (2000).

Net Greenhouse Gas Accounting

Net GHG accounting includes gases other than CO_2 that affect GWP. During a 20-yr period, a molecule of N_2O added to the atmosphere is 275 times more potent than a molecule of CO_2; a molecule of CH_4 is greater by a factor of 62 (IPCC 2001b). During a 100-yr time horizon, GWP values are somewhat different: 23 for CH_4 and 296 for N_2O, reflecting differences in their residence time in the atmosphere (see Chapter 1). Thus small increases in CH_4, and especially N_2O, emissions can offset substantially increases in soil C sequestration. The uncertainty on GWP values is estimated to be approximately ±35% (IPCC 2001b).

Net GHG accounting shows that N management can be as important as C management for GHG mitigation. In five of six different U.S. midwestern cropping systems studied by Robertson, Paul, and Harwood (2000), N_2O flux was the single greatest source of GWP. In their no-till system, N_2O flux offset approximately 60% of the GWP credit that was gained from C sequestration. But N_2O fluxes were similar for both conventional and no-till crops, so that on a relative basis, the GWP of the no-till system was still much less than for conventional management.

Net GHG accounting also can be performed on systems other than cropping systems. Global Warm-

ing Potential analysis of an early successional field such as might be included in the CRP showed a net mitigation potential far higher than that of any cropping system (Robertson, Paul, and Harwood 2000), chiefly because of rapid soil C accumulation, the absence of fertilizer N or other crop inputs, and very low N_2O fluxes (Figure 2.15). In the same study, a late successional forest had a neutral GWP because of stable soil C stores and CH_4 oxidation rates that offset modest N_2O fluxes. In wetland restorations it will be important to know the degree to which the GWP increases because of CH_4 production, attenuating the mitigation gains from new sediment C storage.
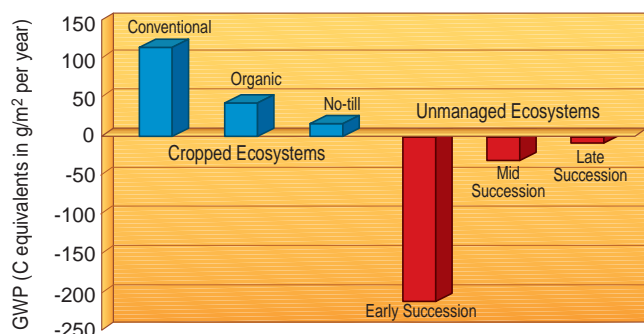

Figure 2.15. Net GHG emissions (given as GWP) for annual crop systems and uncultivated successional ecosystems at the Kellogg Biological Station Long-Term Ecological Research site in southwestern Michigan. The GWP was calculated to include net emission less sequestration for CO_2, CH_4, and N_2O from soil. (Source: Robertson, Paul, and Harwood 2000.)

Especially for livestock operations, GWP analysis should be applied at the whole-farm or landscape scale. The GWP of confined animals in almost all instances will be high as a result of CH_4 and N_2O fluxes associated with the animals themselves or with manure storage. Properly handled, however, manure can contribute to soil C sequestration elsewhere on the farm and also can offset certain fertilizer-N needs. Whole-farm analysis will provide a more complete picture of the GWP costs of these operations and suggest opportunities for mitigation that may not be readily apparent from more narrow analyses.

Biofuels

Expanded use of biofuels would decrease societal use of fossil fuels for energy and chemicals; hence, biofuels can substitute for renewable energy and thus mitigate CO_2 and other GHGs released from fossil sources. Biofuels may be defined as renewable ener-gy sources derived from contemporary biological processes. Biofuels can include dedicated energy crops, agricultural wastes and residues, or energy-rich gases produced by microbial systems processing agricultural wastes. The energy supplied by such systems can be used for power, fuel, or chemical feedstocks, which can supplant current fossil sources and the associated CO_2 emissions of these commodities.

With respect to agricultural production systems, the major opportunities for increased use of biofuels lie in crops and crop residues. These include corn grain produced for conversion to fuel ethanol, oilseed crops to produce biodiesel, cellulosic crops such as trees and grasses, and crop residues such as corn stover and bagasse (sugarcane waste) that can be used to generate energy by either biochemical conversion or combustion. The contributions that biofuels can make to GHG mitigation depend on three factors: (1) whether biofuels can be produced at prices competitive with those of traditional agricultural crops or crop uses (food and fiber); (2) whether energy derived from these crops will be cost competitive with energy from fossil fuels; and (3) whether there exist additional ecological and economic benefits of biofuels that will be factored into the pricing/evaluation equation. Overlying these issues is the efficiency with which biofuels decrease GHGs, which depends on the energy expended in the production, processing, and use of biofuel energy.

To illustrate the interplay of these factors in GHG mitigation, the potential for a dedicated energy crop, switchgrass (*Panicum virgatum*), to achieve wide-scale adoption in biofuels production is examined here (Figure 2.16). The impacts of biofuel production on GHG emissions are compared with those of fossil-based power and fuel systems. Drawing on a similar analysis by McLaughlin and colleagues (2002), two principal areas of influence are examined: (1) displacement of fossil-fuel emissions by using switchgrass as a combustion feedstock and (2) increased soil C sequestration associated with switchgrass production.

Switchgrass is a fast-growing, perennial, warm-season grass selected by the DOE for development as a model bioenergy species (McLaughlin et al. 1999). Resource efficient and widely adaptable to American agricultural systems (Moser and Vogel 1995), switchgrass was an important component of the precolonial tallgrass prairie (Risser et al. 1981). Its native geographical range covers most of the eastern two-thirds of the United States. Switchgrass can be managed and harvested with traditional equipment currently used to harvest 30 million ha of hay annually and thus is compatible with existing farming operations

Figure 2.16. Switchgrass can yield almost twice as much ethanol as corn. A plant geneticist conducts breeding and genetics research on switchgrass to improve its biomass yield and its ability to recycle C as a renewable energy crop. Photo courtesy of B. Hampton, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

throughout much of the United States. A recent national economic analysis (Walsh et al. 1998) indicated that it is the species that will be most competitive as a dedicated energy crop on a national scale. This analysis was based on application of the agricultural sector model POLYSYS (Walsh et al. 1998), which was designed to evaluate competitiveness and market effects of individual agricultural crops within larger integrated agricultural markets. Because POLYSYS models the changes in land use for specific geographic areas, it can determine which types of lands are likely to be converted to switchgrass, as well as the total acreage converted as a function of feedstock price. Both land-use changes and total biofuel production are important factors to consider in determining the GHG mitigation potential of switchgrass as an energy crop.

In Table 2.9, the influence of farmgate feedstock price on adoption of switchgrass for bioenergy produc-

tion is summarized in terms of four preexisting land uses: (1) cropland, (2) CRP, (3) pasture, and (4) other idle land not enrolled in CRP or other government programs. These categories are important because they determine the initial soil C stocks against which any impacts of switchgrass production will be measured. From these analyses, it is evident that significant gains in biofuel production potential occur within a price range between approximately $30 and $41/ tonne. For an $11/tonne increase in feedstock price valuation, the area dedicated to switchgrass increases by more than tenfold (from approximately 1 million ha to 13 million ha). The sensitivity of production levels to feedstock price provides an important framework for calculations of both soil C sequestration per unit of land area and displacement of fossil fuel-derived emissions of GHGs by renewable energy produced on that land.

Land-use types recruited into switchgrass production at a farmgate price of $41/tonne are projected to come primarily from cropland (60%), with approximately 50% of total land converted being located in midwestern states. Approximately 30% (3.8 million ha) would be derived from CRP lands. An important caveat is that this level of CRP participation is based on policy changes that would allow landowners to participate in biofuels production and sales with a stipulated 25% decrease in CRP payments.

To derive the potential GHG decrease from this level of biofuels production, three components are addressed: (1) the potential fuel-biomass and energy production, (2) the net energy gained in the production/conversion life cycle leading to replacement of fossil fuel, and (3) changes in soil C sequestered by switchgrass. The first component of this accounting process is biomass production. An average production of approximately 11 tonnes/ha/yr, a value well supported by switchgrass-production research (McLaughlin et al. 1999), is assumed. Based on a farmgate price of $41.43/tonne (Table 2.9) and the

Table 2.9. Sources and amounts of agricultural land projected to be more profitable to farmers when planted to switchgrass compared with the existing land use. Projections were made with the economic agricultural sector model POLYSYS (Walsh et al. 1998) for 2004 (based on data from Mosier et al. 1998c)

	Millions of hectares in switchgrass for different prices in US$/dry Mg (metric tonne)		
Land-use type	$30.30	$41.30	$52.40
Annual cropland	0.98	8.08	12.07
CRP	<0.01	3.82	5.39
Pasture	0	1.06	2.51
Idle	0	0.51	1.34
Total	0.99	13.47	21.31

assumption that this price is competitive for the end-point energy sector, the production potential of this dedicated energy crop on a national scale would be 150 million tonnes of biomass. At an energy content of 18 gigajoules (GJ = 10^9 joules) per tonne (McLaughlin et al. 1996), this could provide 2.7 X 10^9 GJ (or 2.6 quads) of bioenergy to decrease fossil-derived energy and associated GHG emissions. One quad of energy is equal to 10^{15} British thermal units and represents approximately 1% of the total annual energy consumption of the United States.

Biofuels are not, however, produced, harvested, or collected without incurring energy costs, and these costs must be considered in the derivation of net energy gains. McLaughlin et al. (2002) recently have summarized (Table 2.10) *life cycle analyses* that characterize the complete energy flow and balance in ethanol (Wang, Saricks, and Santini 1999) and in electric power (Hartmann and Kaltshmitt 1999; Mann and Spath 1999) from biofuels. These data compare the decreases in GHGs, expressed as CO_2-equivalent decreases, with the fossil fuels they replace. The analysis shows that the net C emission decrease achieved by production and use of renewable fuels in place of fossil fuels varies widely between biofuels and conversion technologies. Carbon emission decreases for a more energy-intensive annual crop such as corn, when only grain is used to produce ethanol, may decrease GHG by <10% of the mass of the corn used, whereas conversion of cellulosic crops may provide GHG decreases ranging from 23% to a maximum of 93% of the

biomass fuel mass when co-fired with coal. Overall, the data suggest that grass-based systems have the potential to decrease GHG emissions from fossil sources by 0.22 to 0.91 tonne C per tonne of biofuel used.

Conversion of aboveground biomass to energy, however, is only part of the GHG avoidance associated with biofuel production. The replacement of SOC depleted by decades of intensive annual crop production represents another opportunity for effecting net decreases in atmospheric GHGs (CAST 2000; Lal et al. 1998). Perennial grass crops, particularly rapid-growing species such as switchgrass, can restore depleted C pools as a consequence of the high C inputs associated with maintenance of deep, active root systems and lack of soil disturbance (McLaughlin, Bransby, and Parrish 1994). Garten and Wullschleger (2000) recently quantified both annual inputs and annual turnover of SOC pools under switchgrass managed for bioenergy production and estimated annual C gain under switchgrass to be as high as 1.2%/yr for croplands with depleted SOC pools.

Carbon sequestration rates under switchgrass will depend on a variety of factors, including productivity as affected by climate and cultural practices, soil type, and previous land use. For switchgrass systems yielding harvested biomass of 11 tonnes/ha (used in previous calculations), soil C accumulations between 0.8 and 2 tonnes C/ha/yr, during a 30-yr period, have been projected (McLaughlin et al. 2002). The lower rates occur on land with high initial soil C stocks (e.g., pasture and idle land), and the higher rates on land that

Table 2.10. Comparison of reductions in GHG gas emissions per unit of switchgrass, corn grain, or poplar wood used as fuel. Systems compare the production and combustion of fuel ethanol or biomass electrical power with conventional gasoline or hard coal as reference systems (based on McLaughlin et al. 2002)

Fossil-derived energy source	Reduction in GHG emissions with biomass-derived fuel Mg C equivalent per Mg fuel			
	Ethanol source and percentage of fuel mix			
	Corn-E10	Corn-E85	SG-E10	SG-E85
Conventional gasoline[1]	0.074	0.091	0.23	0.24
	Co-firing—% wood with coal			
	5% co-fire	15% co-fire		100% biomass
Coal-fired electric power generation[2]	0.97	1.02		0.52

[1]Calculations based on ethanol life cycle data and analyses performed by Wang, Saricks, and Santini (1999) for comparisons of switchgrass or corn with conventional gasoline. Assumptions: near-term (2005) technology for corn (dry mill) and cellulosic conversion to ethanol at 2.6 gallons/bushel for corn and 80 gallons/ton for switchgrass; fuel efficiency of 25 mpg for gasoline and E-10 and 26.8 mpg for E-85.

[2]Calculations based on data for power production life cycle analyses performed by Mann and Spath (1999). Dedicated biomass power plant uses integrated gasification combined cycle system with an energy conversion efficiency of 37%. Conversion efficiency of 5% and 15% co-firing systems was 31%. Where energy density of switchgrass was substituted with poplar (Mann and Spath 1999) and equal efficiency of energy recovery with original feedstocks was assumed, C offsets using switchgrass for co-firing (data not shown) were approximately 5% lower than for poplar.

has been depleted in soil C through previous management (e.g., annual cropland and recent CRP).

Combining estimates of GHG decreases from fuel replacement (Table 2.10) with the lower estimate of soil C sequestration (0.07 tonne C/ tonne biomass yield) provides a combined estimate of GHG mitigation that might be achieved by the most efficient energy crop system (switchgrass managed as a dedicated energy crop, with whole-plant conversion to energy). Total system-mitigation potential would be in the range of 0.31 to 0.95 tonne C/tonne biofuel when compared with conventional fossil-based gas and power systems. Other biofuel production systems that use only part of the plant (corn grain), harvest a portion of the residue (stover or rice straw), or exclude perennial harvest management strategies very likely will result in much less mitigation per unit of land area. Thus, full implementation of all available land (13.2 million ha) that would produce switchgrass bioenergy at $41.30/Mg could result in 45 to 138 million tonnes C/yr in avoided emissions. At higher prices, land allocation and GHG mitigation would increase (see Table 2.9).

The preceding discussion of switchgrass provides a good example of one type of dedicated energy crop. Other dedicated energy crops, both herbaceous and woody, also are of interest, as are other agricultural sources of biofuels and bioenergy including crop residues, biodiesel, and bioethanol. The best choice of production system will vary geographically and with changes in the supply and demand for different bioenergy products. Ongoing technological improvements are likely to affect further the comparative advantage of different bioenergy production systems. The extent to which bioenergy becomes a significant component of U.S. agriculture will depend on its economic competitiveness with traditional food and fiber crops and with conventional petrochemical feedstocks, as well as on social and political pressures for more renewable energy sources (Paustian et al. 1998).

Similarly, the role of agricultural bioenergy within the overall mix of agricultural mitigation options (such as C sequestration) will vary throughout time and space. In a recent analysis of mitigation alternatives, (McCarl and Schneider 2001) predict that under an emission offset trading system, soil C sequestration in croplands will be a significant low-cost, "early" action. At higher C prices, afforestation and bioenergy production become dominant, because of higher per-ha potentials (but at higher cost), which then supplant an increasing proportion of land previously developed to annual crop production. Thus as energy prices rise, along with desires to decrease fossil C emissions, bioenergy production in agriculture is likely to assume a prominent role.

3 Policy Options and Design

Introduction and Conceptual Framework

Whereas a good understanding exists of the economic trade-offs involved in controlling greenhouse gas (GHG) emissions from major energy and industrial sources (e.g., Falk and Mendelsohn 1993; Nordhaus 1991), relatively little analysis has been devoted to the economics of carbon (C) sequestration in agricultural soils or to the economic trade-offs involved in controlling agricultural GHG emissions (Feng, Zhao, and Kling 2000; McCarl and Schneider 2000a,b; Pautsch et al. 2001). These policy issues are particularly relevant now as the United States and several state governments formulate policies dealing with GHGs. Although the current U.S. policy is based on voluntary measures for GHG mitigation and the U.S. government has decided not to participate in the Kyoto Protocol, it is likely that U.S. policies will continue to evolve throughout time and may include mandatory emission reduction targets and participation in international agreements at some point in the future. In this chapter the authors examine a range of possible policy options and specific issues relating to policy design and implementation.

There are at least three scenarios under which GHG programs targeted at agriculture could be established. First, international agreements, such as the Kyoto Protocol, could allow terrestrial sinks, both forest and agricultural, to count toward a country's international commitment to decrease GHGs. Such a scenario has the potential to create a major role for agricultural C sequestration, including income generation associated with altered farming and land-use practices. Second, even in the absence of credit for agricultural sinks in the international community, the United States might adopt policies encouraging soil C sequestration, because soil C is an indicator of long-term soil productivity and likely is correlated with many beneficial environmental attributes. Implementation of such a domestic policy also may demonstrate the feasibility of managing soil C as a means of decreasing atmospheric concentrations. Depend-

ing on how this policy is implemented, it may have significant income-generating potential for agricultural sources. And third, voluntary arrangements whereby emitters buy offsetting credits from farmers or their representatives may arise if consumers are willing to pay extra for climate change-neutral products. Unless international or national policies generate official credit for C sinks, however, C sequestration probably will not be a major determinant of farming practice or income.

The proper role for government depends critically on which, if any, of these scenarios comes to pass. For example, there is little role, beyond standard market oversight, for government involvement if the third scenario is adopted. Likewise, if international credit for agricultural sinks is not recognized, a GHG marketable credits program is less likely to be worth implementing, inasmuch as trading from energy and other sectors is likely to play a key role in such a market. Nonetheless, many issues must be addressed before any government policy concerning GHG control from agricultural sources can be implemented effectively.

This chapter identifies and discusses certain of these critical issues. Although focusing on the scenario in which agricultural sinks are fully integrated into international GHG treaties, the chapter discusses possibilities related to all three scenarios. Specifically, it identifies and discusses those issues that should be resolved satisfactorily *before* international implementation of agricultural sink policies. Furthermore, agricultural sinks and emission reductions must be considered within the broader context of GHG emissions and abatement strategies for all sectors of the economy. Thus, there is a brief outline of a conceptual framework to guide the subsequent discussion in the chapter.

First, consider the total stock of GHGs in the atmosphere in any given year. The change in the stock of the GHGs can be viewed in terms of *net emissions*— that is, gross emissions minus sinks. These can be separated as emissions and sinks directly related to land-use activities and those because of other human activities. Practices affecting net emissions from soil

broadly comprise land use, crop management, and livestock management, including manure handling. For agriculture, gross emissions include all activities that increase GHGs, such as cultivating soil, burning fuel, and using fertilizer and pesticides. Net emissions may be negative if sufficient C sequestration occurs (i.e., negative emissions mean a net uptake from the atmosphere to the soil).

Emissions and sinks associated with land use vary geographically, depending on climate, soil properties, and the kind of land-use and management activities being practiced. For accounting purposes, one can think in terms of spatial "patches" or land areas representing a particular climate and soil type and with a particular set of management activities that determine the net emission rates. The area and scale of the geographic units considered (e.g., fields, countries, or ecoregions) will vary depending on the objectives and scale of the accounting system. Aggregate net emissions from agriculture for the entire country (or other geographic area) can be estimated as the net GHG emission per hectare (ha) for each different soil C climate-management combination, multiplied by the land area involved. Uncertainty regarding the relationship among activities, environmental variables, and emissions also must be taken into account. Finally, potential damages from GHGs through climate change depend on the accumulated total stock of GHGs and how the stock changes throughout time. Although simple, this outline of GHG emissions identifies several key policy issues that must be resolved before a successful GHG mitigation policy can work in agriculture.

The next section addresses four key issues of policy operation and design that will determine the effectiveness and acceptance of a mitigation program, including accounting, observability, timing, and adoption. This discussion will be followed by a description of the details of institutional structure. The chapter concludes with a summary of how mitigation programs fit with other domestic and international agricultural and environmental policies.

Issues of Operation and Design

Acceptance of agricultural GHG mitigation programs by the international community will increase if the programs address four key concerns. First, because damages from climate change depend on the combined effects of all GHGs, policies will need to account fully for all changes in C emissions in a country, as well as other GHGs (e.g., nitrous oxide [N_2O]

and methane [CH_4]). This requirement will be referred to as *full greenhouse gas accounting*. Second, it will be necessary to quantify all components of net emissions as well as sequestration with an acceptable degree of accuracy and to characterize reasonably the uncertainty associated with emission estimates. Thus, observability is the second issue needing to be addressed. Third, there is an obvious and important time element with respect to GHG stocks, and an understanding of how timing affects value and use of C sinks in agriculture is crucial, especially because agricultural sinks may not be permanent. And fourth, an effective GHG mitigation strategy in agriculture must change the behavior of farmers and other agriculturalists in regard to the adoption of improved conservation and land-use practices. Acceptance of agricultural sinks will require the design of policies that can induce such change convincingly.

Full Land and Greenhouse Gas Accounting

The most effective GHG mitigation policies will be those that include full accounting for GHG emissions in two respects: across GHGs and across locations. In regard to full accounting across gases, agricultural actions may influence more than one GHG. For example, increasing biomass production through fertilizer use may increase C sequestration but also may increase N_2O emission. Ignoring the negative effect of fertilizer use will lead to underestimation of net emissions from a site. Likewise, adoption of conservation tillage can increase soil C and also decrease fuel use, both of which are positive from the standpoint of GHG mitigation.

The second full accounting issue is whether all land use will be accounted for in meeting a country's obligations or whether only a subset of the land participating in a C sequestration program will be included. Clearly, in a national or global accounting system in which the national government is responsible for meeting a target, all land whose net emission is nonzero should be included. Lack of total accounting, or a partial program, will raise accountability issues. For example, it will be difficult for a country to claim GHG credit from a program that credits farmers adopting conservation tillage but does not debit farmers converting grassland to cropland. Total land and GHG accounting may or may not be required for implementation of a domestic agricultural policy or a voluntary policy. If the policy performance measure is not aggregate decrease in GHGs, then partial accounting may be acceptable. For example, a volun-

tary program may be based on the amount of C sequestered by participants, without regard to non-participants.

Full land and GHG accounting is a challenge for implementers of GHG programs targeted at land management. Full land accounting means that any parcel of land where net emissions are not zero must be considered. Information about activities and climatic and environmental conditions must be obtained and reliable estimation procedures established to determine how these variables affect all GHGs. If the GHG program involves incentives, then estimates of net emissions must be translated into appropriate subsidies, taxes, or any other policy variables for each parcel of land. Given the current technology, it is infeasible to consider every parcel of land separately. Rather, the intensity of monitoring and targeting will differ across locations, with the greatest emphasis on those regions and land uses where net emissions are significant. Any such heterogeneous targeting program will need to be justified quantitatively.

Observability

A second substantive issue in developing policy options concerns the ability to observe and to credit sequestered C and decreased GHG emissions appropriately. Land-based emissions of GHGs are considered nonpoint source pollutants, because individual levels of emissions are difficult to observe. Policies for controlling and monitoring nonpoint source pollution with agricultural sources have received considerable attention in regard to water quality (e.g., Braden and Segerson 1993; Griffin and Bromley 1982; Herriges, Govindasamy, and Shogren 1994; Millock, Sunding, and Zilberman 2000; Segerson 1988; Tomasi, Segerson, and Braden 1994). In most of these studies, aggregate pollution levels are observable, but individual contributions are not. Control programs thus could be based on knowledge of aggregate pollution and observable individual actions. In theory, the observability of aggregate pollution allows policymakers to modify the level of individual actions until desired aggregate pollution reduction is achieved (Segerson 1988). Alternatively, direct control, or taxation, of production inputs (e.g., fertilizer) can yield efficient pollution reductions (Griffin and Bromley 1982; Holterman 1976; Shortle and Dunn 1986).

Control of GHGs may be more difficult to achieve than control of other nonpoint pollutants (such as nutrient or pesticide leaching) because of difficulty in observing and measuring aggregate GHG emissions

from land sources. For water pollutants, for example, measurements in stream flow and groundwater can provide an integrated estimate of pollutants released within a specific geographic area. Analogous measurements for GHG emissions (e.g., using micrometeorological techniques, flux chambers, or repeated sampling of biomass and soils [see Chapter 4]) are generally more difficult and more expensive.

An alternative to direct measurement of aggregate emissions is to estimate emissions based on observation of the land-use and management practices that affect emissions. Acceptance of a soil-GHG program will depend heavily on the reliability of such estimates, including the quantification of uncertainty as a function of observed actions (i.e., management practices) and environmental variables that influence the rate and amount of C storage and GHG emissions. Accurate assessment will require reliable estimates of the rate and capacity for soil C storage across a range of conditions. For example, a program may pay a farmer to adopt conservation tillage (Figure 3.1), which is expected to result in a certain level of C sequestration depending on soil type, condition, and climate. The amount of residue at planting time provides an indicator of the type of tillage. This information and information about crop selection, irrigation, and fertilization are other observable actions. The amount of payment (or credit) given the farmer could be based on the translation of these observable actions, interacting with environmental variables, into emission decreases.

For agricultural mitigation to be successful, it will be necessary that the estimates of C sequestration and other GHG emission reductions are deemed reliable enough to serve as the basis for payments and prac-

Figure 3.1. No-till planting on terraces in Montgomery County, Iowa. Photo courtesy of T. McCabe, U.S. Department of Agriculture.

ticable enough so that the costs of including agricultural sinks do not exceed the benefits.

Timing Issues and the Value of Sinks

Despite the potential benefits of C sinks and the fact that the implementation guidelines of the Kyoto Protocol permit their use to offset GHG emissions, there are questions about how effective these sinks can be and whether they can contribute meaningfully to decreases in the buildup of GHGs. After all, as noted in Chapter 2, terrestrial sinks eventually will reach a new equilibrium under a particular land-use and management regime, at which point no more carbon is sequestered unless further management changes are implemented. In addition, unlike permanent abatement measures, C sequestration practices can be reversed, possibly releasing previously stored C. For example, trees can be cut and farming practices such as no-till can be reversed easily. Given the temporary nature and finite holding capacity, what is the value of temporary C storage? This issue will be discussed from two perspectives: pollution damage control and abatement cost savings. A related question that is addressed is how to consider the value of a unit of C sequestered today in relation to a unit sequestered in the future.

Pollution Damage Control

Misconceptions about the value of C sinks are common. One is that temporary storage has value mainly because the public discounts future pollution damage. This argument proceeds by noting that when the pollutant is stored now, current pollution damage decreases, but when the pollutant is released from storage, future pollution damage increases by the same amount. If the damage increase in the future equals the current decrease, storage has no value unless the future damage increase is discounted. Figure 3.2 illustrates this argument: Area A measures current damage decrease; area B, future damage increase. With a positive discount rate, the present value of the increase in future damage is less than the decrease in current damage, so the net value is increasing in the discount rate.

The logic associated with this argument is not flawed inherently, for it would be quite accurate if the damages from GHGs came about primarily from the flow of GHGs (i.e., from the amount produced each year, not from the accumulated amount). Excess GHGs in the atmosphere constitute a stock pollutant,

Figure 3.2.   Intertemporal damage trade-offs from temporary storage of a flow pollutant. (Source: B. Babcock, Ames, Iowa.)

however, and so damage depends on the total stock of GHGs accumulated throughout time. Thus, with a stock pollutant, temporary storage has value even when the social discount rate is zero. The reason for this difference is that a sink decreases the accumulated stock for the period during which pollutants are stored, thereby decreasing damage. Thus, as depicted in Figure 3.3, there clearly is a gain during the period in which the sink is in operation, with no associated damage beyond what would have been present in the absence of the sink when the period ends. Discounting actually decreases the sink's value for a stock pollutant such as C. The primary service of C sinks, therefore, is that when used they can decrease atmospheric CO_2 level for the duration of storage. Even when stored C is released, no more damage is done than in the no-storage scenario. The value of this service simply is decreased damage, from the lowered C level, for the duration of storage.
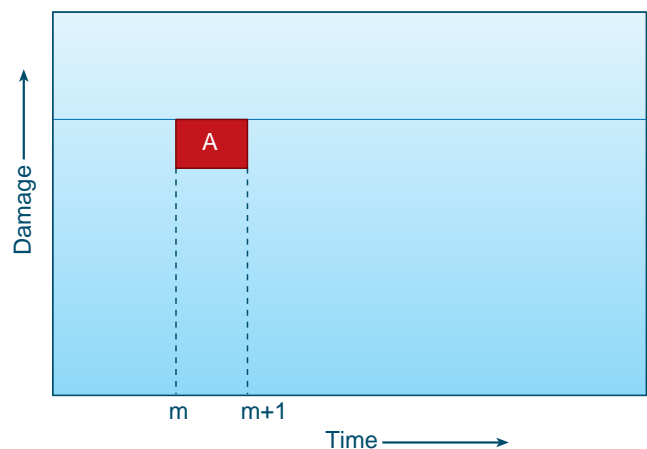

Figure 3.3.   Damage reduction from temporary storage of a stock pollutant. (Source: B. Babcock, Ames, Iowa.)

This same logic can be used to compare the value of a *permanent sink* with the value of a temporary sink. Specifically, the value of a permanent sink is the accumulated value of the sink during all time periods.

Abatement Cost Savings

So far, the value of C sinks in terms of the decreased damage from atmospheric GHG levels has been discussed. Another perspective in assessing the value of sinks is that, at least in the short run, they may exhibit cost advantages over permanent decreases for certain levels of decreased GHG. If the current cost of emission reductions rises rapidly with the size of the reduction, then the use of storage may decrease the total cost of decreasing GHGs. According to this argument, low-cost C sinks may be valuable as permanent tools.

Additionally, from a more dynamic perspective, sinks may provide valuable temporary reductions in atmospheric GHG buildup if technological progress lowers the future costs of emission reductions by industry. This argument often is referred to as "buying time." That is, temporary storage allows time for emitting industries to develop innovative methods of decreasing emissions at lower costs (Figure 3.4). The buying time argument is relevant only if new technologies will be invented and adopted by polluting industries. One concern is that use of C sinks will decrease industry incentives to develop and to adopt newer technologies, although the degree to which this effect may be a factor is difficult to assess.

The net value of agricultural sinks depends on their costs and their benefits. The damages avoided by decreasing GHG emissions by one unit are the same,
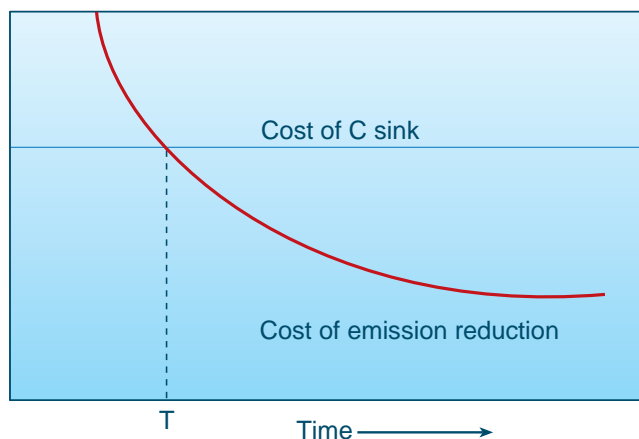
regardless of the source (e.g., agricultural sink, energy conservation, or decreased CH_4 emissions), although the potential number of units from these sources may differ significantly. Thus, the cost per unit of decreasing emissions may be quite different, which makes it desirable to choose the method that results in the most damage avoided per unit of cost.

Value of Greenhouse Gases Sequestered at Different Times

Any GHG policy will need to recognize explicitly that the value of a unit of decreased GHGs can vary dramatically during different time periods (Richards 1997). Thus, a policy that induces adoption of a technology yielding a decrease of 500 million tonnes of CO_2 in 20 years (yr) should not be treated the same as one yielding a decrease of 500 million tonnes today. Not only is it appropriate to discount the costs and the benefits of alternative policies, but also the value of decreasing GHG emissions will change throughout time, depending on the stock of GHGs and the cost of technologies available to combat their production.

The nonpermanence of sinks has been at the center of the debate about incorporating soil C sequestration into the Kyoto Protocol (Chomitz 1998). Numerous studies have investigated the costs of forestry or agricultural sinks empirically (Marland, McCarl, and Schneider 2001; Mitchell et al. 1999; Parks and Hardie 1995; Pautsch et al. 2001; Plantinga, Mauldin, and Miller 1999; Sohngen and Sedjo 2000; Stavins 1999), but discussion regarding how to make sequestration and abatement projects commensurable has been limited. Fearnside (1997) suggested a *tonne-year accounting method*, which also is recommended by Chomitz (1998) in a report to the Carbon Offset Unit of the World Bank. In this accounting method, physical C is discounted so that one unit of decreased C in the future counts less than a one-unit decrease today.

Theoretically, the rate at which CO_2 or other GHGs should be viewed as equivalent across time depends on the present value of the damage from CO_2 emissions. In general, this rate will not be constant. The rate at which a unit of C, sequestered for a finite period of time, should be traded against a unit of C sequestered in perpetuity can be calculated. Although this calculation is conceptually straightforward, this intertemporal trading rate will be difficult to implement in practice. And it is doubtful that all parties to a national or an international agreement will be able to agree easily on intertemporal trading rates.

Figure 3.4.   Effects of time on C abatement costs. (Source: B. Babcock, Ames, Iowa.)

Adoption

Farmers will participate in agricultural soils-sink and/or emission-reduction programs only if sufficient incentives are provided. It is crucial, therefore, for policymakers to understand what motivates the decision making of individual farmers.

Generally, farmers respond to increased profit possibilities. In GHG transactions, profit consists of market revenue less production cost plus net payment from a GHG contract. When a decision affects the future, farmers may evaluate profit streams discounted into current value. Discounting is of special relevance to soil sequestration policymakers because it entails soil C stock management throughout time.

A quantitative understanding of farmers' likely responses to financial incentives to adopt GHG mitigation practices is required. Policymakers and scientists are disappointed sometimes to realize that farmers do not switch completely to practices that evidently are relatively profitable. A wide range of literature explains the factors affecting adoption and diffusion of new technologies (e.g., Sunding and Zilberman 2001). First, it is useful to distinguish between adoption, which is an individual's decision to use the new technology, and diffusion, which is aggregate adoption and the measure (in percentage) of land area or producers adopting the new technology. The first study of adoption conducted by a sociologist (Rogers 1962) recognized that diffusion is an *S*-shaped function of time. There is an early period with a low diffusion rate, followed by a period of take-off, and finally a period of saturation. New technologies rarely are diffused fully. In certain instances, diffusion rate may decline, and such is the case with diffusion of conservation tillage in the United States. After 1995, diffusion rates of this technology have declined, mainly because of lack of sufficient incentives.

The *S*-shaped diffusion curve has two explanations. One approach, introduced by Griliches (1957), views diffusion as a process of imitation whereby farmers observe others' behaviors, and for a while, a high rate of diffusion accelerates imitation and thus the diffusion process. The alternative approach, called the *threshold approach*, views farmers as heterogeneous, and at each moment only the subgroup of farmers for whom the technology is profitable will adopt. Throughout time, the relative profitability of the new technology improves as a result of decreased fixed capital requirements (because of the process of "learning by doing" by producers of equipment) and improved use of the new technology (because of the process of "learning by using" by farmers). The two approaches explaining diffusion are complementary. Farmers learn from and imitate one another, but they are a heterogeneous population, and the optimal timing of adoption depends on individual and region.

Studies of adoption by individual farmers recognize the importance of uncertainty and risk considerations. The performances of new technologies are uncertain to farmers, who view adoption as risky. By studying the results of demonstration plots and experimenting on a small scale themselves, farmers are able to gauge risk better and thus to modify their responses. Just and Zilberman (1983) argue that adoption decisions have two dimensions: first, the decision regarding whether to adopt a new technology; second, the decision regarding to what extent they should adopt. Certain farmers may diversify land management strategies by adopting conservation tillage on only a portion of their lands. They may choose this strategy if they perceive differences between the average profitability and the riskiness of conservation tillage compared with intensive tillage and if no technology dominates the others in terms of average profits and risks.

The riskier a technology is perceived as being, the fewer the adopters and the lower the rate of adoption on each farm. Introduction of various policies can increase adoption rates (e.g., payment for soil C sequestration will increase the adoption rate by increasing profitability of conservation tillage practices). If there are, in addition, programs to protect farmers from significant losses resulting from their use of the technology, adoption levels will rise further. In certain instances, the main obstacle to adoption is the need to invest in new equipment. Here again, programs helping to finance machinery or other inputs facilitating transition to the new technology will lead to increased adoption. Farmers may not know how to use the new technology. In this regard, effective extension programs and reliable extension staff who can educate farmers on the use of alternative soil management practices and assist farmers as they deal with unexpected situations will decrease the uncertainties surrounding new technologies and so contribute to increased adoption (Figure 3.5).

In most instances, extension staff are few in number, and equipment and input dealers and their staffs provide some of the day-to-day support in the introduction of new technologies. Manufacturers of equipment may increase adoption by providing discounts and product warranties. Regarding soil management practices, dealers must be educated in the use of new technologies, yet in certain instances dealers may not

Figure 3.5. A cotton specialist checks rotation plots near Huntsville, Alabama to see which tillage methods and cover crops work best. Photo courtesy of B. Nichols, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

be supportive of adoption of conservation tillage practices because such practices do not serve their business interests. In these instances, extension and the public sector may need to make additional efforts in terms of technological support and education. Further, it may be useful to encourage cooperation among farmers and to build up mutual-support and knowledge-exchange systems capable of decreasing uncertainty about the new technologies, thereby encouraging adoption.

There is ample evidence that financial incentives for new technologies will lead to their adoption. Similarly, insurance, warranties, and other incentives that decrease the risks of new technologies have been shown empirically to be effective in increasing adoption in many situations. Birkhaeuser, Evenson, and Feder (1991) surveyed a large body of literature documenting the effectiveness of extension efforts to usher in new technologies.

Socioeconomic factors also are very important to adoption. Relatively young, educated farmers have a greater tendency to adopt new technologies. In certain agricultural crops in which production and marketing contracts are prevalent (e.g., livestock, fruits, and vegetables), the product buyer may influence grower production practices. If marketers of a certain product consider production with C sequestering technologies an attractive feature that may add value to the final product, these marketers may pay growers to adopt such technologies. Thus, one way to increase adoption of soil C sequestration activities is to label them *green practices*.

Alternative Policy Designs

Many alternative program designs for decreasing GHG emissions by means of agricultural sinks exist. There has been much discussion of "carbon trading" and of the buying and selling of C permits,[4] and various ways in which actual implementation of such trading schemes could be accomplished are recognized. At present, trading arrangements based on voluntary GHG reductions, such as the Chicago Climate Exchange, have begun operating in the United States. Internationally, several countries are in the process of developing C and emission offset trading systems, and the World Bank has initiated a "Carbon Bank" to promote C trading under the Kyoto Clean Development Mechanism. As noted, appropriate design will vary according to the program under which sinks are established. Specifically, significant government-based ventures, such as the design of an official GHG trading program, are less likely to come to pass unless there is an international accord allowing agricultural sinks to count toward treaty commitments to decrease GHG emissions. In contrast, an international accord allowing agricultural sinks to count toward a country's GHG decreases will require adoption and implementation of a national policy that makes sinks a meaningful strategy.

For an effective program to be implemented, many details concerning the mechanisms for implementing an agricultural sink-GHG policy must be worked out. The important dimensions of program design are (1) definition of the commodity to be regulated or targeted; (2) organizational structure of the program; (3) identification and control of GHG "leakage" (unintended GHG emissions resulting from adoption of policies intended to control GHG emissions); (4) enumeration of payment rules, including time frame and reversibility issues; and (5) monitoring and verification that details of program contracts are being fulfilled.

Definition of the Commodity

Farming can affect GHG balance in several ways, including (1) release or sequestration of soil C, (2) use

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[4] See Baumol and Oates (1988), Cropper and Oates (1992), and Tietenberg (1984) for useful reviews of the theory and implementation of marketable permit systems.

of fossil fuels in agricultural production, (3) release of N_2O or other gases in production, and (4) use of fossil fuels in the manufacture of inputs such as fertilizer (McCarl and Schneider 2000a). The exact definition of the commodity at the farm level likely will depend on enumeration of GHG accounting in other economic sectors. As a rule, it is most useful to account for GHG changes whenever they actually occur. In such instances, the farmer is credited or debited for items 1, 2, and 3, but not for item 4.

Another definitional issue is the fact that different kinds of GHG decreases may have different values in a program. For example, the C equivalent of 100 gallons of fuel not used in production of a crop in a year is more valuable than the same amount of C sequestered in the soil. The reason is that avoidance of fuel use is a permanent decrease in the GHG stock, whereas sequestered C may be released in the future. Certain programs may target only a single GHG. Other programs may target according to location. For example, North Carolina might target manure lagoons to decrease CH_4 or N_2O emissions, whereas Iowa might target soil C sequestration. If programs target different emission sources specifically rather than measuring everything in permanent C equivalents, it will be necessary to agree on the rate at which these different sources will be credited toward meeting a nation's requirements regarding the decrease of GHGs.

Organizational Structure

Successful implementation of a GHG mitigation program in agriculture depends crucially on establishing a workable organizational structure. Organizational structure defines the organizations participating in the program and the form of their interactions. Depending on the scenario adopted, different organizational structures will be used. Here, the major structures that might be adopted are outlined, including a form of marketable permits (a trading scheme). In all instances, farmers and their agents will be potential participants. In most instances, there also is some role for government. Certain programs involve traders and brokers as well as nonagricultural buyers. All programs also involve monitoring and recording agencies.

Scenario 1: International Agreement

With international agreements, the government will, because of accounting and other international obligations, play a central role in designing and implementing sequestration programs. The government is the entity signing agreements and is responsible for fulfilling their obligations. A critical objective of government policy will be to provide assurance of system integrity to the international community. The federal government may adopt one of two systems: one placing the government in the central role; the other relying more heavily on market forces (marketable permits or trading).

Central bidding (tax and subsidy system). In this instance, the government owns all permits, and GHG emitters such as energy companies must bid for the right to emit, or pay a tax. Sequesterers, such as farmers who have adopted conservation tillage practices, bid for the right to sell permits, or receive a subsidy. A feasible formulation of this program is that emitters would buy permits only if they emitted more GHGs than have been emitted historically or are allowed legally. Likewise, those who sequestered could sell credits only if they sequestered above a baseline level. Sequestering farmers could bid for the right to sell permits to the government. Buyers could bid directly to the government for the right to emit.

Tradable permits. In this framework, the government would hand out to emitters permits equal to the total amount of GHG emissions allowed the country by treaty agreements. These permits might be based on historical emission levels or on some fraction thereof. Alternatively, they could be allocated randomly, or a bidding system could be set up to acquire them. Permits could be good for a single year or multiple years, again depending on the international treaty. Once permits have been allocated, they are fully tradable. This means that anyone owning a permit can sell it to anyone else who can use it to cover their own emissions. Of course, once a permit is sold, it cannot be used to cover the GHG emissions of the original owner; hence, sellers must decrease their emissions by the amount specified in the permits sold. In addition to the direct trading of permits, emitters could contract with farmers or their agents to purchase GHG reductions to cover emissions. Thus, if an energy company could demonstrate to the government that it had contracted with a group of farmers to sequester 100,000 tonnes of C, it would be allowed to emit 100,000 tonnes beyond the amount for which it held permits. Sequestration effectively generates additional permits, although it is important to recognize that total allowable emissions do not increase as a result because permits are generated by agricultural

activities decreasing GHG levels equal to the number of permits created.

In the United States, there is ample precedence for markets of this sort to be developed and administered by the government (Hahn 1989; Stavins 1999). The flagship example is the sulphur dioxide permit-trading market authorized in the 1990 Clean Air Act Amendments (Ellerman et al. 2000). Numerous examples of transferable rights exist in the water economics literature. For example, California recently introduced an electronic water exchange called Waterlink, whereby farmers can lease water rights to others. This followed the water bank system, in which the State of California served as an intermediary through which farmers with surplus water sold water to drought-affected farmers (Olmstead et al. 1997). Whereas in California water rights can be leased only, in New Mexico and in Chile rights can be sold (Bauer 1997).

In principle, the outcomes of a market-type trading program and the central bidding (tax/subsidy) scheme could be identical. In either instance, ensuring the integrity of the system will require establishing two entities: (1) an independent monitoring organization that verifies actions taken by farmers and other market participants and quantifies these actions in terms of tradable commodities, and (2) a centralized registry in which all transactions are recorded and stored. Monitoring and registering organizations likely will need to be accredited by an international body.

Because of the great number of farmers who will need to participate in a trading program, specialized aggregators are likely to emerge to represent farmer interests. Aggregators in this market may include existing farm organizations, farmer-owned cooperatives, or large producers controlling many farms. By contracting with independent verification and monitoring organizations and by buying insurance that indemnifies buyers if the farmer-end of the contract is not fulfilled, aggregators can increase the value of permits sold. California water districts and water-users' associations around the world play the role of aggregators in water markets.

Scenario 2: Government Agricultural Policy in the Absence of an International Accord

In the absence of participating in an international agreement that includes agricultural sinks as a source of C sequestration, the U.S. government still might adopt a program to encourage reduction in GHGs, as discussed previously. The basic organizational structure might be similar to the central bidding program, in which aggregators representing farmers bid for the right to sell sequestered C. The only difference is that government would not resell the sequestered C to emitters, because emitters of GHGs would have no reason to hold sequestered C credits. A program of this sort could be incorporated as a targeting criteria for the Conservation Reserve Program (CRP), as discussed by Babcock and colleagues (1996) and Wu, Zilberman, and Babcock (2001). Especially if it is to demonstrate the feasibility of using soil sinks to mitigate GHG buildup, a program of this type will require a high-quality monitoring and recording structure.

Scenario 3: Voluntary Mechanism

If the only market for soil GHG sinks exists as an outcome of the desire of companies to be perceived as "green" by consumers, there will be quite a different set of institutions and roles for government than in either of the previous two scenarios. The objective of such an organizational structure will be to provide consumers with the confidence that they are buying products contributing to the mitigation of climate change. The role of the government will be similar to its role in any other market, including the enforcement of property rights and transparent trading rules. For example, C Quest, a brokerage organization attempting to develop private markets for GHG mitigation, has shown that the key elements of such a system are farmers pooling mitigation credits through an aggregator who may work with a broker to sell credits through an exchange to buyers wishing to offset emissions of their consumer or other products. Additionally, independent monitoring and registry are critical steps toward gaining consumer confidence. An insurance policy offsetting mitigation failure also would increase the integrity and feasibility of such a program.

Leakage

The effectiveness of sequestration can be undermined if what has been called "leakage" in the Kyoto context, or "slippage" in the agricultural context, occurs. That is, actions to enhance C storage may alter current or anticipated production levels of different products, which may, in turn, have significant price effects that can induce an increase in emitting activities elsewhere. Leakage means that the net amount of project-based GHG mitigation is decreased because

of expanded GHG emissions elsewhere.

Leakage can manifest itself in a number of ways. Consider an example involving forest C sequestration. Suppose, in the name of C sequestration, the harvest of timber in a significant region in the Pacific Northwest is restricted. Such an action would decrease the amount of saw logs entering the market. Such a project would not, however, change the market demand for saw logs, so consumer prices would rise, and consumers would look for additional logs from other sources. Price effects may be large. For example, Berck and Bentley (1997) estimate that the 1978 expansion of Redwoods National Park in California increased the market price of redwood by 26%, which increased the value of the remaining private stands of redwood by $583 million. Restrictions of timber harvests of Pacific Northwest public lands in the 1990s resulted in accelerated harvest rates in Canada and the southern United States. Wear and Murray (2001) and Murray and McCarl (2001) derived leakage estimates of approximately 85%.

Offsets such as in this example also could occur when pursuing soil C sequestration programs. For example, U.S. policy could convert a significant amount of cropland into grasslands in one region to increase C sequestration. In turn, that conversion would lower crop production and raise prices, which would cause producers in other regions nationally or internationally to meet any associated market shortage, possibly by converting grasslands, forestlands, or wetlands to croplands. Wu (2000) found in the instance of the U.S. CRP that approximately 20% of the acres enrolled in the program were replaced by other acreage, with 9 to 14% of the environmental benefits offset. International effects also may occur. In a modeling study, Lee et al. (2001) show that unilateral implementation of the Kyoto Protocol in the United States would lead to a decline in U.S. exports and an increase in production in the rest of the world. Wu, Zilberman, and Babcock (2001) show how enrollment rules for conservation programs can be enhanced in the presence of leakage and demonstrate that standard cost-benefit criteria for project selection may yield suboptimal environmental outcomes.

Leakage implies that programs need to be evaluated under a broad national and international accounting scheme so that leakage is estimated and the program achieves cost-effective global GHG decreases. Project evaluations need not look myopically at the project only but also at major competitive regions that may be affected when project activities are imposed so that net GHG estimates can be formed.

Payment Rules

Writers of an effective GHG-agricultural mitigation policy must find ways to construct contracts that not only encourage farmer participation but also obtain desired results in terms of decreasing CO_2 and other GHG emissions. This section provides details of contract design incorporating both these features, with an emphasis on (1) choice of level and timing of the benchmark emission, (2) choice of long-term instead of yearly contracts, and (3) impact of reversibility.

Benchmark Emission Level and Timing

Any GHG mitigation program in agriculture must establish an emission level, or benchmark, to serve as a basis for determining payment. For example, the Kyoto Protocol specifies 1990 emissions as a benchmark and establishes targets as percentage decreases from this level. Put in the form of an equation, let e_0 denote the level of net emissions in the benchmark period. If ρ is the per-unit payment, then a farmer in an evaluation or reward year t with emission e_t will receive $(e_0 - e_t)\rho$. Thus, farmers would prefer a higher e_0 and a lower e_t.

One strategy for encouraging farmers to meet mitigation targets could involve establishing a benchmark for each farmer and then rewarding or penalizing accordingly. Any benchmark should be based on past behavior and should be unaffected by current or future behavior (i.e., e_0 should be fixed). In this way, farmers will be prevented from taking actions today to manipulate future payments to them. Under C contracts without a fixed baseline, farmers might find it profitable to deplete soil C levels today to enhance their abilities to sequester more in the future.

Long-Term Versus Yearly Contracts

A related issue is that using a far-in-the-future reward year may give an individual the incentive to deplete soil C today to increase the ability to sequester C in the future (Antle and Mooney 2002). To avoid this moral hazard problem, a reward year as close to the present as possible must be established.

Once a baseline is agreed on, the optimal length of contracts covering GHG sinks must be considered. One approach to designing a GHG contract is to follow the example of CRP contracts, which specify a farmer's future actions in exchange for a fixed annual payment. Because C sequestration occurs during a several-year period, and the time horizon of buyers

is quite long, long-term contracts may be appropriate. But long-term fixed payment contracts have several drawbacks arising from uncertainty about the future value of C sequestration and GHG emission decreases, about the measurement of GHG flux and C sequestered, and about the future behaviors of both parties.

A long-term fixed contract essentially is a series of forward contracts in which the farmer agrees to perform certain actions in exchange for a fixed price. The farmer runs the risk that an action's value will increase in the future, either because demand for mitigation increases or because the quantity of C that can be sequestered increases as a result of new production technologies. Consequently, long-term contracts generally require greater compensation than short-term contracts do. Although running the opposite risk, buyers may find long-term contracts, which enhance the ability to plan far ahead, desirable. This advantage may be especially valuable to energy companies and other industrial sources that might be the primary buyers of agricultural sink credits.

An alternative approach is to pay annually, based on actions and changes in GHG emissions. This approach decreases risk and allows adjustment in behavior to be reflected immediately in payments. This increased flexibility may encourage farmers to participate in a program, much as short-term development easements are much more palatable to them than long-term or perpetual easements. But short-term contracts—and the increases in flexibility—will generate a lower annual price.

Impact of Reversibility

A third issue that must be dealt with is the impact of reversibility. For example, reverting back to intensive tillage after having used no-till for several years can lead to the loss of sequestered soil C. Farmers may choose to switch tillage practice in response to changing weather conditions, input costs, pest problems, or other factors affecting profits. When such changes occur, farmers will have decreased the potential value of previous practices. Likewise, change in crop price could lead to a change in crop grown and thus to a change in the amount of C sequestered, even if production practices are not changed. Contracts should provide for such situations and incorporate appropriate responses.

Feng, Zhao, and Kling (2000) consider three forms of contracts that differ in terms of length, all of which deal with the reversibility issue in one form or another. The researchers refer to these three contracts as the *pay-as-you-go* (PAYG), the *variable length con-*

tract (VLC), and the *carbon annuity account* (CAA) systems. For all systems, a well-functioning external C market setting the price for permanent decrease in C emissions (as CO_2) is assumed.

In a PAYG system, owners of sinks sell (and possibly repurchase) emission credits based on the permanent decrease of C emissions (i.e., through C sequestration). Thus, in the first year a farmer adopting conservation tillage practices on 100 ha may earn 50 permanent C reduction credits, which can be sold at the going rate. If, in the fifth year, the farmer adopts intensive tillage and loses all the stored C, the farmer would be required to purchase C credits from the market at the going price, to cover emissions. Although economically efficient, this system is probably very difficult to implement because of monitoring and enforcement issues.

The VLC system might evolve through independent broker arrangements. If a broker wishes to buy permits from sink producers and then sell them to emitters, a contract must be made with sink producers to achieve a permanent decrease in C. Contracting can be accomplished in various ways; for instance, one farmer could adopt conservation tillage for 3 yr before plowing the field, and a second farmer could plant trees beginning in yr 4 for a certain number of years, and so on. In each period, the broker might offer farmers a menu of prices associated with different contract lengths. In this system, private brokers provide the service of generating "permanent" decreases in C from a series of separate temporary changes.

A CAA system may be fairly straightforward to implement. In this system, the generator of a sink is paid the full value of the permanent decrease in the GHGs stored in the sink, but payment is deposited directly into an annuity account. As long as the sink remains in place, the owner can directly access the earnings of the annuity account, but not the principal. The principal is withdrawn (or confiscated) when and if the sink is removed (i.e., the soil is tilled or another change is made to release stored GHGs). If the sink remains permanent, the owner eventually earns all interest payments, the discounted present value of which equals the principal itself, or the permanent permit price.

All three systems are economically efficient (for formal proofs see Feng, Zhao, and Kling [2000]). Which, if any, of these systems would be used likely will depend on the ease of enforcement and implementation. Nothing prevents all three systems from being used simultaneously. In any event, an important element in the success of any system will be the mon-

itoring and transaction costs associated with it.

Feasibility: Monitoring and Transaction Costs

As has been mentioned, any successful GHG program must have effective monitoring available at reasonable cost. Programs should not be implemented if costs of monitoring and administering exceed program benefits. A hectare of land sequestering 0.5 tonne of C/yr at a price of $20/tonne generates a value of $10/yr as long as the C remains sequestered. This sets an upper limit on the amount that can be spent on monitoring and administrative costs. The value actually paid to a farmer likely would be decreased by these costs. The administrative costs of programs differ widely. For agricultural program transactions involving minimal monitoring, such as delivery of loan deficiency payments, these costs include the additional personnel costs in the county offices of the Farm Services Agency and additional oversight and accounting costs throughout the U. S. Department of Agriculture (USDA), which cannot amount to more than 5 to 10% of total payments. For crop insurance programs, which involve significant marketing expenses and moderate monitoring of crop yields, the USDA reimburses crop insurance companies at a rate of approximately 24% of the value of the insurance premiums collected.

Because the value of a GHG mitigation contract increases with the reliability of monitoring, determination of the optimal amount of monitoring is an economic problem. In the simple instance, in which monitoring costs are subtracted from the contract price, the optimal amount of monitoring occurs where the incremental benefit from extra monitoring is equal to the incremental cost.

Changes in soil C and other GHGs are observable directly only through specialized measurements and subsequent lab analyses. Thus the cost of directly measuring GHG fluxes for an individual contract may be prohibitive, depending on the scale and type of project, and it is likely that monitoring programs will be based largely on the observation of farmers' actions. Clearly, the cost of direct observation depends on the nature of the variable being observed. It is much easier to observe discrete decisions such as crop selection or irrigation technology choice than it is to observe ongoing decisions such as the amount of fertilizer or pesticide applied to a field. Discrete choices can be observed visually, either by remote sensing or direct inspection. Examination of farmers' management records (if such records are kept) and of financial data is required to observe ongoing choices (Segerson 1988). The literature on mechanism design shows how individuals can be induced to tell the truth about ongoing choices, but implementation of such contracts has yet to be demonstrated (Wu and Babcock 1996). Millock, Sunding, and Zilberman (2000) demonstrated in a discussion of monitoring that when farmers are paid a premium for reliable monitoring of their pollution mitigation or, in this instance, sequestration, they may have the incentive to adopt monitoring technologies providing relatively reliable estimates.

Discrete production practices influencing GHG flux include choices of crop, tillage, irrigation technology, and livestock waste-handling system. These choices can be observed by combining remote sensing technologies and manual observations. A monitoring infrastructure must be developed, however. Costs can be decreased by using existing infrastructures such as conservation districts; crop insurance adjusters/agents; federal, state, and local environmental monitoring agencies; and/or tax authorities. Universities may contribute the expertise of faculty and staff for training and certifying program monitors.

Currently, data on farmers' ongoing choices are available minimally. The main source of such information is the USDA's Area Studies data and Agricultural Resource Management Survey data. But access to these data is limited because of confidentiality concerns, and the USDA does not provide individual observations. Only if farmers are required to keep detailed records of their production decisions and report these data to a monitoring agency will it be feasible to monitor ongoing choices. New precision-farming technologies and increased use of crop consultants may increase the availability of ongoing data.

Monitoring costs likely will play an important role in decisions regarding whether agriculture becomes incorporated officially into national GHG mitigation programs. The GHG-related activities of other industries also involve monitoring, but the monitoring cost per unit of GHGs in point-source industries is likely to be lower than that in agriculture. These monitoring costs must be added to the direct cost of mitigation before it can be concluded which industries are low-cost providers of GHG mitigation.

From a global perspective, the monitoring problem in agriculture is even more severe than in the United States, because most countries do not have the infrastructure to monitor farmers' land-use choices and other discrete choices appropriately. Moreover, with respect to ongoing choices, most countries have, at

most, rudimentary systems to track the use of variable inputs. Development of required infrastructure may not be out of the question, however, because labor costs are low in many developing countries, and the increased availability of wireless communication technologies may make the transfer of data from farm fields much less costly than in the past. Of course, development of this infrastructure may provide other benefits such as increased agricultural productivity, and an increased ability to manage environmental programs may make such investments feasible.

Consistency with Agricultural Policy

Ultimately, GHG mitigation policies in agriculture must be consistent with U.S. domestic agricultural policy and acceptable to the international agricultural policy community. In 1985, major environmental provisions were passed in the United States requiring farmers to increase their use of environmentally friendly practices to maintain eligibility for subsidies. These provisions strengthened the role that environmental quality played as an objective for agricultural commodity policy. Programs and practices adopted to improve environmental quality were, in general, GHG friendly, even though decreased GHGs were not a targeting criterion. Key elements of the 1985 farm policy included conservation compliance that increased the use of conservation tillage and the adoption of cross-compliance rules designed to prevent conversion of grassland and wetlands to cultivation. By maintaining soil C stocks, these land-conversion provisions also were GHG friendly. Another component of the 1985 farm policy is the CRP, which retired more than 12.1 million ha of annual cropland. More important, the CRP established a mechanism to convert environmentally sensitive land from traditional practices to conservation uses (Figure 3.6).

Subsequent farm policy legislation has resulted in an administrative capacity to implement environmentally friendly management of soils through various types of incentives. The CRP pays farmers to convert land. The Environmental Quality Incentive Program (EQIP) provides cost-share subsidies to adopt environmentally friendly technologies. Conservation compliance for the first time has made receipt of farm commodity subsidies conditional on implementation of soil conservation practices by farmers.

As the United States has established these mechanisms to improve environmental quality, the use of

other program provisions—especially of insurance and government-guaranteed prices—has intensified. In spite of the stated objective of the 1996 Federal Agriculture Improvement Act that production subsidies be phased out, the current system establishes minimum price guarantees protecting farmers against low prices. Furthermore, current farm policy increasingly uses subsidized crop insurance premiums to transfer income to farmers. Large-premium subsidies as well as price guarantees tend to increase

Figure 3.6. A permanent grass cover established under the Conservation Reserve Program resulted in sequestration of large amounts of C in this northern Minnesota soil. The dark area in this soil profile is very rich in C. Photo courtesy of the Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.

supply and to counteract the environmental provisions of agricultural policy.

The current state of agricultural policy contains multiple objectives working at cross-purposes with regard to GHG mitigation. If GHG mitigation becomes a more prominent policy objective, then current legislative and administrative structures provide many vehicles through which to decrease emissions. Furthermore, crop payments and insurance subsidies could be tied more closely to environmental performance. For example, eligibility for crop insurance

subsidies could be restricted to farmers maintaining conservation compliance plans.

Adoption of the 1985 conservation provisions demonstrates that the cost of inducing adoption of GHG-friendly practices may be shared by programs targeted at other environmental amenities. This trend away from direct subsidies of production toward subsidies enhancing multiple environmental attributes and meeting other social goals (so-called *multifunctionality*) may continue, especially because World Trade Organization (WTO) rules allow farm subsidies to promote environmental protection. These WTO rules may increase support for subsidizing GHG-friendly practices by farmers in other developed countries. Further, education about the link between GHG mitigation and other environmental improvements allowed by WTO rules may increase interest in GHG mitigation programs in agriculture worldwide, enhance the profitability of participating in such programs, and strengthen the political support for them.

Many economists have recognized the need for careful coordination between environmental and agricultural policy programs (Antle and Just 1991; Hrubovcak, LeBlanc, and Miranowski 1990; Just et al. 1991; LeBlanc and Miranowski 1991). To assure the public that programs implemented to decrease GHG emissions in agriculture are not working at cross-purposes, such programs must be designed and carried out in conjunction with other environmental and agricultural goals.

The two basic organizational structures for GHG mitigation programs are government centered and market centered. It is important not to focus solely on government-centered agricultural programs. Because most countries are moving away from government allocation of resources toward increased reliance on market forces, acceptance of market-based structures for allocation of GHG mitigation credits may be preferred by a growing number of countries.

Carbon sequestration is likely to occur as the result of a societal goal to decrease net GHG emissions. In such a setting, C sequestration will be but one of a number of ways to decrease net emissions. Other land-use-related sources for decreases in emissions include agricultural emission control, agricultural emission offsets (i.e., biofuels), and forestry-based sequestration. The effects of international trading also are important: other countries may have sources of emission decreases that would diminish the potential for U.S.-based agricultural actions if global trading of emission offsets were instituted. This report next examines the cost implications of these issues.

Multiple Gas Trade-Offs

Of the main GHGs, three are of importance to agriculture: CO_2, N_2O, and CH_4. Greenhouse gas emissions trading likely will encompass multiple gases. To place all GHGs on an equal footing, the Intergovernmental Panel on Climate Change (IPCC) developed the concept of Global Warming Potential (GWP), which compares the relative abilities of gases to trap heat in the atmosphere (see Chapter 1). For example, during a 100-yr time frame, the effect of CH_4 on greenhouse warming is approximately 23 times that of an equivalent volume of CO_2, and N_2O is approximately 296 times more powerful than CO_2. This means that a strategy increasing C sequestration but using more fertilizer and emitting more N_2O will decrease net emissions only if the C sequestered exceeds the C emitted in fertilizer manufacture plus 296 times the additional N_2O emitted. Furthermore, the net cost increase that can be justified to adopt this strategy cannot exceed the prevailing C price times the net emissions offset, less the value of any external effects (e.g., decreased soil erosion or improved wildlife habitat). Given the complexities of different factors, it has been suggested that evaluating multiple gas trade-offs should be based on cost-benefit studies and not simply on a cost perspective (Hayhoe et al. 1999; Manne and Richels 2001; Reilly et al. 1999).

Agricultural Mitigation of Greenhouse Gases in the Global Context

Comparing Across Mitigation Options

As previously discussed, agricultural mitigation options include not only C sinks but also emission decreases and other emission offsets. Globally, agriculture's share of total anthropogenic emissions is substantial. Approximately 50% of CH_4, 70% of N_2O, and 20% of CO_2 emissions, from all human-caused sources, are attributed to agriculture and land-use change (see Cole et al. 1996; Isermann 1994). Agricultural CO_2 emissions come primarily from deforestation and from land degradation in developing countries with, relatively speaking, minor contributions from fossil energy use in agriculture, especially in developed countries. Emissions of N_2O occur mainly as by-products of crop and livestock production, whereas CH_4 emissions stem from raising livestock, managing manure, and producing rice. McCarl and

Schneider (2000a) have summarized the costs of GHG emission-reduction strategies for agricultural sources described in the literature (Table 3.1). More recently, McCarl and Schneider (2000b, 2001) demonstrated that tillage changes, afforestation, and selected decreases in fertilizer and use of manure would occur at C prices at and below $30/tonne.

Agriculture also can offset emissions through making products that can be substituted for those currently produced using fossil fuels. These products include

Table 3.1. Estimated costs for agricultural GHG in the United States for each of the three major GHGs. Costs are in year 2000 $US, per metric tonne of C equivalent (based on GWPs of 21 for CH_4 and 310 for N_2O) (based on data from McCarl and Schneider 2000a)

Target Gas	Strategy	Cost in $ per tonne C
N_2O	Improved crop nutrient management	-158
	Nitrification inhibitors	164
	N fertilizer tax	370
	No anhydrous N fertilizer	46
	N fertilizer use reduction	56
	Low-protein swine feed	-1,400
	N-reduced poultry feed	1,300
CH_4	Enteric fermentation	-3,700–270
	Liquid manure management	0–200
	Rice cultivation	103–116
	Altered rations	204
	Herd reduction	730
CO_2	Tillage	0–400
	Biofuel for power plants	26–100
	Ethanol	180–300

commodities that displace fossil C-based energy sources (i.e., biofuels) and other nonagricultural sources of GHG, such as the use of GHG-intensive building materials (e.g., wood and bio-based polymers). Table 3.1 summarizes the cost estimates for these offsets that McCarl and Schneider (2000a) computed from the literature.

The other major land-use-based GHG mitigation option is forest C sequestration. Forest C sequestration usually has greater potential per unit of land area than agricultural C sequestration does, because of the large aboveground stocks of C in biomass, which can accrue relatively quickly compared with soil C. Many estimates of C sequestration rates have been made for afforestation and reforestation activities (Table 3.2). Estimates now are arising relative to the potential for C sequestration through improved management of existing forests. For example, in an August 2000 submission to the Kyoto Protocol, the United States claimed that, under a business-as-usual scenario, forests would increase C sequestration by 275 teragrams (Tg)/yr and agricultural soils by 25 Tg/yr for the period 2008 to 2012. Much of this projected forest C sequestration, however, is due simply to forests regrowing from previous periods of greater forest harvest and from forest regrowth on abandoned agricultural lands, rather than from targeted forest management.

Numerous studies of nonagricultural CO_2 mitigation costs have been conducted, with a recent summary provided in a special issue of the *Energy Journal* edited by Weyant and Hill (1999). Table 3.3 contains summary information for estimated U.S. costs per tonne of C for these nonagricultural options. This information is taken from a set of economy-wide estimates constructed by 11 different modeling teams under various assumptions regarding the extent of international C trading. Generally, the more liberal the trading assumption, the lower the cost; many estimates come in at or above $30/tonne, and no estimates come in at below $21/tonne.

International Emissions Trading

The Kyoto Protocol provides for trading among countries to meet emission guidelines. Certainly there will be trading in some form, regardless of the

Table 3.2. Estimates for C sequestration potential (in million metric tonnes [mmt] per year) and costs for afforestation and reforestation activities in the United States (courtesy of B. McCarl)

Study	C sequestered	Comments
Parks and Hardie 1995	36.3 mmt/yr at $23/metric tonne	Assumes all land converts from agriculture to forest if bid value outweighs land rents
Parks and Hardie 1996	3.6 mmt/yr at $23/ metric tonne	Estimates predicted rates of program participation based on data from CRP program
Stavins 1999	120–150 mmt/yr at $23/metric tonne	Net change in forest area (afforestation–deforestation)
Adams et al. 1999	73 mmt/yr at $21/ metric tonne	Afforestation only
McCarl 1998	15.7–52.7 mmt/yr at $25/metric tonne	Afforestation only with quantity dependent on contract terms

Table 3.3. Summary information of average costs ($ per metric tonne of C) for CO_2 reductions equivalent to a 7% reduction from 1990 levels in the United States (i.e., Kyoto Protocol target), estimated from 11 different models, including all economic sectors, with different assumptions regarding the allowable trading of C reduction credits (based on data from Weyant and Hill 1999; adapted by B. McCarl)

Model	No trading	Annex 1 trading	European Union and Annex 1 trading	Full global trading
1	412	227	174	122
2	320	108	103	27
3	265	137		90
4	238	78		30
5	197	82		
6	192	85	61	26
7	174	53		27
8	155	66	46	37
9	137	62		20
10	82	21		
11	80	55	27	20
Average	205	89	82	44

level of emission-reduction targets. The results in Table 3.3 show that trading among the Annex 1 countries, which are mainly developed countries, lowers the cost of meeting an emissions target by 57% on average, whereas including full trading in the European Union lowers costs by an additional 3 to 60%. Free trading among all countries is forecast to result in a drop in prices of, on average, almost 80%. Thus, trading can provide cheaper alternatives for GHG emission reductions in other areas of the world, and this likely will be true with respect to agriculture.

A fundamental principle of economics states that increased flexibility decreases the cost of meeting a given objective. Applying this principle to the cost of decreasing GHG buildup means that a mitigation strategy that only focuses on using soil C sequestration would be more costly than one that allows a fuller range of offsets from multiple agricultural emissions sources. Schneider (2000) and McCarl and Schneider (2001) studied the consequences of relying on a narrow range of strategies and found substantial advantage in considering the entire set of options. Reilly and colleagues (1999) arrived at the same conclusion, in a general economy setting.

4 Monitoring and Verifying Carbon Sequestration and Greenhouse Gas Fluxes

Quantifying net emissions of carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) between soils and the atmosphere presents a number of challenges, many of which are associated with the high degree of spatial and temporal variability in the processes controlling these fluxes. Carbon dioxide emissions and/or carbon (C) sinks for soils can be accounted for as stock changes and/or gaseous CO_2 fluxes, whereas N_2O and CH_4 are quantifiable only as gaseous fluxes. Quantifying stock changes in soil C is complicated by the large amount of C that is relatively inactive on short timescales. Thus changes in soil C on annual timescales will occur primarily in a small portion of the total soil organic matter (SOM), and hence short-term (i.e., <5 years [yr]) changes in total C stocks can be difficult to detect against the large background of total soil C. In addition, there is considerable spatial variability of soil C, because of the interaction of many intrinsically spatially variable environmental and management factors (e.g., Robertson et al. 1997). Although changes in the concentrations of CO_2, N_2O, and CH_4 can be detected readily against background atmospheric concentrations during short timescales, fluxes of these gases can be even more variable than soil C stocks throughout time and space.

These difficulties can be dealt with using a variety of approaches—or ideally, a combined integrated approach—that this report organizes into three categories: (1) direct measurement of C stocks or gas fluxes throughout time to determine net emissions or sinks, (2) extrapolation and interpolation from direct measurements combined with models and ancillary information, and (3) verification of estimated changes. Direct measurements and experiments provide the foundation from which specific, local estimates are extended in space and time, through models, thereby enabling regional and national estimates of changes of C sequestration and greenhouse gas (GHG) fluxes.

Direct Measurements

Soil Carbon Stocks

Measurement Methods

Field measurements of soil C changes have played a fundamental role in understanding how agricultural management affects soil C. Any quantification scheme for emission reduction or C sequestration credits likely will incorporate measurements of changes in soil C, either directly or indirectly. Field measurements of soil C stocks are applicable in a variety of ways. First, measurements of soil C under different management treatments, including before and after implementation of new practices, can be used to estimate changes in soil C at the field or farm scale (Figure 4.1). Second, knowledge from soil C measurements under different agricultural treatments, soil types, climatic conditions, and land-use histories is key to establishing relationships between these variables and soil C. Such generalized knowledge, embodied in statistical and/or dynamic simulation models, can be used to extrapolate local measurements to broad scales (discussed in the next section) and to further refine models of soil C change. Finally, sampling-based inventories of soil C stock changes can provide an independent validation of extrapolation and model-based estimates. This section examines methods of measuring soil C, limitations of those methods, and implications for sampling for each of the purposes just mentioned.

Methods for measuring soil properties have been documented extensively and there are well-developed standards (Carter 1993; Culley 1993; Klute 1986; Magdoff, Tabatabai, and Hanlon 1996; Page, Miller, and Keeney 1982; USDA–NRCS 1996). The most

Figure 4.1. Hydrologic probe equipment such as this is used routinely to extract soil cores that then are taken to the laboratory for processing and measurement of C content. Photo courtesy of S. Sprague, Colorado State University.

commonly used methods for determining soil C are wet digestion (using acids and a strong oxidant such as potassium dichromate [$K_2Cr_2O_7$]) or, increasingly, dry combustion, using automated C and nitrogen (N) analyzers. These instruments are highly accurate and efficient at processing large numbers of samples. All conventional measurement techniques, however, require removal of samples from the field, transport to a laboratory, and sample preparation (e.g., sieving, grinding) before analysis. New in situ methods, using soil probes and in-field measuring devices such as laser-induced breakdown spectroscopy (Cremers et al. 2001), laser-induced fluorescence (Chen, Chadwick, and Lieberman 1997), and other spectrographic methods, offer hope for more rapid, noninvasive, and less expensive measurements in the future. At present, however, these new technologies are not yet at an operational stage for widespread monitoring of soil C stocks.

Estimating Stocks

Laboratory measurements of soil C provide values in terms of concentration (i.e., mass of C/mass of soil [gram (g)/kilogram (kg)]). In C balance calculations, however, C stocks must be reported as soil organic C on a mass per unit area (kg/m^2) or mass per unit volume basis (kg/m^3). Converting estimates of C from concentration to per unit area C stocks requires estimates of soil bulk density (i.e., soil mass per unit volume), rock content, sampling depth, and root components, and how these variables change with depth, for the volume of soil being considered. Each of these variables is accompanied by uncertainties because of measurement error and temporal and spatial variability (Kral and Hawkins 1994). Rock or coarse fragment (>2 millimeter [mm]) content must be subtracted from soil volume before estimating total C on an areal basis. Collecting samples in rocky areas and obtaining adequate numbers of samples present significant challenges (Fernandez, Rustad, and Lawrence 1993; Kral and Hawkins 1994).

Soil physical characteristics can change over time. Bulk density may change because of compaction, bioturbation, cultivation, or structural change resulting from changes in organic C, soil structure, and roots (Oades 1993). Sampling methods that minimize this variability, such as obtaining bulk density samples without compaction and taking samples at the same time of year, are recommended. To compare the C in two soils, the same mass of soil, to a mass equivalent reference depth (e.g., below the plow layer), should be compared rather than the same volume (Ellert and Bettany 1995). If the C content of a soil has increased throughout time, it likely will have a decreased bulk density, so a greater soil depth will need to be sampled than for a comparable soil with no change in bulk density. In this manner, C storage in soils under different management or sampled at different times can be compared directly. The most efficient way of doing this is by sampling several different depth increments to derive a depth profile of soil C concentration and soil mass.

Root content adds another source of variability to C content estimates. Coarse roots are removed in standard soil-sample preparations by passing samples through a 2-mm sieve (USDA–NRCS 1996), but certain fine root and dead root fragments pass through the sieve along with the soil material. The amount of fine root and litter material included in the 2-mm sieved sample will vary according to sample prepa-

ration—whether soils are sieved field-moist versus air-dried and then ground through the sieve. The latter procedure will tend to include more root and litter fragments in the sieved sample. Fragments >2 mm need to be handled consistently; that is, they either must be included or excluded, when comparing between systems or in measurements made throughout time. In certain circumstances, such as conversion of annual cropland to perennial grassland, the buildup of a permanent standing stock of root biomass (including coarse roots) can constitute a significant change in the ecosystem C balance that may need to be accounted for. The uneven and patchy distribution of plants and their associated root systems in forests and grasslands presents sampling difficulties, in addition to affecting SOM spatial heterogeneity (Burke et al. 1999). In annual cropland, row crop spacing, fertilizer application, and tillage all contribute to spatial variability in roots and crop residues, affecting SOM distribution.

Lateral soil movement because of erosion and deposition can confound efforts to monitor changes in soil organic C storage. In research environments, erosion processes (e.g., detachment, transport, and deposition) either can be measured or estimated (Lal 1994). Measurements of erosion, for example by sediment collection from gauged catchments or using sediment tracers (e.g., ^{137}Cs), are usually long term, involved, and costly, and therefore likely to remain in the domain of research. Several erosion models such as the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1991), the Erosion Productivity Impact Calculator (EPIC) (Williams 1995), the Water Erosion Prediction Project (WEPP) (see Figure 4.2; Laflen et al. 1997), and the Wind Erosion Prediction System (WEPS) (Hagen 1991) have been developed and are applied widely in the United States. It is likely that erosion estimates from simulation models, together with information about C content in sediments and sediment enrichment ratios, will be the most practical way to estimate C changes resulting from erosion and deposition in operational projects for C sequestration.

Although total soil C measurements are useful for monitoring C changes, they may not be sufficient to detect significant changes throughout short periods (e.g., less than 3 to 5 yr). Soil organic matter fractions that change more rapidly than bulk soil C content have been isolated and proposed as indicators of changes in total soil C. For example, management-induced changes in light-fraction C (i.e., material

having a lower density, <1.8 g per centimeter [cm], than bulk soil), often are detectable after a few years and may be useful predictors of longer-term changes in total soil C (Janzen et al. 1992). Aerobic incubation is another commonly used technique to estimate labile C that can serve as an indicator for changes in total soil C. Soil incubation consists of measuring CO_2 evolved from soil samples, under constant temperature and moisture conditions in the laboratory, for periods of several weeks to months (Collins et al. 2000; Elliott et al. 1994). Microbial biomass, which usually is measured as the C released after chloroform fumigation (Jenkinson and Powlson 1976; Vance, Brookes, and Jenkinson 1987), also may serve as an early indicator of changes in soil C content (Anderson and Domsch 1989). One or all of these measurements, or ratios with total soil C, may enable earlier detection of changes in soil C in response to management or environmental changes than by measurement of total soil C alone.

Minimum Detectable Differences—Temporal and Spatial

Temporal and spatial variability in the amount of organic matter from roots, litter, and microbial biomass combined with variability in bulk density, coarse fragments, and horizonation contribute to high variability in SOM estimates (Figure 4.3) even within

Figure 4.2. **The rainfall simulator and test plot at Cottonwood, South Dakota, enabled technicians to measure water runoff rates and collect soil samples in a Water Erosion Prediction Project cropland field study. Photo courtesy of T. McCabe, Agricultural Research Service, U.S. Department of Agriculture, Beltsville, Maryland.**
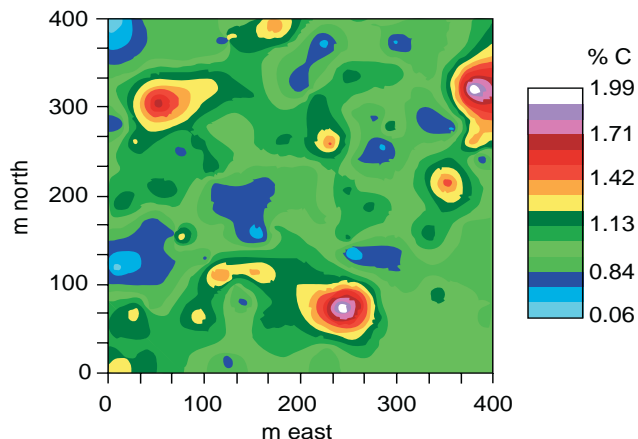
Figure 4.3. Map showing the spatial distribution of soil organic matter C (to a depth of 15 cm) across a 400 X 400-m-square area at the Kellogg Biological Station in southwestern Michigan. The map was generated from more than 128 samples taken in an irregular offset grid and then spatially interpolated using a statistical procedure called "kriging." Sampling was done before establishing long-term experiments on a previously uniformly managed field, cropped to alfalfa, corn, and soybean. (Source: Robertson et al. 1997.)

seemingly uniform sites under long-term cultivation (Robertson et al. 1997). In nonagricultural ecosystems such as forests, spatial variability of soil properties tends to be even greater because of variations in plant density, species type, and disturbance history, among other factors.

A number of studies have examined the variability of soil properties in different ecosystems in order to estimate the sampling requirements for detection of significant differences between systems and/or to detect changes occurring throughout time. For example, Wilding and Drees (1983 [cited in Buol et al. 1997]) reported that, given typical field-scale spatial variability of soil properties, detection of changes within ±10% of the mean ($\alpha = 0.05$) would require on the order of 10 samples for pH, 10 to 25 samples for texture, and 25 or more samples for horizon thickness. Mollitor, Leaf, and Morris (1980) reported that 20 to 30 samples were needed in floodplain forests in order to limit uncertainty to within ±10% of mean soil organic C ($\alpha = 0.05$). Huntington, Ryan, and Hamburg (1988) calculated that changes on the order of 2 to 3 metric tonnes C/hectare (ha) could be detected in the C stocks of surface soils (0 to 10 cm) for a small, forested watershed, with approximately 60 sample locations. Garten and Wullschleger (1999) calculated minimum detectable difference for changes in soil

organic C content after the planting of switchgrass on cropland. They estimated that more than 100 samples would be needed to detect a 2 to 3% change in soil organic C, but that a 10 to 15% change could be detected with a more practicable sample size (n = 16). Similarly, Conant and Paustian (2002) found that a modest number of samples (13 to 28) would be required to detect changes in soil C that could occur throughout a 5-yr period with improved management in cultivated fields and pastures, but many more samples (>100) would be necessary to detect changes of the same magnitude at more-variable forest sites.

Variance estimates for combinations of soil types and environmental conditions are needed, but once the level of C content variation is determined, the number of samples needed to detect a 10%, 25%, or other magnitude of change can be established for a given level of statistical power (see Conant and Paustian 2002; Garten and Wullschleger 1999). Because annual changes are small, even with large changes in SOM input rates, it is infeasible to measure changes in soil organic C on annual or more frequent timescales. Interannual variations in environmental conditions also complicate estimates of C stock changes at all spatial scales. With sufficient time, however (i.e., 3 to 10 yr), statistically significant differences in soil organic C because of land use and management activities have been documented widely in field experiments (see reviews by Grace, Ladd, and Skjemstad 1994; Izaurralde et al. 1998a; Paul et al. 1997; Paustian, Collins, and Paul 1997; Post and Kwon 2000; Smith, Powlson, and Glendining 1996).

Detection limits and sampling requirements also are highly dependent on scale. Because much of the inherent variability of soil C content is expressed at a fairly fine resolution (e.g., field level or smaller), the number of samples needed to detect a given change increases only modestly in going to regional and higher scales to detect the same relative change in C stocks (Figure 4.4). Thus, the number of samples per unit area being considered decreases dramatically as the size of a C sequestration project increases, or for sampling applied to regional or national accounting.

Gas Fluxes

Nitrous oxide and CH_4 emissions and sinks, in contrast to CO_2, cannot be inferred from stock change estimates and thus require direct measurement of gas flux to and/or from the soil. Carbon emissions and uptake as CO_2 also can be determined via gas flux
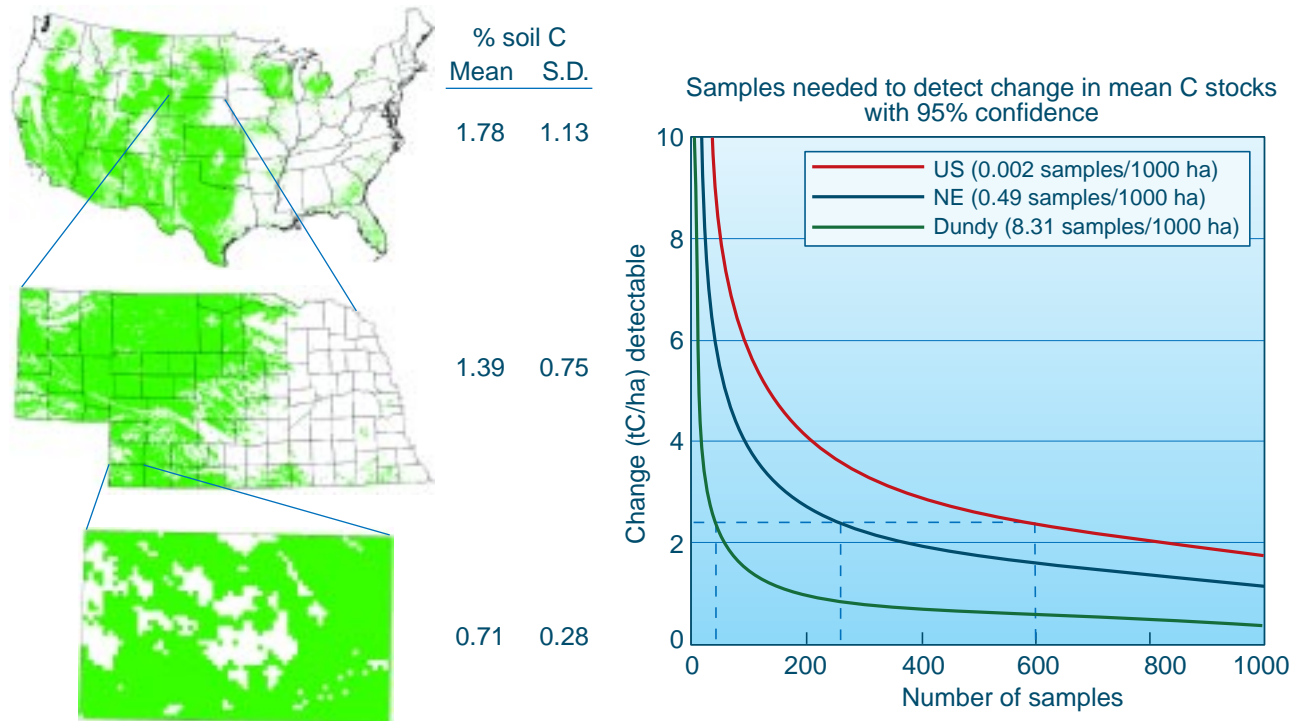
Figure 4.4. Distribution of grassland-derived soils (i.e., Mollisols) at national (U.S.), state (Nebraska), and county (Dundy) scales, together with mean soil organic C concentrations (and standard deviations), taken from the USDA's national soil characterization data set, at each scale. The graph shows the minimum detectable difference in overall C stock change (at a 95% confidence level) as a function of the number of samples needed at each scale to detect an overall increase in grazing land C stocks. As an illustration, sampling requirements to detect a change of 2.3 Mg C/ha (equivalent to an average rate of C change because of improved pasture management of 0.46 Mg C/ha/yr over a 5-yr period) are shown. Courtesy of R. Conant, Colorado State University.

measurements, but typically the simultaneous occurrence of photosynthesis (uptake) and respiration (loss) from both plant and soil sources makes it difficult to measure the individual component fluxes. But micrometeorological techniques (discussed later in this section) are being used increasingly to estimate the net CO_2 flux between the atmosphere and the land surface.

A variety of gas flux measurement approaches have been developed, varying according to the objectives of the study, type of ecosystem, gas species in question, cost, and scientific infrastructure. There are two main types of approaches: chamber-based techniques (Figure 4.5) and micrometeorological techniques (Figure 4.6). Chamber techniques involve small enclosures that are placed on the soil, within which fluxes are measured using conventional gas measurement instrumentation. Micrometeorological methods are based on integrating measurements of gas concentrations with measurements of air movement to derive

estimates of surface-atmosphere fluxes. They provide nondestructive, integrated measurements of gas fluxes over large areas, but generally require large, uniform land areas. In addition, both tower-based and airborne micrometeorological methods require expensive fast-response sensors and logistical support. Extensive reviews have been written describing methods for measuring gas fluxes from terrestrial surfaces, such as chambers (Hutchinson and Livingston 1993; Livingston and Hutchinson 1995; Mosier 1989), micrometeorological (Denmead and Raupach 1993; Desjardins et al. 1993; Fowler and Duyzer 1989), aircraft-based (Desjardins 1992; Desjardins and MacPherson 1989; Desjardins et al. 1993), and convective boundary layer budget methods (Denmead and Raupach 1993; Denmead et al. 1996; Raupach, Denmead, and Dunin 1992). Only a brief overview of these methods, selected from Lapitan, Wanninkhof, and Mosier (1999), will be given here.

Analytical Methods for Measuring Trace Gas Concentrations

Gas chromatography, infrared (IR) gas analysis, and other commonly used analytical methods for measuring atmospheric gases were reviewed comprehensively by Crill and colleagues (1995) (Table 4.1).

Figure 4.5. A static flux chamber measurement being taken in a growing corn crop. Typical designs use a removable cover that is placed on a semipermanent base (can be removed for tillage) having an airtight seal. For soil trace gas measurements, the area within the chamber usually is kept vegetation-free to eliminate CO_2 uptake by plants. For measurements, the cover is sealed and left for several minutes to allow the gas being measured to accumulate in the chamber (or to be consumed where there is uptake by the soil), and a series of gas samples are taken via syringe (for later analysis on a gas analyzer) to determine the change in gas concentration over time, from which flux rates are determined. Photo courtesy of G. P. Robertson, Michigan State University.

Table 4.1. Summary of analytical methods for measuring trace gases (based on data from Denmead 1994; adapted by A. R. Mosier)

Method	Footprint area	Maximum instrument frequency response (Hz)
Chambers	1 m	0.001–0.003
Mass balance	20 m	0.003
Eddy accumulation	0.2–1 km	0.003
Gradient methods	0.2–1 km	0.02
Eddy correlation	0.2–1 km	2.5
Convective boundary layer	30 km	0.02
Aircraft	5–15 km	10

For methods (such as chambers) that do not require fast response times for measuring instantaneous gas concentrations, gas chromatography is used widely for analyzing CO_2, CH_4, and N_2O. In conjunction with micrometeorological methods, however, analytical instruments must have a fast response (i.e., be sensitive and accurate enough to detect one-tenth of the mean concentration difference between updrafts and downdrafts for typical gas flux rates) (Denmead 1979;

Eddy covariance measurements of carbon dioxide and other fluxes   **Measuring components of solar radiation**   **Close-up of Eddy covariance flux sensors**

Figure 4.6. Ground-based micrometeorological methods employ towers instrumented with fast-response sensors to measure gas concentrations as well as three-dimensional air movement at different heights above the plant canopy (or soil surface if no vegetation is present). Measurements are taken on a near-continuous basis (several times per minute) and used to calculate gas fluxes across the boundary between the atmosphere and the plant canopy, thus providing an integrated measurement of net fluxes to or from the total plant/soil system. Photos courtesy of S. Verma, University of Nebraska–Lincoln.

Desjardins et al. 1993). For CO_2, open- and closed-path IR analyzers provide adequate sensitivity and response times for both chamber and micrometeorological applications. For CH_4 and N_2O, in ecosystems where fluxes of these trace gases are low and instantaneous fluctuations of gas concentrations at very fine temporal resolution are required, tunable diode laser (TDL) and Fourier transform infrared (FTIR) spectrometers offer high resolution and fast response time, enabling detection of gas concentrations at parts-per-trillion levels in a second or fraction of a second. An overview of these advanced spectroscopic instruments can be found in Kolb, Wormhoudt, and Zahniser (1995).

Chamber Techniques for Trace Gas Flux Measurements

The chamber method still is considered the method of choice for process-level studies of soil and microbiological factors controlling trace gas fluxes. The current types of chambers used vary in sampling area, from ≤ 1 m^2 to 64 m^2 (Galle, Klemendtson, and Griffith 1994). The two basic chamber designs are closed ("static") and open-top ("dynamic"). Positive and negative aspects of chamber methods are detailed in a number of reviews (e.g., Denmead 1979; Hutchinson and Mosier 1981; Livingston and Hutchinson 1995; Mosier 1990). With closed chambers, the concentration buildup (or drawdown in the instance of sinks) of gases within the chamber is measured throughout time to calculate the flux rate. In the open-top chamber design, the concentration of the gas in the enclosed volume of air is maintained at ambient level through a continual flow of external air, and a steady-state gas concentration gradient at the soil-air interface is established. Open chamber design is preferred over closed chambers if extended, repeated measurements at a fixed location are desired; however, static designs provide better precision in detecting small fluxes of trace gases. Generally errors in flux measurements can be attributed to chamber effects on the natural conditions of the sampling site, modifications of the microclimate, pressure-induced gas flows in open chambers, and inhibiting effects of concentration buildup in closed-chamber designs. Properly designed chamber-sampling arrays, however, can provide area-wide estimates of fluxes that agree closely with measurements using flux-gradient methods (Christensen et al. 1996).

Automated chamber systems have been used effectively to characterize temporal variability of gas fluxes better (Figure 4.7). Such systems have been used to measure emissions semicontinuously (i.e., several times per day) throughout several years in rice (Bronson et al. 1997; Wassmann et al. 2000), field crops and successional plots (Ambus and Robertson 1998), forest ecosystems (Loftfield, Brumme, and Beese 1992), and tropical soils (Weitz et al. 1999). Automated systems provide the frequency of measurements that manual chamber systems cannot provide without a large input in labor, but they require additional logistical support and higher equipment costs.
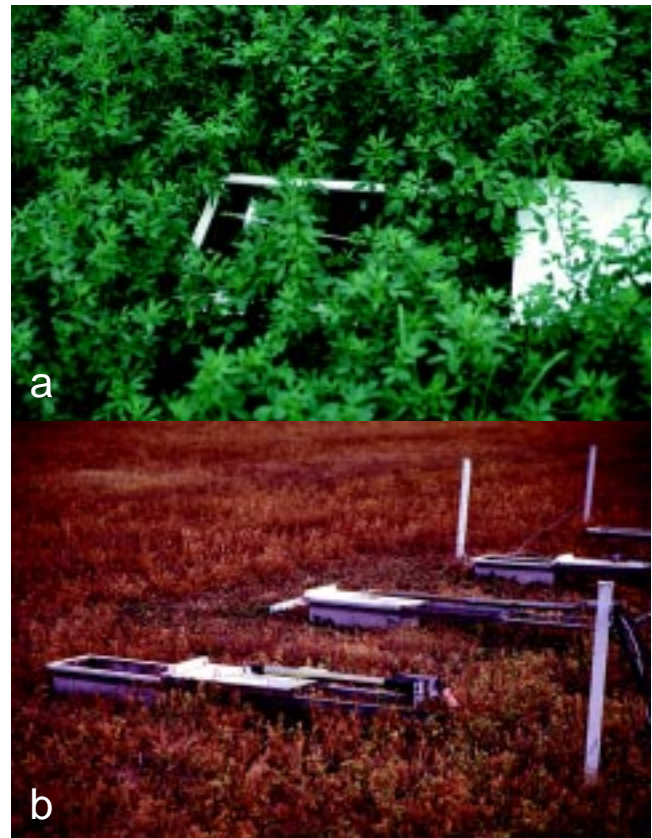
Figure 4.7. Automated chambers are constructed to open and close for short periods, several times daily, to measure trace gas fluxes on a more continuous basis than typically is done using static chamber methods (see Figure 4.5). The principle is similar to that used for traditional static chambers in which a series of measurements are taken after the chamber is closed to determine change in gas concentration over time. In automated systems, however, several chambers are connected directly by sample lines (usually small-diameter plastic tubing) to a computer-controlled valve system and gas measurement devices located in the field. Shown are (a) a single open chamber in a growing alfalfa crop and (b) a chamber array with one chamber closed. Photo courtesy of G. P. Robertson, Michigan State University.

Micrometeorological Techniques for Trace Gas Measurements

Micrometeorological techniques are based on frequent measurements of air movements, energy balance components, and gas concentrations above the vegetation canopy to estimate net gas exchange between the land surface and the atmosphere. Several variations of the same general principles have been developed, and the theories behind each method have been elaborated in the literature (Baldocchi, Hicks, and Meyers 1988; Businger and Oncley 1990; Denmead 1994; Denmead, Simpson, and Freney 1977; Desjardins et al. 1993; Fowler and Duyzer 1989; Lenschow 1995; Verma 1990; Webb, Pearlman, and Leuning 1980). Approaches include both ground-based (i.e., instrumented towers) and airborne facilities.

Micrometeorological techniques require highly accurate instruments with fast response time that make measurements several times per minute to calculate gas fluxes on hourly timescales. The integration of these short-term fluxes carried out through sufficiently long periods can yield estimates of the net exchange between the land surface and the atmosphere of the gas species in question. In addition to the high temporal resolution of the technique, the methods are noninvasive, in that they do not require destructive sampling and they integrate both vegetation and soil sources and sinks.

Micrometeorological approaches, particularly eddy covariance, are used most widely for estimating net CO_2 flux between the land surface and the atmosphere. The net flux is termed "net ecosystem exchange" (NEE); positive values indicate that ecosystems are accumulating C and negative values imply an overall loss of C from vegetation and/or soils. Several networks of ground-based flux measurement sites (Figure 4.8) have become established in recent years, (e.g., AmeriFlux, EUROFLUX, and AsiaFlux), focusing predominantly on forest ecosystems but including certain grassland and cropland sites as well. The two components of terrestrial ecosystems that contribute significantly to NEE are vegetation and soil C stocks. In order to separate the fluxes associated with vegetation and soils, additional measurements are necessary. Because changes in aboveground vegetation C stocks generally are easier to measure, changes in soil organic C stocks (and roots) often are assumed to equal the difference between NEE and aboveground vegetation C stock change. Where multiyear measurements of eddy flux and vegetation stock changes are made, it is desirable to combine these measurements with direct sampling and measuring of soil organic C, to fully resolve the C balance. Possibilities exist for using isotopic measurements such as $^{13}CO_2$, $^{14}CO_2$, and $C^{18}O_2$ to improve the partitioning of NEE fluxes between plants and soils. Coupling eddy covariance flux measurements with ecosystem models that simulate the major components of the ecosystem C cycle also can aid in the partitioning of measured NEE into soil- versus vegetation-derived C fluxes.

Micrometeorological techniques such as eddy covariance have a variety of measurement errors associated with them including systematic errors related to the performance of instruments under different environmental conditions, random errors, and sampling

Figure 4.8. Map showing site locations for flux tower measurements of CO_2 in North America, comprising the AmeriFlux and CanadaFlux networks. (Adapted from AmeriFlux 2000.)

uncertainty because of breakdown or maintenance periods, resulting in gaps in the measurement record (Goulden et al. 1996; Moncrieff, Malhi, and Leuning 1996). Ideally, measurements should be carried out on level and spatially uniform land surfaces, and thus complex hilly terrain is not well suited for the technique. Several studies have noted that the method tends to underestimate nighttime respiration losses of CO_2, which can lead to overestimates of net ecosystem C exchange (Piovesan and Adams 2000). Accordingly, correction factors for nighttime respiration estimates have been proposed (Goulden et al. 1996), but further work is needed to understand better the uncertainties associated with ecosystem flux measurements.

For non-CO_2 gases, there has been less use of micrometeorological approaches to date as compared with CO_2. Advances in real-time spectroscopic methods with high temporal resolution, however, such as

the TDL differential absorption and FTIR spectroscopy, extend significantly the capability of micrometeorological techniques to detect small fluxes and gradients of CH_4 and N_2O trace gases (Kolb, Wormhoudt, and Zahniser 1995). The methods commonly used include the following:

1. **Flux gradient**. Most recently, flux gradient applications have been extended for continual ($\leq$1-hour (hr) interval) flux determinations of CH_4 and N_2O (Wagner-Riddle et al. 1996a,b; Wienhold, Frahm, and Harris 1994) from profile concentrations measured with TDL and FTIR spectrometers.

2. **Eddy correlation**. Recent applications of eddy correlation have coupled this approach to fast response TDL and FTIR spectrometers that can measure sensitively the trace concentrations of CH_4 and N_2O (Christensen et al. 1996; Edwards et at. 1994; Hargreaves et al. 1996; Wagner-Riddle et al. 1996b).

3. **Eddy accumulation/Conditional sampling**. Hargreaves and colleagues (1996) and Christensen and colleagues (1996) used this technique to monitor CH_4 and N_2O fluxes in a grassland system.

4. **Aircraft-based measurements**. Eddy correlation can be applied for aircraft-mounted measurements to provide spatially averaged, aerial assessments of CO_2, CH_4, and N_2O gases (Desjardins et al. 1989; Desjardins and MacPherson 1989).

5. **Convective boundary layer budget**. A way to extrapolate surface fluxes of trace gases to regional scales (10^8 to 10^9 km^2 area) is to quantify directly the changes in the concentration of the gas in the upper, mixed layer of the convective boundary layer of the atmosphere (Denmead and Raupach 1993; Raupach, Denmead, and Dunin 1992).

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Extrapolation and Regional Estimates of Soil Carbon Emissions and Sinks

Regional estimation generally involves the extrap-
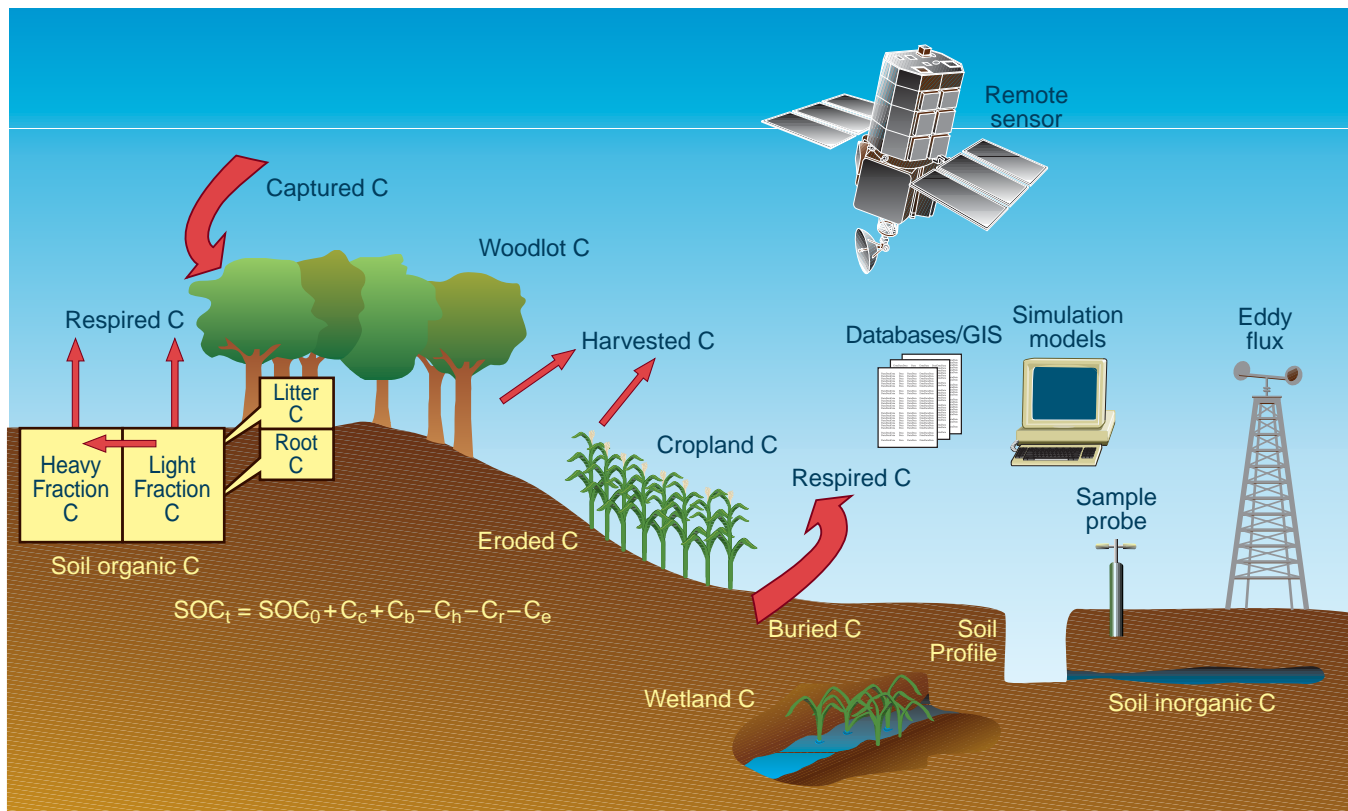
Figure 4.9. Conceptual diagram depicting the integration of measurement and models operating at local to regional scales to estimate changes in soil C stocks. SOC = soil organic C; C_c = C input through photosynthesis; C_b = C deposited and buried through erosion; C_h = C harvested (i.e., removed); C_r = C respired; C_e = C lost through erosion). Courtesy of M. Post (Oak Ridge National Laboratory, Oak Ridge, Tennessee) and N. Rosenberg (Pacific Northwest National Laboratory, Richland, WA).

olation of data gathered at local scales (i.e., point or field-level estimates) (Figure 4.9). Broad-scale soil C inventories derived from sampling could take the form of many soils randomly sampled across a region or country without regard to soil, vegetation, or climatic variability, but the utility of such measurements would be limited. A more informative approach is to use soil C data from samples that can be coupled with information about how soil C contents vary with the factors (so-called "driving variables") that control soil C. Such driving variables include climate, soil physical properties, topography, land use, and management practices. Spatial distribution of soil C and changes in C stocks then can be determined by relating soil C dynamics to those driving variables that are applicable for the region of interest. Direct soil C and flux measurements can be extrapolated using statistical models relating soil C to single or multiple driving variables or by using simulation models (Paustian et al. 1997b) that mechanistically integrate many driving variables and include system feedbacks and temporal and spatial dynamics of soil C. This section includes discussion of how these approaches typically are applied and their limitations, driving data sets that are available, and future data sets that will become available.

Spatial Scaling and Aggregation

When developing regional- or national-level estimates of soil C, the region or area of interest usually is divided into patches that are as homogenous as possible for the suite of driving variables used in the analysis. Estimated values for C stocks and/or stock changes for each patch, as calculated by the statistical or simulation model being used (Figure 4.10), are then multiplied by patch area, and results for several patches are summed to provide regional C stock estimates. Patches may be defined using various types of soil survey, land cover, land management, land-use history, climate, or other spatial datasets in combination. Relationships between environmental factors and soil C may be developed from field and laboratory experiments in agricultural, forest, or ecological research or from nonexperimental field surveys, such as chronosequences (sites that form a sequence in time because of disturbance or management change) and transects across topographic, climatic, or other gradients.

As patch size increases, the assumptions of homogeneity in management and/or environmental conditions become less valid and estimates based on the
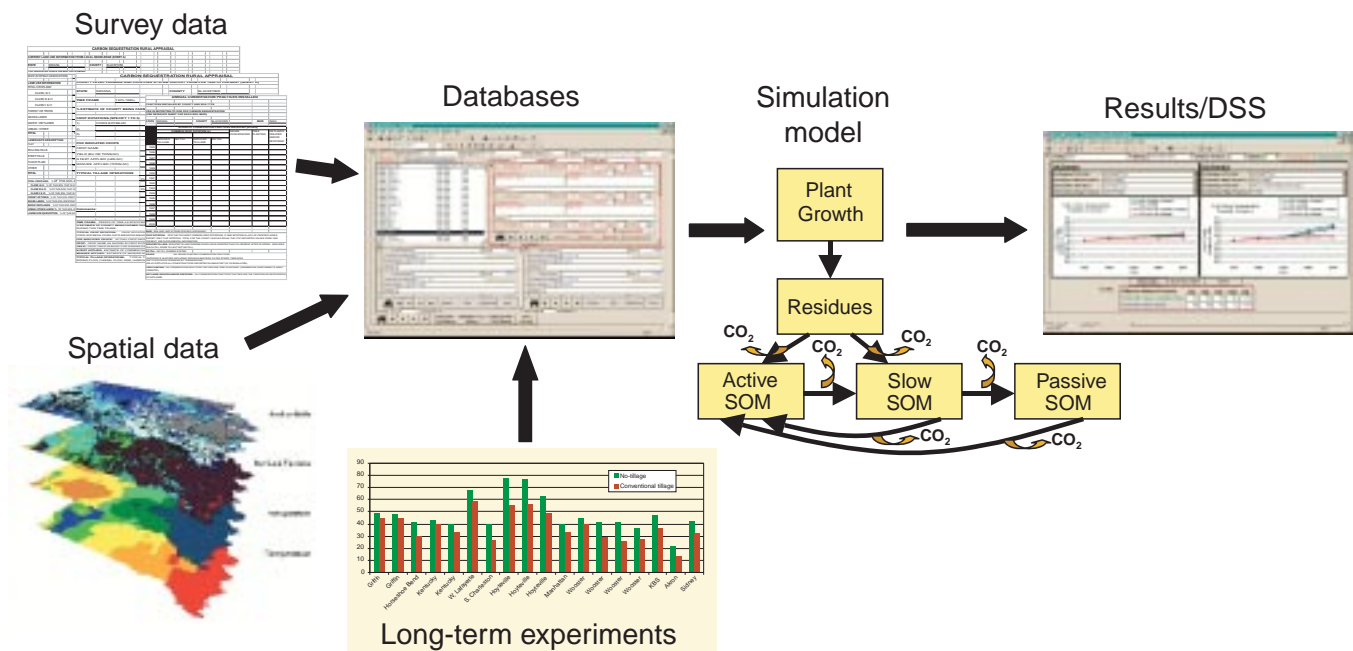
Figure 4.10. Methodology for model-based estimates of soil C and GHG fluxes at regional and national scales showing the components used. Input data from spatially explicit data layers (using Geographic Information Systems) and other data (e.g., statistical surveys of farm practices) are organized in a common database, which then is processed by a computer simulation model. Data from field experiments and/or in-field monitoring are used for validation and quality control. Model outputs can be used to generate maps, table summaries, or other forms of output to meet user requirements. (Source: K. Paustian, Colorado State University.)

average conditions of the patch are subject to increasing error, depending on the form of the relationship between soil C and the driving variable in question. In general, where the response variable, in this instance soil C, varies in a linear fashion with respect to a particular driving variable, then estimates based on averaged values across the patch can provide an unbiased estimate (Paustian et al. 1997b). For example, Burke and colleagues (1990) found net primary production (NPP) of shortgrass prairie in northeastern Colorado to be related linearly to precipitation. Because inputs of C to soil were tied closely to NPP, and soil C was linearly related to C input rates, Burke and colleagues found that predictions of soil C were not biased significantly when averaging across precipitation gradients for different-sized land areas. In contrast, factors for which soil C varies in a nonlinear fashion are subject to increasing error or aggregation bias as the patch size increases. For example, temperature generally is recognized as affecting SOM decomposition rate according to a nonlinear, exponential relationship (e.g., Q_{10} effect). Thus, significant errors can be introduced by averaging across a broad gradient in temperature, and it becomes necessary to subdivide regions into smaller patches or to model explicitly the known gradients affecting soil organic C distribution and change rate (Paustian et al. 1997b). Inappropriate spatial averaging can be minimized by a systematic stratification of environmental factors affecting SOM dynamics.

Regional Data Sources

In the United States, numerous resources are available for constructing spatially detailed maps of variables related to soil organic C content and organic C dynamics. The data sources listed in the following subsections are of interest for making regional and national estimates.

Plant productivity databases. These include data gathered regularly (often annually) as county summaries (e.g., crop production) or from permanent (forest) inventory points. The sources provide information for computing productivity and C inputs to soil on an annual basis, but require conversions from many different units (e.g., bushels of grain, tons of hay, volume of wood, etc.) to standardized units of C mass per unit area. Sources include crop production statistics (National Agricultural Statistics Service, <http://www.usda.gov/nass/>, forest inventories (Forest Inventory and Analysis database, <http://srsfia.usfs.msstate.edu/scripts/ew.htm>, and other

types of productivity data.

Soil geographic databases. Soil mapping data can be used to describe relatively static landscape features and soil physical properties that will not change significantly over periods of several decades. Soil characteristics, such as texture, are included, usually as ranges by soil type. Soil map units often correlate with major land-cover classes but do not reflect impacts of current land use and management. Data are available as generalized soil maps (USDA–NRCS 1994, 1995) for the entire United States (State Soil Geographic database) and as detailed soil survey information (Soil Survey Geographic database) for most counties, available from the National Soil Survey (<http://www.statlab.iastate.edu/soils/nssc/>). Baseline soil C maps for the United States have been developed using these data (e.g., Bliss, Waltman, and Petersen 1995).

Previously published soil C data from a variety of ecosystems. Data from field experiments can be used to develop and/or test models that link soil C to land use, land management, land-use history, climate, or other variables. This includes data from field experimental networks such as SOMNET (<http://www.iacr.bbsrc.ac.uk/aen/somnet/index.htm>), FLUXNET (<http://www-eosdis.ornl.gov/FLUXNET/>), LTER (<http://www.lternet.edu/>), and broadly distributed soil measurements such as the Natural Resources Conservation Service (NRCS) pedon database (NSSC 1997) and others (e.g., Post et al. 1982).

Land-use and land-cover databases. These include spatially resolved data that show how vegetative cover changes throughout time and may contain information applicable for calculating C inputs (e.g., percentage of cover, leaf area index, estimates of NPP). Data are available at multiple scales (30 m to 1 km) and land areas (county maps to national land-cover databases). Examples include land cover and land use from remote sensing (e.g., Loveland et al. 2000; <http://edcdaac.usgs.gov/main.html>).

Land-use and land-management databases. Information on current and past changes in management practices, such as cultivation, forest cutting and regrowth, and erosion are needed to clarify present and potential changes in C stocks and fluxes. Examples include the following:

1. **The National Resources Inventory (NRI)** is a stratified two-stage area sample of more than 1 million points across all nonfederal land in the United States and the Caribbean (Nusser and Goebel 1997). The NRI contains data on land cover and use, crops grown, land class, soil erosion, and other resource variables. Sample

points are cross-linked to soil map unit designations and soil attributes.

2. **The Conservation Technology Information Center (CTIC)** conducts the Crop Residue Management survey to estimate the portion of cropland managed under the various tillage systems. These are annual county-level surveys indicating acres of a specific crop planted by tillage system for each survey year (CTIC 1998).

3. **The Census of Agriculture** is conducted every 5 yr and collects county-level data that include crops grown by area, farm size, and crop yields, and many other types of management-related information.

Topographic databases. Digital elevation data can be used to derive slope, aspect, flow direction, flow accumulation, stream length, and topographic information, useful for modeling erosion and sediment movement as well as soil moisture, which impacts productivity and decomposition. Sources include topographic maps and digital elevation models (DEM) (<http://www.usgs.gov>).

Climatic databases. Climatic information is necessary for crop growth and ecosystem models used to calculate C inputs to soil, and soil C is influenced by both temperature and moisture, which are key variables in SOM models. Examples include the National Climate Data Center at <http://lwf.ncdc.noaa.gov/oa/ncdc.html>, the Parameter-Elevation Regressions on Independent Slopes Model (PRISM) (Daly, Neilson, and Phillips 1994), and the Vegetation/Ecosystem Modeling and Analysis Project (VEMAP) (Kittel et al. 1995).

Despite the relative variety and abundance of existing databases that can be applied at regional and national scales, however, crucial gaps in information remain. Statistically based inventory systems such as the NRI are extremely valuable—yet the NRI lacks key data on management practices such as tillage type and frequency and fertilizer and manure application rates, that if collected could enhance significantly the value of the other information on land use and crop type that are recorded. Even more striking is the relative paucity of regional and spatially explicit data on management practices on grazing lands, which as discussed earlier make up one of the largest areas of managed land in the United States and worldwide. Information on where different grazing systems are applied and on animal stocking rates, for example, as well as use of production inputs such as fertilizer and manure application would improve significantly the understanding of management activities and the im-

plications for C sequestration and GHG mitigation on these lands.

Future Database Improvements

The availability of more comprehensive and detailed spatial and geographic information will improve the accuracy of regional estimates of soil organic C sequestration. Furthermore, sensitivity analyses of C measurements in relation to environmental factors should help reveal the extent to which the accuracy of different spatial data layers is adequate or needs improvement. In this way—by improving the most sensitive spatial data layers—a strategy can be constructed for making more precise regional and national estimates of soil organic C content and changes. Several remote sensors, recently deployed or planned for the near future, will provide new, high-quality information to help assess changes in soil C, especially for regions where detailed geographic information does not exist yet. New products include those designed for producing high-resolution DEMs and for characterizing land-cover/land-use changes. A new global elevation dataset is being developed from the Jet Propulsion Laboratory's Shuttle Radar Topographic Mission, which was launched in September 1999. The mission is projected to create 30- and 100-meter (m) DEMs of the world, between the 60° N and 60° S latitudes, within 2 yr of the shuttle flight (<http://www-radar.jpl.nasa.gov/srtm/>). The MODerate-resolution Imaging Spectroradiometer (MODIS) sensor, on the Terra platform of the U.S. National Aeronautics and Space Administration's Earth Science Enterprise, has been in orbit around Earth since late 1999. This sensor provides additional capabilities for spatial and spectral resolution that are not possible with the current Advanced Very High Resolution Radiometer sensor, the basis for the current global land-cover datasets. Research using the MODIS sensor will enable interpretations of remotely sensed images in terms related to biophysical processes of C accumulation on the landscape (e.g., leaf area index, fraction of photosynthetically active radiation, and NPP).

The U.S. Geological Survey is investigating the hypothesis that erosion and sedimentation contribute to C sequestration. Geographic information (high-resolution DEM, land use, and soil type, in particular) is being incorporated into models to estimate erosion and sedimentation of soil and associated organic C on the landscape scale. A database of reservoir sedimentation is being used to benchmark the extrapolations of limited field data to the Mississippi River

Basin. Soil erosion models such as the WEPP model (Laflen, Lane, and Foster 1991) and the Universal Soil Loss Equation as well as soil organic C models such as CENTURY (Parton, Stewart, and Cole 1988) are being used in this study.

The suggestion has been made to extend the NRCS pedon database (NSSC 1997) to include more sampling of important "benchmark" soils that are spatially extensive and important for their C dynamics. Determinations of the amounts and decomposition rates of various organic matter fractions, such as the light fraction, microbial biomass, or particulate organic C, with distinct dynamic properties, also could be included in future analyses (Paul et al. 1997). Improvements in remote sensing and geographic information will allow for improved modeling of the regional distributions of soil organic C sources and sinks. An overview of current technology for quantifying soil C changes and likely developments during the next 20 years is given in Table 4.2.

Integration and Assessment Using Simulation Models

Soil organic matter process models and ecosystem simulation models provide a means of representing the dynamics of the C cycle (for changes in both time and space). Dynamic models are capable of including the many feedbacks and interactions of soil and plant processes affecting C cycling as well as external driving variables such as climate and management. Consequently, such models are valuable tools in extrapolating estimates of soil C changes across different environments and management conditions and for projecting changes in C stocks in response to possible future climate and management conditions. Dynamic models of soil C have been used in soil science and ecosystem research for more than 50 yr (see reviews by McGill [1996] and Paustian [1994]), but their application for analyzing C sequestration, particularly in a regional context, is more recent. Regional analyses using simulation models have been done in North America and Europe (e.g., Falloon et al. 1998, 2002), and similar efforts are under way elsewhere in the world. Certain specific case studies for regional analyses done in North America are described here.

Donigian and colleagues (1994) used a geographic information system (GIS) database with simulation modeling using the Century model (Parton et al. 1987) to estimate C sequestration potential in agricultural soils in the central United States, under several management change scenarios. They estimated that a continuation of current agricultural practices and trends during the next 30 yr would lead to C sequestration rates on the order of 25 to 50 teragrams (Tg) C/yr on 87 million ha (approximately 65% of U.S. cropland), driven by increases in productivity and trends toward reduced tillage. Further increases in soil C stocks were projected under scenarios of greater no-till and reduced-till adoption and use of cover crops.

A similar approach was used to estimate C stock

Table 4.2. Current and future technologies for monitoring soil C (adapted from Post et al. 1999)[1]

Technology	Current (2001–2003)	Midterm (2003–2007)	Long term (2008–2020)
Soil C measurements	Decrease sampling errors, improve root estimates, development of non-destructive C measurement devices	Nondestructive field measurement (experimentally deployed)	Nondestructive field measurement (routine, low cost)
Eddy flux	ca. 60 sites worldwide	Expanded network to characterize all significant land cover types	Routine, part of automated stations (when weather stations satisfy "upwind fetch" requirements) Verma, S. 1998. Personal communication. (low cost)
Remote sensing	Low-resolution LULC, absorbed PAR, hyperspectral (experimental), SAR (experimental)	High-resolution, satellite-based hyperspectral, SAR, models (experimental)	High-resolution, hyperspectral, SAR, models (routine)
C modeling	Models linked to databases; model intercomparisons	Models driven by RS input (experimental)	Real-time simulation of land processes driven by RS
C accounting	Databases, maps, census, models (experimental)	Databases, maps, census, models, new sensors (refinement)	Databases, maps, census, models, new sensors (operational)

[1]RS = Remote Sensing, LULC = Land Use and Land Cover, SAR = Synthetic Aperture Radar.

changes in Conservation Reserve Progam (CRP) grasslands during the initial 10-yr contract period (1985–1995), for a 16-state region in the central and northwestern United States, containing approximately 70% of the CRP acreage nationwide (Paustian et al. 2001). Climate, soil, and land-use data were used to delineate 59 climate/land-use zones, each including multiple soil textural classes. The model predicted that CRP lands were sequestering C and that regional differences in rates of C storage were controlled mainly by differences in primary production rates and secondarily by abiotic and soil factors influencing decomposition. Simulated rates of SOM C accumulation under CRP ranged from 10 to more than 40 g $C/m^2/$ yr, with the highest rates in the most humid regions. Total belowground C accumulation rates (including roots) ranged from 25 to 135 g $C/m^2/yr$. For the study region, predicted C increases for the first 10-yr period of the CRP were on the order of 25 Tg $(10^{12}$ g) for SOM C alone or up to 69 Tg if plant and litter C stocks were included. The model-based estimates were within the same range as independent estimates based on extrapolations of measured data for CRP grasslands (Paustian et al. 2001).

Current and potential soil C sequestration was assessed for the state of Iowa (Brenner et al. 2001; Paustian et al. 2002) using existing GIS-based resource data, simulation modeling and county-level surveys (Brenner et al. 2002a) of management practices, and land-use histories. The analysis suggested that Iowa's agricultural soils are presently a sink for CO_2, on the order of 3.1 Tg C/yr, primarily because of increasing crop residue inputs, implementation of the CRP, and increased adoption of conservation tillage

practices during the past two decades. Widespread adoption of no-till was predicted to increase soil gains potentially by 3 to 6 Tg C/yr, because more than 50% of the state's annual cropland remains under conventional, intensive tillage practices. Later the same approach was used to analyze current soil C dynamics and potential soil C sequestration in Indiana (Smith et al. 2002) and Nebraska (Brenner et al. 2002b).

Izaurralde and colleagues (2000) evaluated a number of simulation models (Table 4.3) against data from long-term experiments and selected the SOCRATES model (Grace and Ladd 1995) as most suitable to estimate regional soil C sequestration in western Canadian croplands. A major objective of the study was to compare methodologies and scaling effects for model-based regional analyses. Three aggregation procedures were used to compare estimates of C sequestration in two ecodistricts (E_1 and E_2) in Alberta, ranging from 650,000 to 750,000 ha, each characterized by numerous soil C climate-management combinations. A first estimate (M_1) was obtained by simulating changes in C storage for each major soil (14 soils in E_1 and 7 in E_2) and production system (e.g., dairy, cattle, pork, wheat, oilseed, or grain) present in an ecodistrict. The second method of estimation (M_2) simplified the aggregation process by using only the dominant soil of each ecodistrict and applying all management combinations to it. The most aggregated method (M_3) used only the dominant soil type and management system as representative of the entire ecodistrict. For ecodistrict E_1, all three aggregation methods predicted that C was being sequestered. For ecodistrict E_2, there were large discrepancies in mag-

Table 4.3. Evaluation of different scaling and data aggregation methods to estimate changes in soil C storage for two ecodistricts in Alberta, estimated by simulation modeling (from Izaurralde et al. 2000, used with permission)

Scaling approach[1]	Cultivated land	Improved rangeland	Unimproved rangeland	Total
	Change in soil C storage (Mg C/yr)			
Ecodistrict E_1				
M_1	32,600	2,600	-200	35,000
M_2	29,500	2,500	-1,100	30,900
M_3	27,900	2,600	-1,100	29,400
Ecodistrict E_2				
M_1	6,200	NA[2]	-5,200	1,000
M_2	1,100	NA	-5,300	-4,200
M_3	-6,600	NA	-5,300	-11,900

[1]Scaling method M_1 calculates changes for each major soil type and management system in the ecodistrict; method M_2 calculates changes for each management system but applies only the dominant soil type to the area of the ecodistrict; method M_3 calculates changes applying only the dominant management system and soil type to the entire area of the ecodistrict.

[2]NA = not applicable.

nitude and direction of soil C changes (Table 4.3). Each aggregation method was evaluated based on accuracy, sensitivity to nonnormal distributions of soil C climate-management combinations, and analysis effort required. The uneven areal distribution of changes in soil C storage in ecodistrict E_2 was identified as the source of the large discrepancies found among the three methods of aggregation. The study demonstrated that understanding the spatial distribution and interactions of soil C climate-management combinations and their impacts on model outputs is essential in selecting appropriate aggregation levels for regional analyses.

National-Level Accounting of Carbon Sequestration

Net C stock changes and emissions from agricultural soils for the United States have been estimated in order to meet the national GHG inventory reporting requirements for signatories of the United Nations Framework Convention on Climate Change. For national inventory purposes, a default methodology has been developed by the Intergovernmental Panel on Climate Change (IPCC 1997) that estimates C stock changes associated with land-use change and changes in land management. The IPCC method provides for categorizing land areas by climate, soil type, land use, and management system and includes a set of factors related to land-use conversions, tillage management, and plant C input levels that are used as multipliers to estimate soil C stock changes for a 20-yr inventory period. Organic soils are treated separately, with assigned climate and land-use-dependent C emission rates for organic soils that have been converted to agricultural use.

Eve et al. (2001, 2002) estimated net soil C stock changes for U.S. agricultural soils for 1982 to 1997 using the IPCC approach and data derived primarily from the NRI, the CTIC, the PRISM climate database, and the U.S. Department of Agriculture soil pedon database. Based on application of the IPCC inventory methods, Eve and colleagues (2002) estimated net soil C sequestration on U.S. agricultural lands at 21 Tg C/yr average throughout the 15-yr inventory period 1982 to 1997. Of this total, 6 Tg C/yr was sequestered on grazing lands whereas 15 Tg C/yr was stored in cropland soils. These estimates accounted for emissions (negative soil C storage) from organic soils as well as the sequestration (positive soil C storage) occurring on mineral soils (Figure 4.11). The increases in agricultural soil C stocks were attributable prima-

rily to increased adoption of conservation tillage, establishment and maintenance of the CRP, and decreased use of summer fallow in semiarid cropping systems. More recent estimates using revised parameters for the IPCC method, derived from analysis of U.S. long-term studies, suggest lower rates of C sequestration on mineral soils (15–20 Tg/yr) and higher losses from cultivated organic soils (5–10 Tg/yr), yielding net soil C sequestration rates of 5–10 Tg/yr (Ogle et al. 2003).

These estimates of current, on-going sequestration rates can be compared with various estimates of potential rates, given a widespread adoption of best-management practices, which are on the order of 75 to 150 Tg C/yr for cropland (Bruce et al. 1999; Lal et al. 1999) and 30 to 110 Tg C/yr on grazing land (Fol-
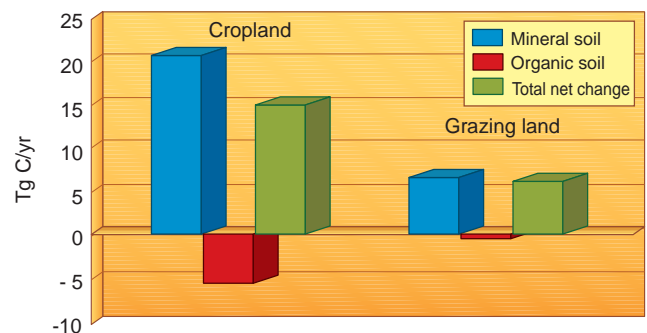
Figure 4.11. Net soil C changes associated with cropland and grazing land management and land use, estimated using the IPCC inventory methodology, the USDA/NRCS National Resources Inventory, and ancillary data sets. Positive values indicate gains of soil C on mineral soils, and negative values are for net C losses from managed organic soils. (Source: Eve et al. 2002.)

lett, Kimble, and Lal 2001). Sperow, Eve, and Paustian (2003) used the IPCC inventory procedure and default factor values to estimate maximum potential sequestration on cropland that could be derived from full adoption of no-till, enrolling all highly erodible lands in CRP, adding a winter cover crop where climatic conditions are favorable, and eliminating the use of bare summer fallow. Their analysis resulted in an estimated potential from these activities of approximately 100 Tg C/yr in the United States (Table 4.4, Figure 4.12) (Sperow, Eve, and Paustian 2003). Comparing actual and potential estimates using the same methodology (i.e., IPCC guidelines) suggests that current rates of C sequestration are less than 10–20% of their potential.

Table 4.4.   Potential C sequestration in U.S. agricultural soils by major management and land-use practices (from Sperow, Eve, and Paustian 2003)

Climate regime[1]	Baseline[2]	Set-aside of HEL cropland[3]	Use of winter cover crops[4]	Elimination of summer fallow[5]	Continuous no-till adoption[6]	Total
			Tg C/yr			
Dry	2.4	3.6	0	1.0	5.4	12.4
Humid	14.7	7.0	22.8	2.2	41.5	88.2
Total	17.1	10.6	22.8	3.2	46.9	100.6

[1]The dry climate regime denotes areas where potential evapotranspiration exceeds precipitation and mainly includes agricultural areas from the Central Great Plains and west to the Pacific Coast, except for the West Coast from central California northward and areas of higher elevation within the western United States. The humid climate regime includes all other areas, mainly consisting of the central and eastern United States.

[2]Baseline C sequestration represents the estimate of 1997 soil C stock changes in U.S. agricultural soils because of all agricultural practices, estimated using the IPCC inventory methodology.

[3]Assumes that all highly erodible land (HEL) not already enrolled in CRP would be set-aside and planted to perennial grasses. Total area would be 26 million hectares.

[4]Assumes use of winter cover crops on all annual cropped areas where sufficient soil moisture is available (the dry climate region was excluded because of limited water).

[5]Assumes all summer-fallow annual cropping (excluding HEL summer-fallow land allocated to set-aside) is converted to continuous annual crop rotations.

[6]Assumes adoption of no-till on all annual cropland (excluding HEL land allocated to set-aside).

Tg C/yr

☐ 0	◻ 1.0 – 1.5
◻ < 0.1	◼ 1.5 – 2.0
◻ 0.1 – 0.5	◼ 2.0 – 2.5
◼ 0.5 – 1.0	◼ > 2.5
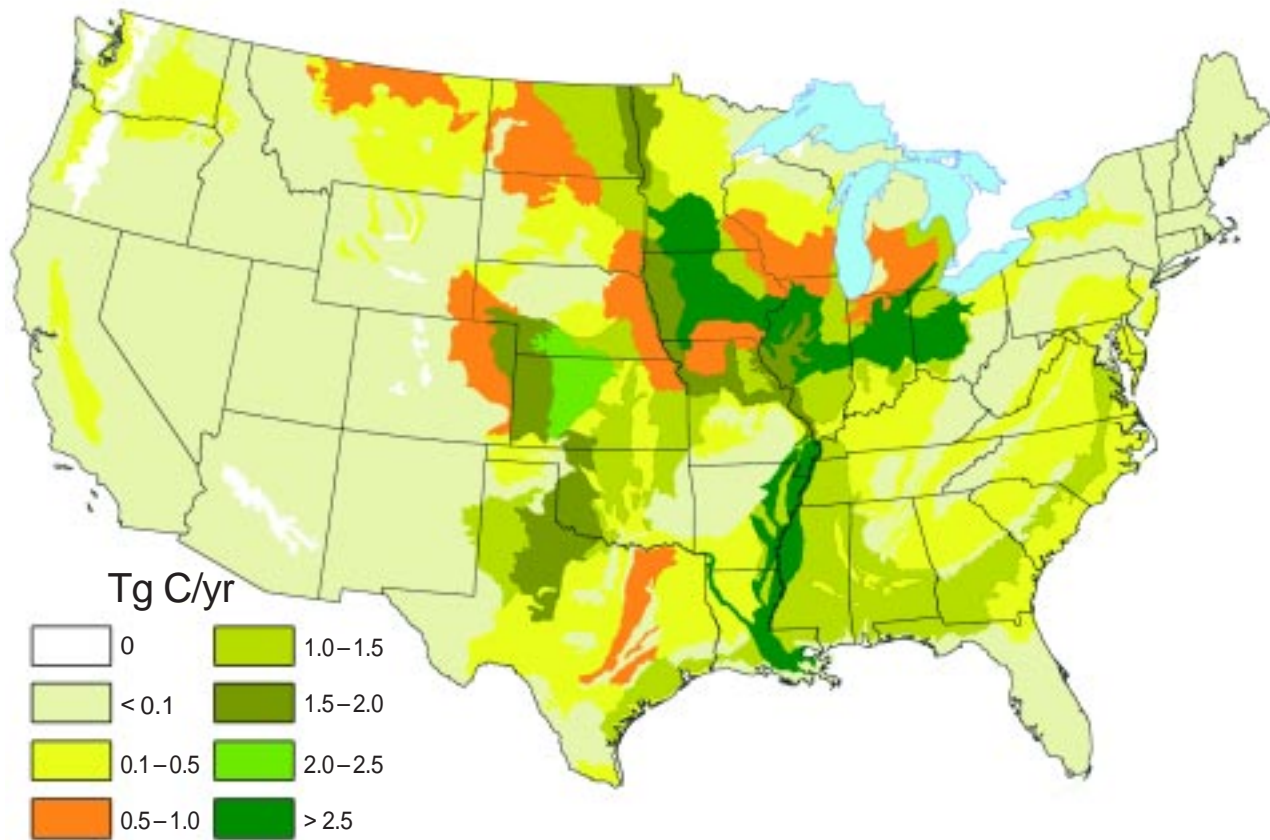
Figure 4.12.   Estimated C sequestration potential on U.S. croplands and its distribution by Major Land Resource Areas (MLRA) using the IPCC national inventory method, assuming widespread implementation of improved management practices for C sequestration (see Table 4.4). (Source: Sperow, Eve and Paustian 2003.)

Verification

For regional-based estimates of soil C emissions/sinks, whether for use as part of GHG inventories or for policy-related assessments of different mitigation strategies, independent verification is a crucial component. Ideally, verification procedures would encompass a comprehensive uncertainty analysis of every component that contributes to regional estimates of changes in C stocks. Combining uncertainty for each type of data included in the analysis can be used for calculating confidence intervals for C stock change estimates. Additionally, there are a number of issues to consider in developing a network of monitoring points for verification. An in-depth discussion of either approach is beyond the scope of this report. Instead, the authors briefly discuss multiple component uncertainty analysis and certain details to consider for development of a sampling-based verification network.

Uncertainty Analysis of Soil Carbon Changes

Although many of the databases (see preceding section) used as inputs for model-based extrapolations and predictions contain no explicit information about variability, methods exist to develop probability density functions from which measures of uncertainty can be developed (Rypdal and Winiwarter 2001; Smith and Heath 2001). Combining the uncertainty of each component used in extrapolation or modeling enables calculation of confidence intervals for estimated changes in C stocks. These calculations also enable investigation of the contribution of each component to overall uncertainty and can be used to target areas that would benefit the most from additional data. One recent example of this type of in-depth uncertainty analysis for C sequestration assessment is the recent U.S. national assessment, which used a revised version of the IPCC model to calculate changes in C stocks in U.S. agricultural lands between 1982 and 1997 (Ogle et al. 2003). The assessment used regression analysis to estimate probability density functions for management factors and reference soil C stocks in the IPCC model. These were combined with error estimates from the NRI, through a Monte Carlo simulation, to estimate 95% confidence limits for soil C changes. Ogle and colleagues (2003) found that improving estimates of tillage impacts on soil C stocks and C loss rates after tillage of organic soils would lead to the largest decrease in uncertainty. Further, decreasing uncertainty around initial C stock values and C input rates would have the smallest impact on over-all uncertainty. Single or multiple database extrapolations of soil C values (e.g., Ogle et al. 2003) or model-based extrapolations all can be evaluated using the established statistical tools of uncertainty analysis. Rigorous analyses will provide assessments of the data products and an evaluation of their utility for various purposes and may lead to ideas about where future sampling should be focused.

Monitoring Soil Carbon Changes

Targeted sampling, whether as part of a national network of permanent monitoring locations or for a specific C sequestration project, likely will be an important component of an overall verification scheme. As mentioned earlier, there are a number of studies and site networks with information on soil C stock changes from sampling (direct soil C sampling or through eddy covariance techniques) that can be used for independent verification of projected or extrapolated changes in soil C stocks. Although many sources of validation data exist, however, new sampling will be required to verify fully the changes in soil C stocks across the landscape. Establishing sampling locations will require careful consideration and may rely heavily on uncertainty analysis similar to that described previously. Frequency of sampling, methods of sample analysis (i.e., total organic C or organic C fractions), and intensity of sampling are important considerations. A recent study (Conant and Paustian 2002) that used C stock variability estimates from the national soil pedon database (NSSC 1997) suggested that for a large aggregate area (i.e., U.S. cropland), modest overall C stock changes (averaging 1.5 Mg C/ha) could be detected with a relatively small number (on a per-area basis) of sample points (n = 1,500). A recently completed study for western Canadian cropland reported statistically significant changes of soil C stocks in Saskatchewan during a 3-yr remeasurement period, based on 137 sampled fields (B. McConkey, personal communication 2002).

An efficient national soil C monitoring program will require a rigorous sampling scheme with well-understood limits before sample collection so that samples could be arrayed to maximize information gained while minimizing the number of samples required. As mentioned, changes in soil C stocks at any scale can be detected through direct measurement, but there is a trade-off between the number of samples collected and costs. A well-designed verification network will strive to maximize confidence in C stock estimates while minimizing costs.

5 Challenges and Opportunities for Agriculture

The debate on climate change—its potential effects and what can or should be done—will continue to evolve and change in the years to come, shaped by scientific discovery, emerging policies, and subsequent changes in public awareness and opinion. But regardless of how science and policy may direct the future discussion of climate change and greenhouse gas (GHG) mitigation, agriculture will be a focal point of concern. Agricultural producers and consumers will need to be informed and to be engaged in the debate.

Although uncertainties remain about both the rate and extent of human-induced climatic change and the types and severity of its impact on agriculture, there is strong consensus on certain points:

- Climate *is* changing, and even more significant changes are on the horizon. As stated in the National Academy of Sciences report to President Bush in 2001, "Greenhouse gases are accumulating in Earth's atmosphere as a result of human activities, causing surface air temperatures and subsurface ocean temperatures to rise."

- The possibility of increased climate variability and more extreme weather events is a significant concern, especially as it interacts with agricultural production. Multiple and interacting effects on crop productivity, livestock, water resources, insects and diseases, and management practices all need to be considered.

- Agriculture as a whole has evolved ways of dealing with variability in weather and climate, but adaptation to change—especially rapid change—may be costly.

There is less consensus on other points:

- Just how will precipitation, climate variability, and extreme events be influenced by increasing GHGs?

- How should the overall impacts of climate change on agriculture be assessed? The pervasive influence of climate on all biological systems makes such an assessment both complex and difficult.

Agriculture's challenge in helping to mitigate the increase in GHGs to the atmosphere is compelling.

- Agriculture itself is one of the primary sources of GHGs—globally, accounting for roughly 20% of the total greenhouse effect. In the United States, agriculture is a less dominant source, but still contributes approximately 7% of total U.S. emissions (from all GHGs, on a carbon [C]-equivalent basis).

- In nearly all cases, GHG emissions represent a loss of energy and/or nutrients that can be directly translated into productivity losses.

  ○ Soil organic matter losses (as carbon dioxide [CO_2]) equate with losses in soil fertility.

  ○ Nitrous oxide (N_2O) emissions represent wasted plant nutrients.

  ○ Livestock methane (CH_4) emissions represent a loss of energy and weight gain.

  ○ In addition, these wasted resources typically contribute to other forms of environmental degradation, such as soil erosion, surface water and coastal eutrophication, diminished groundwater quality, and other manifestations of water and air pollution.

- Reductions in GHG emissions typically will carry with them more efficient resource utilization on the part of producers and a broad array of environmental improvements.

- Practices that can decrease GHG emissions, and even offset emissions from other sectors of the economy (through C sequestration and biofuel production), are relatively well known but only used currently to a limited degree.

- Agricultural research throughout the past several decades has provided an array of field experimental data and basic research on soil C dynamics and soil trace gas emissions.

- Although much remains to be learned about the interactions of environmental and management

factors that control CO_2, N_2O, and CH_4 production, a reasonable accounting of current emissions and sinks at the national level and estimates of potential mitigation levels using the best existing practices is available.

- The potential mitigation from C sequestration, reduction in non-CO_2 GHG emissions, and biofuel offsets together represent a substantial amount, on the order of 100–300 million tonnes of C equivalent per year, equal to a reduction of approximately 5–15% of current total emissions in the United States.

Although technical potentials may be high, achievable mitigation levels are more uncertain, in part because of uncertainty about economic and policy constraints.

- For agricultural mitigation practices to be effective, farmers must adopt them, but there are many impediments to adoption, including:

  ◦ Technical capability.

  ◦ Social/behavioral factors.

  ◦ Economic costs. Simply put, many mitigation practices, under current economic conditions, either increase (or are perceived to increase) risk and/or decrease profits. For example, there is ample experimental as well as anecdotal evidence, in certain locations, of yield declines under the first few years of conversion to no-till practices. Although yields and net returns in these no-till systems may, over time, exceed those under conventional practices, the initial "conversion" costs present a barrier to adoption. In other areas, there is little or no initial yield penalty, and consequently no-till is more widely adopted.

- Reducing nitrogen (N) fertilizer applications in order to cut N_2O emissions likely will be perceived as adding risk. But if crop fertilizer use efficiency can be increased, then yields can be maintained and fertilizer costs decreased, resulting in net economic gains.

- For practices such as the establishment of conservation reserves, cropland is removed from production and income is lost unless producers are compensated directly.

As with many other environmental services, GHG mitigation suffers from the "failure of the commons," in that the atmosphere is treated as a "free-use" public good, where the costs associated with its use are excluded from the costs of production. Hence, public policy, acting in the interest of all of society, is required to value the atmosphere as a "commodity" and, in the instance of GHGs, to reward mitigation and discourage emissions. Such policies can take a variety of forms, from voluntary measures to legally binding limits, and can operate in domestic as well as international venues. Clearly the economic dimensions, and the potential opportunities (and hazards), for agricultural producers will vary significantly depending on the types of policies implemented. As an example, as of this writing (September 2003), C prices in the emissions trading market in the European Union (*Point Carbon* 2003) are approximately $10/tonne CO_2 equivalent. In Europe, companies and governments are operating on anticipated mandatory emission caps specified in the Kyoto Protocol. In the United States, where current policies are based on voluntary emission reductions, initial trades on the Chicago Climate Exchange average < $1/tonne CO_2 (Chicago Climate Exchange 2003).

Because GHGs stem from so many different sources, agriculture must "compete" with other sectors of the economy for mitigation opportunities.

- There is competition either for receiving government payments or in producing tradable emission offsets for the marketplace.

- Although emissions and potential sinks for agriculture are large in aggregate, the amounts for an individual farm are small compared with large point sources such as power plants.

- Emissions and sinks for GHGs in agriculture are characterized by being widely dispersed and diffuse (i.e., nonpoint source), compared with certain of the other major sources such as power plant emissions.

- These attributes have implications in implementing agricultural GHG mitigation strategies, including measurement and monitoring, offset valuation, and contractual arrangements.

Compared with well-defined point sources, there are greater challenges associated with measuring and monitoring agricultural GHG emission levels and emission reductions and in verifying compliance with targeted reductions, but progress is being made on several fronts:

- A variety of methods for measuring soil C sequestration and GHG emissions currently exist, and development and refinement of new methods are occurring at a rapid pace.

- Quantification methods will vary as a function of scale and policy/market designs—in most cases a

hybrid approach of modeling and direct measurements likely will be most successful.

• As quantification methods improve throughout time, with additional research and pilot applications, the uncertainty of estimates will diminish, effectively increasing the value and appeal of agricultural mitigation options.

Other issues that will influence the implementation of agricultural GHG mitigation policies include the following:

• Permanence, leakage, and sink capacity. None of these issues is a "show stopper," and a variety of mechanisms including discounting, annuity and rental contracts, and comprehensive accounting can be used to deal with them.

• Carbon sequestration is by its nature subject to a finite capacity.

• There are biological limits to how much N_2O and CH_4 emissions can be decreased.

Although agricultural biofuels can play a longer-term role, it seems clear that the greatest opportunities for agricultural mitigation of GHGs are in the near term—during the next few decades. Fortunately, this time frame meshes well with the fact that agricultural mitigation practices are among the most readily implementable in the near term, in contrast to the fundamental changes in the energy supply infrastructure ultimately required to replace fossil fuels and stabilize CO_2 concentrations in the atmosphere.

In conclusion, agriculture can and should play a role in the debate about and solutions to climate change and GHG increases. The extent of that role is as yet impossible to predict given that outcomes, to a large degree, depend on economic, social, and political decisions that will be made in the near future and in years to come. But the state of the science and emerging policy environment as well as the enlightened self-interest of the agricultural industry argue for a prominent role for agriculture.

Appendix A: Abbreviations and Acronyms

μg	microgram
μm	micrometer or micron (10^{-6} m)
C	carbon
Ca	calcium
CAA	carbon annuity account
CCC	Canadian Climate Centre
CFC	chlorofluorocarbon
CH_4	methane
cm	centimeter
CO_2	carbon dioxide
CRP	Conservation Reserve Program
CTIC	Conservation Technology Information Center
DEM	digital elevation model
DOC	dissolved organic carbon
DOE	U.S. Department of Energy
EPA	U.S. Environmental Protection Agency
EPIC	erosion productivity impact calculator
FACE	free-air CO_2 enrichment
FTIR	Fourier transform infrared
g	gram (0.002 pounds)
GCM	global climate model; also called general circulation model
Gg	gigagram (10^9 g)
GHG	greenhouse gas
GWP	Global Warming Potential
ha	hectare (2.47 acres)
HFC	hydrofluorocarbon
IPCC	Intergovernmental Panel on Climate Change
IR	infrared
K	potassium
kg	kilogram
m	meter
Mg	magnesium; megagram (10^6 g) = 1 metric tonne
mm	millimeter (10^{-3} m)

mmt	million metric tonne
MODIS	moderate-resolution imaging spectroradiometer
N	nitrogen
N_2	dinitrogen gas
N_2O	nitrous oxide
NEE	net ecosystem exchange
NEP	net ecosystem production
NH_3	ammonia gas
NH_4^+	ammonium ion
NO_3^-	nitrate ion
NO_x	sum of nitric oxide (NO) and nitrogen dioxide (NO_2)
NPP	net primary production
NRCS	Natural Resources Conservation Service
NRI	National Resources Inventory
O_2	oxygen gas
O_3	ozone
OH	hydroxyl radical
PAYG	pay-as-you-go
PFC	perfluorocarbon
Pg	petagram (10^{15} g) = 1 billion metric tonnes
ppb_v	parts per billion volume
ppm_v	parts per million volume
RUSLE	Revised Universal Soil Loss Equation
SF_6	sulfur hexafluoride
SOC	soil organic carbon
SOM	soil organic matter
TDL	tunable diode laser
Tg	teragram (10^{12} g) = 1 million metric tonnes
USDA	U.S. Department of Agriculture
VLC	variable length contract
WEPP	Water Erosion Prediction Project
WEPS	Wind Erosion Prediction System
WTO	World Trade Organization
yr	year

Appendix B: Glossary

Adoption. Individual use of a new technology; usually implies changing from an existing management practice to a newer one.

Banded. Placed in the soil, within crop rows (e.g., fertilizer).

Biofuels. Renewable energy sources derived from contemporary biological processes.

C_3. Denotes plants having the dominant photosynthetic pathway in which CO_2 is initially fixed into a 3-carbon compound (3-Phosphoglycerate) as a precursor to glucose synthesis. These plants include most trees and the majority of crop plants, cool-season grasses, and legumes. The C_3 pathway has a lower saturation threshold for CO_2 (compared with the C_4 pathway) and hence has a high direct response of photosynthesis to increased CO_2 concentrations.

C_4. Denotes plants having a photosynthetic pathway in which CO_2 initially is fixed into a 4-carbon compound (oxaloacetate) as a precursor to glucose synthesis. These plants include certain crops of tropical origin (e.g., maize, sorghum) and warm-season grasses. C_4 plants show a lesser photosynthetic response to increased CO_2 concentration than C_3 plants do.

Carbon saturation. A theoretical point at which mineral soil could not increase organic carbon stocks even if rates of addition of organic residues were increased.

CO_2 fertilization effect. Process whereby the elevated atmospheric CO_2 level stimulates photosynthesis, which in turn increases plant growth rates.

Denitrification. Reduction of soil NO_3^- to gaseous N compounds (e.g., N_2O, N_2) by microorganisms using NO_3^-, in the absence of O_2, as an electron acceptor during respiration.

Diffusion. Aggregate adoption of a new technology.

Equilibrium state. The state wherein C input balances losses such that the rate of change of soil organic C is zero and soil organic C remains constant if conditions governing organic matter input rates and decomposition rates remain constant.

Forcing (radiative). Externally imposed perturbations of the Earth's radiative energy balance because of secular changes in factors such as concentrations of greenhouse gases and aerosols, changes in incident solar irradiance, and changes in the radiative energy absorbed by the Earth's surface.

Full greenhouse gas accounting. Accounting for net emissions and sinks from all major GHGs (CO_2, N_2O, and CH_4).

Full land accounting. Accounting for any parcel of land where land use and/or management activities affect GHG emissions or sinks.

Global mass balance of gas. The difference between emissions into the atmosphere and losses to terrestrial and ocean sinks from the atmosphere.

Global Warming Potential. A measure of the cumulative radiative forcing of various GHGs relative to CO_2 (as a reference gas), throughout a specific time horizon such as 20 or 100 years. Usually expressed as CO_2 equivalents.

Greenhouse effect. Retention of long-wave radiation from the Earth's surface by absorption and reemission by atmospheric trace constituents, thus heating the lower atmosphere and Earth surface, causing it to become warmer than it would be if direct solar radiation were the only source of energy.

Greenhouse gases. Trace gases found in the atmosphere that absorb and emit infrared radiation, resulting in the greenhouse effect. These gases include naturally occurring gases such as carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), water vapor (H_2O), and ozone (O_3), and several man-made gases such as chlorofluorocarbons (CFCs), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulphur hexafluoride (SF_6).

Gross primary production. Total CO_2 plant-uptake rate.

Halocarbon. Carbon compounds that contain fluorine, chlorine, bromine, or iodine that can exist as trace gases in the atmosphere (including CFCs, HFCs, and PFCs). Although it does not contain carbon, SF_6 often is included in this group when discussing man-made GHGs.

Humify. The conversion by soil organisms of organic residues into more stable organic compounds.

Life cycle analysis. A comprehensive, "cradle-to-grave" accounting of all process outputs (e.g., energy production) and environmental impacts of a product or technology, typically done as a side-by-side comparison of alternative technologies (e.g., fuel or energy produced from biomass versus the same commodities produced from petroleum).

Methanogen. A type of bacteria that can produce CH_4 under anaerobic conditions.

Methanotroph. A type of bacteria that can consume CH_4 under aerobic conditions.

Multifunctionality. The quality of enhancing multiple environmental attributes and meeting other social goals.

Net Ecosystem Production. A measure of the net rate of C accumulation of an ecosystem, calculated as net primary production minus C losses from decomposition.

Net Primary Production. The fraction of gross primary produc-

tion remaining after CO_2 is returned to the atmosphere by plant respiration, equivalent to the rate at which photosynthetically fixed C is incorporated into plant tissues.

Nitrification. The aerobic oxidation of soil NH_4^+ to nitrite and then to NO_3^- by microorganisms.

Nitrous oxide mole fraction. The proportion of denitrification end products emitted as nitrous oxide, expressed as $N_2O/[N_2O+N_2]$.

Permanence. A relative measure of the time period during which C removed from the atmosphere will remain in place and not be reemitted to the atmosphere as well as the relative ease by which C taken up by a sink can be reemitted.

Photosynthesis. The process by which CO_2 is assimilated by plants, using energy from sunlight, and then is converted to organic compounds used to synthesize biomass.

Precision farming. Management that matches resource application (e.g., fertilizer and pesticide rates) and agronomic practices with soil attributes and crop requirements as they vary across a field. This typically involves use of global positioning satellite (GPS) technology for spatial referencing of equipment, detailed mapping and data collection (e.g., yield monitors on harvest equipment) to quantify within-field variability, and equipment designed for variable rate application of management inputs.

Radiative forcing. *See* **Forcing**

Rumen. The large forestomach of certain grazing animals, where anaerobic bacteria help to break down ingested plant material.

Sink. Biological and chemical processes that remove a gas from the atmosphere.

Sink permanence. *See* **Permanence**

Soil respiration. The rate of emission of CO_2 produced by soil organisms and plant roots.

Stomates. Small openings in leaf surfaces through which CO_2 is absorbed and H_2O released.

Threshold approach. View of farmers as heterogeneous; an assertion that at each moment only the subgroup of farmers for whom a technology is profitable will adopt it.

Time horizon. The time period during which the effects of increased greenhouse gases on radiative forcing are integrated (i.e., in calculating Global Warming Potential).

Tonne-year accounting method. One tonne of carbon (or carbon equivalent) held out of the atmosphere for 1 year. This measure serves as a basis for a system that can place comparable values on carbon emissions avoided today or in the future and carbon stored for differing lengths of time.

Trace gases. Greenhouse gases (e.g., water vapor, carbon dioxide, nitrous oxide, methane, and ozone).

Literature Cited

Achorn, F. P. and M. F. Broder. 1984. Mechanics of applying nitrogen fertilizer. Pp. 493–506. In R. D. Hauck (ed.). *Nitrogen in Crop Production.* Soil Science Society of America, Madison, Wisconsin.

Acock, B. and L. H. Allen, Jr. 1985. Crop responses to elevated carbon dioxide concentrations. Pp. 53–97. In B. R. Strain and J. D. Cure (eds.). *Direct Effects of Increasing Carbon Dioxide on Vegetation.* DOE/ERB0238. U.S. Department of Energy, Washington, D.C.

Adams, D. M., R. J. Alig, B. A. McCarl, J. M. Callaway, and S. M. Winnett. 1999. Minimum cost strategies for sequestering carbon in forests. *Land Econ* 75(3):360–374.

Adams, R. M., B. H. Hurd, and J. Reilly. 1999. *Agriculture and Global Climate Change: A Review of Impacts to U.S. Agricultural Resources.* Pew Center on Global Climate Change, Washington, D.C.

Adams, R. M., B. Hurd, S. Lenhart, and N. Leary. 1998. The effects of global warming in agriculture: An interpretive review. *J Climate Res* 11:19–30.

Adams, R. M., C. Rosenzweig, R. M. Peart, J. T. Richie, B. A. McCarl, J. D. Clyer, R. B. Curry, J. W. Jones, K. J. Boote, and L. H. Allen. 1990. Global climate change and U.S. agriculture. *Nature* 345:219–224.

Adu, J. K. and J. M. Oades. 1978. Physical factors influencing decomposition of organic materials in soil aggregates. *Soil Biol Biochem* 10:109–115.

Agriculture and Agri-Food Canada (AAFC). 1999. *The Health of Our Air. Toward Sustainable Agriculture in Canada.* P. 40. In H. H. Janzen, R. L. Desjardins, J. M. R. Asselin, and B. Grace (eds.). Publication 1981/E. Agriculture and Agri-Food Canada, Ottawa, Ontario, Canada.

Allen, L. H., Jr., K. J. Boote, J. W. Jones, P. H. Jones, R. R. Valle, B. Acock, H. H. Rogers, and R. C. Dahlman. 1987. Response of vegetation to rising carbon dioxide: Photosynthesis, biomass, and seed yield of soybean. *Global Biogeochem* Cycles 1:1–14.

Allison, F. E. 1973. *Soil Organic Matter and Its Role in Crop Production.* Developments in Soil Science 3. Elsevier, Amsterdam. 637 pp.

Allmaras, R. R., H. H. Schomberg, C. L. Douglas, Jr., and T. H. Dao. 2000. Soil organic carbon sequestration potential of adopting conservation tillage in U.S. croplands. *J Soil Water Conserv* 55:365–373.

Ambus, P. and G. P. Robertson. 1998. Automated near-continuous measurement of carbon dioxide and nitrous oxide fluxes from soil. *Soil Sci Soc Am* 62:394–400.

AmeriFlux. 2000. Homepage, <http://public.ornl.gov/ameriflux/Participants/Sites/Map/index.cfm> (17 December 2003)

Anderson, T. H. and K. H. Domsch. 1989. Ratios of microbial biomass carbon to total organic carbon in arable soils. *Soil Biol Biochem* 21:471–479.

Andrews, J. A. and W. H. Schlesinger. 2001. Soil CO_2 dynamics, acidification and chemical weathering in a temperate forest with experimental CO_2 enrichment. *Global Biogeochem Cycles* 15:149–162.

Antle, J. and R. Just. 1991. Effects of commodity program structure on resource use and the environment. Pp. 97–127. In N. Bockstael and R. E. Just (eds.). *Commodity and Resource Policies in Agricultural Systems.* Springer-Verlag, New York, New York.

Antle, J. and S. Mooney. 2002. Designing efficient policies for agricultural soil carbon sequestration. Pp. 323–336. In J. M. Kimble, R. Lal, and R. F. Follett (eds.). *Agriculture Practices and Policies for Carbon Sequestration in Soil.* CRC Press, Boca Raton, Florida.

Babcock, B. A., J. Wu, P. G. Lakshminarayan, and D. Zilberman. 1996. The economics of a public fund for environmental amenities. *Am J Agric Econ* 78:961–971.

Baldocchi, D. D., B. B. Hicks, and T. P. Meyers. 1988. Measuring biosphere-atmosphere exchanges of biologically related gases with micrometeorological methods. *Ecology* 69:1331–1340.

Baldwin, R. L., J. M. Thornley, and D. E. Beever. 1987. Metabolism of the lactating cow. II. Digestive elements of a mechanistic model. *J Dairy Res* 54:107–131.

Banerjee, N. K. and A. R. Mosier. 1989. Coated calcium carbide as a nitrification inhibitor in upland and flooded soils. *J Indian Soc Soil Sci* 37:306–313.

Bauer, C. 1997. Bringing water markets down to earth: The political economy of water rights in Chile, 1976–95. *World Dev* 25:639–656.

Baumol, W. and W. Oates. 1988. *The Theory of Environmental Policy.* Cambridge University Press, Cambridge, U.K.

Beare, M. H., M. L. Cabera, P. F. Hendrix, and D. C. Coleman. 1994. Aggregate-protected and unprotected organic matter pools in conventional- and no-tillage soils. *Soil Sci Soc Am J* 58:787–795.

Berck, P. and W. R. Bentley. 1997. Hotelling's theory, enhancement, and the taking of the Redwood National Park. *Am J Ag Econ* 79:287–298.

Birkhaeuser, D., R. E. Evenson, and G. Feder. 1991. The economic impact of agricultural extension: A review. *Econ Dev Cult Change* 39:607–650.

Black, A. L. and D. L. Tanaka. 1997. A conservation tillage-cropping systems study in the Northern Great Plains of the United States. Pp. 335–342. In E. A. Paul, K. Paustian, E. T. Elliot, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America.* CRC Press, Boca Raton, Florida.

Bliss, N. B., S. W. Waltman, and G. W. Petersen. 1995. Preparing a soil carbon inventory for the United States using geographic information system. Pp. 275–295. In R. Lal, J. Kimble, E. Levine, and B. A. Stewart (eds.). *Advances in Soil Science: Soil and Global Change.* CRC Press, Lewis Publishers, Boca Raton, Florida.

Bohn, H. 1978. On organic soil C and CO_2. *Tellus* 30:472–475.

Boone, D. R. 1991. Biological formation and consumption of methane. Pp. 102–127. In M. A. K. Khalil (ed.). *Atmospheric Methane: Sources, Sinks and Role in Global Change*. North Atlantic Treaty Organization Advanced Science Institutes Series, Vol. 13. Springer-Verlag, Berlin, Germany.

Bouwman, A. F. 1996. Direct emission of nitrous oxide from agricultural soils. *Nutr Cycling Agroecosystems* 46:53–70.

Bouwman, A. F. and W. G. Sombroek. 1990. Inputs to climate change by soils and agriculture related activities: Present status and possible future trends. Pp. 15–30. In H. W. Scharpenseel, M. Schomaker, and A. T. Ayoub (eds.). *Soils on a Warmer Earth*. Special Volume. Developments in Soil Science 20. Elsevier, Amsterdam, The Netherlands.

Bowden, R. D., J. M. Melillo, and P. A. Steudler. 1991. Effects of nitrogen additions on annual nitrous oxide fluxes from temperate forest soils in the northeastern United States. *J Geophys Res* 96:9321–9328.

Braden, J. and K. Segerson. 1993. Information problems in the design of nonpoint-source pollution policy. Pp. 1–36. In C. S. Russell and J. F. Shogren (eds.). *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution*. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Branine, M. E. and D. E. Johnson. 1990. Level of intake effects on ruminant methane loss across a wide range of diets. *J Anim Sci* 68:509–510.

Brenner, J., K. Paustian, G. Bluhm, J. Cipra, M. Easter, E. T. Elliott, T. Kautza, K. Killian, J. Schuler, and S. Williams. 2001. *Quantifying the Change in Greenhouse Gas Emissions Due to Natural Resource Conservation Practice Application in Iowa*. Final report to the Iowa Conservation Partnership. Colorado State University Natural Resource Ecology Laboratory and the U.S. Department of Agriculture Natural Resources Conservation Service, Fort Collins, Colorado.

Brenner, J., K. Paustian, G. Bluhm, K. Killian, J. Cipra, B. Dudek, S. Williams, and T. Kautza. 2002a. Analysis and reporting of carbon sequestration and greenhouse gases for conservation districts in Iowa. Pp. 127–134. In J. M. Kimble, R. Lal, and R. F. Follett (eds.). *Agriculture Practices and Policies for Carbon Sequestration in Soil*. CRC Press, Boca Raton, Florida.

Brenner, J., K. Paustian, G. Bluhm, J. Cipra, M. Easter, R. Foulk, K. Killian, R. Moore, J. Schuler, P. Smith, and S. Williams. 2002b. Quantifying the change in greenhouse gas emissions due to natural resource conservation practice application in Nebraska. Final report to the Nebraska Conservation Partnership. Colorado State University Natural Resource Ecology Laboratory and USDA Natural Resources Conservation Service, Fort Collins, Colorado.

Bronson, K. F. and A. R. Mosier. 1991. Effect of encapsulated calcium carbide on dinitrogen, nitrous oxide, methane and carbon dioxide emissions from flooded rice. *Biol Fertil Soils* 11:116–120.

Bronson, K. F. and A. R. Mosier. 1993. Nitrous oxide emissions and methane consumption in wheat and corn-cropped systems. Pp. 133–144. In L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston (eds.). *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*. American Society of Agronomy Special Publication 55. American Society of Agronomy/ Crop Science Society of America/Soil Science Society of America, Madison, Wisconsin.

Bronson, K. F., H. U. Neue, U. Singh, and E. B. Abao, Jr. 1997. Automated chamber measurements of methane and nitrous oxide flux in flooded rice soils. I. Fallow period emissions. *Soil Sci Soc Am J* 61:988–993.

Brown, R. A. and N. J. Rosenberg. 1999. Climate change impacts on the potential productivity of corn and winter wheat in their primary United States growing regions. *Clim Change* 41:73-107.

Brown, S. and A. E. Lugo. 1990. Changes in carbon and nitrogen contents of tropical soils from forest clearing and succession in Puerto Rico and U.S. Virgin Islands. *Plant Soil* 124:53–64.

Bruce, J. P., M. Frome, E. Haites, H. Janzen, R. Lal, and K. Paustian. 1999. Carbon sequestration in soils. *J Soil Water Conserv* 54:382–389.

Brumme, R. and F. Beese. 1992. Effects of liming and nitrogen fertilization on emissions of CO_2 and N_2O from a temperate forest. *J Geophys Res* 97:851–858.

Buckman, H. O. and N. C. Brady. 1969. *The Nature and Properties of Soils*. 7th ed. Collier-Macmillan, London, England. 653 pp.

Buol, S. W., F. D. Hole, R. J. McCracken, and R. J. Southard. 1997. *Soil Genesis and Classification*. 4th ed. Iowa State University Press, Ames.

Buringh, P. 1984. Organic carbon in soils of the world. Pp. 41–109. In G. M. Woodwell (ed.). *The Role of Terrestrial Vegetation in the Global Carbon Cycle*. SCOPE 23, J. Wiley and Sons, Chichester, U.K.

Burke, I. C., C. M. Yonder, W. J. Parton, C. V. Cole, K. Flach, and D. S. Schimel. 1989. Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils. *Soil Sci Soc Am J* 53:800–805.

Burke, I. C., D. S. Schimmel, C. M. Yonker, W. J. Parton, L. A. Joyce, W. K. Lauenroth. 1990. Regional modeling of grassland biogeochemistry using GIS. *Landscape Ecol* 4:45–54.

Burke, I. C., W. K. Lauenroth, R. Riggle, P. Brannen, B. Madigan, and S. Beard. 1999. Spatial variability of soil properties in a shortgrass steppe: The relative importance of topography, grazing, microsite, and plant species in controlling spatial patterns. *Ecosystems* 2:422–438.

Businger, J. A. and S. P. Oncley. 1990. Flux measurements with conditional sampling. *J Atmos Ocean Tech* 7:349–352.

Buyanovsky, G. A. and G. H. Wagner. 1998. Carbon cycling in cultivated land and its global significance. *Global Change Biol* 4:131–142.

Campbell, C. A., G. P. Lafond, A. P. Moulin, L. Townley-Smith, and R. P. Zentner. 1997. Crop production and soil organic matter in long-term crop rotations in the sub-humid northern Great Plains of Canada. Pp. 297–315. In E. A. Paul, K. Paustian, E. T. Elliott, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Carter, M. R. (ed.). 1993. *Soil Sampling and Methods of Analysis*. Canadian Society of Soil Science, Lewis Publishers, Ann Arbor, Michigan.

Carter, M. R. 1994. Strategies to overcome impediments to adoption of conservation tillage. Pp. 3–19. In M. R. Carter (ed.). *Conservation Tillage in Temperate Agroecosystems*. CRC Press, Boca Raton, Florida.

Cavigelli, M. A. and G. P. Robertson. 2000. The functional significance of denitrifier community composition in a terrestrial ecosystem. *Ecology* 81:1402–1414.

Chen, C. C. and B. A. McCarl. 2001. Pesticide usage as influenced by climate: A statistical investigation. *Climatic Change* 50:475–487.

Chen, C. C., D. Gillig, and B. A. McCarl. 2001. Effects of climatic change on a water dependent regional economy: A study of the Texas Edwards Aquifer. *Climatic Change* 49:397-409.

Chen, C. C., B. A. McCarl, and R. M. Adams. 2001. Economic implications of potential climate change induced ENSO frequency and strength shifts. *Climatic Change* 49:147–159.

Chen, R. F., D. B. Chadwick, and S. H. Lieberman. 1997. The application of time-resolved spectrofluorometry to measuring benthic fluxes of organic compounds. *Org Geochem* 26:67–77.

Chicago Climate Exchange. 2003. Homepage, <http://www.chicagoclimatex.com/> (9 December 2003)

Chomitz, K. M. 1998. *Baselines for Greenhouse Gas Reductions: Problems, Precedents, Solutions*. Carbon Offsets Unit, World Bank, Washington, D.C.

Christensen, S., P. Ambus, J. R. M. Arah, H. Clayton, B. Galle, D. W. T. Griffith, K. J. Hargreaves, L. Klemedtson, A.-M. Lind, M. Maag, A. Scott, U. Skiba, K. A. Smith, M. Welling, and F. G. Wienhold. 1996. Nitrous oxide emission from an agricultural field: Comparison between measurements by flux chamber and micrometeorological techniques. *Atmos Environ* 30:4183–4190.

Cicerone, R. J., J. D. Shetter, and C. C. Delwiche. 1983. Seasonal variation of methane flux from a California rice paddy. *J Geophys Res* 88:7203–7209.

Coakley, S. M., H. Scherm, and S. Chakraborty. 1999. Climate change and plant disease management. *Annu Rev Phytopathol* 37:399–426.

Cole, C. V., K. Paustian, E. T. Elliott, A. K. Metherell, D. S. Ojima, and W. J. Parton. 1993. Analysis of agroecosystem carbon pools. *Water Air Soil Pollut* 70:357–371.

Cole, C. V., C. Cerri, K. Minami, A. Mosier, N. Rosenberg, D. Sauerbeck, J. Dumanksi, J. Duxbury, J. Freney, R. Gupta, O. Heinemeyer, T. Kolchugina, J. Lee, K. Paustian, D. Powlson, N. Sampson, H. Tiessen, M. van Noordwijk, and Q. Zhao. 1996. Agricultural options for mitigation of greenhouse gas emissions. Pp. 744–771. In R. T. Watson, M. C. Zinyowera, and R. H. Moss (eds.). *Climate Change, 1995. Impacts, Adaptations, and Mitigation of Climate Change. Scientific-Technical Analyses*. Contribution of Working Group II to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.

Cole, C. V., J. Duxbury, J. Freney, O. Heinmeyer, K. Minami, A. Mosier, K. Paustian, N. Rosenberg, N. Sampson, D. Sauerbeck, and Q. Zhao. 1997. Global estimates of potential mitigation of greenhouse gas emissions by agriculture. *Nutr Cycling Agroecosyst* 49:221–228.

Collins, H. P., E. T. Elliott, K. Paustian, L. G. Bundy, W. A. Dick, D. R. Huggins, A. J. M. Smucker, and E. A. Paul. 2000. Soil carbon pools and fluxes in long-term Corn Belt agroecosystems. *Soil Biol Biochem* 32:157–168.

Conant, R. T. and K. Paustian. 2002. Spatial variability of soil organic carbon in grasslands: Implications for detecting change at different scales. *Environ Pollut* 116:413–422.

Conant, R. T., K. Paustian, and E. T. Elliott. 2001. Grassland management and conversion into grasslands: Effects on soil carbon. *Ecol Appl* 11:343–355.

Conservation Technology Information Center (CTIC). 1998. *1998 Crop Residue Management Executive Summary*. Conservation Technology Information Center, West Lafayette, Indiana.

Council for Agricultural Science and Technology (CAST). 1992. *Preparing U.S. Agriculture for Global Climate Change*. Task Force Report 119. Council for Agricultural Science and Technology, Ames, Iowa. 96 pp.

Council for Agricultural Science and Technology (CAST). 2000. *Storing Carbon in Agricultural Soils to Help Mitigate Global Warming*. Issue Paper 14. Council for Agricultural Science and Technology, Ames, Iowa. 8 pp.

Cremers, D. A., M. H. Ebinger, D. D. Breshears, P. J. Unkefer, S. A. Kammerdiener, M. J. Ferris, K. M. Catlett, and J. R. Brown. 2001. Measuring total soil carbon with laser-induced breakdown spectroscopy (LIBS). *J Environ Qual* 30:2202–2206.

Crill, P. M., J. H. Butler, D. J. Cooper, and P. C. Novelli. 1995. Standard analytical methods for measuring trace gases in the environment. Pp. 164–205. In P. A. Matson and R. C. Harris (eds.). *Biogenic Trace Gases: Measuring Emissions from Soil and Water*. Blackwell Scientific Publishing, London.

Cropper, M. and W. Oates. 1992. Environmental economics: A survey. *J Econ Lit* 30:675–740.

Crutzen, P. J., I. Aselmann, and W. Seiler. 1986. Methane production by domestic animals, wild ruminants, other herbivorous fauna, and humans. *Tellus* 38:271–284.

Culley, J. L. B. 1993. Density and compressibility. Pp. 529–539. In M. R. Carter (ed.). *Soil Sampling and Methods of Analysis*. Canadian Society of Soil Science, Lewis Publishers, Ann Arbor, Michigan.

Cure, J. D. and B. Acock. 1986. Crop responses to carbon dioxide doubling: A literature survey. *Agric For Meteorol* 38:127–145.

Dahlsten, D. L. and R. Garcia (eds.). 1989. *Eradication of Exotic Pests: Analysis with Case Histories*. Yale University Press, New Haven, Connecticut.

Daly, C., R. P. Nielson, and D. L. Phillips. 1994. A statistical-topographic model for mapping climatological precipitation over mountainous terrain. *J Appl Meteorol* 33:140–158.

Darwin, R., M. Tsigas, J. Lewandrowski, and A. Raneses. 1995. World agriculture and climate change: Economic adaptations. Agricultural Economic Report 703. Natural Resources and Environmental Division, Economic Research Service, U.S. Department of Agriculture, Washington, D.C.

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

Delmas, R. 1993. An overview of present knowledge of methane emissions from biomass burning. Pp. 81–91. In A. R. van Amstel (ed.). *Methane and Nitrous Oxide*. RIVM Report 481507003. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Denmead, O. T. 1979. Chamber method for measuring nitrous oxide emissions from soils in the field. *Soil Sci Soc Am J* 43:89–95.

Denmead, O. T. 1994. Measuring fluxes of CH_4 and N_2O between agricultural systems and the atmosphere. Pp. 209–234. In K. Minami, A. Mosier, and R. L. Sass (eds.). *CH_4 and N_2O: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*. National Institute for Agro-Environmental Science, Yokendo Publishers, Tokyo.

Denmead, O. T. and M. R. Raupach. 1993. Methods for measuring atmospheric gas transport in agricultural and forest systems. Pp. 19–43. In L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston (eds.). *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*. American Society of Agronomy Publication 55. American Society of Agronomy/Crop Science Society of America/Soil Science Society of America, Madison, Wisconsin.

Denmead, O. T., J. R. Simpson, and J. R. Freney. 1977. A direct field measurement of ammonia emission after injection of anhydrous ammonia. *Soil Sci Soc Am J* 41:827–828.

Denmead, O. T., M. R. Raupach, F. X. Dunin, H. A. Cleugh, and R. Leuning. 1996. Boundary layer budgets for regional estimates of scalar fluxes. *Global Change Biol* 2:255–264.

Desjardins, R. L. 1992. Review of techniques to measure CO_2 flux

densities from surface and airborne sensors. Pp. 1–23. In G. Stanhill (ed.). *Monograph on Advances in Bioclimatology*. Vol. 1. Springer-Verlag, New York, New York.

Desjardins, R. L. and J. I. MacPherson. 1989. Aircraft-based measurements of trace gas fluxes. In M. O. Andreae and D. S. Schimel (eds.). *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere*. John Wiley and Sons, Chichester, U.K.

Desjardins, R. L., J. I. MacPherson, P. H. Schuepp, and F. Karanja. 1989. An evaluation of aircraft measurement of CO_2, water vapor, and sensible heat. *Boundary-Lay Meteorol* 47:55–70.

Desjardins, R. L., P. Rochete, J. I. MacPherson, and E. Pattey. 1993. Measurements of greenhouse gas fluxes using aircraft- and tower-based techniques. Pp. 45–62. In L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston (eds.). *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*. Agronomy Society of America Publication 55. Agronomy Society of America/Crop Science Society of America/Soil Science Society of America, Madison, Wisconsin.

Dick, W. A., W. M. Edwards, and E. L. McCoy. 1997. Continuous application of no-tillage to Ohio soils: Changes in crop yields and organic matter-related soil properties. Pp. 171–182. In E. A. Paul, K. Paustian, E. T. Elliott, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Dick, W. A., R. L. Blevins, W. W. Frye, S. E. Peters, D. R. Christensen, F. J. Pierce, and M. L. Vitosh. 1998. Impacts of agricultural management practices on C sequestration in forest-derived soils of the eastern Corn Belt. *Soil Till Res* 47:235–244.

Dlugokencky, E. 2000. Carbon cycle greenhouse gases. National Oceanic and Atmospheric Association Climate Monitoring and Diagnostic Laboratory, Boulder, Colorado, <http://www.cmdl.noaa.gov/ccgg/gallery/index.php?currDir=./Data_Figures&pageType=image&image=ch4trend_global.jpg>

Donigian, A. S., Jr., T. O. Barnwell, Jr., R. B. Jackson, IV, A. S. Patwardhan, K. B. Weinrich, A. L. Rowell, R. V. Chinnaswamy, and C. V. Cole. 1994. Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States. EPA/600/RB94B067. Environmental Research Laboratory, U.S. Environmental Protection Agency, Athens, Georgia. 193 pp.

Donigian, A. S., A. S. Patwardhan, R. V. Chinnaswamy, and T. O. Barnwell. 1997. Modeling soil carbon and agricultural practices in the central U.S.: An update of preliminary study results. Pp. 449–518. In R. Lal, J. M. Kimble, R. F. Follett, and B. A. Stewart (eds.). *Soil Processes and the Carbon Cycle*. CRC Press, Boca Raton, Florida.

Doran, J. W., L. N. Mielke, and S. Stamatiadis. 1988. Microbial activity and N cycling as regulated by soil water-filled pore space. Paper 132. Pp. 49–54. *In Proceedings of the 11th International Soil Tillage Research Organization*, Edinburgh, Scotland, July 11–15, 1988.

Dyal, R. S., F. B. Smith, and R. V. Allison. 1939. The decomposition of organic matter in soils at different initial pH. *J Am Soc Agron* 31:841–850.

Edwards, G. C., H. H. Nuemann, G. den Hartog, G. W. Thurtell, and G. E. Kidd. 1994. Eddy correlation measurements of methane fluxes using a tunable diode laser at the Kinosheo Lake tower site during the Northern Wetlands Study (NOWES). *J Geophys Res* 99:1511–1517.

Ellerman, A., P. Joskow, R. Schmalensee, J.-P. Montero, and E. Bailey. 2000. *Markets for Clean Air: The U.S. Acid Rain Program*. Cambridge University Press, New York, New York.

Ellert, B. H. and J. R. Bettany. 1995. Calculation of organic matter and nutrients stored in soils under contrasting management regimes. *Can J Soil Sci* 75:529–538.

Elliott, E. T., H. W. Hunt, and D. E. Walter. 1988. Detrital food web interactions in North American grassland ecosystems. *Agric Ecosyst Environ* 24:41–56.

Elliott, E. T., I. C. Burke, C. A. Monz, S. D. Frey, K. H. Paustian, H. P. Collins, E. A. Paul, C. V. Cole, R. L. Blevins, W. W. Frye, D. J. Lyon, A. D. Halvorson, D. R. Huggins, R. F. Turco, and M. V. Hickman. 1994. Terrestrial carbon pools in grassland and agricultural soils: Preliminary data from the Corn Belt and Great Plains Regions. Pp. 179–192. In J. W. Doran, D. C. Coleman, D. F. Bezdicek, and B. A. Stewart (eds.). *Defining Soil Quality for a Sustainable Environment*. Soil Science Society of America Special Publication 35. Soil Science Society of America/Agronomy Society of America, Madison, Wisconsin.

Eve, M. D., K. Paustian, R. F. Follett, and E. T. Elliott. 2001. An inventory of carbon emissions and sequestration in the United States. In R. Lal (ed.). *Soil Carbon Sequestration and the Greenhouse Effect*. Soil Science Society of America Special Publication 57. Soil Science Society of America, Madison, Wisconsin.

Eve, M. D., M. Sperow, K. Paustian, and R. F. Follett. 2002. National-scale estimation of changes in soil carbon stocks on agricultural lands. *Environ Pollut* 116:431-438.

Falk, I. and R. Mendelsohn. 1993. The economics of controlling stock pollutants: An efficient strategy for greenhouse gases. *J Environ Econ Manage* 25:76–88.

Fallon, P. D., P. Smith, J. U. Smith, J. Szabó, K. Coleman, and S. Marshall. 1998. Regional estimates of carbon sequestration potential: Linking the Rothamsted Carbon Model to GIS databases. *Biol Fertil Soils* 27:236–241.

Fallon, P., P. Smith, J. Szabó and L. Pasztor. 2002. Comparison of approaches for estimating carbon sequestration at the regional scale. *Soil Use Manage* 18:164-174.

Fearnside, P. M. 1997. Monitoring needs to transform Amazonian forest maintenance into a global warming-mitigation option. *Mit Adapt Strategies Global Change* 2:285–302.

Feng, H., J. Zhao, and C. Kling. 2000. *Carbon Sequestration in Agriculture: Value and Implementation*. Working Paper 00–WP 25. Center for Agricultural and Rural Development, Iowa State University, Ames.

Fenn, M. E., M. A. Poth, J. D. Aber, J. S. Baron, B. T. Bormann, D. W. Johnson, A. D. Lemly, S. G. McNulty, D. F. Ryan, and R. Stottlemyer. 1998. Nitrogen excess in North American ecosystems: Predisposing factors, ecosystem responses, and management strategies. *Ecol Appl* 8:706–733.

Fernandez, I. J., L. E. Rustad, G. B. Lawrence. 1993. Estimating total soil mass, nutrient content, and trace-metals in soils under a low elevation spruce-fir forest. *Can J Soil Sci* 73:317–328.

Firestone, M. K. and E. A. Davidson. 1989. Microbiological basis of NO and N_2O production and consumption in soil. Pp. 7–22. In M. D. Andreae and D. S. Schimel (eds.). *Trace Gas Exchange Between Terrestrial Ecosystems and the Atmosphere*. John Wiley, Berlin, Germany.

Fisher, M. J., I. M. Rao, M. A. Ayarza, C. E. Lascano, J. I. Sanz, R. J. Thomas, and R. R. Vera. 1994. Carbon storage by introduced deep-rooted grasses in the South American savannas. *Nature* 317:236–238.

Follett, R. F. 1997. CRP and microbial biomass dynamics in temperate climates. Pp. 305–322. In R. Lal, J. M. Kimble, R. F. Follett, and B. A. Stewart (eds.). *Advances in Soil Science: Management of Carbon Sequestration in Soil*. CRC Press, Boca Raton, Florida.

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

Follett, R. F., J. M. Kimble, and R. Lal. 2001. *The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect.* CRC Press, Boca Raton, Florida.

Folorunso, O. A. and D. E. Rolston. 1984. Spatial variability of field-measured denitrification gas fluxes. *J Soil Sci Soc Am* 48:1214–1219.

Fowler, D. and J. H. Duyzer. 1989. Micrometeorological techniques for the measurement of trace gas exchange. Pp. 189–207. In M. O. Andreae and D. S. Schimel (eds.). *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere.* John Wiley and Sons, Chichester, England.

Frolking, S. E., A. R. Mosier, D. S. Ojima, C. Li, W. J. Parton, C. S. Potter, E. Priesack, R. Stenger, P. Haberbosch, P. Dorsch, H. Flessa, and K. A. Smith. 1998. Comparison of N_2O emissions from soils at three temperate agricultural sites: Simulations of year-round measurements by four models. *Nutr Cycling Agroecosyst* 52:77–105.

Frye, W. W. 1984. Energy requirement in no-tillage. Pp. 127–151. In R. E. Phillips and S. H. Phillips (eds.). *No Tillage Agricultural Principles and Practices.* Van Nostrand Reinhold, New York, New York.

Gadeken, D., D. Rath, and D. Sauerbeck. 1990. Methane production in ruminants. Pp. 36–38. In *Greenhouse Gas Emissions from Agricultural Ecosystems.* Vol. 1. U.S. Environmental Protection Agency Report 20P–2005. Office of Policy Analysis, Washington, D.C.

Galle, B., L. Klemendtson, and D. W. Griffith. 1994. Application of an FTIR system for measurements of N_2O fluxes using micrometeorological methods, an ultralarge chamber system, and conventional field chambers. *J Geophys Res* 99:16575–16583.

Ganzhara, N. F. 1998. Humus, soil properties and yield. *Eurasian Soil Sci* 31:738–745.

Garten, C. T. and S. D. Wullschleger. 1999. Soil carbon inventories under a bioenergy crop (Switchgrass): Measurement limitations. *J Environ Qual* 28:1359–1365.

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

Gebhart, D. L., H. B. Johnson, H. S. Mayeux, and H. W. Polley. 1994. The CRP increases soil organic carbon. *J Soil Water Conserv* 49:488–492.

Gibbs, M. J. and R. A. Leng. 1993. Methane emissions from livestock. Pp. 73–79. In A. R. van Amstel (ed.). *Methane and Nitrous Oxide.* Royal Institute of Veterinary Medicine Report 481507003. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Gleick, P. H. 1987. Regional hydrologic consequences of increases in atmospheric CO_2 and other trace gases. *Clim Change* 10:137–161.

Goulden, M. L., J. W. Munger, S. M. Fan, B. C. Daube, and S. C. Wofsy. 1996. Measurements of carbon sequestration by long-term eddy covariance: Methods and a critical evaluation of accuracy. *Global Change Biol* 2:169–182.

Grace, P. R. and J. N. Ladd. 1995. *SOCRATES v2.00 User Manual.* Cooperative Research Centre for Soil and Land Management, Glen Osmond, South Australia.

Grace, P. R., J. N. Ladd, and J. Skjemstad. 1994. The effect of management practices on soil organic matter dynamics. Pp. 162–171. In C. Pankhurst, B. M. Doube, V. V. S. R. Gupta, and P. R. Grace (eds.). *Management of Soil Biota in Sustainable Farming Systems.* Commonwealth Scientific and Industrial

Organization, Melbourne, Australia.

Griffin, R. and D. Bromley. 1982. Agricultural runoff as a nonpoint externality: A theoretical development. *Am J Agric Econ* 64:547–552.

Griliches, Z. 1957. Hybrid corn: An exploration in the economics of technological change. *Econometrica* 25:501–522.

Groffman, P. M. and J. M. Tiedje. 1989. Denitrification in north temperate forest soils: Spatial and temporal patterns at the landscape and seasonal scales. *Soil Biol Biochem* 21:613–620.

Gupta, R. K. and D. L. N. Rao. 1994. Potential of wastelands for sequestering carbon by reforestation. *Curr Sci* 66:378–380.

Haas, H. J., C. E. Evans, and E. F. Miles. 1957. Nitrogen and carbon changes in Great Plains soil as influenced by cropping and soil treatments. U.S. Department of Agriculture Technical Bulletin 1164:1–111. U.S. Department of Agriculture, Washington, D.C.

Hagen, L. J. 1991. Wind erosion prediction system to meet user needs. *J Soil Water Conserv* 46:106–111.

Hahn, G. L. and J. A. Morgan. 2000. Potential consequences of climate change on ruminant livestock production. Draft report. U.S. Meat Animal Research Center, Agricultural Research Service, U.S. Department of Agriculture, Clay Center, Nebraska.

Hahn, R. 1989. Economics prescriptions for environmental problems: How the patient followed the doctor's order. *J Econ Perspect* 3:95–114.

Hansen, J. E. and M. Sato. 2001. Trends in measured climate forcing agents. *Proc Natl Acad Sci USA* 98(26):14778–14783.

Hargreaves, K. J., F. G. Wienhold, L. Klemedtson, J. R. M. Arah, I. J. Beverland, D. Fowler, B. Galle, D. W. T. Griffith, U. Skiba, K. A. Smith, M. Welling, and G. W. Harris. 1996. Measurement of nitrous oxide emission from an agricultural land using micrometeorological methods. *Atmos Environ* 30:1563–1571.

Harper, L. A., O. T. Denmead, J. R. Freney, and F. M. Byers. 1999. Direct measurements of methane emissions from grazing and feedlot cattle. *J Anim Sci* 77:1392–1401.

Hartmann, D. and M. Kaltshmitt. 1999. Electricity generation from solid biomass via co-combustion with coal. Energy and emission balances from a German case study. *Biomass Bioenergy* 16:397–406.

Hassink, J. and A. P. Whitmore. 1997. A model of the physical protection of organic matter in soils. *Soil Sci Soc Am J* 61:131–139.

Havlin, J. L., D. E. Kissel, L. D. Maddux, M. M. Claassen, and J. H. Long. 1990. Crop rotation and tillage effects on soil organic carbon and nitrogen. *Soil Sci Soc Am J* 54:448–452.

Hayhoe, K., A. Jain, H. Pitcher, C. McCracken, M. Gibbs, D. Wuebbles, R. Harvey, and D. Kruger. 1999. Costs of multi-greenhouse gas reduction targets for the USA. *Science* 286:905–906.

Hendrey, G. R., K. F. Lewin, and J. Nagy. 1993. Free air carbon-dioxide enrichment: Development, progress, results. *Vegetation* 104:17–31.

Herriges, J., R. Govindasamy, and J. Shogren. 1994. Budget-balancing incentive mechanisms. *J Environ Econ Manage* 27:275–285.

Hogan, K. B. 1993. Methane reductions are a cost-effective approach for reducing emissions of greenhouse gases. Pp. 187–201. In A. R. van Amstel (ed.). *Methane and Nitrous Oxide: Methods in National Emissions Inventories and Options for Control.* Royal Institute of Veterinary Medicine Report 481507003. National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Holland, E. A., B. H. Braswell, J. F. Lamarque, A. Townsend, J. Sulzman, J. F. Muller, F. Dentener, G. Brasseur, H. Levy, J.

Penner, and G. J. Roelofs. 1997. Variations in the predicted spatial distribution of atmospheric nitrogen deposition and their impact on carbon uptake by terrestrial ecosystems. *J Geophys Res* 102:15849–15866.

Holterman, S. 1976. Alternative tax systems to correct for externalities and the efficiency of paying compensation. *Economica* 43:1–16.

Hooper, D. U., D. E. Bignell, V. K. Brown, L. Brussaard, J. M. Dangerfield, D. H. Wall, D. A. Wardle, D. C. Coleman, K. E. Giller, P. Lavelle, W. H. Van Der Putten, P. C. De Ruiter, J. Rusek, W. L. Silver, J. M. Tiedje, and V. Wolters. 2000. Interactions between aboveground and belowground biodiversity in terrestrial ecosystems: Patterns, mechanisms, and feedbacks. *BioScience* 50:1049–1061.

Houghton, R. A. 1995. Changes in the storage of industrial carbon since 1850. Pp. 45–65. In R. Lal, J. M. Kimble, E. Levine, and B. A. Stewart (eds.). *Soils and Global Change*. CRC/Lewis Publishers, Boca Raton, Florida.

Houghton, R. A. 1999. The annual net flux of carbon to the atmosphere from changes in land use from 1850 to 1990. *Tellus* 51B:298–313.

Howarth, R. W., G. Billen, D. Swaney, A. R. Townsend, N. Jaworski, K. Lajtha, J. A. Downing, R. Elmgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch, and Z. Zhao-Liang. 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainage to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry* 35:75–139.

Hrubovcak, J., M. LeBlanc, and J. Miranowski. 1990. Limitations in evaluating environmental and agricultural policy coordination benefits. *Am Econ Rev* 80:208–212.

Huggins, D. R., G. A. Buyanovsky, G. H. Wagner, J. R. Brown, R. G. Darmody, T. R. Peck, G. W. Lesoing, M. B. Vanotti, and L. G. Bundy. 1998. Soil organic C in the tallgrass prairie-derived region of the Corn Belt: Effects of long-term crop management. *Soil Till Res* 47:219–234.

Hunt, H. W., M. J. Trlica, E. F. Redente, J. C. Moore, J. K. Detling, T. G. F. Kittel, D. E. Walter, M. C. Fowler, D. A. Klein, and E. T. Elliott. 1991. Simulation model for the effects of climate change on temperate grassland ecosystems. *Ecol Model* 53:205–246.

Huntington, T. G., D. F. Ryan, and S. P. Hamburg. 1988. Estimating soil nitrogen and carbon pools in a northern hardwood forest ecosystem. *Soil Sci Soc Am J* 52:1162–1167.

Hutchinson, G. L. and G. P. Livingston. 1993. Use of chamber systems to measure trace gas fluxes. Pp. 63–78. In L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston (eds.). *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*. American Society of Agronomy Special Publication 55. American Society of Agronomy/Crop Science Society of America/Soil Science Society of America, Madison, Wisconsin.

Hutchinson, G. L. and A. R. Mosier. 1981. Improved soil cover method for field measurement of nitrous oxide fluxes. *Soil Sci Soc Am J* 45:311–316.

Huyler, M. T., K. A. Johnson, H. H. Westerberg, B. K. Lamb, and P. Zimmerman. 1993. A method for quantifying ruminant methane emissions from free-range ruminants. *J Anim Sci Suppl* 1:269.

Intergovernmental Panel on Climate Change (IPCC). 1994. *Radiative Forcing of Climate Change: The 1994 Report of the Scientific Assessment Working Group of the Intergovernmental Panel on Climate Change. Summary for Policymakers*. World Meteorological Organization/United Nations Environmental Program, Geneva, Switzerland. 28 pp.

Intergovernmental Panel on Climate Change (IPCC). 1995. Technical summary. Intergovernmental Panel on Climate Change, World Meteorological Organizations, Geneva. 44 pp.

Intergovernmental Panel on Climate Change (IPCC). 1996a. J. T. Houghton, L. B. Meira Filho, B. A. Callander, N. Harris, A. Kattenberg, and K. Maskell (eds.). *Climate Change 1995: The Science of Climate Change*. Intergovernmental Panel on Climate Change Second Assessment Report, Vol. 1. Cambridge University Press, Cambridge, U.K. 572 pp.

Intergovernmental Panel on Climate Change (IPCC). 1996b. R. T. Watson, M. C. Zinyowera, and R. H. Moss (eds.). *Climate Change 1995: Impacts, Adaptations and Mitigation of Climate Change: Scientific-Technical Analyses*. Chaps. 13 and 23. The Intergovernmental Panel on Climate Change Second Assessment Report, Vol. 2. Cambridge University Press, Cambridge, U.K. 878 pp.

Intergovernmental Panel on Climate Change (IPCC). 1997. *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories*. Intergovernmental Panel on Climate Change, Organization for Economic Cooperative Development, and International Energy Agency. The Organization for Economic Cooperative Development, Paris.

Intergovernmental Panel on Climate Change (IPCC). 2000. *Land Use, Land Use Change, and Forestry Intergovernmental Panel on Climate Change Special Report*. Oxford University Press, Oxford, U.K. 377 pp.

Intergovernmental Panel on Climate Change (IPCC). 2001a. J. T. Houghton and D. Yihui (eds.). *Climate Change 2001: The Scientific Basis*. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K. 881 pp.

Intergovernmental Panel on Climate Change (IPCC). 2001b. J. McCarthy and O. Canziani (eds.). *Climate Change 2001: Impacts, Adaptation, and Vulnerability*. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.

Intergovernmental Panel on Climate Change (IPCC) 2001c. R. T. Watson (ed.). *Climate Change 2001: Synthesis Report*. Cambridge University Press, Cambridge, U.K.

Isermann, K. 1994. Agriculture's share in the emission of trace gases affecting the climate and certain cause-oriented proposals for sufficiently reducing this share. *Environ Pollut* 83:95–111.

Izaurralde, R. C., M. Nyborg, E. D. Solberg, H. H. Janzen, M. A. Arshad, S. S. Malhi, and M. Molina-Ayala. 1998a. Carbon storage in eroded soils after five years of reclamation techniques. Pp. 369–386. In R. Lal, J. Kimble, R. Follett, and B. A. Stewart (eds.). *Soil Processes and the Carbon Cycle*. Advances in Soil Science series. CRC Press, Boca Raton, Florida.

Izaurralde, R. C., W. B. McGill, A. Bryden, S. Graham, M. Ward, and P. Dickey. 1998b. Scientific challenges in developing a plan to predict and verify carbon storage in Canadian prairie soils. Pp. 433–446. In R. Lal, J. Kimble, R. Follett, and B. A. Stewart (eds.). *Management of Carbon Sequestration in Soil*. Advances in Soil Science series. CRC Press, Boca Raton, Florida.

Izaurralde, R. C., K. L. Hauren-Kozyra, D. C. Jans, W. B. McGill, R. F. Grant, and J. C. Hiley. 2000. Soil C dynamics: Measurement, simulation and site-to-region scale-up. Pp. 553–575. In R. Lal, J. Kimble, R. Follett, and B. Stewart (eds.). *Assessment Methods for Soil C Pools*. Lewis Publishers, Boca Raton, Florida.

Janzen, H. H., C. A. Campbell, S. A. Brandt, G. P. Lafond, and L.

Townley-Smith. 1992. Light-fraction organic matter in soil from long-term rotations. *Soil Sci Soc Am J* 56:1799–1806.

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

Jenkinson, D. S. and S. Powlson. 1976. The effects of biocidal treatments on metabolism in soil. V. A method for measuring soil biomass. *Soil Biol Biochem* 8:209–213.

Johnson, D. E., T. M. Hill, G. M. Ward, K. A. Johnson, M. E. Branine, B. R. Carmean, and D. W. Lodman. 1993. Ruminants and other animals. Pp. 199–229. In M. A. K. Khalil (ed.). *Atmospheric Methane: Sources, Sinks, and Role in Global Change*. Springer-Verlag, New York, New York.

Jones, O. R., B. A. Steward, and P. W. Unger. 1997. Pp. 335–342. In E. A. Paul, K. Paustian, E. T. Elliott, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Just, R., N. Bockstael, R. Cummings, J. Miranowski, and D. Zilberman. 1991. Problems confronting the joint formulation of commercial agricultural and resource policies. Pp. 3–16. In R. E. Just and N. Bockstael (eds.). *Commodity and Resource Policies in Agricultural Systems*. Springer-Verlag, New York, New York.

Just, R. E. and D. Zilberman. 1983. Stochastic structure, farm size, and technology adoption in developing agriculture. *Ox Econ Pap* 35:307–328.

Kaiser, H. M., S. J. Riha, D. S. Wilks, D. G. Rossier, and R. Sampath. 1993. A farm-level analysis of economic and agronomic impacts of gradual warming. *Am J Agric Econ* 75:387–398.

Karlen, D. L., M. J. Mausbach, J. W. Doran, R. G. Cline, R. F. Harris, and G. E. Schuman. 1997. Soil quality: A concept, definition and framework for evaluation. *Soil Sci Soc Am J* 61:4–10.

Keerthisinghe, D. G., J. R. Freney, and A. R. Mosier. 1993. Effect of wax-coated calcium carbide and nitrapyrin on nitrogen loss and methane emission from dry-seeded flooded rice. *Biol Fertil Soils* 16:71–75.

Keller, M., M. E. Mitre, and R. F. Stallard. 1990. Consumption of atmospheric methane in soils of central Panama: Effects of agricultural development. *Global Biogeochem Cycles* 4:21–27.

Keller, M., T. J. Goreau, S. C. Wofsy, W. A. Kaplan, and M. B. McElroy. 1983. Production of nitrous oxides and consumption of methane by forest soils. *Geophys Res Lett* 10:1156–1159.

Keller, M., E. Veldkamp, A. M. Weitz, and W. A. Reiners. 1993. Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica. *Nature* 365:244–246.

Khalil, M. A. K. 1999. Non-CO_2 greenhouse gases in the atmosphere. *Annu Rev Energy Environ* 24:645–661.

Kimball, B. A. 1983. Carbon dioxide and agricultural yield: An assemblage and analysis of 430 prior observations. *Agron J* 75:779–788.

Kimball, B. A. and S. B. Idso. 1983. Increasing atmospheric CO_2. Effects on crop yield, water use and climate. *Agric Water Manage* 7:55–72.

Kirkegaard, J. A. 1995. A review of trends in wheat yield responses to conservation cropping in Australia. Pp. 35–48. In R. J. Martin, P. R. Grace, C. A. Anderson, and J. C. Fegent (eds.). *Long-Term Agronomic Experiments for Productive and Sustainable Farming Systems*. CSIRO Publishing, East Melbourne, Australia.

Kirschbaum, M. U. F. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warm-ing on soil organic C storage. *Soil Biol Biochem* 27:753–760.

Kittel, T. G. F., N. A. Rosenbloom, T. H. Painter, and D. S. Schimel. 1995. The VEMAP integrated database for modeling United States ecosystem/vegetation sensitivity to climate change. *J Biogeogr* 22:857–862.

Klute, A. (ed.). 1986. *Methods of Soil Analysis*. Part 1. Agronomy Monograph 9. American Society of Agronomy, Madison, Wisconsin.

Knapp, A. K., J. M. Briggs, D. C. Hartnett, and S. L. Collins (eds.). 1998. *Konza Prairie Long-term Ecology Research*. Oxford Press, New York, New York.

Knowles, R. 1993. Methane: Processes of production and consumption. Pp. 145–156. In L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston (eds.). *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change*. American Society of Agronomy, Madison, Wisconsin.

Kolb, C. E., J. C. Wormhoudt, and M. S. Zahniser. 1995. Recent advances in spectroscopic instrumentation for measuring stable gases in the natural environment. Pp. 259–290. In P. A. Matson and R. C. Harriss (eds.). *Biogenic Trace Gases: Measuring Emissions from Soil and Water*. Blackwell Scientific Publishing, London.

Kral, D. M. and S. L. Hawkins. 1994. *Erosion and Productivity of Soils Containing Rock Fragments*. Soil Science Society of America Special Publication 13. Soil Science Society of America, Madison, Wisconsin 103 pp.

Kroeze, C. and A. R. Mosier. 2000. New estimates for emissions of nitrous oxide. Pp. 45–64. In J. E. A. van Ham (ed.). *Non-CO_2 Greenhouse Gases: Scientific Understanding, Control and Implementation*. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Kroeze, C. and S. P. Seitzinger. 1998. Nitrogen inputs to rivers, estuaries and continental shelves and related nitrous oxide emissions in 1990 and 2050: A global model. *Nutr Cycling Agroecosyst* 52:195–212.

Kroeze, C., A. Mosier, and L. Bouwman. 1999. Closing the global N_2O budget: A retrospective analysis. 1500–1994. *Global Biogeochem Cycles* 13:1–8.

Laflen, J. M., L. J. Lane, and G. R. Foster. 1991. WEPP: A new generation of erosion prediction technology. *J Soil Water Conserv* 46:34–38.

Laflen, J. M., W. J. Elliot, D. C. Flanagan, C. R. Meyer, and M. A. Nearing. 1997. WEPP—Predicting water erosion using a process-based model. *J Soil Water Conserv* 52:96–102.

Lal, R. 1987. *Tropical Ecology and Physical Edaphology*. J. Wiley and Sons, Chichester, U.K.

Lal, R. 1994. Soil erosion by water and wind: Problems and prospects. Pp. 1–10. In R. Lal (ed.). *Soil Erosion Research Methods*. Soil Water Conservation Society, St. Lucie Press, Delray Beach, Florida.

Lal, R. 1999. Soil management and restoration for carbon sequestration to mitigate the accelerated greenhouse effect. *Prog Environ Sci* 1:307–326.

Lal, R., J. M. Kimble, R. F. Tolbert, and C. V. Cole. 1998. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*. Ann Arbor Press, Chelsea, Michigan. 128 pp.

Lal, R., R. F. Follett, J. Kimble, and C. V. Cole. 1999. Managing U.S. cropland to sequester carbon in soil. *J Soil Water Conserv* 54:374–381.

Lapitan, R. L., R. Wanninkhof, and A. R. Mosier. 1999. Methods for stable gas flux determination in aquatic and terrestrial systems. Pp. 27–66. In A. F. Bouwman (ed.). *Approaches to Scal-*

ing of Trace Gas Fluxes in Ecosystems. *Developments in Atmospheric Science 24.* Elsevier, Amsterdam, The Netherlands.

Larson, W. E. and F. J. Pierce. 1994. The dynamics of soil quality as a measure of sustainable management. Pp. 37–51. In J. W. Doran, D. C. Coleman, D. F. Bezdicek, and B. A. Stewart (eds.). *Defining Soil Quality for a Sustainable Environment.* Soil Science Society of America Special Publication 35. Soil Science Society of America, Madison, Wisconsin.

Lashof, D. and B. Hare. 1999. The role of biotic carbon stocks in stabilizing greenhouse gas concentrations at safe levels. *Environ Sci & Policy* 2:101–110.

Lee, H. C., B. A. McCarl, U. A. Schneider, and C. C. Chen. 2001. Economic implications of international participation alternatives for agricultural greenhouse gas emission mitigation. Unpublished paper. Department of Agricultural Economics, Texas A & M University, College Station, <http://ageco.tamu.edu/faculty/mccarl/papers/823.pdf> (20 May 2001)

Leng, R. A. 1991. Improving ruminant production and reducing methane emissions from ruminants by strategic implementation. Report 400/1B91/004. Office of Air and Radiation, Washington, D.C.

Lenschow, D. H. 1995. Micrometeorological techniques for measuring biosphere-atmosphere trace gas exchange. Pp. 126–163. In P. A. Matson and R. C. Harriss (eds.). *Biogenic Trace Gases: Measuring Emissions from Soil and Water.* Blackwell Scientific Publishing, Limited, London.

Levine, J. S., W. R. Cofer, and F. P. Pinto. 1993. Biomass burning. Pp. 230–253. In M. A. K. Khalil (ed.). *Atmospheric Methane: Sources, Sinks and Role in Global Change.* Springer-Verlag, Berlin.

Lewandrowski, J. and D. Schimmelpfennig. 1999. Economic implications of climate change for U.S. agriculture: Assessing recent evidence. *Land Econ* 75:39–57.

Li, Z. and Q. Zhao. 1998. Carbon dioxide fluxes and potential mitigation in agriculture and forestry of tropical and subtropical China. *Clim Change* 40:119–133.

Lin, E., H. Dong, and Y. Li. 1994. Methane emissions of China: Agricultural sources and mitigation options. Pp. 115–120. In *Agricultural Sources and Mitigation Options. Proceedings of the Symposium on Non-CO$_2$ Greenhouse Gases.* Kluwer Academic Publishers, Dordrecht, The Netherlands.

Lincoln, D. E., N. Sionit, and B. R. Strain. 1984. Growth and feeding response of *Pseudoplusia includens* (Lepidoptera: Noctuidae) to host plants grown in controlled carbon dioxide atmospheres. *Environ Entomol* 13:1527–1530.

Lindau, C. W., P. K. Bollich, R. D. DeLaune, A. R. Mosier, and K. F. Bronson. 1993. Methane mitigation in flooded Louisiana rice fields. *Biol Fertil Soils* 15:174–178.

Lindstrom, J. J., S. C. Gupta, C. A. Onstad, W. E. Larson, and R. F. Holt. 1979. Tillage and crop residue effects on soil erosion in the Corn Belt. *J Soil Water Conserv* 34:80

Livingston, G. P. and G. L. Hutchinson. 1995. Enclosure-based measurements of trace gas exchange: Applications and sources of errors. Pp. 14–51. In P. A. Matson and R. C. Harriss (eds.). *Biogenic Trace Gases: Measuring Emissions from Soil and Water.* Blackwell Scientific Publishing, London.

Loftfield, N. S., R. Brumme, and F. Beese. 1992. Automated monitoring of nitrous-oxide and carbon-dioxide flux from forest soils. *Soil Sci Soc Am J* 56:1147–1150.

Loveland, T. R., B. C. Reed, J. F. Brown, D. O. Ohlen, Z. Zhu, L. Yang, and J. W. Merchant. 2000. Development of a global land cover characteristics database and IGBP DISCover from 1-km AVHRR Data. *Int J Remote Sens* 21:1303–1330,

<http://edcdaac.usgs.gov/main.html> (30 December 2003)

Magdoff, F. R., M. A. Tabatabai, and E. A. Hanlon, Jr. (eds.). 1996. *Soil Organic Matter: Analysis and Interpretation.* Soil Science Society of America Special Publication 46. Soil Science Society of America, Madison, Wisconsin. 67 pp.

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

Mann, L. K. and P. L. Spath. 1999. The net CO$_2$ emissions and energy balances of biomass and coal-fired power systems. Pp. 379–385. In R. P. Overend and E. Chornet (eds.). *Proceedings of the Fourth Biomass Conference of the Americas,* Oakland, California. 29 August–2 September 1999.

Manne, A. S. and R. G. Richels. 2001. An alternative approach to establishing trade-offs among greenhouse gases. *Nature* 410:675–677.

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

Matthews, R. B., R. Wassmann, J. W. Knox, and L. V. Buendia. 2000. Using a crop/soil simulation model and GIS techniques to access methane emissions from rice fields in Asia. IV. Upscaling to national levels. *Nutrient Cycling Agrosys* 58:201–217.

Matson, P. A. and P. M. Vitousek. 1987. Cross-system comparisons of soil nitrogen transformations and nitrous oxide flux in tropical forest ecosystems. *Global Biogeochem Cycles* 1:163–170.

McCarl, B. A. 1998. Carbon sequestration via tree planting on agricultural lands: An economic study of costs and policy design alternatives. Paper presented at the Energy Modeling Forum, Snowmass, Colorado, 3–11 August.

McCarl, B. A. and U. Schneider. 2000a. Agriculture's role in a greenhouse gas emission mitigation world: An economic perspective. *Rev Agric Econ* 22:134–159.

McCarl, B. A. and U. Schneider. 2000b. Economic potential of biomass for greenhouse gas emission reductions: Comparative role in agriculture. In *Proceedings of the Conference on Sustainable Energy: New Challenges for Agriculture and Implications for Land Use.* Wageningen University Press, Wageningen, The Netherlands.

McCarl B. A. and U. A. Schneider. 2001. Greenhouse gas mitigation in U.S. agriculture and forestry. *Science* 294:2481–2482.

McCarl, B. A., R. M. Adams, and B. Hurd. In press. Global climate change and its impact on agriculture. In C. C. Chang and S. Hsu (eds.). *Encyclopedia of Life Support Systems.* Institute of Economics, Academia Sinica and the United Nations Educational, Scientific, and Cultural Organization, Taipei, Taiwan.

McGill, W. B. 1996. Review and classification of ten soil organic matter models. Pp. 111–132. In D. S. Powlson, P. Smith, and J. U. Smith (eds.). *Evaluation of Soil Organic Matter Models Using Existing Long-Term Datasets.* Springer-Verlag, Heidelberg.

McGuire, A. D., S. Sitch, J. S. Clein, R. Dargaville, G. Esser, J. Foley, M. Heimann, F. Joos, J. Kaplan, D. W. Kicklighter, R. A. Meier, J. M. Melillo, B. Moore, III, I. C. Prentice, N. Ramankutty, T. Reichenau, A. Schloss, H. Tian, L. J. Williams, and U. Wittenberg. 2001. Carbon balance of the terrestrial biosphere in the twentieth century: Analysis of CO$_2$, climate and land use effects with four process-based ecosystem models. *Global Biogeochem Cycles* 15:183–206.

McLaughlin, S. B., D. I. Bransby, and D. Parrish. 1994. Perennial grass production for biofuels: Soil conservation consideration. Pp. 359–370. In *Proceedings of Bioenergy '94,* Reno, Nevada, October 2–6, 1994.

McLaughlin, S. B., R. Samson, D. Bransby, and A. Weislogel. 1996.

Evaluating physical, chemical, and energetic properties of perennial grasses biofuels. Pp. 1–8. In *Proceedings of Bioenergy '96*, Nashville, Tennessee, September 15–20, 1996. Southeastern Regional Biomass Energy Program, Muscle Schools, Alabama.

McLaughlin, S. B., J. Bouton, D. Bransby, B. Conger, W. Occumpaugh, D. Parrish, C. Talliaferro, K. Vogel, and S. Wullschleger. 1999. Developing switchgrass as a bioenergy crop. Pp. 282–299. In J. Janick (ed.). *Proceedings of the Fourth National New Crops Symposium*. Phoenix, Arizona, November 8–11, 1998.

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

McTaggart, I. P., H. Clayton, J. Parker, L. Swan, and K. A. Smith. 1997. Nitrous oxide emissions from grassland and spring barley, following N fertilizer application with and without nitrification inhibitors. *Biol Fertil Soils* 25:261–268.

Megraw, S. R. and R. Knowles. 1987. Methane production and consumption in a cultivated humisol. *Biol Fertil Soils* 5:56–60.

Melillo, J. M., J. D. Aber, and J. F. Muratore. 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. *Ecol* 63:621–626.

Millock, K., D. Sunding, and D. Zilberman. 2000. *Regulating Pollution with Endogenous Monitoring*. Working Paper. Department of Agricultural and Resource Economics, University of California, Berkeley.

Minami, K. 1993. Methane from rice production. Methane and nitrous oxide. Pp. 143–162. In A. R. van Amstel (ed.). *Proceedings of the International Panel on Climate Change Workshop*, Bilthoven, The Netherlands, February 3–5, 1993.

Mitchell, P. D., P. G. Lakshminarayan, T. Otake, and B. A. Babcock. 1999. The impact of soil conservation policies on carbon sequestration in agricultural soils of the central United States. Chap. 10. In R. Lal (ed.). *Management of Carbon Sequestration in Soil*. CRC Press, Boca Raton, Florida.

Moe, P. W. and H. F. Tyrell. 1979. Methane production in dairy cows. *J Dairy Sci* 62:1583–1586.

Mollitor, A. V., A. L. Leaf, and L. A. Morris. 1980. Forest soil variability on northeastern flood plains. *Soil Sci Soc Am J* 44:617–620.

Moncrieff, J. B., Y. Malhi, and R. Leuning. 1996. The propagation of errors in long-term measurements of land-atmosphere fluxes of carbon and water. *Global Change Biol* 2:231–240.

Morgan, J. A., A. R. Mosier, D. G. Milchunas, D. R. LeCain, J. A. Nelson, and W. J. Parton. 2004. CO_2 enhances productivity, alters species composition, and reduces digestibility of shortgrass steppe vegetation. *Ecol Applications* 14:208–219.

Morison, J. I. L. 1985. Sensitivity of stomata and water use efficiency to high CO_2. *Plant, Cell, Environ* 8:467–474.

Moser, L. E. and K. P. Vogel. 1995. Switchgrass, big bluestem, and indiangrass. Pp. 409–420. In R. F. Barnes, D. A. Miller, C. J. Nelson (eds.). *Forages*. Vol. 1. *An Introduction to Grassland Agriculture*. Iowa State University Press, Ames.

Mosier, A. R. 1989. Chamber and isotope techniques. Pp. 175–187. In M. O. Andreae and D. S. Schimel (eds.). *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere*. John Wiley and Sons, Chichester, U.K.

Mosier, A. R. 1990. Gas flux measurement techniques with special reference to techniques suitable for measurements over large ecologically uniform areas. Pp. 289–301. In A. F. Bouwman (ed.). *Soils and the Greenhouse Effect*. John Wiley and Sons, Chichester, U.K.

Mosier, A., D. Schimel, D. Valentine, K. Bronson, and W. Parton. 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. *Nature* 350:330–332.

Mosier, A. R., J. M. Duxbury, J. R. Freney, O. Heinemeyer, and K. Minami. 1996. Nitrous oxide emissions from agricultural fields: Assessment, measurement and mitigation. *Plant Soil* 181:95–108.

Mosier, A. R., W. J. Parton, D. W. Valentine, D. S. Ojima, D. S. Schimel, and O. Heinemeyer. 1997. CH_4 and N_2O fluxes in the Colorado shortgrass steppe. 2. Long-term impact of land use change. *Global Biogeochem Cycles* 11:29–42.

Mosier, A. R., J. M. Duxbury, J. R. Freney, O Heinemeyer, and K. Minami. 1998a. Assessing and mitigating N_2O emissions from agricultural soils. *Climatic Change* 40:7–38.

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput. 1998b. Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle: Organization for Economic Cooperation and Development/Intergovernmental Panel on Climate Change/International Energy Agency phase II development of IPCC guidelines for national greenhouse gas inventory methodology. *Nutr Cycling Agroecosyst* 52:225–248.

Mosier, A. R., J. M. Duxbury, J. R. Freney, O. Heinemeyer, K. Minami, and D. E. Johnson. 1998c. Mitigating agricultural emissions of methane. *Clim Change* 40:39–80.

Murray, B. C. and B. A. McCarl. 2001. *General Measures for Leakage in Forest Sequestration Policies*. Working Paper. Environmental and Natural Resource Economics Program, Research Triangle Institute, Raleigh, North Carolina.

National Academy of Sciences (NAS). 2000. *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution*. National Academy Press, Washington, D.C.

National Assessment Synthesis Team (NAST), U.S. Global Change Research Program. 2000. *Climate Change Impacts on the United States: The Potential Consequences of Climate Variability and Change. Overview Report*. Cambridge University Press, New York, New York.

National Assessment Synthesis Team (NAST), U.S. Global Change Research Program. 2001. *Climate Change Impacts on the United States. Foundation Report*. Cambridge University Press, New York, New York.

National Soil Survey Center (NSSC). 1997. *National Soil Survey Laboratory Characterization Data*. Digital Data. National Resources Conservation Service, U.S. Department of Agriculture, Lincoln, Nebraska.

Nelson, D. W. 1982. Gaseous losses of nitrogen other than through denitrification. Pp. 327–363. In F. J. Stevenson (ed.). *Nitrogen in Agricultural Soils*. American Society of Agronomy, Madison, Wisconsin.

Nesbit, S. P. and G. A. Breitenbeck. 1992. A laboratory study of factors influencing methane uptake by soil. *Agric Ecosyst Environ* 41:39–54.

Neue, H. U. 1992. Agronomic practices affecting methane fluxes from rice cultivation. In D. S. Ojima and B. H. Svensson (eds.). *Trace Gas Exchange in a Global Perspective. Ecol Bull (Copenhagen)* 42:174–182.

Niblack, T. L. (ed.). 1999. Soybean cyst nematode management guide. North Central Soybean Research Program, University of Missouri Printing Service, Columbia.

Nordhaus, W.D. 1991. A sketch of the economics of the greenhouse effect. *Am Econ Rev* 81:146–150.

Nouchi, I., S. Mariko, and K. Aoki. 1990. Mechanisms of methane transport from the rhizosphere to the atmosphere through rice

plants. *Plant Physiol* 94:59–66.

Nusser, S. M. and J. J. Goebel. 1997. The National Resources Inventory: A long-term multi-resource monitoring programme. *Environ Ecol Stat* 4:181–204.

Nyakatawa, E. Z., K. C. Reddy, and J. L. Lemunyon. 2001. Predicting soil erosion in conservation tillage cotton production systems using the revised universal soil loss equation (RUSLE). *Soil Till Res* 57:213–224.

Oades, J. M. 1993. The role of biology in the formation, stabilization and degradation of soil structure. *Geoderma* 56:377–400.

Oenema, O., G. Gebauer, M. Rodriguez, A. Sapek, S. C. Jarvis, W. J. Corre, and S. Yamulki. 1998. Controlling nitrous oxide emissions from grassland livestock production systems. *Nutr Cycling Agroecosyst* 52:141–149.

Ogle, S. M., F. J. Breidt, M. D. Eve, and K. Paustian. 2003. Uncertainty in estimating land use and management impacts on soil organic carbon storage for U.S. agricultural lands between 1982 and 1997. *Global Change Biology* 9:1521–1542.

Ojima, D. S., W. J. Parton, D. S. Schimel, and C. E. Ownesby. 1990. Simulated impacts of annual burning on prairie ecosystems. Pp. 118–132. In S. L. Collins and L. L. Wallace (eds.). *Fire in North American Tallgrass Prairies*. University of Oklahoma Press, Norman.

Ojima, D. S., B. O. M. Dirks, E. P. Glenn, C. E. Ownesby, and J. O. Scurlock. 1993a. Assessment of C budget of grasslands and dry lands of the world. *Water Air Soil Pollut* 70:95–109.

Ojima, D. S., W. J. Parton, D. S. Schimel, J. M. O. Scurlock, and T. G. F. Kittel. 1993b. Modeling the effects of climatic and CO_2 changes on grassland storage of soil C. *Water Air Soil Pollut* 70:643–657.

Olmstead, J., D. Sunding, D. Parker, R. Howitt, and D. Zilberman. 1997. Water marketing in the '90s: Entering the electronic age. *Choices* (Third Quarter):24–28.

Owensby, C. E., J. N. Ham, A. K. Knapp, D. Bremer, and L. M. Auen. 1996. Water vapor fluxes and their impact under elevated CO_2 in C_4 tallgrass prairie. *Global Change Biol* 3:189–195.

Owensby, C. E., J. M. Ham, A. K. Knapp, and L. M. Auen. 1999. Biomass production and species composition change in a tallgrass prairie ecosystem after long-term exposure to elevated atmospheric CO_2. *Global Change Biol* 5:497–506.

Page, A. L., R. H. Miller, and D. R. Keeney (eds.). 1982. *Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties*. 2nd ed. Agronomy Monograph 9. American Society of Agronomy/Soil Science Society of America, Madison, Wisconsin.

Panek, J. A., P. A. Matson, I. Ortiz-Monasterio, and P. Brooks. 2000. Distinguishing nitrification and denitrification sources of N_2O in a Mexican wheat system using ^{15}N as a tracer. *Ecol Appl* 10:506–514.

Parkin, T. B. 1987. Soil microsites as a source of denitrification variability. *Soil Sci Soc Am J* 51:1194–1199.

Parks, P. J. and I. W. Hardie. 1995. Least cost forest carbon reserves: Cost-effective subsidies to convert marginal agricultural lands to forests. *Land Econ* 71(1):122–136.

Parks, P. J. and I. W. Hardie. 1996. Forest carbon sinks: Cost and effects of expanding the Conservation Reserve Program. *Choices* Second Quarter:37–39.

Parton, W. J., J. A. Morgan, and R. H. Kelly. 2000. Soil C in grassland systems: Responses to environmental change. Pp. 371–398. In Follett, F. R., J. M. Kimble, and R. Lal (eds.). *Carbon Sequestration Potential of U.S. Grazing Land*. Ann Arbor Press, Chelsea, Michigan.

Parton, W. J., J. W. B. Stewart, and C. V. Cole. 1988. Dynamics of

C, N, P and S in grassland soils: A model. *Biogeochemistry* 5:109–131.

Parton, W. J., D. S. Schimel, C. V. Cole, and D. S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Sci Soc Am J* 51:1173–1179.

Patterson, D. T. 1993. Implications of global climate change for impact of weeds, insects, and plant diseases. Pp. 273–280. In *International Crop Science I*. Crop Science Society of America, Madison, Wisconsin.

Paul, E. A. and F. E. Clark. 1989. *Soil Microbiology and Biochemistry*. Academic Press, San Diego, California. 275 pp.

Paul, E. A., K. Paustian, E. T. Elliot, and C. V. Cole (eds.). 1997. *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Paustian, K. 1994. Modeling soil biology and biogeochemical processes for sustainable agriculture research. Pp. 182–196. In C. Pankhurst, B. M. Doube, V. V. S. R. Gupta, and P. R. Grace (eds.). *Management of Soil Biota in Sustainable Farming Systems*. CSIRO Publishers, Melbourne, Australia.

Paustian, K., H. P. Collins, and E. A. Paul. 1997. Management controls on soil carbon. Pp. 15–49. In E. A. Paul, K. Paustian, E. T. Elliott, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Paustian, K., W. J. Parton, and J. Persson. 1992. Modeling soil organic matter in organic-amended and N-fertilized long-term plots. *Soil Sci Soc Am J* 56:476–488.

Paustian, K., E. T. Elliott, G. A. Peterson, and K. Killian. 1996. Modeling climate, CO_2 and management impacts on soil carbon in semi-arid agroecosystems. *Plant Soil* 187:351–365.

Paustian, K., O. Andren, H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. van Noordwijk, and P. Woomer. 1997a. Agricultural soil as a C sink to offset CO_2 emissions. *Soil Use Manage* 13:230–244.

Paustian, K., E. Levine, W. M. Post, I. R. Ryzhova. 1997b. The use of models to integrate information and understanding of soil C at the regional scale. *Geoderma* 79:227–260.

Paustian, K., C. V. Cole, D. Sauerbeck, and N. Sampson. 1998. CO_2 mitigation by agriculture: An overview. *Clim Change* 40:135–162.

Paustian, K., K. Killian, J. Cipra, E. T. Elliott, G. Bluhm, and J. L. Smith. 2001. Modeling and regional assessment of soil carbon: A case study of the Conservation Reserve Program. Pp. 207–225. In R. Lal and K. McSweeney (eds.). *Soil Carbon Sequestration and the Greenhouse Effect*. Special Publication No. 52. Soil Science Society of America, Madison, Wisconsin.

Paustian, K., J. Brenner, K. Killian, J. Cipra, S. Williams, E. T. Elliott, M. D. Eve, T. Kautza, and G. Bluhm. 2002. State-level analyses of C sequestration in agricultural soils. Pp. 193–204. In J. M. Kimble, R. Lal, and R. F. Follett (eds.). *Agriculture Practices and Policies for Carbon Sequestration in Soil*. CRC Press, Boca Raton, Florida.

Pautsch, G., L. Kurkalova, B. Babcock, and C. Kling. 2001. The efficiency of sequestering carbon in agricultural soils. *Contemp Econ Pol* 19:123–134.

Petchawee, S. and W. Chaitep. 1995. Organic matter management in upland systems in Thailand. Pp. 21–26. In R. D. B. Lefroy, G. J. Blair, and E. T. Crawell (eds.). *Organic Matter Management in Upland Systems in Thailand*. Australian Center for International Agricultural Research, Proceedings 56:21–26, Canberra, Australia.

Peterson, G. A. and D. G. Westfall. 1997. Management of dryland

agroecosystems in the central Great Plains of Colorado. Pp. 335–342. In E. A. Paul, K. Paustian, E. T. Elliott, and C. V. Cole (eds.). *Soil Organic Matter in Temperate Agroecosystems: Long-Term Experiments in North America*. CRC Press, Boca Raton, Florida.

Peterson, G. A., A. D. Halvorson, J. L. Havlin, O. R. Jones, D. J. Lyon, and D. L. Tanaka. 1998. Reduced tillage and increasing cropping intensity in the Great Plains conserves soil C. *Soil Till Res* 47:207–218.

Piovesan, G. and J. M. Adams. 2000. Carbon balance gradient in European forests: Interpreting EUROFLUX. *J Veg Sci* 11:923–926.

Plantinga, A. J., T. Mauldin, and D. J. Miller. 1999. An econometric analysis of the costs of sequestering carbon in forests. *Am J Agric Econ* 81:812–824.

Point Carbon. 2003. Carbon Market Europe, September 26, 2003, <http://www.pointcarbon.com/wimages/Carbon_Market_Europe_26_September_2003.pdf> (9 December 2003)

Poorter, H. 1993. Interspecific variation in the growth response of plants to an elevated ambient CO_2 concentration. *Vegetatio* 104/105:77–97.

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

Post, W. M., W. R. Emanuel, P. J. Zinke, and A. G. Stangenberger. 1982. Soil carbon pools and world life zones. *Nature* 298:156–159.

Post, W. M., R. C. Izaurralde, L. K. Mann, and N. Bliss. 1999. Monitoring and verifying soil organic carbon sequestration. Pp. 49–66. In N. J. Rosenberg, R. C. Izuarralde, and E. L. Malone (eds.). *Carbon Sequestration in Soils: Science, Monitoring, and Beyond*. Proceedings of St. Michaels Workshop, December 1998. Battelle Press, Columbus, Ohio.

Poth, M. and D. D. Focht. 1985. [15]N kinetic analysis of N_2O production by *Nitrosomonas europaea* and examination of nitrifier denitrification. *Appl Environ Microbiol* 49:1134–1141.

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

Prather, M., R. Derwent, D. Ehhalt, P. Frawer, E. Sanhueza, and X. Zhou. 1995. Other trace gases and atmospheric chemistry. Pp. 73–126. In J. Houghton, L. G. Meira, E. Haites, N. Harris, and K. Maskell (eds.). *Climate Change 1994: Radiative Forcing of Climate Changes and an Evaluation of the International Panel on Climate Change IS92 Emission Scenarios*. Cambridge University Press, New York, New York.

Raich, J. W. and C. S. Potter. 1995. Global patterns of carbon dioxide emissions from soils. *Global Biogeochem Cycles* 9:23–36.

Rasmussen, P. E., R. R. Allmaras, C. R. Rohde, and N. C. Roager, Jr. 1980. Crop residue influences on soil carbon and nitrogen in a wheat-fallow system. *Soil Sci Soc Am J* 44:596.

Raupach, M. R., O. T. Denmead, and F. X. Dunin. 1992. Challenges in linking atmospheric CO_2 concentrations to fluxes at local and regional scales. *Austr J Bot* 40:697–716.

Reeburgh, W. S., S. C. Whalen, and M. J. Alperin. 1993. The role of methylotrophy in the global methane budget. Pp. 1–14. In J. C. Murrell and D. P. Kelly (eds.). *Microbial Growth on C-1 Compounds*. Intercept, Andover, U.K.

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

Reicosky, D. A. and M. J. Lindstrom. 1995. Impact of fall tillage on short-term carbon dioxide flux. Pp. 177–187. In R. Lal, J. Kimble, E. Levine, and B. A. Stewart (eds.). *Soils and Global Change*. CRC Press, Boca Raton, Florida.

Reilly, J. M. and K. O. Fuglie. 1998. Future yield growth in field crops: What evidence exists? *Soil Till Res* 47:275–290.

Reilly, J., N. Hohmann, and S. Kane. 1994. Climate change and agricultural trade: Who benefits? Who loses? *Global Environ Change* 4:24–36.

Reilly, J., R. Prinn, J. Harnisch, J. Fitxmaurice, H. Jacoby, D. Kickleighter, J. Melillo, P. Stone, A. Sokolov, and C. Wang. 1999. Multigas assessment of the Kyoto Protocol. *Nature* 401:549–555.

Reilly, J., J. Graham, J. Hrubovcak (team leaders), D. Abler, R. Darwin, S. Hollinger, C. Izaurralde, S. Jagtap, J. Jones, J. Kimble, B. McCarl, L. Mearnes, D. Ojima, E. Paul, K. Paustina, S. Riha, N. Rosenberg, C. Rosenzweig, and F. Tubiello. 2002. *Changing Climate and Changing Agriculture: Report of the Agriculture Sector Assessment Team, U.S. National Assessment*. U.S. Global Change Research Program, National Assessment of Climate Variability, Cambridge University, Cambridge, U.K.

Reilly, J., F. Tubiello, B. McCarl, D. Abler, R. Darwin, K. Fuglie, S. Hollinger, C. Izaurralde, S. Jagtap, J. Jones, L. Mearnes, D. Ojima, E. Paul, K. Paustian, S. Riha, N. Rosenberg, and C. Rosenzweig. 2003. U.S. agriculture and climate change: New results. *Climatic Change* 57:43–69.

Renard, K. G., G. R. Foster, G. A. Weesies, and J. P. Porter. 1991. RUSLE: Revised Universal Soil Loss Equation. *J Soil Water Conserv* 46:30–33.

Rice, C. W. and C. E. Owensby. 2001. Fire and grazing impacts on soil carbon in rangelands. Pp. 323–342. In R. Follett, J. M. Kimble, and R. Lal (eds.). *Carbon Sequestration Potential of U.S. Grazing Lands*. Ann Arbor Press, Chelsea, Michigan.

Rice, C. W., T. C. Todd, J. L. Blair, T. Seastedt, R. A. Ramundo, and G. W. T. Wilson. 1998. Belowground biology and processes. Pp. 244–264. In A. K. Knapp, J. M. Briggs, D. C. Hartnett, and S. L. Collins (eds.). *Konza Prairie Long-term Ecology Research*. Oxford Press, New York, New York.

Rice, C. W., A. B. Omay, C. J. Dell, M. A. Williams, and Y. Espinoza. 1999. Soil organic matter in grasslands: Response to climate and land management. Global change and terrestrial ecosystems. P. 44 In *Focus 3 Conference on Food and Forestry: Global Change and Global Challenges*. University of Reading, Reading, U.K., September 20–23, 1999.

Richards, K. R. 1997. The time value of carbon in bottom-up studies. *Crit Rev Environ Sci and Tech* 27:S279–S292.

Risser, P. G., E. C. Birney, H. D. Blocker, S. W. May, W. J. Parton, and J. A. Wiens. 1981. *The True Prairie Ecosystem*. US/IBP Synthesis Series 16. Hutchinson Ross Publishers, Stroudsburg, Pennsylvania.

Robertson, G. P. 1997. Nitrogen use efficiency in row-crop agriculture: Crop nitrogen use and soil nitrogen loss. Pp. 347–365. In L. Jackson (ed.). *Ecology in Agriculture*. Academic Press, New York, New York.

Robertson, G. P. 1999. Denitrification. Pp. 181–190. In M. E. Sumner (ed.). *Handbook of Soil Science*. CRC Press, Boca Raton, Florida.

Robertson, G. P. 2004. Abatement of nitrous oxide, methane, and the other non-CO_2 greenhouse gases. Pp. 493–506. In C. B. Field (ed.). *Towards CO_2 Stabilization*. Island Press, Washington, D.C.

Robertson, G. P., E. A. Paul, and R. R. Harwood. 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. *Science*

289:1922–1925.

Robertson, G. P., K. M. Klingensmith, M. J. Klug, E. A. Paul, J. R. Crum, and B. G. Ellis. 1997. Soil resources, microbial activity, and primary production across an agricultural ecosystem. *Ecol Appl* 7:158–170.

Robles, M. D. and I. C. Burke. 1998. Soil organic matter recovery on conservation reserve program fields in southeastern Wyoming. *Soil Sci Soc Am J* 62:725–730.

Rogers, E. 1962. *Diffusion of Innovations*. Free Press of Glencoe, New York, New York.

Rosenzweig, C. and D. Hillel. 1998. *Climate Change and the Global Harvest: Potential Impacts of the Greenhouse Effect on Agriculture*. Oxford University Press, New York, New York. 324 pp.

Rosenzweig, C. and M. L. Parry. 1994. Potential impact of climate change on world food supply. *Nature* 367:133–138.

Rosenzweig, C., A. Iglesias, X. B. Yang, P. Epstein, and E. Chivian. 2000. *Climate Change and U.S. Agriculture: The Impacts of Warming and Extreme Weather Events on Productivity, Plant Diseases, and Pests*. Center for Health and the Global Environment, Harvard Medical School, Boston, Massachusetts.

Rosenzweig, C., F. N. Tubiello, R. Goldberg, E. Mills, and J. Bloomfield. 2002. Increased crop damage in the US from excess precipitation under climate change. *Global Environmental Change* 12:197-202.

Roy, K. W., J. C. Rupe, D. E. Hershman, and T. S. Abney. 1997. Sudden death syndrome. *Plant Dis* 81:1100–1111.

Rozanov, B. G., V. Targulian, and D. S. Orlov. 1990. Soils. Pp. 203–214. In B. L. Turner, W. C. Clark, R. W. Kates, J. F. Richards, J. T. Mathews, and W. B. Meyer (eds.). *The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere Over the Past 300 Years*. Cambridge University Press, Cambridge, U.K.

Russell, J. S. 1960. Soil fertility changes in long-term experimental plots at Kybybolite, South Australia. I. Changes in pH, total nitrogen, organic carbon, and bulk density. *Aust J Agric Res* 11:902–926.

Rypdal, K. and W. Winiwarter. 2001. Uncertainties in greenhouse gas emission inventories: Evaluation, comparability, and implications. *Environ Sci Pol* 4:107–116.

Safley, L. M., M. E. Casada, J. W. Woodbury, and K. F. Roos. 1992. Global methane emissions from livestock and poultry manure. USEPA report 400/1B91/048. Office of Air and Radiation, U.S. Environmental Protection Agency, Washington, D.C.

Salt, D. T., B. L. Brooks, and J. B. Whitaker. 1995. Elevated carbon dioxide affects leaf-miner performance and plant growth in docks (*Rumex* spp.). *Global Change Biol* 1:153–156.

Sass, R. L. 1994. Short summary chapter for methane. Pp. 1–8. In K. Minami, A. Mosier, and R. Sass (eds.). *CH_4 and N_2O: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*. National Institute for Agro-Environmental Sciences, Yokendo Publishers, Tokyo.

Sass, R. L., F. M. Fisher, Y. B. Wang, F. T. Turner, and M. F. Jund. 1992. Methane emission from rice fields: The effect of floodwater management. *Global Biogeochem Cycles* 6:249–262.

Scharpenseel, H. W. and P. Becker-Heidmann. 1992. Twenty-five years of radiocarbon dating of soils: Paradigm or error and learning. *Radiocarbon* 34:541–549.

Schimel, D., I. G. Enting, M. Heimann, T. M. L. Wigley, D. Raynaud, D. Alves, and U. Siegenthaler. 1995. CO_2 and the carbon cycle. P. 41. In J. T. Houghton, L. G. Meira Filho, J. Bruce, H. Lee, B. A. Callander, E. Haites, N. Harris, and K. Maskell (eds.). Climate Change 1994. *Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios*. Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.

Schlesinger, W. H. 1986. Changes in soil carbon storage and associated properties with disturbance and recovery. Pp. 194–220. In J. R. Trabalka and D. E. Reichle (eds.). *The Changing Carbon Cycle: A Global Analysis*. Springer-Verlag, New York, New York.

Schlesinger, W. H. 1990. Evidence from chronosequences studies for a low carbon-storage potential of soils. *Nature* 348:232–234.

Schlesinger, W. H. 1995. An overview of the carbon cycle. Pp. 9–25. In R. Lal, J. Kimble, E. Levine, and B. A. Stewart (eds.). *Advances in Soil Science: Soils and Global Change*. CRC Press, Lewis Publishers, Boca Raton, Florida.

Schlesinger, W. H. 1997. *Biogeochemistry: An Analysis of Global Change*. Academic Press, San Diego, California.

Schlesinger, W. H. 1999. Carbon sequestration in soils. *Science* 284:2095.

Schlesinger, W. H. 2000. Carbon sequestration in soils: Some cautions amidst optimism. *Agric Ecosyst Environ* 82:121–127.

Schneider, U. 2000. Greenhouse gas emission mitigation in the U.S. agricultural sector, an economic assessment. Ph.D. diss., Texas A&M University, College Station.

Schnitzer, M. 1991. Soil organic matter: The next 75 years. S*oil Sci* 75:41– 58.

Schnürer, J., M. Clarholm, and T. Rosswall. 1985. Microbial biomass and activity in an agricultural soil with different organic matter contents. *Soil Biol Biochem* 17:611–618.

Scholes, R. J. and M. R. van der Merwe. 1996. Sequestration of carbon in savannas and woodlands. *Environ Profess* 18:96–103.

Schuman, G. E., J. E. Herrick, and H. H. Janzen. 2000. Soil carbon dynamics in rangelands. Pp. 267–290. In R. F. Follett, J. M. Kimble, and R. Lal (eds.). *Carbon Sequestration Potential of U.S. Grassland*. Ann Arbor Press, Chelsea, Michigan.

Schutz, H., A. Holzapfel-Pschron, R. Conrad, H. Rennenberg, and W. Seiler. 1989. A 3-yr continuous record on the influence of daytime, season and fertilizer treatment on methane emission rates from an Italian rate [sic] paddy. *J Geophys Res* 94:16405–16416.

Seastedt, T. R., B. P. Hayden, C. E. Owensby, and A. K. Knapp. 1998. Climate change, elevated CO_2, and predictive modeling. Pp. 283–300. In A. K. Knapp, J. M. Briggs, D. C. Hartnett, and S. L. Collins (eds.). *Konza Prairie Long-term Ecology Research*. Oxford Press, New York, New York.

Segerson, K. 1988. Uncertainty and incentives for nonpoint pollution control. *J Environ Econ Manage* 15:87–98.

Seiler, W., R. Conrad, and D. Scharffe. 1984. Field studies of methane emission from termite nests into the atmosphere and measurements of methane uptake by tropical soils. *J Atmos Chem* 1:171–186.

Sexstone, A. J., T. B. Parkin, and J. M. Tiedje. 1986. Denitrification response to soil wetting in aggregated and unaggregated soil. *Soil Biol Biochem* 20:767–769.

Shibata, M. 1994. Methane production in ruminants. Pp. 105–115. In K. Minami, A. Mosier, and R. Sass (eds.). *CH_4 and N_2O: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*. National Institute for Agro-Environmental Sciences, Yokendo Publishers, Tokyo.

Shibata, M., F. Terada, M. Kurihara, and K. Iwasaki. 1993. Estimation of methane production in ruminants. *Anim Sci Technol* 64:790–795.

Shoji, S., J. A. Delgado, A. Mosier and Y. Miura. 2001. Use of control release fertilizers and nitrification inhibitors to increase

nitrogen use efficiency and to conserve air and water quality. *Commun Soil Sci Plant Anal* 32:1051–1070.

Shortle, J. and J. Dunn. 1986. The relative efficiency of agricultural source water pollution control policies. *Am J Agric Econ* 68:87–98.

Smith, J. E. and L. S. Heath. 2001. Identifying influences on model uncertainty: An application using a forest carbon budget model. *Environ Manage* 27:253–267.

Smith, K. A., I. P. McTaggart, K. E. Dobbie, and F. Conen. 1998. Emissions of N_2O from Scottish agricultural soils, as a function of fertilizer N. Pp. 77–105. In A. Mosier, G. Abrahamsen, L. Bouwman, O. Bockman, H. Drange, S. Frolking, R. Howarth, C. Kroeze, O. Oenema, K. Smith, and M. A. Bleken (eds.). *Nutrient Cycling in Agroecosystems*. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Smith, P., D. S. Powlson, and M. Glendining. 1996. Establishing a European GCTE soil organic matter network (SOMNET). Pp. 81–97. In D. S. Powlson, P. Smith, and J. U. Smith (eds.). *Evaluation of Soil Organic Matter Models Using Existing Long-term Datasets*. North Atlantic Treaty Organization Advanced Science Institutes Series. Vol. 38. Springer-Verlag, Berlin.

Smith, P., J. Brenner, K. Paustian, G. Bluhm, J. Cipra, M. Easter, E.T. Elliott, K. Killian, D. Lamm, J. Schuler and S. Williams. 2002. *Quantifying the Change in Greenhouse Gas Emissions Due to Natural Resource Conservation Practice Application in Indiana*. Final report to the Indiana Conservation Partnership. Colorado State University Natural Resource Ecology Laboratory and USDA Natural Resources Conservation Service, Fort Collins, Colorado.

Smith, R. C. 1954. An analysis of 100 years of grasshopper population in Kansas (1854–1954). *Kansas Acad Sci* 57:397–433.

Sobecki, T. M., D. L. Moffitt, J. Stone, C. D. Franks, and A. G. Mendenhall. 2001. A broad-scale perspective on the extent, distribution, and characteristics of U.S. grazing lands. Pp. 21–63. In R. F. Follett, J. M. Kimble, and R. Lal (eds.). *Carbon Sequestration Potential of U.S. Grazing Land*. Ann Arbor Press, Chelsea, Michigan.

Sohngen, B. and R. Sedjo. 2000. Potential carbon flux from timber harvests and management in the context of a global timber market. *Clim Change* 44:151–172.

Sommerfeld, R. A., A. R. Mosier, and R. C. Musselman. 1993. CO_2, CH_4, and N_2O flux through a Wyoming snowpack and implications for global budgets. *Nature* 361:140–142.

Sowden, F. J. and H. J. Atkinson. 1968. Effect of long-term additions of various organic amendments on the organic matter of a clay and a sand. *Can J Soil Sci* 48:323–330.

Sperow, M., M. D. Eve, and K. Paustian. 2003. Potential soil C sequestration on U.S. agricultural soils. *Climatic Change* 57:319–339.

Squires, V., E. P. Glenn, and A. T. Ayoub. 1995. Combatting global climate change by combatting land degradation. *Workshop Proceedings of the United Nations Environmental Programme (UNEP)* Nairobi, Kenya, September 4–8, 1995.

Stavins, R. 1999. The costs of carbon sequestration: A revealed preference approach. *Am Econ Rev* 89:994–1009.

Steudler, P. A., R. D. Bowden, J. M. Melillo, and J. D. Aber. 1989. Influence of nitrogen fertilization on methane uptake in temperate forest soils. *Nature* 341:314–316.

Stevens, R. J., J. R. Laughlin, and R. C. Hood. 1997. Measuring the contribution of nitrification and denitrification to the flux. *Soil Biol Biochem* 29:139.

Stinner, B. R., R. A. J. Taylor, R. B. Hammond, F. F. Purrington, D. A. McCartney, N. Rodenhouse, and G. W. Barrett. 1989.

Potential effects of climate change on plant-pest interactions. Pp. 8-1–8-35. In J. B. Smith and D. A. Tirpak (eds.). *The Potential Effects of Global Climate Change in the United States*. EPA-230-05-89-053. Appendix C Agriculture, Vol. 2. U.S. Environmental Protection Agency, Washington, D.C.

Striegl, R. G., T. A. McConnaughey, D. C. Thorstenson, E. P. Weeks, and J. C. Woodward. 1992. Consumption of atmospheric methane by desert soils. *Nature* 357:145–147.

Strzpek, K. M., D. C. Major, C. Rosenzweig, A. Iglesias, D. N. Yates, A. Holt, and D. Hillel. 1999. New methods of modeling water availability for agriculture under climate change: The U.S. cornbelt. *J Am Water Res Assoc* 35:1639–1655.

Sunding, D. and D. Zilberman. 2001. The agricultural innovation process: Research and technology adoption in a changing agricultural sector. Pp. 207–261. In B. Gardner and G. C. Rausser (eds.). *Handbook of Agricultural and Resource Economics*. North Holland, Amsterdam.

Sutherst, R. W. 1990. Impact of climate change on pests and diseases in Australia. *Search* 21:230–232.

Swift, M. J., O. W. Heal, and J. M. Anderson. 1979. *Decomposition in Terrestrial Ecosystems*. Blackwell, Oxford, U.K. 372 pp.

Tans, P. P., I. Y. Fung, and T. Takahashi. 1990. Observational constraints on the global atmospheric carbon dioxide budget. *Science* 247:1431–1438.

Tate, R. L., III. 1987. *Soil Organic Matter: Biological and Ecological Effects*. John Wiley and Sons, New York, New York. 291 pp.

Thornley, J. H. M. and M. G. R. Cannell. 1997. Temperate grassland responses to climate change: An analysis using the Hurley Pasture Model. *Ann Bot* 80:205–221.

Tian, G., B. T. Kang, and L. Brussard. 1992. Biological effects of plant residues with contrasting chemical compositions under humid tropical conditions: Decomposition and nutrient release. *Soil Biol Biochem* 24:1051–1060.

Tietenberg, T. 1984. Marketable emission permits in principle and practice. Presented at the 75th Anniversary Conference of the Stockholm School of Economics, Stockholm, Sweden, August 1984. Resources for the Future, Washington, D.C. 123 pp.

Timmermann, A., J. Oberhuber, A. Bacher, M. Each, M. Latif, and E. Roeckner. 1999. Increased El-Niño frequency response in a climate model forced by future response to greenhouse warming. *Nature* 398:694–697.

Tisdall, J. M. and J. M. Oades. 1982. Organic matter and water-stable aggregates in soils. *J Soil Sci* 33:141–163.

Tomasi, T., K. Segerson, and J. Braden. 1994. Issues in the design of incentive schemes for nonpoint source pollution control. Pp. 1–37. In C. Dosi and T. Tomasi (eds.). *Nonpoint Source Pollution Regulation: Issues and Analysis*. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Tubiello, F. N., C. Rosenzweig, R. A., Goldberg, S. Jagtap, and J. W. Jones. 2002. Effects of climate change on U.S. crop production: Simulation results using two different GCM scenarios. Part I. Wheat, potato, maize, and citrus. *Clim Res* 20:259–270.

United Nations Food and Agricultural Organization (FAO). 2000. FAO agricultural data bases are obtainable on the world wide web, <http://www.fao.org>

United Nations Framework Convention on Climate Change (UNFCCC). 2000. United States submission on land-use, land-use change, and forestry. Pp. 84–144. In *United Nations Framework Convention on Climate Change, Subsidiary Body for Scientific and Technological Advice. Methodological Issues: Land-Use, Land-Use Change and Forestry, Submissions from Parties*, <http://unfccc.int/resource/docs/2001/sbsta/

misc06a01.pdf> (7 October 2003)

U.S. Department of Agriculture (USDA). 1997. *Agricultural Resources and Environmental Indicators, 1996–97.* Economic Research Service, U.S. Department of Agriculture, Washington, D.C.

U.S. Department of Agriculture–Natural Resources Conservation Service (USDA–NRCS). 1994. State soil geographic (STATSGO) data base. Data use information. Miscellaneous Publication 1492. Natural Resources Conservation Service, U.S. Department of Agriculture, Ft. Worth, Texas.

U. S. Department of Agriculture–Natural Resources Conservation Service (USDA–NRCS). 1995. Soil survey geographic (SSURGO) data base. Data use information. Miscellaneous Publication 1527. Natural Resources Conservation Service, U.S. Department of Agriculture, Ft. Worth, Texas.

U.S. Department of Agriculture–Natural Resources Conservation Service (USDA–NRCS). 1996. *Soil Survey Laboratory Methods Manual.* Soil Survey Investigations Report 42. Version 3.0. U.S. Government Printing Office, Washington, D.C.

U.S. Department of Agriculture–Natural Resources Conservation Service (USDA–NRCS). 1997. National Resources Inventory, <www.nrcs.usda.gov/technical/NRI> (3 July 2003)

U.S. Department of Energy (DOE). 1999. Emission and reduction of greenhouse gases from agriculture and food manufacturing. A Summary White Paper. DOE/GOB10099B646. U.S. Department of Energy/Office of Energy Efficiency and Renewable Energy/Office of Industrial Technologies, Washington, D.C. 35 pp, <http://www.oit.doe.gov/agriculture/pdfs/greenhousegases.pdf>

U.S. Environmental Protection Agency (USEPA). 1992. Anthropogenic methane emissions in the U.S. Estimates for 1990. Report Number 430BRB93B003. Office of Air and Radiation, U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1993a. Methane emissions from livestock manure. In *Global Methane Emissions. Report to the Congress.* Prepared by Climate Change Division, Office of Policy, Planning, and Evaluation, U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1993b. Options for reducing methane internationally. Vol. II. International Opportunities. USEPA 430R93B006B. U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 1994. Assessment of alternative management practices and policies affecting soil carbon in agroecosystems of the central United States. EPA/600/R–94/067. U.S. Environmental Protection Agency, Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 2002. Inventory of U.S. greenhouse gas emissions and sinks: 1990–2000. EPA 430-R-02-003. U.S. Environmental Protection Agency, Office of Policy, Planning and Evaluation, Washington, D.C.

van Aardenne, J. A., C. Kroeze, and L. Hordijk. 1998. Analysis of the uncertainties in the IPCC default method for estimate N_2O emissions from agricultural soils. Pp. 305–308. In K. Chan, S. Tarantola, and F. Campolongo (eds.). Second International Symposium in Sensitivity Analysis of Model Output. EURreport 177758. Venice, Ca'Dolphin, Italy.

Vance, E. D., P. C. Brookes, and D. S. Jenkinson. 1987. An extraction method for measuring soil microbial biomass. *Soil Biol Biochem* 19:703–707.

Velthof, G. L., M. L. van Beusichem, and O. Oenema. 1998. Mitigation of nitrous oxide emission from dairy farming systems. *Environ Pollut* 102:173–178.

Verma, S. B. 1990. Micrometeorological methods for measuring surface fluxes of mass and energy. In N. S. Goel and J. M. Norman (eds.). *Instrumentation for Studying Vegetation Canopies for Remote Sensing in Optical and Thermal Infrared Regions.* Special Issue, *Remote Sens Rev* 5:99–115.

Wagner-Riddle, C., G. W. Thurtell, G. E. Kidd, G. C. Edwards, and I. J. Simpson. 1996a. Micrometeorological measurements of trace gas fluxes from agricultural and natural systems. *Infrared Phys Tech* 37:51–58.

Wagner-Riddle, C., G. W. Thurtell, K. M. Ling, G. E. Kidd, and E. G. Beauchamp. 1996b. Nitrous oxide and carbon dioxide fluxes from a bare soil using a micrometeorological approach. *J Environ Qual* 25:989–907.

Wallace, A. 1994. Soil organic matter must be restored near original levels. *Comm Soil Sci Plant Anal* 25:29–35.

Walsh, M. E., D. D. T. Ugarte, S. Slinsky, R. I. Graham, H. Shapouri, and D. Ray. 1998. *Economic Analysis of Energy Crop Production in the U.S.* Great Lakes Regional Biomass Energy Program, Chicago, Illinois.

Wang, M., C. Saricks, and D. Santini. 1999. *Effects of Fuel Ethanol Use on Fuel-Cycle Greenhouse Gas Emissions.* Center for Transportation Research. Report ANL/ESD-38. Argonne National Laboratory, Argonne, Illinois.

Wassmann, R., H. U. Neue, R. S. Lantin, L. V. Buendia, and H. Rennenberg. 2000. Characterization of methane emissions from rice fields in Asia. I. Comparison among field sites in five countries. *Nutrient Cycling Agroecosys* 58:1-12.

Watson, R. T., L. G. Meira Filho, E. Sanhueza, and T. Janetos. 1992. Greenhouse gases: Sources and sinks. Pp. 25–46. In J. T. Houghton, B. A. Callander, and S. K. Varney (eds.). *Climate Change 1992. The Supplementary Reports to the International Panel on Climate Change Scientific Assessment.* Cambridge University Press, Cambridge, U.K.

Wear, D. N. and B. C. Murray. 2001. A tale of two birds: Spotted owls, trade hawks, and their impacts on U.S. softwood markets. Working Paper. Environmental and Natural Resource Economics Program, Research Triangle Institute, Research Triangle Park, North Carolina..

Webb, E. K., G. I. Pearlman, and R. Leuning. 1980. Correction of flux measurements for density effects due to heat and water vapor transfer. *Quart J Roy Meteorol Soc* 106:85–100.

Weitz, A. M., M. Keller, E. Linder, and P. M. Crill. 1999. Spatial and temporal variability of nitrogen oxide and methane fluxes from a fertilized tree plantation in Costa Rica. *J Geophys Res* 104:30097–30107.

West, T. O. and G. Marland. 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: Comparing tillage practices in the United States. *Agric Ecosyst Environ* 91:217–232.

Weyant, J. and J. N. Hill. 1999. The costs of the Kyoto Protocol: A multimodel evaluation. Introduction and overview: Special issue. *Energy Journal* [unnumbered]:vii–xliv.

Whalen, S. C., W. S. Reeburgh, and K. S. Kizer. 1991. Methane consumption and emission by taiga. *Global Biogeochem Cycles* 5:261–273.

Wienhold, F. G., H. Frahm, and G. W. Harris. 1994. Measurements of N_2O fluxes from fertilized grassland using a fast response tunable diode laser spectrometer. *J Geophys Res* 99:16557–16567.

Wilding, L. P. and L. R. Drees. 1983. Spatial variability and pedology. Pp. 83–116. In L. P. Wilding, N. W. Smeck, and G. F. Hall (eds.). *Pedogenesis and Soil Taxonomy. I. Concepts and Interactions.* Elsevier, Amsterdam.

Williams, J. R. 1995. The EPIC model. Pp. 909–1000. In V. P.

Singh (ed.). *Computer Models of Watershed Hydrology*. Water Research Publications, Highlands Ranch, Colorado.

Williams, R. E., B. W. Allred, R. M. DeNio, and H. E. Paulsen, Jr. 1968. Conservation, development, and use of the world's rangelands. *J Range Manage* 21:355–360.

Wu, J. 2000. Slippage effects of the Conservation Reserve Programs. *Am J Agric Econ* 82:979–992.

Wu, J. and B. A. Babcock. 1996. Contract design for the purchase of environmental goods from agriculture. *Am J Agric Econ* 78:935–945.

Wu, J., D. Zilberman, and B. A. Babcock. 2001. Environmental and distributional impacts of conservation targeting strategies. *J Environ Econ Manage* 41:333–350.

Yagi, K. and K. Minami. 1990. Effect of organic matter application on methane emission from some Japanese paddy fields. *Soil Sci Plant Nutr* 36:599–610.

Yang, X. B. and H. Scherm. 1997. El Niño and infectious disease. *Science* 275:739.

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