TECHNICAL WORKING GROUP ON AGRICULTURAL GREENHOUSE GASES (T-AGG) REPORT

Assessing Greenhouse Gas Mitigation Opportunities and Implementation Strategies for Agricultural Land Management in the United States

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NICHOLAS INSTITUTE FOR ENVIRONMENTAL POLICY SOLUTIONS
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What is T-AGG?

The **Technical Working Group on Agricultural Greenhouse Gases (T-AGG)** began work in November 2009 to assemble the scientific and analytical foundation for implementation of high-quality agricultural GHG mitigation activities. Mitigation activities that increase carbon storage in soil or reduce methane and nitrous oxide emissions could be an important part of U.S. and global climate change strategies. Despite the significant potential for GHG mitigation within agriculture, only a very few high-quality and widely approved methodologies for quantifying agricultural GHG benefits have been developed for various mitigation programs and markets. Much of the focus to date has been around forests on agricultural lands and manure management, rather than on production agriculture or grazing lands, where we focus our attention. However, there are now a number of new agricultural protocols under development.

T-AGG is coordinated by a team at the Nicholas Institute for Environmental Policy Solutions at Duke University with partners in the Nicholas School of the Environment at Duke and at Kansas State University, and regularly engages the expertise of a science advisory committee and cross-organizational advisory board (details below). The work was made possible by a grant from the David and Lucile Packard Foundation.

The project will produce a series of reports which survey and prioritize agricultural mitigation opportunities in the U.S. and abroad to provide a roadmap for protocol development, and provide in-depth assessments of the most promising approaches for protocol development. Experts and scientists are providing guidance throughout the process, through the advisory groups, experts meetings, and individual outreach. We will also involve the agricultural community in order to gain their feedback and guidance on the approaches assessed in our reports. We hope these reports will be of use to private or voluntary markets and registries as well as regulatory agencies that may oversee similar programs or the development of regulatory carbon markets. ***We intend for these reports to provide the fundamental information necessary for the development or review of protocols designed for agricultural GHG mitigation projects or for the design of broader programs intended to address GHG mitigation (e.g., Farm Bill)***.

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ACRONYMS AND ABBREVIATIONS

ACR	American Carbon Registry
ARMS	Agricultural Resource Management Survey of the U.S. Department of Agriculture
BAU	business as usual
BP	biophysical potential
C	carbon
CO_2	carbon dioxide
CO_2e	carbon dioxide equivalent
C-AGG	Coalition on Agricultural Greenhouse Gases
CAR	Climate Action Reserve
CCX	Chicago Climate Exchange
CDM	Clean development mechanism
CH_4	methane
CP	competitive economic potential
CRP	Conservation Reserve Program of the U.S. Department of Agriculture
EP	economic potential
EPA	U.S. Environmental Protection Agency
EQIP	Environmental Quality Incentives Program of the U.S. Department of Agriculture
GHG	greenhouse gas
GM	genetically modified
GPG	Good Practice Guidance (of the Intergovernmental Panel on Climate Change)
Gt	gigatonne (10^9 tonnes)
FASOMGHG	Forest and Agricultural Sector Optimization Model with Greenhouse Gases
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organization
LCA	life-cycle assessment
MAC	marginal abatement cost
M-AGG	Market Mechanisms for Agricultural Greenhouse Gases
Mha	million hectares
Mt	megatonne (million tonnes)
N	nitrogen
N_2O	nitrous oxide
NASS	National Agricultural Statistical Service of the U.S. Department of Agriculture
NRCS	Natural Resources Conservation Service of the U.S. Department of Agriculture
NRI	National Resources Inventory of the U.S. Department of Agriculture
REDD	reduced emissions from deforestation and degradation
SOC	soil organic carbon
SRWC	short-rotation woody crops
SSR	greenhouse gas source, sink, or reservoir
t	tonne
T-AGG	Technical Working Group on Agricultural Greenhouse Gases
USDA	U.S. Department of Agriculture
VCS	Verified Carbon Standard

EXECUTIVE SUMMARY

Agriculture currently contributes approximately 6% of total greenhouse gas (GHG) emissions in the United States (USDA 2011). Although increases in efficiency and improvements in management reduce emissions per unit of production (Burney 2010), the demand for increased production will likely outpace these improvements, leading to a net rise in emissions, without additional investment. A wide range of on-farm management practices can help to reduce these emissions and generate significant increases in carbon sequestration. Government, industry, and voluntary efforts are under way to incentivize such practices by creating new business opportunities or revenue for farmers and ranchers. The hoped-for outcome is accelerated innovation and adoption of practices that simultaneously mitigate emissions, improve resilience to climate change, and support the nutritional and energy needs of a growing population.

To achieve a balance of increased production and reduced environmental impacts, government programs and corporate supply-chain initiatives seek to motivate the use of increasingly efficient, intensive, and sustainable agricultural practices. New initiatives and programs that target GHG mitigation are considering market mechanisms (e.g., emission offsets) or other performance-based metrics (e.g., life-cycle analysis) for tracking success and making payments or purchases contingent on environmental outcomes. These initiatives and programs require information on the crops, management practices, and new technologies that can enhance GHG mitigation—information such as their viability in different regions, their economic costs or savings, their effect on production, and their net GHG emissions. In addition, performance-based approaches require quantification and verification of outcomes.

The Technical Working Group on Agricultural Greenhouse Gases (T-AGG) was formed to help assemble and provide this basic information. This assessment reviews a wide range of agricultural practices for principal cropping systems in the United States. It provides a roadmap and resource for programs and initiatives that are designing protocols, metrics, or incentives to engage farmers and ranchers in large-scale efforts to enhance GHG mitigation on working agricultural land in the United States.

In assembling information about agricultural GHG mitigation, T-AGG takes into account an evolving range of government and business policy and program options, from cap-and-trade laws to voluntary market and federal payment programs and corporate supply-chain requirements. This assessment provides a side-by-side comparison of net biophysical GHG mitigation potential (soil carbon [C], land emissions of methane [CH_4] and nitrous oxide [N_2O], and upstream or process emissions) for 42 agricultural land management activities synthesized from existing research. It also summarizes a survey that assesses the scientific community's confidence in the mitigation potential of these activities, given often limited data and highlights research coverage and gaps.

This assessment identified 28 agricultural land management activities likely to be beneficial for GHG mitigation. Five have relatively high mitigation potential due to land use changes and are applicable in only some regions. Fifteen tend to have lower mitigation potential, do not shift land use, and are applicable in many U.S. regions. The other eight have significant data gaps and need additional research. These activities include increased cropping intensity, agroforestry, histosol management, and rotational grazing for soil C sequestration or conservation, as well as irrigation improvements and improved manure application for N_2O emission reduction. Rotational grazing on pasture lands is particularly interesting. While the C sequestration potential from this practice seems positive, its broader impact on the efficiency of livestock production and the potential for broader mitigation effects is even more promising.

For the fourteen remaining activities, mitigation potential was uncertain, low, or negative. Six of these activities may deserve additional attention as they have been little studied or studies have yielded variable results. Seven of these activities have low or negative net GHG mitigation potential. The final activity, biochar application, may have significant potential, but research on the magnitude of this potential and on life-cycle implications is needed.

The adoption of these management practices primarily depends on their economic potential, given the opportunity cost of various cropping and management options, the costs and benefits of adoption, and other socioeconomic variables. With a limited land base and a large suite of management options, producers must choose what works best for them. This assessment summarizes studies in the published literature that document the economic and competitive potential of select management practices at various C prices. Only a limited suite of activities has been covered in these studies, which focus on fallow lands and tillage reduction, conversion of cropland to permanent grass or other forage, and afforestation. Higher payments for carbon tend to generate more GHG mitigation and cause shifts in the activities.

Reduced-tillage practices are incentivized at lower prices; conversion of cropland to forest or perennial grass becomes more prevalent only when prices rise (even though biophysical potential is greater per unit area, compared with tillage changes). Although model predictions can provide useful guidance, they cannot fully account for transaction costs, farm-level adoption barriers, and environmental co-benefits, all of which can affect the willingness of producers to shift various management practices.

Measuring GHG outcomes from agricultural management projects in a manner that fosters confidence but keeps costs low has been a significant challenge. Field-based sampling is appealing in its tangibility and is likely the best approach for programs focused on innovation. But variability (within soils and across fields, seasons, and rainfall events) and technical limitations can make achieving sufficient levels of certainty relatively expensive. Thus, scientific experts suggest that modeling is a better approach for large-scale implementation of known and tested management activities. Modeling options range from simple, national default factors and regional or ecozone-specific factors to the detailed, farm-level application of process models. The United States has enough data and sufficiently well-calibrated and -tested process-based models to apply regional or farm-scale approaches for most activities supported by moderate levels of research. Regional-scale approaches are less complex to implement but are less flexible than farm-based approaches.

Process-based biogeochemical models can simulate GHG dynamics under a range of changing environmental (soil physical properties, climate, topography) and management (cropping, livestock, manure, grazing practices) variables, while capturing temporal and spatial variability. These models are based on and calibrated with field research and data, but they are sometimes limited in their accuracy due to research gaps or insufficient calibration with existing research. But they can be refined as research evolves. Due to the complexity of the models, user-friendly and application-specific versions, such as COMET-Farm, will be needed for consistent and verifiable use in protocols and programs.

GHG accounting frameworks for many protocols and programs will require clear guidance for calculating baselines, determining additionality, accounting for leakage, addressing reversals in C storage, and monitoring and verifying outcomes. Standardized approaches for baseline and additionality, which use data on national, regional, or sectoral trends, are commonly used by programs in the United States because they reduce transaction costs and increase transparency. These approaches require aggregated data on agricultural management practices, which may not always be available at the level of detail needed. If farm- or project-scale approaches for baseline determination are used instead, their application must be consistent and their results verifiable. Meeting these requirements may be possible with the development of standardized user interfaces for process-based models. As long as farm-level management data can be gathered in a low-cost and verifiable manner, process-based models can produce low-cost and transparent farm-scale baseline estimates.

Bad or negative leakage will be an issue where management practices—such as reducing nitrogen fertilizer rates, reducing animal stocking rates on grazing lands, switching from annual to perennial crops, or setting aside cropland—could reduce productivity. Given the loss of profits that may come with reduced productivity, these activities may be less viable choices for producers unless greater compensation is available. Many methods, such as leakage discounts, can be used to address leakage impacts. Output- and yield-based performance accounting methods (e.g., tonnes of GHGs per tonne of corn) incorporate both positive and negative leakage and reward improvements in production efficiency (Murray and Baker 2011).

For many of the reviewed agricultural practices, reversals are either not an issue (e.g., for avoided N_2O and CH_4 emissions), or they are only a short-term concern (e.g., elimination of cover crops for a single year) for which management can compensate. Cessation of management practices that sequester soil C tend to be intentional—as when conventional tillage is reintroduced on land not tilled for many years—and the loss of stored carbon tends to occur slowly. Only those few practices that involve aboveground biomass, such as windbreaks, can result in significant immediate unintentional releases, such as those typical of forestry projects. How programs will handle the uncertain effects of climate change on the risks of reversals of stored carbon remains unclear. Despite this and other uncertainties, the work of T-AGG suggests the knowledge, data, and methods are sufficient to move forward on a number of options for mitigating GHG emissions on agricultural lands in the United States.

INTRODUCTION

This section describes the purpose of this report, the role that agriculture can play in greenhouse gas mitigation, and the policy and market drivers that may stimulate mitigation. It provides an overview of existing programs or protocols for GHG mitigation in agriculture. In addition, it describes the approach used for assessing agricultural management activities for their mitigation potential, the standardized accounting procedures that guide GHG mitigation, and the activities covered in this report.

Purpose of This Assessment

This assessment provides information on the potential and feasibility of greenhouse gas (GHG) mitigation from agricultural land management activities in the United States. Its aim is to provide a foundation and roadmap for the inclusion of agricultural practices in government programs, industry sustainability policy, or market-based accounting protocols. Prior to this effort, the debate over agriculture's role in climate policies and programs had been limited in scope. This assessment should promote a broader and more informed discussion on how to include agricultural land management in GHG mitigation.

This report provides a side-by-side comparison of the GHG mitigation potential of numerous agricultural management practices. It identifies gaps in data and research on GHG mitigation in agricultural systems in the United States, and offers insights into the state of measurement and modeling capabilities for quantifying changes in greenhouse gases. It also reviews options for establishing baselines, monitoring, verification, estimating leakage, and handling the risk of carbon sequestration reversals. The intended audience for this report includes legislative bodies that are designing new policies, government agencies that are developing new programs, carbon market registries or certification programs that are developing new mitigation protocols, supply chain initiatives that will include agricultural products, and agricultural industry representatives investigating how shifting practices can affect GHG mitigation.

Why Is Agriculture Important for GHG Mitigation?

Around the world, countries and companies continue to expand efforts to reduce greenhouse gas emissions and slow global climate change. Over the last few years attention has shifted toward agriculture and forestry, which together generate approximately one-third of global emissions. Programs and policies and funding for reduced deforestation have been moving forward, while efforts on agriculture have lagged behind. Given that agricultural expansion has been the greatest driver of deforestation and that agriculture is an equally large source of emissions, T-AGG and others are working to enhance discussion of agriculture's potential role in mitigation. Because demand for food and fiber continues to grow in response to population growth, new efficiencies and new approaches to production are needed. Moving forward multiple known approaches together could generate positive feedbacks for sustainability and food production; however, breakthrough innovation is still needed.

Box 1. GHG units and GWP

In this report, mitigation refers to metric tonnes (t) of CO_2 or CO_2 equivalent (CO_2e) of emissions avoided or emissions removed from the atmosphere. GHGs other than CO_2 are translated into CO_2e according to their 100-year global warming potential (GWP), as in IPCC AR4 (Solomon et al. 2007) where $CO_2 = 1$, $CH_4 = 25$, and $N_2O = 298$. Area units are in hectares (ha). The units are consistent with ISO 14064-2 standardization.

Over the last century human population growth and expansion have altered between a third and a half of the earth's land surface through crop production, grazing, and urbanization (Vitousek et al. 1997). Removal of native vegetation and tillage of the soil for agricultural production has released large quantities of carbon dioxide (CO_2) into the atmosphere, contributing GHG emissions on a scale similar to fossil fuel burning (Olofsson and Hickler 2008; Paustian et al. 1998). Altering current agricultural activities—changing tillage or cropping patterns—can restore some of this lost carbon and reduce emissions of nitrous oxide (N_2O) and methane (CH_4), two other significant greenhouse gases.

Agricultural lands (cropland, grazing land, agroforestry, and bioenergy crops) cover 37% of Earth's land surface (Smith et al. 2008) and account for 13.5% of GHG emissions contributed by human activity as of 2004 (IPCC 2007). This conservative estimate does not include fuel use, transportation, buildings, and deforestation associated with agriculture. As of 2000, agriculture accounted for 52% of human-contributed global methane emissions (3.1 Gt CO_2e/yr),[1] and 84% of human-contributed global nitrous oxide emissions (2.6 Gt CO_2e/yr) (Smith et al. 2008; U.S. EPA 2006). In the United States, the net GHG flux from agriculture in 2008 (462 Mt CO_2e) was approximately 6% of national GHG emissions (USDA 2011).[2]

1. The term *tonne* (abbreviated *t*) in this report refers to the metric ton. One tonne = 1 megagram (Mg) = 1,000 kg = 2,204.62 lbs. Hence, the abbreviations *Mt* and *Gt* denote the megatonne (1 million tonnes) and the gigatonne (1 billion tonnes), respectively. One tonne of CO_2 = 0.27 tonnes of carbon; 1 tonne of carbon = 3.66 tonnes of CO_2.

2. Emissions of nitrous oxide and methane from management on existing croplands and carbon dioxide from conversion of other lands to cropland totaled 502 Mt CO_2e. Sequestration of carbon on active croplands, on existing grasslands, and from converting other land to grasslands totaled 40 Mt CO_2e. These totals do not include conversion of croplands to forest, which contributes significant additional carbon sequestration.

Many options exist for GHG mitigation in agriculture, including improved crop and grazing land management (e.g., nutrient use; tillage, rotation, and residue use; water and drainage), land use changes (e.g., set-aside lands, forested buffers, agroforestry), and improved livestock management (e.g., alternative feeds, selection for feed efficiency, manure management). Shifts in land management can increase sequestration of carbon in soils and plants and can reduce emissions of nitrous oxide and methane. Sequestration of carbon in soils provides almost 90% of the global potential for agricultural mitigation (Smith et al. 2008).

Enhancing carbon sequestration represents by far the largest opportunity for increasing GHG mitigation on agricultural lands in the United States,[3] and potentially low-cost opportunities for improved nitrogen management (nitrous oxide emissions reduction) and methane emission reduction also exist. Numerous management practices can help mitigate agricultural greenhouse gases in the United States, but those that can be applied most broadly, to major cropping systems, have relatively low rates of carbon sequestration (many are within a range from 0.5 to 1.5 t CO_2e ha^{-1} yr^{-1}) and therefore must be implemented over large areas. Overall, agriculture can help mitigate 5%–14% of U.S. GHG emissions (Murray et al. 2005; Paustian et al. 2006), and many of the management activities that will mitigate emissions can create significant co-benefits by enhancing soil quality and thus farm and ranch productivity. Moreover, many have broader environmental benefits such as reduced nutrient runoff into waterways.

Agricultural innovation can also have GHG-mitigating effects. Increased production efficiency boosts total yield without increasing land used, and it can result in lower GHG emissions per unit of production. Burney et al. (2010) estimated that agricultural intensification and industrialization have avoided up to 590 Gt CO_2e of emissions since 1961, after accounting for the increase in greenhouse gases from the intensification of fertilizer use and production. Therefore, those practices that minimize environmental impacts, while maximizing yield per hectare, will be the most valuable in net terms.

Agricultural mitigation is cost competitive relative to other mitigation options in the energy and transportation sectors. Models suggest 70% of GHG mitigation potential from agriculture is in developing countries, 20% in developed countries, and the remaining 10% in emerging economies (IPCC 2007).[4] Given the cost competitiveness and potential co-benefits of agricultural GHG mitigation, numerous efforts are under way to create incentives for agriculture to engage in mitigation programs. These efforts include certification schemes,[5] government education and incentive programs, and voluntary carbon markets.[6] Agriculture might also be included in future mandatory markets as offsets in regional or national cap-and-trade climate policy.[7]

Policy and Market Drivers

Cap-and-trade policy for climate change in the United States, as a potential tool to reduce GHG emissions through a combination of regulatory mandates and market flexibility, was a topic of congressional debate a few years ago. All the main policy proposals included programs to increase GHG mitigation from agriculture and other "uncapped" sectors through some combination of new emissions trading markets (carbon offsets markets) and more traditional incentive programs (American Clean Energy and Security Act 2009). The offsets market programs were designed to play a critical role in containing costs (Murray and Jenkins 2009) and thus appear likely to reemerge if cap-and-trade policy is reinvigorated. When included in models of U.S. cap-and-trade programs, offsets for domestic agriculture appear to be economically beneficial for producers, outweighing potential cost increases stimulated by the cap (Baker et al. 2010). As of the publication of this report, action on a national cap-and-trade climate policy was stalled, but other programs to promote GHG mitigation continue to move forward and develop.

3. Agricultural land comprises 45% of the total land area of the contiguous 48 states (20% nongrazing cropland and 25% grazing land) (USDA 2007). As of 2007, total harvested cropland amounted to 124 million hectares (Mha).

4. Models suggest a global GHG mitigation potential of 1500–1600 Mt CO_2e/yr from agricultural activities at carbon prices around 20 $US/t CO_2e (IPCC 2007).

5. One is the Rainforest Alliance Certified Farms SAN standard (http://www.rainforest-alliance.org/agriculture/documents/sust_ag_standard_july2010.pdf).

6. Examples of these are Chicago Climate Exchange (http://www.chicagoclimatex.com/index.jsf), Verified Carbon Standard (http://www.v-c-s.org), American Carbon Registry (http://www.americancarbonregistry.org), and Climate Action Reserve (http://www.climateactionreserve.org).

7. See proposed federal policies such as the Waxman-Markey and Kerry-Lieberman bills of 2009–10; regional policies like Alberta's emissions trading system (http://environment.alberta.ca/0923.html), the Western Climate Initiative (http://www.westernclimateinitiative.org), and Regional Greenhouse Gas Initiative (http://www.rggi.org/market/offsets); and state-level cap-and-trade programs such as under California's Assembly Bill 32 (AB 32), the Global Warming Solutions Act of 2006 (http://www.arb.ca.gov/cc/cc.htm).

Many state and regional climate programs are under development. California, under AB 32, is developing regulations for a small offset program that may include agriculture (CA ARB 2009). The Western Climate Initiative continues to develop a regional offset program, but whether it will include agriculture other than manure management remains unclear (WCI 2010). Also uncertain is whether the Midwest Greenhouse Gas Reduction Accord will include development of a program for agricultural offsets (Midwestern GHG Reduction Accord 2010). The Regional Greenhouse Gas Initiative in the northeast currently allows manure management and afforestation within its cap-and-trade program, but it has insufficient demand to support offsets (RGGI 2010).

A number of non-compliance-based carbon offset markets and registries are tracking carbon projects and selling carbon credits in the United States.[8] Many of these are developing protocols for agricultural practices. Although California is the only compliance market in the United States that is likely to include agriculture, buyers continue to purchase credits for other reasons: to quantify performance of stewardship activities, to improve a corporate image, or to offset the environmental impacts of events or activities (e.g., flights).

Key congressional leaders are already looking toward the next farm bill, which could be debated soon. The farm bill provides substantial revenue for various farm programs, some of which already have GHG mitigation benefits. While new programs specifically targeting GHG mitigation have been of interest, a tight federal budget makes this seem unlikely.

Industry supply chain initiatives are another emerging driver of GHG mitigation in agriculture. Consortia of retail buyers are developing metrics to track sustainability, including the GHG impacts of the agricultural resources that go into products.[9] Moreover, the federal government has issued an executive order (Executive Order 13514 2009) requiring all government suppliers to report on their GHG emissions. The supply chains of these major retailers (e.g., Walmart, Tesco, and McDonalds) and federal procurement are global. Thus any requirements set by these buyers have substantial potential to influence agricultural producers, but their impact will depend on the objectives and approach selected by the buyers, which are still evolving.

Existing Protocols

Offset or carbon (or more broadly, GHG) credit activity has been occurring in North America in one form or another since the mid-1990s.[10] In the early, speculative days, when no standardized procedure for quantifying the offsets and little transparency existed, it was buyer *and* seller beware. With the advent of the Clean Development Mechanism (CDM), created under Article 12 of the 2007 Kyoto Protocol, a more systematic and rigorous accounting framework for carbon offsets was implemented, and standardized methodologies were developed literally from the bottom up by project developers investing in their own quantification methods. The CDM Executive Board began assessing and approving or rejecting these methodologies in a relatively transparent fashion. Due to limits on allowable activities in Article 3.4 of the Kyoto Protocol, the CDM has had little activity in agricultural GHG mitigation and no soil carbon sequestration projects.

Today, many programs—some regulatory-based and others for the voluntary market—are developing rigorous and robust protocols for GHG mitigation in agriculture.[11] The initiatives and protocols in Table 1 were in use or under development at the time of writing. Further details on each protocol and initiative can be found in reports recently compiled by the Market Mechanisms for Agricultural Greenhouse Gases (M-AGG) initiative (M-AGG 2010). For agriculture, much of the early quantification work focused on soil carbon sequestration. Now, protocol developers and programs are beginning to focus on N_2O and CH_4. They also are requiring assessment of all three gases (CO_2, N_2O, and CH_4) in their quantification approaches.

One of the earliest compliance-based GHG offset programs in North America was established by the province of Alberta in July 2007. Alberta allowed large regulated emitters to purchase offset credits as an option for compliance. The offsets are provided by voluntary, verified carbon sequestration or other GHG reduction projects within the province, including tillage reduction on agricultural land. In the United States, the Chicago Climate Exchange (CCX), which is no longer active, developed some of the earliest agricultural protocols. Many of the programs currently developing protocols in

8. See note 6.

9. One is the Sustainability Consortium (http://www.sustainabilityconsortium.org).

10. In 1995, electrical utility companies contracted Canadian farmers to plant shrubs in depressional areas on their fields. The Iowa Farmer-GEMCo farm deal occurred in 1997, and the Pacific Northwest Direct Seeder-Energy trade started in 2001.

11. Most programs/regulatory frameworks apply the ISO 14064-2 process standard as the framework for quantification/protocol development.

the United States are building on the lessons learned in the Alberta Offset System and CCX.

Table 1. Agricultural GHG mitigation protocols and methodologies by sponsoring organization

Protocol/Initiative	Emissions scope	Status
Alberta Offset System (Canada)[a]		
Tillage System Management	All GHGs[c]	Approved, 2009
Continuous Cropping (Reduced Fallow)[b]	All GHGs	In development
Nitrous Oxide Emission Reduction	N_2O, Fuel	Approved, 2010
Livestock Feeding (1 Pork, 3 Beef, 1 Dairy)	All GHGs	Approved, 2008–10
American Carbon Registry (U.S. and International)[d]		
Emissions Reductions through Changes in Fertilizer Management[e]	N_2O, Fuel	Approved, 2010
N_2O Emissions Reductions through Fertilizer Rate Reduction[f]	N_2O	In peer review
GHG Emission Reductions through Rice Management[g]	CO_2, CH_4, N_2O	In peer review
Improved Grazing Land Management	CO_2, CH_4, N_2O	In development
Livestock Feeding (Beef and Dairy)	CO_2, CH_4, N_2O	In development
Canadian Offset System (Canada)[h]	Same as Alberta	Described above
Chicago Climate Exchange (United States)[i]		
Continuous Conservation Tillage and Conversion to Grassland	CO_2	Approved (not active)
Sustainably Managed Rangeland Soil Carbon Sequestration	CO_2	Approved (not active)
Climate Action Reserve (United States)[j]		
Cropland Management	Soil C	In development
Nutrient Management	All GHGs; N_2O primary	In development
Rice Cultivation	All GHGs; CH_4 primary	In development
U.S. DOE 1605 (b)[k]		
COMET-VR Online Reporting Tool	CO_2	Approved
Novecta Standard[l]	N/A	Pending
Verified Carbon Standard (United States and International)[m]		
BioCarbon Fund SALM[n]	All GHGs	Pending
Grassland Management[o]	All GHGs	Pending
N_2O Emission Reduction through N Fertilizer Rate Reduction[p]	N_2O	Pending
Emissions Reductions in Rice Systems[q]	All GHGs; CH_4 primary	Pending
CDM Small-Scale Biological Fixation (International)[r]	All GHGs	Approved

a. See http://www.carbonoffsetsolutions.ca.
b. This reduced-fallow protocol is intended to be implemented with the tillage system management protocol.
c. All GHGs have been considered, but only those relevant have been selected for individual assessment, in accordance with the ISO framework. Therefore, in most cases only CO_2, N_2O, and CH_4—the three main GHGs in agriculture—are specifically quantified.
d. See http://www.americancarbonregistry.org/carbon-accounting.
e. http://www.americancarbonregistry.org/carbon-accounting/emissions-reductions-through-changes-in-fertilizer-management.
f. http://www.americancarbonregistry.org/carbon-accounting/methodology-for-n2o-emission-reductions-through-fertilizer-rate-reduction.
g. http://www.americancarbonregistry.org/carbon-accounting/emission-reductions-in-rice-management-systems.
h. See Environment Canada draft rules; http://www.ec.gc.ca/creditscompensatoires-offsets.
i. CCX soil carbon standards include conservation tillage, grassland planting, and rangeland management; http://www.chicagoclimateexchange.com/docs/offsets/CCX_Rulebook_Chapter09_OffsetsAndEarlyActionCredits.
j. See http://www.climateactionreserve.org/how/protocols.
k. See U.S. DOE Technical Guidelines: http://www.eia.doe.gov/oiaf/1605/PartHAgriculturalAppendix[1].pdf.
l. See Novecta's draft Agricultural Soil Credit Standard (soil C sequestration and GHG emissions reductions). Quantification methods in Part C are currently left "undefined"; http://www.novecta.com/documents/Carbon-Standard.pdf.
m. On March 1, 2011, the Voluntary Carbon Standard became the Verified Carbon Standard.
n. See the Biocarbon Fund Sustainable Agricultural Land Management (SALM) at http://www.v-c-s.org/methodology_salm.html.
o. See Adoption of Sustainable Grassland Management through Adjustment of Fire and Grazing at http://v-c-s.org/methodology_alma.html.
p. See Quantifying N_2O Emissions Reductions in U.S. Agricultural Crops through N Fertilizer Rate Reduction at http://www.v-c-s.org/methodology_qn2o.html.
q. See Calculating Emissions reductions in Rice Management Systems at http://www.v-c-s.org/methodology_cerrms.html.
r. http://cdm.unfccc.int/UserManagement/FileStorage/BZG8LM2WO95IDQJCF634VUYTPNEKRX.

The U.S. government has gained experience with GHG reporting through the voluntary greenhouse gas reporting program initiated by the Energy Policy Act of 1992, section 1605(b), under the direction of the Department of Energy. The program, which is administered through the Energy Information Administration, tracks GHG emissions and reductions as well as carbon sequestration activities on an annual basis relative to a 1987–1990 baseline emissions period. The program guidelines include methodologies from the U.S. Department of Agriculture for estimating GHG emissions and carbon sequestration from the forest and agriculture sectors. The final guidelines (EIA 2006) give many of the methodologies for agricultural sources a "B" rating due to simplified default methodologies. These methodologies relate to livestock (enteric fermentation and livestock waste) or crop production (rice cultivation, N_2O emissions from agricultural soils, lime application, CO_2 emissions from and sequestration in mineral soils, and CO_2 emissions from cultivation of organic soils). COMET-VR, a calculator and self-reporting tool for agricultural land use, was developed for the 1605(b) program. Although useful for information gathering, the program incorporates no incentives for reducing net GHG emissions.

Our Approach

This report provides a side-by-side assessment of the biophysical GHG mitigation potential—sequestration and avoided emissions—of more than 40 agricultural practices and combinations of practices (see Table 2). It accounts for the three primary GHGs affected by agricultural activities (carbon dioxide, nitrous oxide, and methane) by synthesizing empirical measurements and estimates from the scientific literature. In addition to land-based emissions, it includes upstream as well as process fuel and energy emissions, in keeping with the streamlined life-cycle analysis outlined by ISO 14064-2 (see "ISO principles" on page 13 for more detail). By assessing mitigation potential on an area basis and estimating total applicable area, the report identifies practices that could be nationally important (when applied to large land areas) as well as those that may have significant potential in particular regions or cropping systems over smaller areas (high-per-hectare mitigation potential). It then assesses the scientific foundations for the estimates of mitigation potential, quantifying the number of comparisons for various practices in the scientific literature. A survey of experts provides context for the synthesis of the scientific data, highlighting mechanisms of GHG mitigation that are well understood even if data are lacking and concerns over offsite or upstream impacts that outweigh confidence in site-level data. These assessments pinpoint critical data gaps and research needs. In addition, the report assesses potential environmental and soil quality co-benefits, which tend to make some mitigation activities more attractive, both to society and to individual landowners. Figure 1 presents an overview of this report.

Economic, market design, technical, and farm-level factors will affect the costs and other barriers to the implementation of these management activities. With a limited land base, producers need to choose between different crops or livestock for market and between different management activities. This report reviews the literature on economic models for GHG mitigation in agriculture, some that are activity-specific and limited to specific regions and others that consider multiple activities and competition among these activities. Transaction costs, farm-level barriers, and other adoption issues can be difficult to predict and are not always examined in the economic models, but are discussed in this report.

The final section of this report reviews the tools and data needed for implementation of projects and programs to assess possible barriers or limitations to implementation. It assesses a number of approaches for quantifying changes in greenhouse gases, which range from the standard methods put forth by the Intergovernmental Panel on Climate Change (IPCC), which were designed for country-level inventories, to field measurement and complex biogeochemical process models. The capacity of these various quantification tools varies by type of agricultural practice. The report also explores the data and tools available to develop accounting procedures for establishing baselines, assessing leakage, and estimating risks of carbon sequestration reversals.

Agricultural practices included in the assessment

The agricultural management practices covered here (see Table 2) were selected because they are potentially applicable at large scale for agriculture and grazing lands in the United States. They were also selected because for several reasons they were missing from early efforts to develop GHG protocols. First, they require large land areas and thus may require engagement of numerous landowners in order to achieve appreciable impacts. Second, their impact on greenhouse gases was considered uncertain. Third, regional differences in this impact can be significant and may not be well understood. Although the practices are discussed as separate activities in the discussion below, many of them can be implemented in combination.

Manure storage or treatment and forest management—including afforestation of agricultural land—are not covered in this report. These activities have protocols in place and projects under development. Efforts to reduce CH_4 emissions from enteric fermentation (mainly in cattle) are also not addressed here. The net GHG effects of the switch from grain-finished cattle to grass-fed cattle and of reduction of total meat production may be worth exploring but were beyond the extent of this assessment. A subsequent T-AGG report will explore mitigation opportunities for livestock management in the United States.

Figure 1. Overview of this report

Physical Potential

- Net GHG/ha, total ha available, and over what time frame
- Significant upstream or downstream GHG impacts (life-cycle analysis)

Scientific Certainty

- Is information sufficient by practice and geography?
- Does directional certainty exist for net GHGs?

Economic Potential

- Costs for management shifts (e.g., opportunity costs, break-even price, yield impacts)

Possible Barriers

- Economic – Capital costs
- Technical – Monitoring, adoption, or production barriers
- Social – Negative community or farmer impacts, resistance to change
- Negative ecological impact

Implementation and Accounting

- Quantification – Are there good methods for measuring or modeling GHG outcomes on a project scale?
- Monitoring and verification
- Additionality
- Baseline – Are there viable approaches for setting baseline? Sufficient data?
- Leakage risk – Is there leakage risk (life-cycle analysis)? Can it be accounted for? Output-based approaches?
- Reversal risk – Is there risk? Can it be estimated? Is it too high?

Significant Co-Benefits?

Consider other social, economic or environmental co-benefits

This assessment of GHG mitigation potential for agricultural land management activities focuses on research and data for the conterminous 48 U.S. states, supplemented by data from other parts of the world where U.S. data are limited. Hawaii and Alaska are generally excluded as they comprise a very small share of total U.S. agriculture (less than 0.5% of total farms and 0.05% of total agricultural land). The covered agricultural activities can be divided into three main categories: (1) management practices on active cropland, (2) management practices on active grazing land, and (3) activities that relate to land use change, such as conversion of cropland to grazing land or set-aside. Within these broad categories, most activities can be assigned to one of the three major GHGs, targeting CO_2 by sequestering carbon in the soil, N_2O by reducing emissions, and CH_4 by reducing emissions or increasing uptake in the system.

Table 2. Agricultural land management activities assessed for GHG mitigation potential

	GHG	Management activity	Activity subset
Cropland management	CO_2	Implement conservation tillage	Switch to no-till
			Switch to strip tillage, ridge tillage, or other conservation tillage
		Reduce fallow	Reduce or eliminate summer fallow
			Use winter cover crops
		Change crop rotations	Diversify annual crop rotations
			Double-crop or otherwise increase cropping intensity
			Incorporate perennials into crop rotations
			Replace annuals with herbaceous perennial crops (not grazing)
		Plant trees	Switch to short-rotation woody crops
			Establish agroforestry (windbreaks, alley cropping, etc.)
		Apply organic material	Apply manure, compost, etc.
			Apply biochar
		Convert dry land to irrigated land	
		Reduce chemical use (other than N, which is covered below)	
		Manage farmed histosols (organic soils)	
	N_2O	Improve N use efficiency and reduce losses	Reduce fertilizer N application rates by 15%
			Switch fertilizer N source from ammonium-based to urea
			Switch to slow-release fertilizer N source
			Change fertilizer N placement
			Change fertilizer N application timing (e.g., fall vs. spring, split application)
			Use nitrification inhibitors
		Improve manure application management	
		Improve irrigation management	Introduce drip irrigation, reduce irrigation intensity
		Drain agricultural land in humid areas	
	CH_4	Manage rice production	Adjust rice water management (e.g., midseason drainage)
			Plant rice cultivars that produce less CH_4
			Reduce rice area by switching to other crops
Grazing land management	CO_2	Improve grazing management on rangeland	Adjust stocking intensity, prevent overgrazing, and reduce degradation
		Improve grazing management on pasture	Adjust stocking intensity
		Introduce rotational grazing	Introduce rotational grazing on rangeland
			Introduce rotational grazing on pasture
		Establish agroforestry	Establish silvopasture systems
		Manage species composition	Plant grass or legume species with higher productivity or deeper roots
		Fertilize grazing land	
		Irrigate grazing land	
		Introduce fire management	
	N_2O	Improve N use efficiency of fertilizer and manure	
Land use change	CO_2 N_2O CH_4	Convert cropland to pasture	
		Set aside cropland	Set aside histosol cropland
			Set aside sensitive croplands or plant herbaceous buffers
		Set aside grazing land	
		Restore wetlands	

Agricultural land management activities other than those in Table 2 have been suggested in different contexts. For example, a 2009 list compiled by the U.S. Geological Survey (USGS) includes organic farming, promotion of urban agriculture, biotechnology applications, and programs to support local farming and purchasing (USGS 2009). Some of these activities are incorporated as components of one or more of the activities covered in this report. Others are not included for specific reasons. Urban and locally oriented agriculture may contribute to GHG mitigation by reducing transportation costs, but most benefits would likely be difficult to quantify (small areas, highly variable production). Biotechnology has implications for many the activities examined, for example, weed control in no-till areas and nitrogen use efficiency. Many biotechnology impacts will be realized through crop-breeding advances whereby crop yield increases and intensification lead to fewer GHG emissions per unit of production. Technological advancement will likely be important in mitigation, but its impact is realized in many different ways and thus difficult to isolate and quantify. Organic agriculture is not addressed as a specific activity in this overview because it incorporates many practices that vary significantly across farms. Research comparing organic and conventional systems has found significantly greater soil organic carbon (SOC) accumulation in the organic systems, both in the United States (Clark et al. 1998; Lockeretz et al. 1981; Pimentel et al. 2005) and abroad (Freibauer et al. 2004). In these systems, C sequestration is enhanced through field application of manure and compost, planting of winter cover crops, high crop-rotation intensity, and maintenance of forested areas for ecological diversity. Many of these practices typical of organic production are assessed individually in this report.

In major cropping systems, fertilizer N application increases yield and soil organic carbon (Varvel 2006). As a result, that application has been proposed as a potential GHG mitigation technique (Snyder et al. 2009). But given that the majority of field crops in the United States already receive fertilizer N (or organic N)—at rates that may exceed crop demand—increasing application above the baseline rate is unlikely to have any major C sequestration impact. Recent studies have found that additional fertilizer N application has little to no impact on SOC or CO_2 fluxes (Alluvione et al. 2009; Mosier et al. 2006) and even a negative impact when fertilizer N promotes organic matter decomposition (Khan et al. 2007). Additionally, the corresponding risk of increasing N_2O emissions with greater applications of fertilizer N generally outweighs any potential GHG mitigation benefit. Therefore, this activity is not explored for its GHG mitigation potential.

To account for the regulatory uncertainty of GHG mitigation programs and policies for the United States, this report considers energy and fuel emissions as well as land-based emissions and sequestration. Under the economy-wide coverage proposed in many of the cap-and-trade policies debated in the U.S. Congress, mitigation used as carbon offsets would not be credited or debited for any changes in GHG fluxes associated with changes in energy and transportation because those sectors' emissions were covered under the cap. In contrast, in a farm bill program, in supply chain reporting requirements, or in voluntary markets, agriculturally related changes in energy and transportation could and probably should count.[12] For example, an offset project that involved elimination of tillage to increase carbon sequestered in soils would also reduce fuel use by decreasing tractor use. In a compliance-based offsets program, only the land-based mitigation would count;[13] in a farm bill program, voluntary market, or certification program, the reduced emissions from lower fuel use might also receive credit. Given uncertainty in future policies and programs, this report assesses outcomes with and without energy and transportation emissions when considering the relative GHG mitigation potential of various agricultural practices.

ISO principles

This assessment of agricultural management activities for GHG mitigation will follow guidance from ISO 14064-2 (2006), which is titled *Specification with guidance at the project level for quantification, monitoring and reporting of greenhouse gas emission reductions or removal enhancements*. This international standard was developed to enhance credibility, consistency, and transparency and is used by GHG mitigation projects, programs, and protocols around the world. The quantification framework is policy-neutral (i.e., can be used in a variety of policy situations), has clear and verifiable principles and requirements, and provides international consistency (Government of Canada 2005). Other related standards provide guidance at the organizational level for quantifying and reporting GHG emissions reductions and removals (ISO 14064-1 2006), guidance for validation and verification of GHG assertions (ISO 14064-3 2006), and requirements for the accreditation of GHG validation and verification bodies (ISO 14065 2007).

12. Though in practice they are sometimes excluded if it can be demonstrated that particular GHG emission sources are either de minimis or conservative to exclude (i.e., excluding tends to underestimate net GHG reductions).

13. Although the change in fuel use would not count toward an offset credit in the proposed cap-and-trade systems, farmers should benefit from cost savings on fuel.

With clear guiding principles, ISO 14064-2 (2006) is expected to improve environmental integrity, speed implementation of GHG projects, and facilitate the crediting and trade of GHG emissions reductions or sequestration (credits or offsets). To promote shared wording and connectivity with other related projects, the T-AGG assessment process incorporates these six guiding principles:

- ***Relevance***—selecting the GHG sources, sinks, and reservoirs (SSRs) appropriate in each situation
- ***Completeness***—identifying all GHG sources and sinks controlled by, related to, or affected by the specified activity in order to identify and categorize sources and sinks
- ***Consistency***—ensuring meaningful comparisons of GHG-related information
- ***Accuracy***—reducing bias and uncertainties as much as possible (and practical)
- ***Transparency***—disclosing sufficient and appropriate information to decision makers and others
- ***Conservativeness***—using conservative assumptions to ensure that claims are not overestimated

The ISO 14064-2 series of procedures begin with a full description of the project that includes identification of relevant GHG sources, sinks, and reservoirs (SSRs), according to materials and energy flows on site (controlled) and upstream and downstream of the project (related) and any impacts the project may have on off-site GHG SSRs due to market changes, activity shifts, or other leakage-related activities (affected). The standard then establishes procedures for determining the baseline scenario(s) and calls for identification of GHG SSRs for the baseline activities. Through this life-cycle assessment (LCA, see Box 2), baseline and project GHG SSRs are compared, allowing for inclusion of only the SSRs that will change as a result of the project. If an SSR in the baseline scenario remains unchanged in the project scenario, it can be excluded from the assessment, given adequate justification. This comparison can greatly simplify the quantification exercise. Other LCAs, which determine an environmental footprint of a product or process, allow comparisons among options rather than between the baseline and the project scenarios.

Box 2. Life-cycle analysis

Life-cycle analysis (LCA) of the environmental impacts of a product or a practice evaluates resource extraction, production, use, and waste disposal. In essence, LCA tracks the effects of a product or practice from "cradle to grave." Increasingly, LCA is critical in comparing the GHG and other environmental implications of different consumer products (World Resources Institute 2009), biofuel options (Groom et al. 2008), and livestock products (de Vries and de Boer 2010).

Maintaining a focus on GHG impacts, the ISO 14064-2 (2006) process of streamlined LCA ensures complete consideration of all sources, sinks, and reservoirs of GHGs affected by an activity as well as upstream and offsite impacts. For example, an activity that sequesters soil C may also affect N_2O emissions, so a net GHG impact is used to account for all emission impacts. Life-cycle quantification of GHG impacts can be considered, along with other environmental impacts such as water quality or other pollutants. A commonly used framework, such as one expressed in economic terms, facilitates comparison of all decision-informing factors (Pearce and Atkinson 1995).

The streamlined LCA also determines which SSRs are included in the assessment. Generally, all "controlled" and most "related" SSRs are included, with clear documentation, whereas "affected" impacts are outside the project boundary and thus considered leakage. Once leakage impacts are identified, risk management strategies can be devised for their control or discounting strategies can be employed.

Tonnes are the standard unit of measure for GHG emissions or reductions, and all GHGs are converted into equivalent CO_2 global warming potential (CO_2e).[14] ISO 14064-2 requires *functional equivalence* or common metrics of comparison of baseline and project calculations to ensure valid comparisons of net GHG reductions and removals. Functional equivalence ensures that a project is not rewarded offsets simply for reducing the level of an activity or production of a good (decommissioning a feedlot, for example, would reduce beef production, resulting in "artificial" emissions savings). The goal is to provide the same level of production with fewer total GHG emissions. Essentially, the project must be able to deliver the same types and levels of products or services as the baseline level of activity or address leakage in other ways. This goal necessitates output-based metrics for meaningful comparisons (see Murray and Baker 2011). Data quality should be maintained and monitoring criteria and procedures followed, with appropriate documentation, verification, and reporting.

14. In this report, one divergence from ISO 14064-2 is use of the IPCC 2007 Assessment Report Four (AR4) Global Warming Potentials for CH_4 and N_2O (25 and 298, respectively). ISO 14064-2 (2006) uses the 1996 values from the Second Assessment Report (SAR) (21 and 310 for CH_4 and N_2O, respectively).

ASSESSING MITIGATION POTENTIAL

This section reviews the scientific literature to provide a side-by-side comparison of the biophysical mitigation potential of a wide range of agricultural practices, explores scientific confidence in this research, highlights critical research gaps, and tracks important environmental co-effects. The section also reviews literature on the economic potential of greenhouse gas mitigation through changes in agricultural management.

Biophysical Greenhouse Gas Mitigation Potential

Methods

Measurements and estimates of the biophysical GHG effects of agricultural land management activities from the scientific literature were reviewed and collated for this report. Any study that included an activity listed in Table 2 that recorded changes in soil C or in GHG (CO_2, CH_4, and N_2O) fluxes was included. For many of these activities, data are too scattered and incomplete for formal meta-analysis, which would provide a robust assessment of mitigation potential and the factors that affect variability. Despite these limitations, programs and protocols will move forward, and a compilation of the best-available information will help inform these early efforts. This review and collection of the available data and expert estimates offers an indication of average mitigation potential by activity, provides a literature review resource, and presents a thorough assessment of research gaps. Caution should be used in interpreting the mitigation potentials, particularly those with few research comparisons. Researchers in the USDA-ARS and universities are currently conducting meta-analyses to assess key issues such as the soil C response to tillage changes as affected by sampling depth, region, soil type, and other factors.[1] An additional meta-analysis by the T-AGG team is examining tillage intensity and soil C saturation.

For practices that have been widely adopted for other ecological or production efficiency reasons—such as conservation tillage, cover crops, and fallow management—long-term field studies provide significant data. This assessment used existing syntheses to identify original field comparison studies and supplemented them with newer research. When U.S. field studies provided fewer than 30 observations, studies from Canada (many within 200 kilometers of the U.S. border), and, if needed, from other international research were used. Modeled values or estimates based on expert opinion were used when fewer than nine field studies were available. A companion report, *Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States: A Synthesis of the Literature* (hereafter the *Synthesis Report*), lists all the references used to calculate the national GHG mitigation potential of each examined activity (Eagle et al. 2011). A library of all of the references is available online.[2]

Most activities *target* a single greenhouse gas: carbon dioxide for C sequestration or nitrous oxide or methane for emission reduction. However, impacts on all *controlled* (changes in soil C and land emissions of N_2O and CH_4) and *related* (upstream and process) GHG fluxes are also quantified, where data are available. These data are combined and presented as a national average net GHG change per activity in tonnes of CO_2e (per hectare, per year). The majority of these land management activities target CO_2, or soil C, where implementation would be expected to increase soil C storage or reduce losses. In the average estimates calculated below, we assume that soil carbon is in the nonsaturated state and one can assume this rate for 30 to 50 years. Data on soil C changes span multiple decades, but N_2O and CH_4 flux impacts of soil C-targeted activities have been monitored only in the last 5–10 years).

Three activities—conversion to no-till, summer fallow elimination, and diversification of annual crop rotations—have at least five field observations in all applicable regions,[3] allowing calculation of regionally weighted averages. For this assessment, regional GHG impact (per hectare) was estimated and then scaled up to a national average based on the cropland area in each region. In all but these three cases, experimental data were too sparse to calculate regionally specific estimates of mitigation potential. Thus, where nine or more field comparisons were available,[4] the average was calculated as the mean of these comparisons. Significant outliers were removed from these analyses to avoid skewing of the results. For each activity, experimental data points that had a modified z-score of more than 3.5 were eliminated before calculation of the mean and range (Peat and Barton 2005). A visual check of the eliminated outliers suggested that they were likely erroneous and not simply indicative of a large range. For each of these activities, the reported range includes 80% of observed experimental results, and thus provides the best possible picture of anticipated GHG effects across the nine U.S. agricultural regions.

1. C. Rice, personal communication, January 2011; S. Ogle, personal communication, March 2011.

2. The reference library from the Synthesis Report is available in endnote format at http://nicholasinstitute.duke.edu/ecosystem/t-agg.

3. The 48 coterminous states are divided into 9 generalized agricultural regions as described in the section on scientific certainty in this report.

4. When 10 or more observations were available for an activity, they tended to be consistent with one another—hence, the choice of nine observations as the dividing line. There are two exceptions to this rule for which only a range of observed values is reported, with no mean. "Improve grazing management on rangeland" is retained in the "Uncertainty due to lack of data or high variability" category because the available observations seem to be inconsistent; "Apply organic material" has life-cycle concerns and is so in that category; and "Set aside grazing land" has low or negative GHG mitigation potential.

Where field comparisons were fewer than nine, expert estimates and model results from the peer-reviewed literature were used as a proxy for additional observations. The resulting national estimates of GHG flux effect tend to contain more inherent uncertainty and so are expressed only as a range from the minimum to the maximum observed or estimated values. No national average was calculated.

For reductions in fertilizer N application rates, many estimates of N_2O emissions reductions in the literature sources are expressed per unit of fertilizer N rate change or as a proportion of the baseline emissions. GHG implications of other activities that target N_2O emissions are also often reported as a proportion of the baseline. To facilitate side-by-side comparisons of activities, we calculate GHG flux effect here per unit area. For fertilizer N rate change, the calculations assume a 15% reduction from the national average application rate of 103 kg N ha^{-1} yr^{-1}. When absolute values are not reported for other N_2O-emission-reducing activities, the proportional reduction is multiplied by a baseline national average 2008 emission rate from U.S. fields of 215.9 Mt CO_2e (U.S. EPA, 2010b) divided by total cropland area of 124 Mha (USDA NASS 2007). For implementation, fertilizer N rate reduction and other N_2O emission reduction expectations would need to be tailored to cropping systems and region.

Upstream and process emission changes are a result of changes in fertilizer N rates or adjustments in fuel use (i.e., field operations and irrigation). These emissions have been directly estimated in the scientific literature for only four of the activities assessed here: conversion to no-till, introduction of conservation tillage, irrigation of dry land, and reduction of nonfertilizer chemical application. For other activities, this assessment estimates the GHG flux effect of fertilizer N rate changes (e.g., reduced fertilizer N needed for winter cover crops and perennial legume crops) and shifts in fuel use (e.g., fewer tillage operations for perennial crops) from national averages found in two sources: the scientific literature and cost-and-return reports published by cooperative extensions in different states.

The national averages were calculated as follows: Assuming that U.S. agricultural fuel use (total amount from Schnepf 2004) is equally allocated to all 124 Mha of U.S. cropland (USDA NASS 2007), the average fuel use for agricultural field operations emits an estimated 0.36 t CO_2e ha^{-1} yr^{-1}.[5] The carbon cost of fertilizer N (for manufacture, distribution, and transportation) is approximately 3.2–4.5 t CO_2 per tonne of fertilizer N manufactured (Izaurralde et al. 1998; West and Marland 2002). Therefore, if the total fertilizer N consumption of 13.6 Mt N yr^{-1} (Millar et al. 2010; USDA ERS 2010b) is equally allocated to all U.S. cropland, the average fertilizer N application is 103 kg N ha^{-1}, and estimated process emissions equal 0.39 t CO_2e ha^{-1} yr^{-1}.

These national averages were used to calculated upstream and process emissions effects. As an example, for activities that involve setting aside cropland, these averages were used to calculate reduced emissions due to elimination of field operations and fertilizer N application. For other management changes that reduce fertilizer use or require adjustments in field operations, the averages were multiplied by the estimated proportion of change. Emissions reductions for an individual project will depend on the baseline cropping system.[6] Throughout this process, all attempts were made to maintain conservative assumptions, as per ISO 14064-2 (2006) standards.

The maximum applicable land area (over and above current adoption rates, i.e., the baseline area) was also calculated from the literature and survey data.[7] This potential applicable land area is affected by crop type, current management practice, and regional and climate variation as well as by implementation and opportunity costs (economic potential is less than biophysical potential), which are not taken into account in these estimates. In addition, multiple activities may compete for the same land area. Therefore, the realizable application area—competitive potential—will be less than the sum of its parts, which is also not addressed here. Relationships among biophysical, economic, and competitive potential are discussed below (see Figure 6).

5. Conversions of gasoline and diesel to CO_2 equivalents are drawn from the U.S. Energy Information Administration: http://www.eia.doe.gov/oiaf/1605/coefficients.html (accessed September 23, 2010).

6. Fuel-related emissions during field operations vary significantly from crop to crop. California crop production data indicate that these emissions range from 0.13 t CO_2e ha^{-1} yr^{-1} for corn to 0.71 t CO_2e ha^{-1} yr^{-1} for wheat. These emissions are calculated from the carbon content of fuel and from crop production cost reports published by the University of California Cooperative Extension (http://coststudies.ucdavis.edu).

7. Total crop areas and relevant survey data were taken from the U.S. Agricultural Census. Current implementation rates from various sources were used to determine the applicable crop area for each activity.

Mitigation potential by activity

The following summaries of 42 management practices are presented in three groups: those affecting (1) cropland soil C, (2) N_2O and CH_4 emissions, and (3) grazing land soil C. Land use changes were incorporated in each of these groups, depending on the target greenhouse gas and the starting land use (cropland or grazing land). Where first introduced in this summary, each individual activity is noted in ***bold italics***. For a more detailed review of the original research underlying the calculated estimates of mitigation potential, including a list of all references, see the T-AGG report *Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States: A Synthesis of the Literature*.[8]

Each summary notes when management activities can significantly affect yield or production (e.g., crop mix change, fertilizer rate reduction, or change in animal numbers), because the GHG impacts beyond the field or plot should be considered. These impacts, which were not incorporated in estimates of GHG mitigation potential, are known as *leakage*. Leakage is positive (or "good") when activities increase productivity or otherwise indirectly reduce GHG emissions in other locations. Negative (or "bad") leakage occurs when activities cause shifts in production that result in higher emissions elsewhere. Based on the ISO principle of conservativeness, GHG programs and protocols have generally not assigned credit for positive leakage, but have required quantification and deduction of negative leakage where it is estimated to exceed specified *de minimis* thresholds.

The studies reviewed in this assessment and the resulting synthesis of biophysical mitigation potential assume the status quo for crop areas and environmental factors, both of which may shift with anticipated climate change (i.e., CO_2 concentration increases, changes in temperature and rainfall, storm severity, and so on). Rising CO_2 concentrations can lead to soil moisture conservation (Prior et al. 2010) and increased plant growth and C input (Gill et al. 2002; Lichter et al. 2008), although the latter may be limited by nitrogen and other nutrient availability (van Groenigen et al. 2006). The impact on soil C decomposition rates is less certain. Soil respiration rates increased in a Colorado shortgrass steppe (Mosier et al. 2003), but Lichter et al. (2008) found no impact of elevated CO_2 on decomposition rates or chemical composition in a forest system. Hungate et al. (2009) used meta-analysis to conclude that net soil C accumulation was likely with elevated CO_2, at least when N availability is not limiting. Elevated CO_2 has also been associated with increased N_2O emissions from well-fertilized grass systems (Baggs et al. 2003) and decreased N_2O emissions under less fertilized conditions (Mosier et al. 2003). Many researchers have observed soil C decline in grassland and cropping systems related to average temperature increases over the last two to three decades (Karhu et al. 2010; Senthilkumar et al. 2009; Yang et al. 2009). Therefore, the interacting factors of moisture, N availability, and air temperature will affect the net GHG flux response to climate change over longer time horizons. Uncertainties about these effects, when potentially significant, should be considered in program design.

Cropland

Long-term studies of management activities that increase soil C generally indicate declining sequestration rates over time (Varvel 2006), so that a new equilibrium is expected after 30–50 years (Sauerbeck 2001; West and Post 2002). Timing (i.e., when a saturation point may be reached), may differ by activity (West and Post 2002),[9] but is most likely affected by climate and soil type. In contrast, some reports suggest that, under certain conditions, soil C storage rates may continue in a linear fashion for up to 60 years (Novak et al. 2007; Potter et al. 1999). This presents significant uncertainty in the scientific literature, and further research could be directed to determine regional or other factors that cause the discrepancies. However, commonly used models of agricultural GHGs assume that soil C will only accumulate to a significant degree for 30–50 years after a management change, and most research covers up to 30 years at most. Emission reductions of N_2O and CH_4 would not be subject to any saturation, and thus have ongoing benefits.

Implement conservation tillage

By reducing soil disturbance, ***implementing conservation tillage*** slows decomposition of organic matter. Conservation tillage, according to USDA-NRCS, is any tillage reduction practice that leaves at least 30% of the soil surface covered by residue (i.e., mulch tillage, ridge tillage, strip tillage, and no tillage). Since ***no-till*** is the most commonly studied and implemented GHG-mitigating agricultural land management practice, this assessment treats it separately from other conservation tillage practices. Many experiments have demonstrated significant increases in total soil C from both

8. Available at http://nicholasinstitute.duke.edu/ecosystem/t-agg.

9. West and Post (2002) suggest that a new equilibrium for NT would occur after 15 to 20 years, and the time frame would extend to 40 to 60 years for a rotation complexity change.

conservation tillage and no-till practices (Franzluebbers 2010; Six et al. 2004; West and Post 2002). However, no-till management in cool, wet climates with poorly aerated soils can reduce crop yields, resulting in lower organic matter inputs and related decreases in soil C (Gregorich et al. 2005; Rochette 2008). Nitrous oxide emissions tend to be little affected in conservation tillage (Johnson et al. 2010; Lee et al. 2009b; Venterea et al. 2005) except for when tillage is eliminated in areas with poorly aerated soils (MacKenzie et al. 1998; Rochette 2008; Six et al. 2002). Decreases in nitrous oxide fluxes have been documented with no-till in drier and warmer regions (Halvorson et al. 2010; Omonode et al. 2011).

This assessment compiled nearly 250 field comparisons of soil C response to use of no-till to calculate a mean increase (regionally weighted) of 1.3 t CO_2e ha^{-1} yr^{-1} (range of -0.4 to 3.6 t CO_2e ha^{-1} yr^{-1}). With slight decreases in N_2O and process emissions and no effect on CH_4, the net GHG mitigation potential attributable to use of no-till is 1.5 t CO_2e ha^{-1} yr^{-1}. Using data from 65 field comparisons, the soil C sequestration potential of other conservation tillage practices averages 0.4 t CO_2e ha^{-1} yr^{-1} (a range of -0.5 to 1.4 t CO_2e ha^{-1} yr^{-1}). Slight decreases in N_2O and process emissions yield a net GHG mitigation potential of 0.6 t CO_2e ha^{-1} yr^{-1}.

Both no-till and other conservation tillage practices could be applied throughout the United States. The primary exception is use of no-till in the cool, wet soils of the Northeast, where crop yields and soil C tend to decrease, while N_2O emissions tend to increase. However, these soils comprise less than 5% of total U.S. cropland area. No-till data are available from all U.S. regions except the Pacific Northwest and Southwest; data for conservation tillage are available from all regions except the Lake States and South Central region.

Conservation tillage and no-till were initially implemented to reduce soil erosion and fuel costs. As a result, some form of reduced tillage is now used on more than 40% of U.S. cropland, with 24%–35% of U.S. cropland under no-till management (CTIC 2008; Horowitz et al. 2010). Therefore, conservatively estimated, the maximum area to which conservation tillage and no-till are applicable is 72 Mha[10] and 94 Mha, respectively. These figures are not additive, since land intended for conservation tillage would no longer be available for no-till. While the scientific research tends to focus exclusively on continuous no-till, some of the area counted as no-till in surveys of farm practice may be tilled at some point in a generally no-till crop rotation.[11] Therefore, many existing no-till fields may not be achieving the full GHG potential estimated from the research, and shifting from intermittent no-till to permanent or semi-permanent no-till may open up more opportunities for mitigation.

Reduce fallow

Fallow periods can be reduced or managed to increase soil C stocks. Summer fallow is used to conserve water on 20 Mha of U.S. cropland in rain-fed wheat production systems (Sperow et al. 2003), primarily in the Great Plains, the Pacific Northwest, and the Rocky Mountain regions. However, given sufficient moisture, ***eliminating summer fallow*** can increase total plant productivity and storage of soil C. In some cases, reductions in fallow are preferable to complete fallow elimination, both for yield and soil C storage reasons, but soil C data are limited. Soil C response from summer fallow elimination is greater in no-till systems than in other systems, likely as a result of better water conservation (Potter et al. 1997; Sainju et al. 2006). With 33 data points the regionally weighted average soil C sequestration rate for summer fallow elimination is 0.6 t CO_2 ha^{-1} yr^{-1} (a range of -0.2 to 1.2 t CO_2 ha^{-1} yr^{-1}). Small increases in upstream and process emissions come from increased fertilizer N use for the crop that replaces the fallow,[12] resulting in an average net GHG potential of 0.4 t CO_2e ha^{-1} yr^{-1}. In most cases, total crop production will increase,[13] which may generate positive leakage.

Winter cover crops add biomass to a field during the normally fallow winter season, increasing total primary productivity and generating an average soil C sequestration rate of 1.3 t CO_2 ha^{-1} yr^{-1} (31 observations ranging from -0.1 to 3.2 t CO_2 ha^{-1} yr^{-1}). Winter cover crops also tend to reduce fertilizer N needs and N_2O emissions by capturing and

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10. These figures and subsequent ones throughout the report assume total U.S. cropland area of 124 Mha (USDA NASS 2007).

11. In the Mississippi River basin, the NRI-CEAP multiyear cropland study found that only 50% of the reportedly untilled corn and soybean crop area was continuously untilled during the three-year study period (Horowitz et al. 2010). In the remaining area, tillage was eliminated for only one or two crop cycles.

12. We assume no effect on on-farm fuel use; traditional summer fallow uses ~4 tillage operations (Jones et al. 2005), not significantly different from the fuel requirements of planting, maintaining, and harvesting a crop in replacement.

13. Any summer fallow elimination that is not accompanied by an increase in total productivity is less likely to achieve soil C gains and would be economically inefficient and impractical. Therefore, the whole system should maintain or increase sufficient yield.

recycling mobile nutrients and, for legumes, by fixing N from the atmosphere (Alluvione et al. 2010; Delgado et al. 2007; Gregorich et al. 2005). Therefore, planting of cover crops has a net GHG mitigation potential of 2.0 t CO_2e ha^{-1} yr^{-1}. Experts estimate that this activity can be implemented in most or all moist regions of the United States—a total land area of 51 Mha to 99 Mha of U.S. cropland (Donigian et al. 1995; Lal et al. 1999; Sperow et al. 2003). Four percent of U.S. cropland is currently in winter cover crops. Excluding dry regions (Rocky Mountains, Great Plains, and Pacific Southwest), the area in winter wheat, and that already in cover crops, at most an estimated 66 Mha of additional cropland could be planted to winter cover crops. These crops may be less feasible in regions with shorter growing seasons, but crop development and experimentation have shown benefits even in large areas of North Dakota.[14] Cover crops under irrigation may also have some GHG mitigation potential, although care must be taken to ensure that the net GHG impacts of irrigation do not negate it. Incorporating cover crops into a system also may require changes in harvest or planting times, variety selection, and other factors for the main crop, and the GHG implications of such adaptation could be significant, but are currently unknown. For example, earlier harvest of a grain crop in order to plant a cover crop in the fall may necessitate fuel expenditure for grain drying.

Change crop rotations

Research is scant on the soil C effects of ***increasing cropping intensity*** by planting more crops within a yearly cycle (e.g., double- or triple-cropping). Where the growing season allows this activity, greater primary productivity could yield soil C gain, suggesting a useful area for research investment. In addition, positive leakage might occur if total crop yields are increased.

Diversifying annual crop rotations to incorporate plants with greater sequestration potential (through increased root or residue biomass production, increased root exudates, or slower decomposition) could capture additional soil C. Despite almost 90 field comparisons, data are regionally limited (only available for the Great Plains, Rocky Mountains, Corn Belt, and South Central regions), even though this activity could be implemented throughout the U.S. cropland area. The average soil C change is near zero (-1.7 to 1.7 t CO_2e ha^{-1} yr^{-1}), but some rotations appear more likely to increase soil C than others, and net primary productivity (and amount of residue) plays a significant role. For example, switching from continuous corn planting to a corn-soybean rotation results in an average soil C decrease of 0.2 t CO_2 ha^{-1} yr^{-1}, although the lower fertilizer N requirements and reduced N_2O emissions may offset this decrease in terms of net GHG emissions. Other types of crop diversification realize an average soil C gain of 0.1 t CO_2 ha^{-1} yr^{-1}. Nitrous oxide emissions reductions may be of greater benefit, leading to average net GHG mitigation for all annual crop diversification of 0.2 t CO_2e ha^{-1} yr^{-1}.

If crop rotation diversification is to be considered for GHG mitigation, the specific conditions must be carefully researched. Scattered data on baseline continuous crop (mono-crop) production in certain states and counties suggests that from 5%–25% of area planted to corn is not in rotation with other crops (Boryan et al. 2009). We estimate that between 25% and 50% of U.S. annual crop production (~46 Mha) could be further diversified, potentially with soil C sequestration gain. However, other environmental and productivity issues (e.g., weeds and diseases) may provide incentive for diversification.

Incorporating perennials into crop rotations and ***replacing annuals with perennial crops*** have greater soil C sequestration potential than changes in annual crops alone. Compared with annual crops, perennial crops (especially grasses) tend to allocate a relatively high proportion of C underground and have a greater number of days per year of active plant primary productivity, resulting in more potential biomass production and SOC storage. However, perennials may be viable only in moist regions because of greater water requirements (which may also mean increased fossil fuel emissions to supply water in dry regions). Incorporating one to three years of a perennial crop such as alfalfa or grass hay into annual crop rotations is estimated (on the basis of 28 field observations) to capture soil C at an average rate of 0.5 t CO_2 ha^{-1} yr^{-1} (a range of 0 to 1.2 t CO_2 ha^{-1} yr^{-1}), whereas fully replacing annuals with perennials, including biofuel grasses, is estimated (on the basis of 17 observations) to average 0.7 t CO_2 ha^{-1} yr^{-1} (a range of -0.9 to 2.0 t CO_2 ha^{-1} yr^{-1}). Reduced need for fertilizer N, fewer field operations, and some associated N_2O emissions reductions result in a net GHG mitigation of 0.7 t CO_2e ha^{-1} yr^{-1} when perennials are included in annual rotations and of 1.4 t CO_2e ha^{-1} yr^{-1} when perennials replace annual crops. An estimated 56 Mha of land (the moist regions) is estimated to be available for incorporation of perennials in existing crop rotations, and 13 Mha of land may be available for conversion from annual

14. S. Samson-Liebig, personal communication, March 2011.

to perennial crops (Lemus and Lal 2005).[15] Data on these activities in the Corn Belt, Lake States, Pacific Northwest, Northeast, South Central, and Southeast regions are limited or unavailable. Any increase in perennial crop area at the expense of key annual crops (e.g., wheat and corn) could have negative leakage implications.

Plant trees

Incorporating trees into cropland management by ***switching to short rotation woody crops*** (SRWCs) or by ***establishing agroforestry*** could serve both agricultural and carbon sequestration objectives. SRWCs, with rotation lengths of fewer than 30 years, are estimated (on the basis of 35 observations) to sequester an average of 2.5 t CO_2 ha^{-1} yr^{-1} (a range of -7.3 to 13.3 t CO_2 ha^{-1} yr^{-1}). Up to 40 Mha of highly eroded, degraded, or mined lands could be planted to SRWCs with limited negative impact on the production of key food and fiber crops (Tuskan and Walsh 2001). On cropland, SRWCs could also generate substantial reductions in fertilizer and fuel use, further reducing GHG emissions for a net GHG mitigation estimated at 3.9 t CO_2e ha^{-1} yr^{-1}, but the leakage implications may be significant. This assessment does not consider end use (e.g., biofuels), the GHG effects of which are under debate. Reducing productive cropland area would have negative leakage implications.

Agroforestry—incorporating trees into annual or perennial cropping systems—has the potential to sequester between 0.8 and 6.9 t CO_2 ha^{-1} yr^{-1} on up to 10 Mha of land (Bailey et al. 2009; Dixon et al. 1994; Lal et al. 2003; Nair and Nair 2003)[16] through alley cropping, windbreaks, or riparian buffers. Up to 21 Mha of land could be planted to trees in this manner (Nair and Nair 2003). Fertilizer and fuel reductions could yield significant GHG benefits, but data on both soil C implications and effects on N_2O and CH_4 are insufficient to estimate net GHG impact. The C sequestration potential varies widely, depending on the specific practice. In general, alley cropping and silvopasture have the potential to sequester more per hectare than windbreaks and riparian buffers, because they are more intensively planted with trees. As with SRWCs, leakage of displaced crop production into other areas may negatively affect the realized GHG mitigation potential. The end use of the trees (if harvested) is not considered in these calculations.

Apply organic material

Applying organic material, such as livestock and poultry manure, to cropland has mixed results for sequestering carbon and offsetting N_2O emissions from fertilizer use. Baseline conditions of fertilizer use as well as temperature must carefully be considered if this practice is to have a positive effect. Warmer climates lead to greater decomposition rates and less sequestration. Application of organic amendments can sequester between 0.2 and 5.1 t CO_2 ha^{-1} yr^{-1} in soil, with some added GHG benefit from reduced upstream emissions of ~0.4 t CO_2e ha^{-1} yr^{-1} if all fertilizer N is replaced by manure. However, any soil C increase and fertilizer benefit may be negated by the loss of soil C or increase in N fertilization at the location no longer receiving the organic material. Therefore, unless the change in location actually results in different (i.e., reduced) decomposition rates, the net soil C effect will be near zero. Nutrient management improvements, to reduce over-application and associated water and air pollution, are likely to be better reasons for improving manure management. Approximately 8.5 Mha of land (an area calculated on the basis of estimates in Gollehon et al. 2001) could accommodate the manure currently applied in excess of optimal rates on and near the originating farms.

Biochar is another organic soil amendment that is generating attention for its potential to replace synthetic fertilizers, possibly increase biomass productivity, and sequester C from source biomass. Biochar organic C is highly stable, at least in tropical locations, but long-term studies under U.S. conditions have yet to be documented, and the technical details of large-scale application are somewhat uncertain. More important, the life-cycle GHG issues for manure also apply to biochar, and any soil C gains with biochar application (which experts suggest could range from 0.6 to 19.6 t CO_2e ha^{-1} yr^{-1}) could come at the cost of soil C at another location. The very low decomposition rates of biochar when compared with raw biomass may yield net GHG mitigation benefits, but lack of data makes a clear conclusion difficult. Also uncertain are the GHG implications of biochar production processes, which are variable. Recent studies suggest additional potential GHG benefits from biochar, including reduced N_2O emissions (Singh et al. 2010; Taghizadeh-Toosi et al. 2011), replacement of fossil fuels (Woolf et al. 2010) and reduced fertilizer N requirements (Lehmann et al. 2003). Exploration of these possibilities is under way. If sufficient feedstock were available, biochar could be applied to the majority of U.S. cropland, although transportation and other issues must be considered.

15. This estimate reflects the total land area that might be available for conversion to perennial biofuel crops.

16. The mitigation potential of agroforestry and other activities for which that potential is low or data are lacking is expressed as a range of field data, model estimates, or expert estimates.

Set aside cropland or convert cropland to pasture

In general, conversions of cropland to alternate nontree uses (pasture or set-aside land) have soil C sequestration potential, largely due to the cessation of tillage, but also to the higher primary productivity of perennial grasses and other plants. ***Setting aside cropland*** may be an option for approximately 14 Mha of erosion- or flood-prone lands (Lal et al. 1999) as well as herbaceous buffers in between cropped portions of fields. Available data are regionally dispersed, as is the potential for implementation. Conversion of boundary cropland to natural areas or unharvested vegetation has the potential to sequester 2.0 t CO_2 ha^{-1} yr^{-1} (on the basis of 28 observations, ranging from -0.4 to 5.1 t CO_2 ha^{-1} yr^{-1}) and to significantly reduce N_2O emissions and upstream and process emissions for a net GHG mitigation potential of 3.6 t CO_2e ha^{-1} yr^{-1}.

With less significant (or at least different) leakage implications, ***converting cropland to pasture*** can generate a somewhat greater soil C sequestration rate: 2.4 t CO_2e ha^{-1} yr^{-1} (on the basis of 26 observations, ranging from 0.4 to 4.2 t CO_2 ha^{-1} yr^{-1}). When the reduced N_2O and process and upstream emissions are also counted, the average net GHG mitigation potential is 3.1 t CO_2e ha^{-1} yr^{-1}. No estimates of applicable area are available for conversion of cropland to grazing land. Any conversion of cropland to other land uses has leakage implications, because production pressure will increase for other locations.

Unlike most soils, which are nearly all mineral material (sand, silt, clay), histosols are composed of at least 20% organic matter and thus are considered to be organic soils. Because histosols have unique soil properties, croplands on them tend to have greater GHG mitigation potential than other croplands, when set aside. The most detailed estimates indicate that 0.8 Mha of histosols are currently farmed (Morgan et al. 2010).[17] The farming of these soils causes rapid decomposition of soil organic C (between 2 and 73 t CO_2e ha^{-1} yr^{-1}). ***Setting aside histosol cropland*** (i.e., converting them from agriculture back to their natural state) slows or stops this decomposition. Farmed histosols can generate high N_2O emissions, which can be significantly reduced by setting these lands aside. Methane emissions may be slightly increased as a result, especially with maintenance of a higher water table, but these increased emissions are about an order of magnitude less than the reductions in CO_2 and N_2O (in CO_2 equivalents). Setting aside this cropland also eliminates fuel, upstream, and process emissions, although reductions in agricultural production may have negative leakage effects. Because histosols in some regions (e.g., the Sacramento-San Joaquin delta in California) produce important vegetable crops, the practical implications require particular attention.

Restore wetlands

Not all former wetland soils have enough organic material to be classified as histosols; nevertheless, many of these soils store significant amounts of carbon, which is emitted during cultivation. Experts have estimated that ***restoring wetlands*** can generate soil C sequestration at a rate of about 1.5 t CO_2e ha^{-1} yr^{-1} (IPCC 2000; Lal et al. 2003). However, data from more recent field comparisons suggest that the rate may average 6.5 t CO_2e ha^{-1} yr^{-1} (a range of -1.0 to 9.9 t CO_2e ha^{-1} yr^{-1}) in the Prairie Pothole region of southwest Minnesota, North and South Dakota, northwest Iowa, and northeast Montana (Badiou et al. 2011; Euliss et al. 2006; Gleason et al. 2009), where 3.8 Mha of former wetlands are currently cultivated for crops (Euliss et al. 2006). These studies compared restored wetlands with cropland (8 comparisons, Gleason et al. 2009) and with undrained wetlands (62 comparisons, Badiou et al. 2011; Euliss et al. 2006). Significant variability in wetland types means significant variability in soil C changes. For example, Euliss et al. (2006) found high soil C sequestration rates in semipermanent wetlands, but little to no C accrual in seasonal wetlands. Data on wetlands in other U.S. regions are unavailable. Setting cropland aside for wetland restoration could also have leakage implications.

Manage farmed histosols (organic soils)

If set-aside is not a viable option, ***managing farmed histosols*** also has potential to reduce CO_2 emissions by up to 15.0 t CO_2e ha^{-1} yr^{-1}. Options for improved management include reducing tillage (including deep plowing), switching to less intensively managed crops, allowing a shallower water table, and converting cropland to grassland. These changes could also reduce N_2O and CH_4 emissions, although more research is needed to confirm this impact.

Convert dry land to irrigated land

Changes in irrigation management can affect soil C as well as CO_2, N_2O, and upstream GHG emissions. ***Converting dry land to irrigated land*** increases biomass productivity (with consequent leakage implications), with soil C effects

17. Morgan et al. (2010) indicate that the total U.S. histosol area is 10 Mha, of which 7.5% is in agricultural production. Lal et al. (2003) report that 15 Mha and 19 Mha of land area are available for improved histosol management (the text indicates one figure and a table indicates the other figure), but whether either figure is the total histosol area or the total farmed histosol area remains unclear.

ranging from -0.6 to 2.8 t CO_2e ha^{-1} yr^{-1} (Bordovsky et al. 1999; Entry et al. 2002; Lal et al. 1999; Liebig et al. 2005). But increased GHG emissions from energy inputs (Follett 2001; West and Marland 2002), degassing that releases CO_2 from irrigation water (Martens et al. 2005), and increased N_2O emissions (Bremer 2006; Liebig et al. 2005; Rochette et al. 2008b) most often outweigh these benefits. The ecological tradeoffs of using more water in water-scarce regions may also be significant. Therefore, an increase in irrigation area is not a promising option for GHG mitigation.

Nitrous oxide and methane emission reduction

Reducing fertilizer N application rates, changing fertilizer sources, and improving application timing, placement, and efficiency can significantly reduce N_2O emissions from nitrification and denitrification of mineral nitrogen on agricultural land. Research studies for mitigation tend to focus on opportunities for reducing direct N_2O emissions—that is, fluxes from the field area. Offsite or indirect emissions—generally originating from nitrogen in fields contained in leaching or runoff water—can also be significant and are influenced by land management. Minamikawa et al. (2010) report that indirect N_2O emissions from soybean-wheat and upland rice systems comprised 34%–40% of total N_2O emissions. Reay et al. (2009) report similar findings from grazed pasture in the United Kingdom, where indirect emissions from leaching comprised 25% of total N_2O emissions. Simulated emissions from the DAYCENT model show indirect N_2O emissions to be 39% of total annual N_2O emissions for all cropland in the United States during the 1990–2003 period (Del Grosso et al. 2006).

Any activity that reduces direct N_2O emissions without decreasing total N losses from the system carries the risk of increasing indirect N_2O emissions (Reay et al. 2009), but activities that improve N use efficiency allowing lower N additions are likely to reduce both direct and indirect emissions. In contrast, activities that affect water use or soil quality, for example, may reduce direct emissions but possibly not indirect emissions. In fact, the latter may increase. The estimates of mitigation potential described in this report account only for changes in direct N_2O emissions. Therefore, the report may underestimate potential where indirect emissions are likely to decline and overestimate them where they may increase.

Improve N use efficiency and reduce losses

Most U.S. cropland has some potential for improved N use efficiency. The potential for ***reducing fertilizer N application***[18] exists, although given the paucity of baseline data, accurately assessing this potential is difficult. Under experimental conditions, research has found that typical fertilizer N application levels could be decreased by 15%–20% (and even below USDA-recommended levels) without significant yield losses, a strategy feasible, many experts believe, in U.S. field-scale production (Bausch and Delgado 2005; Millar et al. 2010; Smith et al. 2008). Others note that excess nitrogen can be an important risk reduction strategy, and therefore persuading farmers to adopt it may not be easy. As yet, no consensus exists as to whether, where, and how much excess fertilizer is being used at the farm or field level. One of the only data sources to inform this debate is a recent assessment of N balance at a county and regional scale for the United States (IPNI 2010). Any yield decreases from reduced fertilizer will have leakage implications, and increase the cost of associated GHG mitigation.

Decreased fertilizer N rates are, in most cases, associated with lower N_2O emissions, but more so in moist than in dry climates, with respective means of 0.6 t CO_2e ha^{-1} yr^{-1} (McSwiney and Robertson 2005; Millar et al. 2010) and 0.05 t CO_2e ha^{-1} yr^{-1} (Bremer 2006; Halvorson et al. 2008; Mosier et al. 2006), a 10-fold difference. However, U.S. data are limited to the Corn Belt, Lake States, Great Plains, and Rocky Mountains. Process and upstream emissions reductions generate some additional GHG emissions savings, and offsite N_2O emissions reductions are also likely, leading to an average GHG mitigation potential of 0.3 t CO_2e ha^{-1} yr^{-1} (range from 0.1 to 0.9 t CO_2e ha^{-1} yr^{-1}). However, reduction of fertilizer N rates will likely be part of larger strategy to improve N use efficiency that results in decreased fertilizer N needs. This strategy would include adjustments in fertilizer source, placement, and timing as well as rate.

Nitrogen fertilizer is applied on nearly all U.S. cropland, and rate reductions may be possible on much of this area. Soil sampling helps farmers know fertilizer needs, often resulting in lower fertilizer application rates,[19] but only 50% of corn cropland is tested for soil N availability (Paustian et al. 2004). If 50%–60% of cropland is overfertilized, and the recommended rate may be higher than needed in some cases, we estimate that rate reductions could be implemented on approximately 68 Mha.

18. Here we assume a 15% reduction, which is consistent with the data available.
19. Of the corn farmers who test for soil N availability, 80% apply the recommended rate (Paustian et al. 2004).

The N_2O emissions from urea-based N fertilizers have been significantly lower than those from ammonia-based N fertilizer in a few studies from Minnesota and Tennessee (Thornton et al. 1996; Venterea et al. 2005; Venterea et al. 2010); crop type and tillage regime affected the magnitude of the difference. On the other hand, researchers in Manitoba found no emission differences between these fertilizer types (Burton et al. 2008). Therefore, while 15 field observations indicate a promising average GHG mitigation of 0.5 t CO_2e ha^{-1} yr^{-1} (range from 0.02 to 1.0 t CO_2e ha^{-1} yr^{-1}) for ***switching fertilizer N source from ammonia-based to urea***, the applicable conditions for switches in fertilizer forms must be more clearly established. Data are not available for nontarget GHG categories, and little effect is expected. Manufacturing related emissions do not vary much among fertilizer types. Given that 20% of U.S. fertilizer is sourced as anhydrous ammonia, a switch in fertilizer form could be implemented on up to 37 Mha of cropland.

Switching to a slow-release fertilizer N source (including controlled-release and stabilized N fertilizer) can also reduce N_2O emissions by improving the synchronization of fertilizer N with plants' N uptake needs. Experimental data indicate emissions reductions of about 0.1 t CO_2e ha^{-1} yr^{-1} (on the basis of 14 observations ranging from 0 to 0.2 t CO_2e ha^{-1} yr^{-1}). In addition, the enhanced N use efficiency could permit lower N application rates, leading to further emissions reductions and lower process and upstream emissions for a net GHG mitigation potential of 0.2 t CO_2e ha^{-1} yr^{-1}. Because EEF use is low, the majority of N-fertilized area (conservatively, 93 Mha or 75% of cropland area) could theoretically be improved in this manner.

Changing fertilizer N placement and application timing can also effectively reduce N_2O emissions by improving N use efficiency and generating lower total N losses. By placing fertilizer N in bands on or under the soil surface, rather than equally across a field (at the same application rate), farmers can deploy it close to the zone of active root uptake and reduce N_2O emissions by an average of 0.3 t CO_2e ha^{-1} yr^{-1} (on the basis of 31 observations ranging from 0 to 0.9 t CO_2e ha^{-1} yr^{-1}). However, the impact of placement depth seems to be affected by climate and soil type, with reduced emissions following shallow placement in Ontario's moist, cool climate (Drury et al. 2006), and lower emissions for deeper placement in Colorado's drier, warmer climate (Liu et al. 2006). In the 1990s, fertilizer was applied by banding on 40% of U.S. corn land area (Paustian et al. 2004). If applied similarly on other cropland, an estimated 60% of U.S. cropland (63 Mha) could experience improved fertilizer N placement. Site-specific fertilizer N placement with GPS technology that matches application rates to crop yields can also reduce N_2O emissions by up to 2.3 t CO_2e ha^{-1} yr^{-1} in low-yielding field zones (Sehy et al. 2003).

Synchronizing fertilizer application with crops' N uptake could also reduce N losses, including N_2O emissions. Shifting fertilizer application from the fall to the spring or from single to split (multiple) applications can reduce losses from both leaching and denitrification. All available research on fertilizer N timing is from Canada (Burton et al. 2008; Hao et al. 2001; Hultgreen and Leduc 2003), and it indicates an average GHG mitigation potential of only 0.2 t CO_2e ha^{-1} yr^{-1} (on the basis of 33 observations ranging from 0 to 0.5 t CO_2e ha^{-1} yr^{-1}). Further research is needed to examine how N_2O flux varies across management combinations and soil types. Given that 30% of U.S. corn is fertilized in the fall (Paustian et al. 2004), and additional land could benefit from split fertilizer application, improvements in fertilizer N timing could be deployed on an estimated 50% (53 Mha) of U.S. cropland.

Using nitrification inhibitors can improve N use efficiency and reduce leaching losses and fertilizer N rates. Current research suggests that significant N_2O emissions reductions—0.6 t CO_2e ha^{-1} yr^{-1} (on the basis of 26 observations ranging from 0 to 1.6 t CO_2e ha^{-1} yr^{-1})—are possible (Bhatia et al. 2010; McTaggart et al. 1997; Snyder et al. 2009). Further research is needed to examine nitrification inhibitor interactions with different fertilizers, timing, placement, depth, soil temperature, and pH. Nitrification inhibitors are currently utilized on only 3.4 Mha (USDA ERS 2010a), and because 90% of commercial fertilizer is urea- or ammonium-based, 92 Mha of the total N area are available for nitrification inhibitor application.

Improve manure application management

Other sources of nitrogen—such as residue, manure, or compost—also affect N_2O emissions. With high ammonium concentrations, manure can also generate NH3 emissions, which can be an indirect source of N_2O emissions when redeposited on soil or plants. ***Improving manure management on cropland***, including reducing total application rates, applying solid rather than liquid manure, using nitrification inhibitors, and applying manure to dry rather than wet areas when air temperatures are relatively low, can reduce N_2O emissions by 0.4 to 1.2 t CO_2e ha^{-1} yr^{-1} (Gregorich et al. 2005; Paustian et al. 2004; Rochette et al. 2000). While adjusting synthetic fertilizer application rates to account for manure additions is uncommon, it could further decrease N_2O emissions. Manure management for N_2O emission reduction is possible on at least a portion of the 12 Mha of U.S. cropland currently receiving manure applications. Improved manure and mineral N management on grazing land, which can include reducing the N content in animal feed (Mosier et al. 1998), may also lower N_2O emissions, but a lack of data prevents us from making any estimate of impact.

Improve irrigation management and drain agricultural land in humid areas

Irrigation efficiency improvements can create upstream energy (and emissions) savings and reduce N_2O emissions because less water is available to cause anaerobic conditions conducive to denitrification. Examples of ***improving irrigation management*** include conversion from less efficient furrow irrigation to central-pivot systems or drip irrigation. Buried drip irrigation leaves a dry soil surface, and better aeration of surface soils can reduce N_2O emissions significantly (Kallenbach et al. 2010). Total N_2O emissions reductions from irrigation improvements are estimated to be between 0.1 and 0.9 t CO_2e ha^{-1} yr^{-1} (Rochette et al. 2008b; Scheer et al. 2008).

In one global review comparing 193 poorly drained soils with 460 well-drained soils, Bouwman et al. (2002) found lower N_2O emissions in the well-drained soils (equal to a difference of 0.19 t CO_2e ha^{-1} yr^{-1}). However, these were not side-by-side comparisons, and we were unable to find other information about the potential of ***draining agricultural land in humid areas*** for N_2O emission reduction.

Manage rice production

Methane from rice production is a significant source of greenhouse gases worldwide, but it represents a small component of U.S. agricultural emissions because the United States has only 1.3 Mha of rice cropland. Nevertheless, potential CH_4 emission reductions per unit area are substantial. GHG mitigation potential in rice systems varies dramatically by management practice and geography, but ***adjusting rice water management*** and ***planting rice cultivars that produce less CH_4*** are two promising emissions-reducing activities. Single or multiple midseason drainages could generate significant CH_4 emission reductions (Li et al. 2005b; Sass and Fisher 1997), averaging 2.0 t CO_2e ha^{-1} yr^{-1} (on the basis of 26 observations ranging from 0.1 to 5.3 t CO_2e ha^{-1} yr^{-1}). This strategy can be effective as long as farmers plant varieties that reflect the climate to avoid yield losses that are common when rice is subjected to low nighttime temperatures, which are normally moderated by flooded conditions. The effectiveness of drainage or other reduced flood time (including shifting pre-harvest drainage dates earlier and shifting from wet to dry seeding) is also dependent on concurrent N_2O emission increases, which can be an issue for soils that have relatively high organic C content. The net GHG mitigation potential of rice water management is estimated to be 1.1 t CO_2e ha^{-1} yr^{-1}. More research on reduction of methane emissions from rice in the U.S. context is needed, and some promising work is under way in California rice systems (De Gryze et al. 2009; Salas 2010).

Conversion to high-yield rice varieties can direct more carbon to grain production rather than root processes, thus reducing the root respiration and exudation rates that increase CH_4 production. Given other CH_4 emission differences among cultivars (Setyanto et al. 2000; Wassmann et al. 2002), appropriate selection could generate mitigation of approximately 1.0 t CO_2e ha^{-1} yr^{-1} (on the basis of 19 observations ranging from 0.1 to 1.9 t CO_2e ha^{-1} yr^{-1}). No data are available for other GHG categories. More research on how specific rice cultivars affect emissions rates in different soils and regions is needed.

Reducing rice area by switching to other crops could reduce CH_4 emissions, but the overall GHG implications depend on subsequent land use and displacement of production elsewhere (leakage), likely eliminating any substantial mitigation potential. Other rice management activities, not assessed in this report but the subject of ongoing research, include reducing the duration and frequency of winter flooding (e.g., by staggering flooding over time and across fields); changing rice residue management (e.g., by removing rice straw prior to flooding); shifting from contour levees to precision- or zero-grade systems that reduce water consumption; and upgrading pumping systems to improve diesel efficiency. These activities may deliver GHG reductions through a combination of reduced anaerobic conditions and reduced process emissions from fossil fuel use for water pumping. Research is ongoing to quantify GHG potential and economic and operational feasibility.

Reduce chemical use

Reducing chemical use other than fertilizer N may also have positive GHG impacts. Although pesticide production uses two to five times more energy (on a per-weight basis) than fertilizer N production, the GHG impacts (on a per-hectare basis) of pesticides are relatively small because much less pesticide than fertilizer is applied per hectare. No soil C or other land emission impact of pesticide reduction is anticipated, and the process and upstream effect ranges from 0.03 to 0.06 t CO_2e ha^{-1} yr^{-1}, very small in comparison to other activities explored in this report.

Grazing Land

Improve grazing management on rangeland

Grazing management on rangeland (grazing land without tillage, seeding, or irrigation inputs) can be improved by reducing stocking rates on overgrazed land, avoiding grazing during drought conditions, and changing the timing and frequency of grazing. ***Improving grazing management on rangeland*** is expected to capture a significant amount of carbon in the United States; broad reviews indicate potential soil C sequestration rates of between 0.6 and 1.3 t CO_2e ha^{-1} yr^{-1} (Conant and Paustian 2002; Conant et al. 2001; Follett et al. 2001). These studies suggest that reduced stocking rate is the primary driver for soil C sequestration. Indeed, Fuhlendorf (2002) measured significant decline in soil C concentration on the Great Plains as stocking rates increased above the USDA-recommended rate, compared with less intense grazing. However, our review of 10 studies of reduced stocking rates on North American rangelands—for which soil C was quantified[20]—showed extremely variable results, but suggests an average soil C *decrease* of about 1 t CO_2e ha^{-1} yr^{-1} (Frank et al. 1995; Liebig et al. 2010c; Manley et al. 1995; Naeth et al. 1991; Reeder et al. 2004; Schuman et al. 1999; Smoliak et al. 1972).

Most of the studies of reduced stocking rates were conducted on research sites with well-managed range, where grazing tends to stimulate plant growth and increase soil C more than setting land aside (Derner and Schuman 2007; Liebig et al. 2005). The broad reviews mentioned above may be considering degraded rangeland, where overgrazing over long periods can decrease productivity (Schuman et al. 1999), with associated soil C effects. Accurate assessment of baseline rangeland health is difficult to achieve because of lack of field data, high rainfall variability, and the large area involved (Herrick et al. 2010), hence the proportion of rangeland in degraded condition that may have increased soil C sequestration potential with stocking rate reductions is unknown. Nonfederal grazing land area in the United States (i.e., owned privately or by state and other governments) is between 176 Mha (Lubowski et al. 2006b) and 214 Mha (USDA NRCS 2007), with an additional 62 Mha of federal grazing land (Lubowski et al. 2006b). Of this land, up to 48 Mha is pasture, that is, grazing land with tillage, seeding, or irrigation inputs (USDA NRCS 2007). The government-owned land is primarily unimproved rangeland, mostly in the western states. Therefore, there may be up to 227 Mha of total rangeland with potential for improved management.[21] Further work is necessary to identify the land area most likely to benefit from management changes as well as the changes with the greatest sequestration potential. Also needed is research on or direct assessment of grazing management that accounts for drought or makes adjustments in timing and frequency.

Improve grazing management on pasture

Pasturelands are grazing lands with tillage, seeding, and/or irrigation inputs. ***Improving grazing management on pasture***, with possible C sequestration benefits, can also involve reduced stocking rates, especially where land has been overgrazed (Conant and Paustian 2002; Franzluebbers and Stuedemann 2009). However, as with rangeland, stocking rate reductions on pasture do not always improve soil C retention rates; grazing can stimulate incorporation of above-ground litter into the soil. Therefore, increased grazing pressure may increase (Schnabel et al. 2001) or have no effect on (Franzluebbers et al. 2001) soil C storage. In addition, grazing activity can alter species composition in ways that increase soil C storage. A literature review shows that reducing grazing pressure on pasture has highly variable soil C

20. That is, these studies measured mass of soil C, not just concentration.

21. This is equal to 165.6 Mha of nonfederal rangeland (USDA NRCS 2007) plus 61.5 Mha of federal grazing land (Lubowski et al. 2006b), assuming that all federal grazing land is range.

implications (ranging from a loss of 3.0 to a gain of 4.8 t CO_2e ha^{-1} yr^{-1}), and improved management must be tailored to individual conditions. Improved pasture management may have some potential on all 48 Mha of U.S. pasture.[22]

Introduce rotational grazing

In ***rotational grazing of pasture***, animals are shifted across the pasture—intensifying grazing in a smaller area over a shorter period of time—to allow optimum plant growth. The few expert assessments and surveys on this activity, which holds particular promise in more humid regions, estimate that it has the potential to sequester between 0 and 2.9 t CO_2e ha^{-1} yr^{-1} (Bosch et al. 2008; Conant et al. 2003; Lynch et al. 2005). Rotational grazing tends to improve forage quality and increase total forage (and thus beef or dairy) production per unit area (DeRamus et al. 2003; Jacobo et al. 2006; Oates et al. 2011), and therefore positive leakage may be significant. If rotational grazing has been implemented on 13% of current pasture area, as is the case in the Northeast (Winsten et al. 2010), an estimated 42 Mha of pasture remain available for adoption of that activity.

On U.S. ***rangeland, rotational grazing*** tends to be less productive than continuous grazing (Briske et al. 2008; Derner et al. 2008), with consequent reductions of up to 5.3 t CO_2e ha^{-1} yr^{-1} in soil C (Manley et al. 1995). Other recent research noted a soil C gain of 1.9 t CO_2e ha^{-1} yr^{-1} (Teague et al. 2010). There may be a differential impact of rotational grazing given differences in water availability, with moist pasture responding more favorably in terms of overall forage production and soil C than drier rangeland. Further research on this relationship is needed.

Manage species composition

Managing species composition on grazing land by establishing seeded pasture or interseeding the land with alfalfa, other legumes, or improved grasses increases productivity and has significant potential to sequester soil C at an average rate of 1.5 t CO_2e ha^{-1} yr^{-1} (on the basis of nine observations ranging from 0.2 and 3.1 t CO_2e ha^{-1} yr^{-1} (Conant et al. 2001; Liebig et al. 2010c; Lynch et al. 2005; Mortenson et al. 2004). An estimated 80 Mha (both pasture and rangeland) are potentially available for improved species management.

Establish agroforestry

Establishing silvopasture systems (trees planted on grazing land—i.e., agroforestry) may also have GHG mitigation potential on up to 70 Mha of grazing land (Nair and Nair 2003), through soil C and aboveground C storage. With little field research data, the estimated soil C sequestration rates of between 0.5 to 3.6 t CO_2 ha^{-1} yr^{-1} (Dixon 1995; Nair and Nair 2003; Sharrow and Ismail 2004) are largely based on expert opinion. Therefore, further assessments are warranted, including the effects on life-cycle GHG balance.

Set aside grazing land

Setting aside grazing land in the United States—that is, excluding grazing animals from grassland (both rangeland and pasture)—tends to result in loss of soil C at an average rate of 0.5 t CO_2e ha^{-1} yr^{-1} (on the basis of 27 field observations ranging from -2.8 to 0.8 t CO_2e ha^{-1} yr^{-1}), when compared to land that is well-managed for grazing (Conant and Paustian 2002; Conant et al. 2001; Derner and Schuman 2007; Liebig et al. 2005; Martens et al. 2005). Grazing can reduce soil C and increase erosion in some situations (Jones 2000), and such degraded land generally needs some management to restore vegetation—for example, rest periods without animal activity. However, the research regarding set-aside tends to compare exclusions with well-managed moderately grazed treatments, from which there appears to be little soil C benefit to set-aside (Katsalirou et al. 2010). The impact of set-aside on degraded grazing lands is not well known. Set-aside from grazing also has significant negative leakage implications, shifting pressure for increased production elsewhere. Therefore, setting aside of well-managed grazing land is not likely a viable option for GHG mitigation, except in limited cases such as very sensitive coastal marshland soils (Reeder and Craft 1999, cited in Franzluebbers 2005).

Other grazing land management activities

Fertilizing and ***irrigating grazing land*** can improve plant productivity and thus increase soil C and cause positive leakage. For fertilizer application, the soil C increase ranges from 0.4 to 5.9 t CO_2e ha^{-1} yr^{-1}, and for irrigation, the soil C gain ranges from 0 to 1.8 t CO_2e ha^{-1} yr^{-1}. However, when associated N_2O and upstream and process emissions are counted, the net impact tends to be negative with both fertilization (Schnabel et al 2001) and irrigation (Follett 2001; Lal 2004a; Schlesinger 2000).

Lack of data prevents estimate of GHG effects for other grazing land management activities. ***Improving N use efficiency***

22. Pasturelands are primarily privately owned.

of fertilizer and manure on grazing land may also have potential to reduce N_2O emissions, but there are few, if any, data available for quantification. ***Introducing fire management on grazing land*** entails negative co-effects associated with burning (methane, smoke, aerosols), a number of which are also linked to climate change, making it even less attractive as a GHG mitigation option (Smith et al. 2008). Therefore, such evidence and the lack of side-by-side comparison data make rangeland fire management a poor candidate for GHG mitigation.

Combining Practices

In many cases, multiple management activities are implemented on one parcel of land. In these cases, interactions among practices may modify the biophysical potential of each practice, generating results that differ from the simple sum of the individual C storage or GHG flux effects. Studies have documented the GHG implications of some of these interactions: conversion to no-till and fallow reduction (Sainju et al. 2006), conservation tillage and introduction of cover crops (Franzluebbers 2010; Parkin and Kaspar 2006), tillage and crop diversification (Dick et al. 1986, redrawn by Lal et al. 1999; Sainju et al. 2006), and crop diversification that includes winter cover crops (Liebig et al. 2010a). Other more complex studies measure the GHG effects of numerous combined activities, complicating assessment of the contributions from and multiple interactions among activities. For example, Drinkwater et al. (1998) examined three systems with different crop rotations, fertilizer N sources, and chemical application rates, with and without cover crops. Wagner-Riddle et al. (2007) compared two systems that differed in tillage, N rate, N timing, and cover crop use. Although the existing body of research is insufficient to provide estimates of net GHG potential for many combinations of practices, biogeochemical models (described in this report) can provide estimates of their GHG fluxes. The existing research that assesses combined practices is important for calibrating and testing the accuracy of these models' estimates.

Specialty Crops

The majority of research on agricultural GHG mitigation is specific to major crops such as corn, soybean, wheat, and grasses for pasture. Specialty crops, which include fruits and vegetables, tree nuts, dried fruits, and nursery crops, also have the potential to mitigate greenhouse gases on a smaller scale.[23] These crops comprise only 4% of U.S. cropland, but they make up 23% of the agricultural sector market. The presence of similar biogeochemical processes in plants and soil suggest that many of the management activities discussed above could apply across crop types. However, differences in residue production, tillage requirements, and perhaps crop-specific impacts may necessitate model or default factor development that considers each crop or crop type individually. On the basis of limited available data, it appears that—for California, at least—cover crops and fertilizer N management have some significant potential but that tillage management is less feasible (J. Six, personal communication).

Biotechnology and Other Agricultural Advances

Agricultural biotechnology—practices ranging from traditional crop breeding to genetic modification (GM)—can contribute to GHG mitigation by increasing crop yields, reducing soil C loss related to tillage, expanding the use of cover crops, intensifying crop rotations, and increasing nitrogen and water use efficiency. Given food demands of a growing population, the GHG emissions associated with agriculture would have been significantly greater without the agricultural intensification and yield increases experienced since the mid-1900s (Burney et al. 2010). Such improvements are often associated with increased efficiency of fertilizer N use, resulting in lower N_2O emissions per unit of output even though fertilizer N application overall has increased. Because food demands will continue to rise, further intensification may be necessary to conserve land and other resources. Output-based GHG calculation metrics that consider GHG intensity per unit of crop produced can appropriately credit this strategy for its GHG benefits to the system as a whole (Murray and Baker 2011).

With a focus on grain crops, traditional breeding and GM development have led to improved plant stem strength, disease and pest resistance, and water use efficiency. Herbicide resistance of crop varieties through genetic modification has also allowed greater adoption of no-till practices. Continued developments in biotechnology may further contribute to GHG mitigation through many of the management practices described above. Because the introduction of new genetic variants can have a range of impacts (both environmental and social), special attention should be paid to possible co-effects (positive or negative). Development of new equipment can also advance mitigation activities; for

23. The high value of specialty crops may make incentive payments for GHG mitigation less attractive than for the major grain and oil-seed crops, simply because the value of GHG mitigation per hectare would be a lesser proportion of total income.

example, direct-seeding equipment for no-till land or global positioning system (GPS) technology for precision fertilizer application.

Research coverage and scientific certainty

Decisions about which agricultural practices to include in mitigation programs should reflect scientific certainty regarding estimates of GHG-flux effects. Some activities have been well studied over multiple regions with field data on multiple GHGs; studies of other activities have far fewer available data, little regional coverage, and limited information on nontarget GHG fluxes. This report assesses the quantity and quality of the data on GHG-mitigating agricultural activities by documenting the research coverage in the requisite data and by asking a panel of scientific experts about their confidence in the available data.

Although research on the GHG impacts of agricultural land management strategies in the United States is impressive, it contains many critical gaps. For most practices, data are insufficient to tease out how mitigation potential changes across cropping systems, soils, and climate. However, data are sufficient to sketch a roadmap of practices with significant potential, those with little or perhaps negative potential, and those with unknown potential. Given the remaining uncertainty and data gaps, this assessment used expert input to help gauge scientific certainty regarding the mitigation potentials synthesized from the scientific literature.

Many data gaps were noted in the discussion of management activities above. Scientific certainty is also affected by a lack of research into different combinations of activities and variations in different activities (e.g., different tillage intensities in the baseline scenario) as well as a lack of data on non-CO_2 greenhouse gases. Appendix B summarizes the data and research gaps that, if addressed, would significantly improve understanding and implementation of GHG-mitigating agricultural activities.

Research coverage

Scientific certainty is enhanced when many field and laboratory comparisons report similar results and when differences in results can be explained by regional or other situational circumstances. Broad regional coverage creates greater confidence that study results will be the same across the country. If all the studies of a specific practice were performed at two similar research stations, confidence in extending the results to other regions may be low.

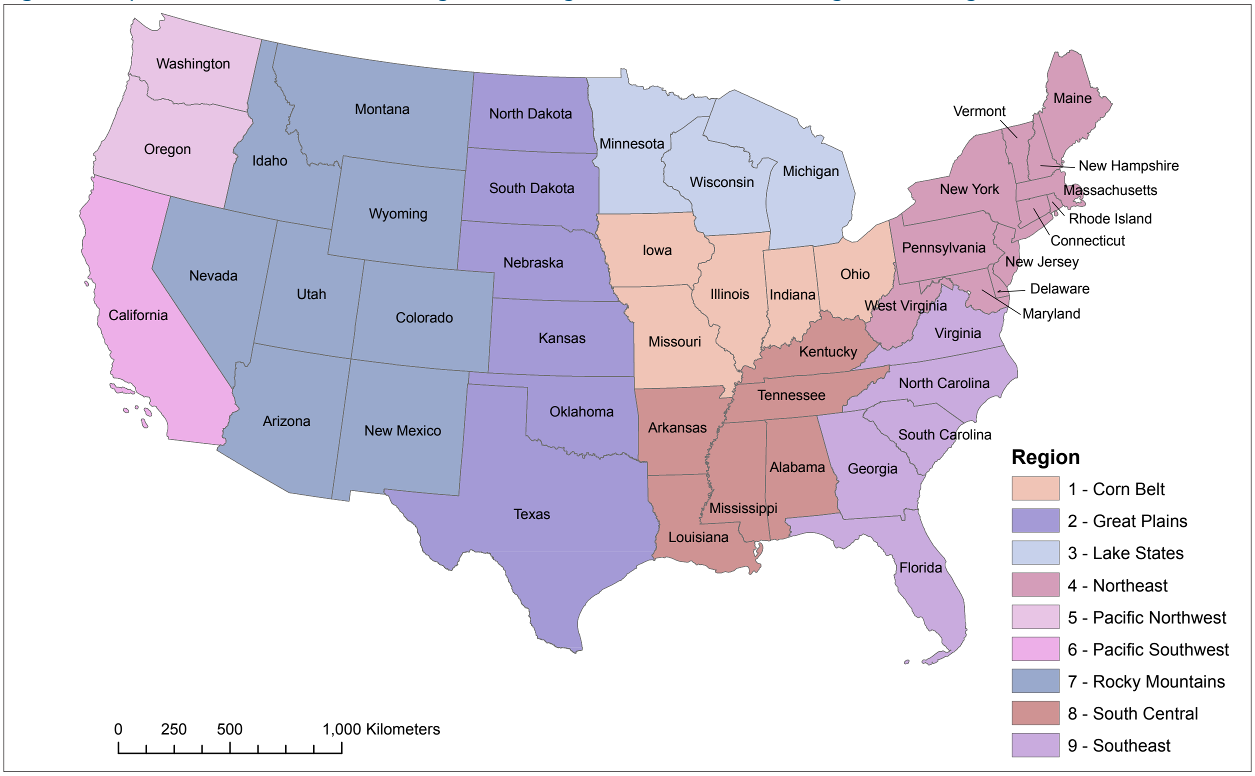
Table 3 summarizes the research coverage for each reviewed land management activity. It reports the number of data sources used to calculate the national estimate of GHG flux effect for each activity's target greenhouse gas (soil C, land emissions of CH_4 and N_2O, or upstream or process emissions). For activities with more than eight side-by-side field comparisons of the GHG flux differences between the activity and conventional practice,[24] the national estimate reflects only data from those comparisons. This estimate is the only number reported in Table 3's first two categories: positive mitigation potential and significant or moderate research. For all other activities, the national estimate reflects the available data plus expert and modeled estimates.[25] For these activities, Table 3 reports three numbers: the number of side-by-side field comparisons, expert estimates, and modeled estimates. It also reports the regional coverage for all contributing data (within the United States and elsewhere when few U.S. values are available). The 48 conterminous states are divided into 9 broad regions on the basis of shared climate and cropping characteristics (see Figure 2).

Many of the management activities that can sequester soil C have been studied broadly. These include tillage, cover crops, and fallow reduction. Other activities—biochar, agroforestry, histosol management, and manure management for N_2O reduction, for example—have little field testing. Estimates of the GHG mitigation potential of these activities rely more heavily on expert opinion.

24. These comparisons tend to be replicated and peer reviewed.

25. Expert estimates are not directly calculated from (although they may refer to) other studies but are estimates based on experience and scientific understanding. Modeled estimates have been calibrated with empirical field data and consider biogeochemical processes common to many or most applicable cropping systems.

Figure 2. Map of the United States indicating the nine regions used to determine regional coverage of scientific data



Survey of experts on scientific certainty

Previous sections of this report have summarized, based on the scientific literature, the biophysical GHG mitigation potential of 42 agricultural land management activities that could be implemented in cropping and grazing systems in the United States. Even when the scientific literature presents robust data, uncertainties about this potential can remain—for example, the yield impacts of reducing fertilizer N rates are unknown, as is the impact of tillage on soil C at depths below the plow layer. Apparent inconsistencies among measurements that are not fully understood and explained can further reduce confidence in the expected GHG impacts.

Because broadly applied meta-analysis is not possible for the wide range of management activities of interest, this assessment uses consultation with experts to gauge confidence in the mitigation potentials found in the scientific literature and to identify regional issues or other caveats in using these numbers to guide development of GHG mitigation policies or programs. A key issue for each activity is whether scientists think that implementation is likely to result in a net positive GHG outcome when all gases and upstream impacts are considered. Accordingly, the panel of experts was asked to qualitatively assess scientific certainty about the biophysical GHG mitigation potentials reported in the literature. For this survey, the original set of 42 activities was reduced to 28 that the literature suggest have positive GHG mitigation potential, broad regional applicability, and high mitigation potential per unit area even if applicability is limited.

A full report on the survey, *T-AGG Survey of Experts: Scientific Certainty Associated with GHG Mitigation Potential of Agricultural Land Management Practices,* is available from T-AGG.[26] The survey took the form of five 90-minute webinar sessions in which an average of 10 scientists participated. Survey sessions were organized according to topic area (soil carbon on cropland, N_2O emissions reductions on cropland, grazing land management, and CH_4 or multiple GHGs emissions reductions). A discussion of qualifiers and caveats for each activity preceded anonymous voting on questions of scientific certainty and level of supporting evidence. The voting options for confidence and evidence were explicitly defined. For example, *medium* and *high* confidence ratings were equated with positive directional certainty—that is, experts had confidence that the proposed activity offered positive GHG mitigation (although the magnitude of mitigation may not be well defined). High confidence meant that the experts thought the value of the mitigation potential

26. See http://nicholasinstitute.duke.edu/ecosystem/t-agg.

was within 20% of the estimate in the literature. Low confidence meant that the estimated GHG mitigation potential was an educated guess, without directional certainty. Voting on evidence was similar: *medium* evidence indicated that sufficient data were available to support the conclusion, even if they didn't cover all regions.

Of the 28 activities surveyed, 13 were accorded medium or high confidence (see Table 3), which means they likely will result in a net reduction in greenhouse gases (positive mitigation outcome). The experts indicated that the available evidence supports that conclusion for seven activities: using no-till, including perennial crops in rotation, switching from an annual to a perennial crop, setting cropland aside, adjusting rice water management, developing rice varieties, converting cropland to pasture, and improving grazing management on rangeland. Despite little supporting evidence, experts expressed confidence in a positive mitigation outcome for four activities: introducing short-rotation woody crops, managing or setting aside farmed histosols, improving grazing management on pasture, and managing species composition on grazing land. The experts had a clear understanding of the mechanism behind the GHG flux impacts of these activities. They had low confidence in the GHG mitigation potential of conservation tillage and application of organic materials. They concluded that the definition of conservation tillage has been too variable for broad-sweeping application. In addition, they noted that in many cases data on this activity have not been sufficiently segregated from data on no-till, making determination of the soil C impact from conservation tillage alone unclear. Experts indicated that application of organic materials has many unresolved life-cycle GHG issues, because such application generally means that another piece of land is no longer receiving the materials. All remaining activities with low levels of certainty were also associated with low levels of supporting evidence.

In many cases, the experts indicated lower confidence than anticipated, given existing research. In these cases, the experts suggested that specific data gaps needed to be filled to increase certainty. For example, rotational grazing on pasture is expected to increase soil C as a result of productivity, but a lack of data in U.S. pasture systems resulted in low confidence. Similarly, high variability in N_2O flux and measurement challenges led the experts to conclude that more data were needed to ensure that N management techniques would consistently reduce N_2O emissions, especially in lesser-studied regions. Thus, a valuable output of the survey was to identify areas in which understanding is weakest and research is most justified. Appendix A discusses these research and data gaps.

This assessment combined the survey findings with the research coverage and mitigation potential summarized from the scientific literature to place the assessed agricultural practices into categories representing a hierarchy of viability. Table 3 reflects these categories as well as special categories for activities with few field data and activities posing life-cycle GHG concerns.

Table 3. Assessed land management activities arranged according to mitigation potential and research coverage (highest to lowest)

Activity	Target GHG	Estimates used in calculations[a]	Regional coverage of data[b]	Scientific certainty[c]
Positive mitigation potential – significant research				
Switch to no-till	Soil C	246	1, 2, 3, 4, 7, 8, 9	Medium
Switch to other conservation tillage	Soil C	65	1, 2, 4, 5, 6, 7, 9	Low
Eliminate summer fallow	Soil C	33	2, 5, 7 (+ Canada)	n/a
Use winter cover crops	Soil C	31	1, 3, 6, 8, 9	Low
Diversify annual crop rotations	Soil C	87	1, 2, 7, 8	Low
Incorporate perennials into crop rotations	Soil C	28	1, 2, 4 (+ Canada)	Medium
Switch to short-rotation woody crops	Soil C	35	1, 2, 3, 9	Medium
Convert cropland to pasture	Soil C	26	2, 7, 9	High
Set aside cropland or plant herbaceous buffers	Soil C	28	1, 2, 3, 7, 9 (+ Canada)	Medium
Reduce fertilizer N application rate by 15%	N_2O	32	1, 2, 3, 7	Low
Adjust rice water management to reduce CH_4	CH_4	26	2[d] (+Asia)	Medium
Positive mitigation potential – moderate research				
Replace annuals with perennial crops	Soil C	17	2, 3, 9 (+ Canada)	Medium
Restore wetlands	Soil C	(70)[e]	2 (+ Canada)	Low
Manage species composition on grazing land	Soil C	9	2, 9 (+ Canada & Australia)	Medium
Switch fertilizer N source from ammonium-based to urea	N_2O	15	3, 8 (+ Canada)	Low
Switch to slow-release fertilizer N source	N_2O	14	3, 7 (+ Japan)	Low

Activity	Target GHG	Estimates used in calculations[a]	Regional coverage of data[b]	Scientific certainty[c]
Change fertilizer N placement	N_2O	31	7 (+ Canada & Europe)	Low
Change fertilizer N application timing	N_2O	33	Canada	Low
Use nitrification inhibitors	N_2O	26	1, 7 (+ International)	Low
Plant rice cultivars that produce less CH_4	CH_4	19	2 (+ Asia)	Medium
Likely positive, but significant data gaps				
Increase cropping intensity	Soil C	no data	n/a	n/a
Establish agroforestry on cropland (windbreaks, buffers, etc.)	Soil C	3/3/0	1 (+ U.S. general)[f]	Low
Improve irrigation management (e.g., drip)	N_2O	4/1/0	Canada and Asia	Low
Improve manure management to reduce N_2O	N_2O	1/3/0	Canada (+ U.S. general)	n/a
Manage farmed histosols	Soil C	2/5/0	1 (+ Europe)	Medium
Set aside histosol cropland	Soil C	3/10/0	6, 9 (+ U.S. general, Canada & Europe)	Medium
Introduce rotational grazing on pasture	Soil C	4/1/1	9 (+ Canada)	Low
Establish agroforestry on grazing land	Soil C	1/3/0	5 (+ U.S. general)	n/a
Significant potential but life-cycle effects uncertain				
Apply biochar to cropland	Soil C	0/5/0	U.S. general and U.K.	No vote[g]
Uncertainty due to lack of data or high variability				
Drain agricultural land in humid areas	N_2O	no data	n/a	n/a
Improve grazing management on rangeland	Soil C	10/3/0	2, 7 (+ U.S. general & Canada)	Medium
Improve grazing management on pasture	Soil C	5/1/0	9 (+ U.S. general & Canada)	Medium
Introduce rotational grazing on rangeland	Soil C	3/0/0	2, 7	n/a
Improve N use efficiency of fertilizer and manure on grazing land	N_2O	no data	n/a	n/a
Fire management on grazing land	Soil C	no data	n/a	n/a
Life-cycle GHG effects/concerns				
Apply organic material (e.g., manure)	Soil C	28/1/2	1, 2, 4, 5, 6, 8, 9 (+U.S. general)	Low
Convert dry land to irrigated	Soil C	11/2/0	2, 7 (+U.S. general & global)	n/a
Fertilize grazing land	Soil C	7/2/1	2, 7, 9 (+U.S. general, Canada & global)	n/a
Irrigate grazing land	Soil C	8/1/0	7 (+Australia & New Zealand)	n/a
Reduce rice area	CH_4	n/a*	n/a	No vote[h]
Low or negative GHG mitigation for target GHG				
Reduce chemical use (other than N)	Upstream emissions	n/a*	n/a	n/a
Set aside grazing land	Soil C	30	2, 5, 7, 9 (+ Canada)	n/a

* National emissions estimates divided by cropland area yielded the GHG mitigation potential per unit area.

a. For the first two groups (and "set aside grazing land"), average national GHG mitigation potential was calculated from field comparisons only (mostly side-by-side), which is the single number in this column. The remaining activities had fewer field observations, so expert and model estimates were also used to determine the range of mitigation potential; therefore the three values in this column are, respectively, the number of "field comparisons/expert estimates/model estimates."

b. Regions are as follows (also see Figure 2): 1–Corn Belt, 2–Great Plains, 3–Lake States, 4–Northeast, 5–Pacific Northwest, 6–Pacific Southwest, 7–Rocky Mountains, 8–South Central, and 9–Southeast.

c. The scientific certainty results indicate the average confidence rating expressed by experts. "n/a" means the activity was not included in the survey.

d. The U.S. rice water management and cultivar comparison research was conducted in Texas, of which the entire state is included in the "Great Plains" region for this assessment. This broad regional division is not perfect, however, since rice-growing regions in Texas more closely resemble the South Central region to the east.

e. The 70 comparisons were not side-by-side experiments with randomly assigned treatments. Instead, the research groups compared restored wetlands with currently cropped land or undrained, virgin wetlands. Because data from side-by-side comparisons are unavailable, wetland restoration was placed in the "moderate" research coverage category.

f. For activities with expert or model estimates, "U.S. general" is used to indicate estimates that applied broadly to the whole country.

g. Experts determined that the available information on biochar was insufficient for a vote.

h. Experts determined that given its production implications, reduced rice acreage was not a viable activity, and therefore they did not vote on it.

Roadmap for agricultural mitigation

This assessment identified 28 agricultural land management activities likely to be beneficial for GHG mitigation (Figure 3). Five have relatively high mitigation potential due to land use changes and are applicable in only some regions (Figure 4). Fifteen tend to have lower mitigation potential, do not shift land use, and are applicable in many U.S. regions (Figure 5). The remaining eight have significant data gaps and need additional research. These activities include increased cropping intensity, agroforestry, histosol management, and rotational grazing for soil C sequestration or conservation, as well as irrigation improvements and improved manure application for N_2O emission reduction. Rotational grazing on pasture lands is particularly interesting. While the C sequestration potential from this practice seems positive, its broader impact on the efficiency of livestock production and the potential for broader mitigation effects is even more promising.

For the fourteen remaining activities, mitigation potential was uncertain, low, or negative. Six of these activities may deserve additional attention as they have been little studied or studies have yielded variable results. Seven of these activities have low or negative net GHG mitigation potential. The final activity, biochar application, may have significant potential, but research on the magnitude of this potential and on life-cycle implications is needed.

Figure 3. Mitigation potential of agricultural management practices included in this report

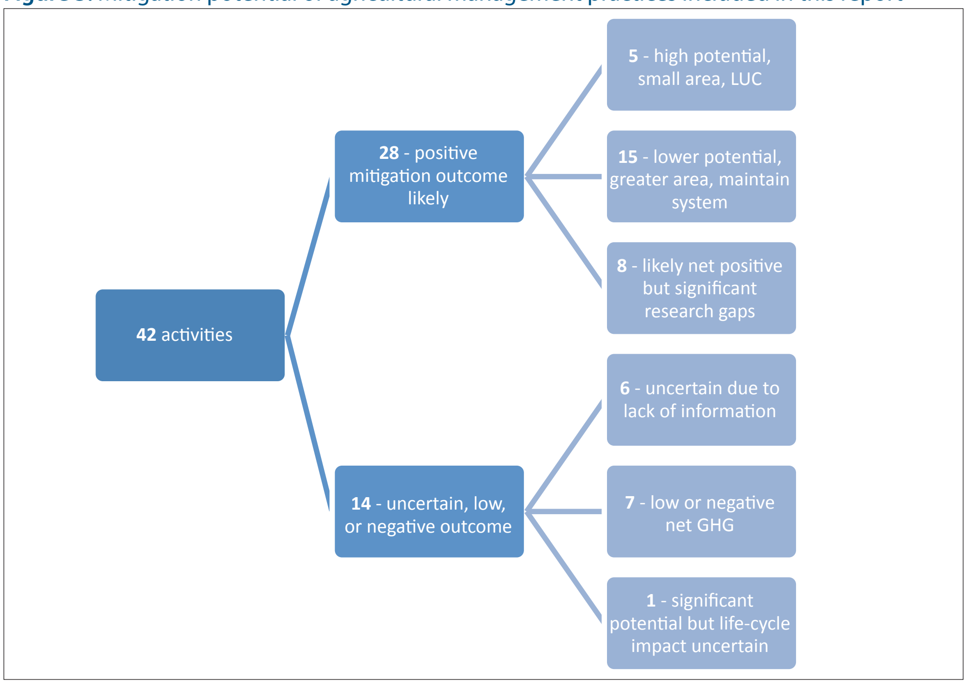


Table 4 reports national mean estimates for changes in soil C, emissions of N_2O and CH_4, and upstream and process emissions for the 20 activities with sufficient research available to calculate means. These activities are divided into two groups based on the amount of available research and related scientific confidence. Eleven activities have significant research (more than 25 consistent field observations) and the rest have moderate supporting research (between 9 and 24 observations) with some critical research gaps for the target GHG. This assessment presents these numbers to provide a side-by-side comparison, but the estimates are not suitable for use as emissions factors without further assessment and customization by region, crop type, and other factors during program or project development. Table 4 also reports an estimate of the *maximum* U.S. area to which each activity is applicable. Less land is likely to be converted to the specified management activity, given a limited land base, competing land uses (many activities possible on same land), transaction costs, and social barriers (see the section "Economic Potential for GHG Mitigation" for more detail). Hence, potential per hectare is not multiplied by maximum area to calculate the maximum national potential for each activity. The reported range for the target GHG indicates the 10th and 90th percentiles (thus containing 80% of field comparisons) of available data, illustrating the range of values that could be expected across regions, soil types, and

Table 4. U.S. agricultural land management activities with positive GHG mitigation potential and significant to moderate research coverage

Activity	Soil carbon	N_2O emissions	CH_4 emissions	Process & upstream emissions	National total	Max area	Comments
	mean (range); t CO_2e ha^{-1} yr^{-1}					Mha	
Significant research							
Switch to no-till	1.27* (-0.43–3.62)	0.11	0.01	0.12	1.50 (-0.20–3.85)	94	N_2O emissions, which are well studied, depend on soil and climate.
Switch to other conservation tillage	0.38 (-0.51–1.36)	0.18	0.00	0.08	0.63 (-0.25–1.61)	72	Soil C change varies by region.
Eliminate summer fallow†	0.60* (-0.22–1.20)	-0.03	0.00	-0.12	0.44 (-0.38–1.05)	20	Process and upstream emissions depend on N fertilizer rates for crop replacing fallow.
Use winter cover crops	1.34 (-0.07–3.22)	0.20	no data	0.46	2.00 (0.59–3.89)	66	This activity can reduce need for fertilizer N, but it may require timing changes for the main crop.
Diversify annual crop rotations	0.00* (-1.69–1.66)	0.17	0.00	0.00	0.17 (-1.52–1.83)	46	Net primary productivity is the key factor.
Incorporate perennials into crop rotations	0.52 (-0.01–1.20)	0.03	0.00	0.17	0.71 (0.19–1.39)	56	
Switch to short-rotation woody crops‡	2.51 (-7.34–13.26)	0.76	no data	0.65	3.92 (-5.93–14.67)	40	Upstream emissions do not include end use. Negative soil C results are limited to studies of less than six years.
Convert cropland to pasture‡	2.39 (0.40–4.18)	0.46	-0.25	0.45	3.06 (1.07–4.85)	no data	The total area is uncertain.
Set aside cropland or plant herbaceous buffers‡	1.98 (-0.37–5.07)	0.84	0.00	0.74	3.57 (1.22–6.66)	17	This activity excludes histosols. Differences in types of land for restoration result in a wide range of mitigation potential.
Reduce fertilizer N application rate by 15%‡	no data	0.28 (0.03–0.82)	no data	0.06	0.33 (0.08–0.88)	68	
Adjust rice water management to reduce CH_4	-0.04	-0.79	1.97 (0.08–5.31)	no data	1.14 (-0.75–4.48)	1.3	U.S. studies are augmented with international data.
Moderate research							
Replace annuals with perennial crops‡	0.67 (-0.86–2.00)	0.24	0.00	0.52	1.43 (-0.10–2.76)	13	
Restore wetlands‡	6.52 (-0.96–9.89)	0.00	-3.33	0.74	3.94 (-3.54–7.31)	3.8	
Manage species composition on grazing land†	1.46 (0.18–3.12)	-0.86	-0.03	no data	0.57 (-0.71–2.23)	80	Emissions of N_2O and CH_4 are based on one study.
Switch fertilizer N source from ammonium-based to urea	no data	0.49 (0.02–1.04)	no data	no data	0.49 (0.02–1.04)	37	
Switch to slow-release fertilizer N source	no data	0.11 (0.04–0.20)	no data	0.06	0.17 (0.09–0.26)	93	Assuming less fertilizer N is used, upstream emissions will be reduced.
Change fertilizer N placement	no data	0.33 (0.0–0.91)	no data	no data	0.33 (0.00–0.91)	63	
Change fertilizer N application timing	no data	0.16 (-0.01–0.50)	no data	no data	0.16 (-0.01–0.50)	53	
Use nitrification inhibitors	no data	0.64 (0.03–1.57)	no data	no data	0.64 (0.03–1.57)	92	
Plant rice cultivars that produce less CH_4	no data	0.00	0.97 (0.06–1.87)	0.00	0.97 (0.06–1.87)	1.3	U.S. studies are augmented with international data.

Note: The mean for the target gas reflects the average mitigation estimate from field comparisons. The mean for other GHG classes relies on field comparisons as well as expert and model estimates. The range for the target gas indicates the 10th and 90th percentiles of the data (80% of observations within the range). This range is used for the national total (net GHG balance).

* These means are regionally weighted. All others are the mean of available observations, given that regionally representative data were insufficient.

† These activities may increase agricultural productivity in the project/program area and thus result in positive leakage.

‡ These activities may decrease productivity in the project/program area and thus result in negative leakage (production shifts elsewhere).

other characteristics. Because most variability is expected to originate from the target greenhouse gas,[27] and variability for other GHG categories is difficult to determine due to lack of data, the reported range for the net GHG impact is based on the variability of the target gas. Depending on the activity, this variability may be a result of regional, soil, climate, or crop differences, or it may be related to uncertainty in existing measurements or other determinations of soil carbon or GHG flux.

For activities that maintain current cropping systems, tillage changes and use of winter cover crops have the most potential and the largest applicable area. Nitrogen management, through which multiple activities could be integrated for additional emissions reductions, is also promising, at least in moist regions with relatively high background emission rates. Rice water management and variety development also have significant mitigation potential, although their applicable area is about 70 times smaller than that for tillage changes. The land-use change practices with the greatest GHG mitigation potential are use of short-rotation woody crops, the setting aside of cropland, conversion of cropland to pasture, and restoration of wetlands.

Figure 4. Mitigation potential in terms of net greenhouse gases per hectare per year for practices that (1) result in land use changes or significant crop mixture changes; (2) are backed by significant research, about which scientific certainty is moderate to high; and (3) are likely to result in a net GHG reduction

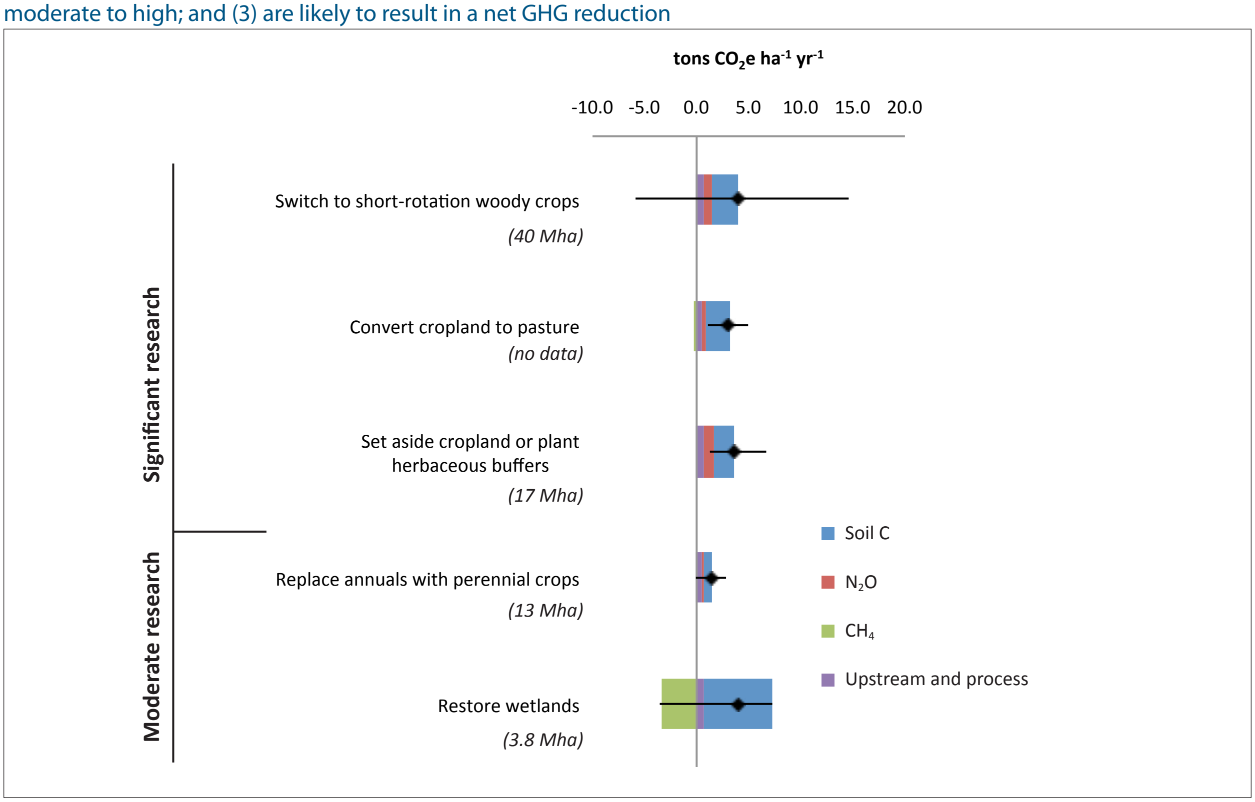
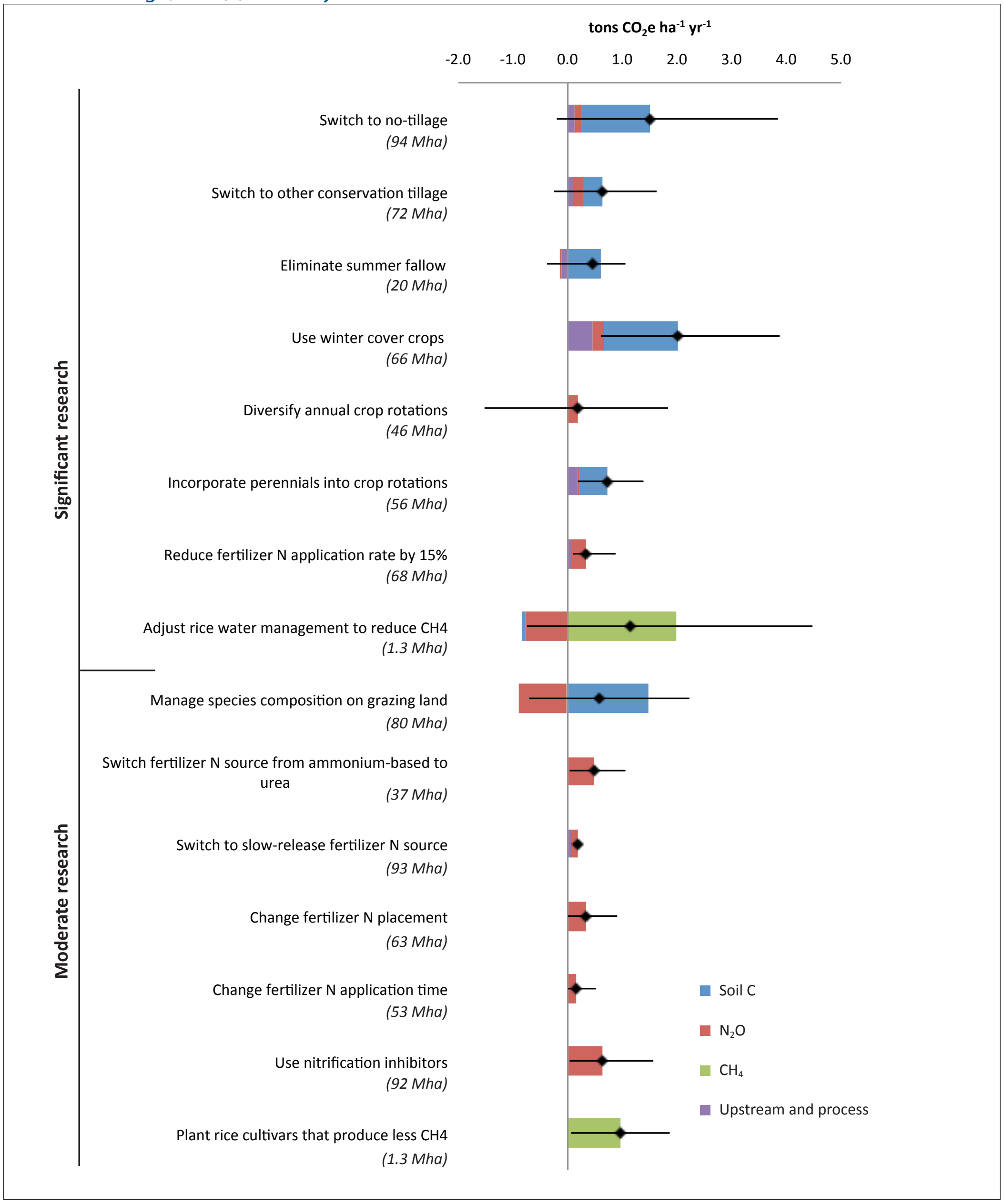


Table 5 presents the remaining activities, for which data were insufficient for estimating a mean mitigation potential. We report a range for the target GHG, which constitutes the minimum and maximum observed values or estimates from the supporting literature. These activities are grouped into categories, where mitigation potential is (1) likely positive but significant data gaps remain, (2) likely significant but uncertainty in life-cycle effects remains, (3) uncertain due to lack of data, (4) uncertain due to life-cycle GHG concerns, and (5) very low or negative.

27. The flux change of nontarget greenhouse gases tends to be much less than that of the target gas, and any variability that significantly affects net GHG mitigation would most likely result in removal of problematic regions or cropping systems from program consideration.

Figure 5. Mitigation potential in terms of net greenhouse gases per hectare per year for practices that (1) do not result in land use changes or significant crop mixture changes; (2) are backed by significant research, about which scientific certainty is moderate to high; and (3) are likely to result in a net GHG reduction



Activities in the first two categories are likely candidates for focused research efforts in the near term, since initial exploration suggests some significant environmental benefits. Despite life-cycle concerns, biochar may have very large potential for C storage and perhaps also for additional N_2O emission reductions. But uncertainty in biochar's life-cycle impacts remains given the significant variability in life-cycle analyses within the scientific literature. For many others, lack of data prevents even tentative conclusions. If resources are available, grazing management (reduced intensity) on rangelands may be worth further examination, since experts have suggested significant soil C storage capability, even though our review suggests otherwise. All activities for which there are life-cycle GHG concerns, except biochar applications, offer little net GHG mitigation benefit. For example, fuel and N_2O emissions prompted by irrigation may offset any soil C gains, and manure application in many cases just moves C from one location to another. Thus these may not warrant further assessment for GHG mitigation.

Table 5. GHG mitigation potential for U.S. agricultural land management activities with significant research gaps, life-cycle GHG concerns, and low or negative GHG mitigation implications

Activity	Target	GHG benefits mean (range)	Max area	Comments
		t CO_2e ha^{-1} yr^{-1}	Mha	
Likely positive, but significant data gaps				
Increase cropping intensity*	soil C	no data	unknown	Using winter cover crops and eliminating summer fallow are treated separately as two unique examples of increasing intensity. Data on other options are not available.
Establish agroforestry on cropland (windbreaks, buffers, etc.)†	soil C	0.84–6.87	21	Total potential is for area in trees alone, and does not include aboveground C storage.
Improve irrigation management (e.g., drip)	N_2O	0.14–0.94	20	Irrigation improvements may also significantly reduce process and upstream emissions if total irrigation water is reduced.
Improve manure management to reduce N_2O	N_2O	0.37–1.22	12	This activity includes applying manure to dry areas rather than wet ones, using solid instead of liquid manure, and reducing application rates.
Manage farmed histosols	soil C	0.00–15.03	0.8	Total area farmed is highly variable in the literature.
Set aside histosol cropland†	soil C	2.20–73.33	0.8	Total area farmed is highly variable in the literature.
Introduce rotational grazing on pasture*	soil C	-0.05–2.90	42	With increased forage production per unit area, this activity can have positive leakage effects. However, it may also increase enteric emissions because more cattle can graze on a given area.
Establish agroforestry on grazing land	soil C	0.47–3.63	70	
Significant potential but life-cycle effects uncertain				
Apply biochar to cropland	soil C	0.63–19.57	124	Biochar application raises concerns about effects on the source location and biochar production raises concerns about GHG balance. Recent research suggests the application has the potential to reduce N_2O emissions.
Uncertainty due to lack of data or high variability				
Drain agricultural land in humid areas	N_2O	no data	unknown	
Improve grazing management on rangeland	soil C	uncertain (see text)	227	Expert assessment indicates positive potential for soil C increase, especially on overgrazed land. Research comparisons demonstrate that soil C loss is common with reduced grazing pressure (likely on well-managed rangeland).
Improve grazing management on pasture	soil C	-2.97–4.76	48	
Introduce rotational grazing on rangeland	soil C	-5.27–1.90	unknown	
Improve N use efficiency of fertilizer and manure on grazing land	N_2O	no data	unknown	
Introduce fire management on grazing land	soil C	no data	unknown	
Life-cycle GHG effects/concerns				
Apply organic material (e.g., manure)	soil C	0.18–5.10	8.5	This activity raises concerns about effects on the source location. Improved manure nutrient distribution might reduce N fertilizer needs (thus lowering upstream emissions).
Convert dry land to irrigated land*	soil C	-0.55–2.82	n/a‡	GHG costs of irrigation equipment and pumping may negate soil C gains. N_2O emissions are also higher with irrigated land.
Fertilize grazing land*	soil C	0.37–5.86	n/a	GHG emissions from fertilizer production may negate soil C gains.
Irrigate grazing land*	soil C	0.00–1.83	n/a	GHG costs of irrigation equipment and pumping may negate soil C gains. N_2O emissions are also higher with irrigated land.
Reduce rice area†	CH_4	2.32–10.26	1.3	Impacts depend on subsequent land use and conditions for displaced rice production.

Activity	Target	GHG benefits mean (range)	Max area	Comments
		t CO_2e ha^{-1} yr^{-1}	Mha	
Low or negative GHG mitigation for target GHG				
Reduce chemical use (other than N)	upstream/ process emissions	0.03–0.06	122	
Set aside grazing land†	soil C	-2.76–0.98§	unknown	Soil C response data are highly variable.

Note: The range indicates the minimum and maximum values for the target gas from field comparisons, expert estimates, and model estimates, as available.
* These activities may increase agricultural productivity in the project/program area and thus result in positive leakage.
† These activities may decrease productivity in the project/program area and thus result in negative leakage (production shifts elsewhere).
‡ The total area is not estimated for activities for which the net GHG effect is negative.
§ The 80% range of 28 field comparisons is presented. The mean is -0.51 t CO_2e $acre^{-1}$ yr^{-1}.

Non-GHG Benefits and Tradeoffs

Many of the agricultural management practices reviewed in this report are used by producers because they can reduce costs or improve yields and productivity. Some co-benefits of GHG mitigation are less direct but sufficiently valuable that offsite beneficiaries are willing to pay for them through public incentive programs or private programs like voluntary markets. On the other hand, some of the reviewed practices can have negative productivity or environmental outcomes or side effects. This review helps identify the potential broader impacts on producers and the environment of new incentives that would encourage adoption of agricultural management practices for their GHG benefits.

Agricultural land management activities that have significant GHG mitigation potential can provide a wide range of benefits, including improved farm production and resilience, by enhancing soil quality and soil biological activity and by encouraging beneficial insects (Table 6). Some activities can also reduce risks of damages from flood and other major weather events, which may be more frequent due to climate change. For example, herbaceous buffers can reduce water runoff and sediment runoff by an average of 45% and 76%, respectively (Arora et al. 2010); this conservation of water and soil serves to maintain long-term productivity. Practices that increase soil C improve soil quality, because higher levels of organic C in the soil improve soil tilth and fertility and reduce compaction. Higher SOC and increased residue cover have also been shown to decrease soil erosion (Ernst and Siri-Prieto 2009; Govaerts et al. 2009; Li et al. 2007), which is expected to become more problematic with erratic rainfall and increased flooding. Water-holding capacity and water availability to plants increase with the elimination of tillage and other SOC-increasing management (Bosch et al. 2005; Franzluebbers 2002), reducing yield risk during drought years. Crop diversification (depending on crop choice) can increase the abundance of crop pest predators (Sunderland and Samu 2000), thus reducing the need for pesticides and providing an economic benefit for the producer and a broader environmental benefit.

Many practices reviewed in this report also provide broader benefits to the environment (Table 6). For example, conservation tillage and no-till reduce soil erosion and help decrease sediment and nutrient loading in streams, rivers, and ground water. Conservation tillage also improves habitat for ground-dwelling birds and other wildlife. Lowering fertilizer N application rates or increasing N use efficiency for N_2O emission control reduces N releases into waterways (as nitrate) and air (as ammonia), with significant impacts on water quality and coastal dead zones (see Box 3). Cover cropping reduces erosion, improves water quality in runoff, and generates other positive environmental services (Dabney et al. 2001). By providing vegetative cover over a longer period, and often with deep roots, plants can consume available soil nitrogen and reduce N leaching into waterways.

Negative environmental outcomes (tradeoffs) have also been noted (Table 6). One problem associated with no-till is the increased use of chemical herbicides for weed control, which has been accompanied by development of herbicide-resistant weed populations (see Box 10) and related environmental quality concerns. Research on and development of alternative weed control can modify this negative effect. For example, banded herbicide applications can maintain crop yields with an approximate 60% reduction in total herbicides (Eadie et al. 1992). Another risk, which can likely be managed, is that the additional nitrogen fixed by legume cover crops will increase nitrate leaching rates (Pimentel et al. 2005).

Table 6. Potential co-benefits and tradeoffs of agricultural GHG mitigation practices

GHG Mitigation Practice	Biodiversity	Water conservation	Water quality	Air quality	Soil quality	Food security
Switch to no-till or other conservation tillage	Improved habitat for ground-nesting birds and other animals Increased herbicide use	Reduced irrigation need	Reduced sedimentation Increased herbicide use and potential for runoff	Reduced emissions from tractor use, reduced respirable and total dust	Reduced erosion, increased SOM	Increased reliance on GM seeds and homogenization
Eliminate summer fallow	Improved habitat for ground-nesting birds and other animals	Increased water use	Reduced nitrate leaching, but increased fertilizer N needs		Increased SOM	
Add winter cover crop	Increased biodiversity	Improved soil water holding capacity Increased water use	Reduced nitrate leaching	Increased emissions from tractor use	Increased SOM	Increased yield through improved fertility and reduced insect and pathogen damage
Diversify annual crop rotations, increase intensity	Increased biodiversity (native and crop species), possibly detrimental to wildlife (e.g., bird diversity)	Reduced or increased water use	Disease-suppressive soils reduce pesticide and herbicide use or increased inputs and erosion	Increased emissions from tractor use	Improved soil quality	Increased yields, improved disease resistance
Include or substitute perennial crops in rotations, SRWCs	Increased biodiversity	Decreased water use and increased soil water holding capacity or possible increase in water use	Potentially decreased sedimentation and herbicide/ pesticide use	Reduced emissions from tractor use	Improved soil quality, reduced erosion, increased SOM	Decreased overall production of main grain crops
SRWCs, agroforestry, herbaceous buffers	Increased biodiversity	Flood control	Reduced sedimentation and improved filtration	Reduced emissions from tractor use	Reduced erosion, increased SOM	Land taken out of production
Irrigation improvements (e.g., drip, supplemental)		Improved water use efficiency	Reduced sedimentation and nutrient runoff			Increased yields
Application of organic materials, biochar	Increased soil microbiota	Improved soil water-holding capacity	Reduced or increased runoff	Possible reduction of trace gases, increased or reduced storage and handling of emissions	Improved soil quality (e.g., structure), increased SOM	Increased yields
Improved fertilizer N management	Decreased crop foliar disease, improved aquatic habitat		Reduced nitrogen runoff	Reduced trace gas and ammonia emissions		Potential yield tradeoff
Reduce chemical use (other than N)	Reduced chemical impact on biodiversity (e.g., insects and birds)		Reduced chemical runoff	Reduced upstream emissions	Improved soil quality (e.g., less chemical residue)	Potential yield tradeoff
Improved grazing management, species composition	Increased or decreased biodiversity		Improved nutrient management and reduced or increased runoff		Improved soil quality	Improved or reduced grazing intensity
Land use change (e.g., cropland to pasture, wetlands restoration)	Increased biodiversity, restored habitat	Improved flood control	Improved water filtration		Improved soil quality, increased SOM	Displaced cropland

Green = benefit; red = tradeoff.

Sources: Asbjornsen et al.; Baker et al. 2005; Benton et al. 2003; Brookes and Barfoot 2010; Chamberlain and Siriwardena 2000; Chase and Duffy 1991; Dabney et al. 2001; Delgado et al. 2001; Dong et al. 2003; Firbank et al. 2008; Foley et al. 2005; Glover et al. 2009; Hallam et al. 2001; Hansen et al. 2001; Hargrove 1991; Henderson et al. 2009; Johnson et al. 2009; Krebs et al. 1999; Laird et al. 2008; Laird et al. 2010; Lal 2004b; Laub and Luna 1992; Machado et al. 2006; Mannering and Fenster 1983; McLaughlin and Mineau 1995; Novak et al. 2009; Oehl et al. 2003; Piñeiro et al. 2009; Rands 1986; Schulte et al. 2006; Shipitalo and Owens 2006; Smith et al. 2007b; Snapp et al. 2005; Sperow et al. 2003; Stetler and Saxton 1996; Stivers and Shennan 1991; Sunderland and Samu 2000; Teasdale et al. 2000; Tonitto et al. 2006 1115; Zentner et al. 2002

Economic Potential for GHG Mitigation

Modeling the economic potential of agricultural mitigation strategies

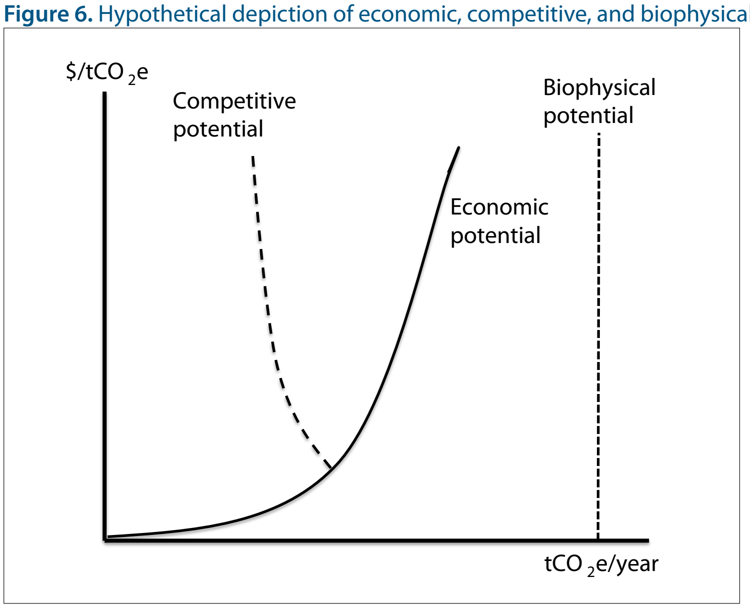
Estimates of the technical or biophysical GHG mitigation potential of various agricultural activities do not factor in economic forces that can influence land management decisions. Smith et al. (2007b) show that achievable global mitigation could be less than 30% of total biophysical potential at low GHG price incentives (<\$20 t CO_2e^{-1}), but higher incentives could boost agricultural mitigation significantly (upward of 74% of biophysical potential at ~\$100 t CO_2e^{-1}). When deciding to undertake a mitigation activity, landowners must consider direct costs (such as the investment and operating costs needed to change production practices) and possible indirect opportunity costs (such as forgone yield). When faced with multiple abatement options, landowners also must compare the expected benefits, costs, and risks (e.g., yield risks) of each option to those of the other options and to the expected returns under conventional cropping systems.

Distinguishing among full biophysical or technical potential, economic potential, and competitive economic potential (Figure 6) is critical. The technical or biophysical potential (BP) of agricultural GHG mitigation generally exceeds the economic potential (Bangsund and Leistritz 2008). Opportunity costs of adoption reach a threshold that often falls short of the biophysical potential for mitigation. Nonprice limitations, such as social, institutional, political, and educational factors, can also affect adoption rates (Smith et al. 2007b).

Economic potential refers to the costs of implementing a mitigation activity on the margin and to the increasing monetary incentive necessary to supply additional GHG mitigation. Economic potential can be thought of as the supply curve of a particular mitigation strategy at some spatial scale. The economic potential of an activity is ultimately less than the total biophysical potential, which is calculated by aggregating mitigation potential across an entire land base without accounting for the costs of wide-scale adoption. Competitive economic potential reflects the reality that mitigation activities will compete not only with other mitigation strategies, but also with the demands for food, bioenergy, or both within an economic system. As adoption of the mitigation activity increases, it can change market prices, thereby changing economic potential relative to competing activities.

Figure 6 provides a conceptual diagram of how biophysical, economic, and competitive mitigation potential can vary for one activity. It depicts mitigation potential in tonnes of CO_2e per year at a given CO_2 price. Total biophysical potential (denoted by the vertical line on the right-hand side of the figure) represents the maximum obtainable mitigation from the terrestrial system, with no regard for the costs of adoption or other socioeconomic barriers that could prevent such a shift. Economic potential follows an upward-sloping supply schedule, indicating higher levels of abatement for greater CO_2 price incentives.

Figure 6. Hypothetical depiction of economic, competitive, and biophysical potential of a mitigation activity



Adapted from Murray et al. (2005) and McCarl and Schneider (2001).

Notice that competitive potential exhibits a "backward-bending" shape, which is unorthodox for a supply curve. The implication is that at some CO_2 price threshold, other abatement options are much more attractive at higher prices (due to higher GHG returns). This threshold makes adoption of the original abatement activity less likely, and it can even reduce the activity's net GHG offset supply at higher CO_2 prices. Previous studies have shown that the competitive potential of soil carbon sequestration through tillage change could diminish because higher CO_2 price incentives could stimulate afforestation or use of bioenergy feedstocks as a source of abatement in lieu of enhanced soil C stocks, thereby shifting land away from conventional agriculture (McCarl and Schneider 2001; Murray et al. 2005).

Economic studies of agricultural GHG mitigation

Many economic tools have been used to estimate the GHG mitigation potential of agricultural land management activities. This report draws from select studies at different spatial scales to estimate the economic potential, competitive potential, or both of agricultural mitigation.[28] In this way, it illustrates how different modeling techniques can yield divergent estimates of mitigation potential and how economic constraints that affect abatement potential vary by region and spatial scale. In the context of competitive potential, the report shows how changing market conditions, opportunity costs, and land use competition can influence the GHG abatement portfolio in agriculture.

Box 3. Water quality as a co-benefit of GHG mitigation

Excess nutrient release into rivers and oceans creates dead zones, a problem that climate mitigation activities could address. Each summer, the dead zone associated with the Mississippi River covers as much as 8,000 square miles along the U.S. Gulf Coast. Nitrogen and phosphorus runoff from agriculture causes eutrophication, leading to low-oxygen concentrations along the coasts of Louisiana, Mississippi, and eastern Texas. These hypoxic zones kill fish, shrimp, and other organisms in a region that supplies more than half of all U.S.-harvested oysters and shrimp. Those zones associated with coastal eutrophication are found periodically throughout the Gulf of Mexico and the Atlantic coast, most notably in the Chesapeake Bay (Diaz and Rosenberg 2008).

Many of the same practices that produce GHG benefits will also help control the severity and extent of the dead zone associated with the Mississippi River and other rivers. A broad opportunity for co-benefits is evident in approaches that combine GHG mitigation with water quality improvements. For example, increases in conservation tillage, decreases in fertilizer use, and consolidation of animal feeding operations in Ohio and Indiana watersheds have reduced reactive nitrogen and phosphorus compounds in rivers there (Renwick et al. 2008).

Local water quality improvements resulting from GHG mitigation activities can also provide economic benefits, such as cost savings for water treatment and other cleanup. In North Carolina, Elsin et al. (2010) recently estimated that the mean net present value of savings for the entire Neuse Basin ranged from $2.7 million to $16.6 million for a 30% improvement in water quality over a 30-year period. Savings associated with improved water quality are also the driving force for changes in agricultural practices in Europe.

Cooperative agreements provide a voluntary tool for improving water delivery and quality as well as other ecosystem services. The Water Framework Directive (WFD) and the reform of the Common Agricultural Policy (CAP) is one recent attempt to improve the status of water in the European Union by 2015. Under this framework, municipalities and companies can provide payments or other financial incentives to help agricultural producers change production methods and modernize farm equipment (Heinz 2008). For instance, the German town of Viersen in the Bundesland North Rhine-Westphalia saved approximately €250 000 per year through a cooperative agreement that helped farmers reduce nitrate concentrations in runoff and ground water, allowing the town to avoid construction of additional water treatment facilities. A similar agreement in Holsterhausen/Üfter Mark had a water company paying farmers to tailor the amounts and timing of pesticide and fertilizer use to the needs of the plants. The farmers used technically advanced pesticide sprayers, intercropping, and better applications of semi-liquid manure. Consequently, the concentration of nitrate in drinking water decreased from 13 mg/l on average to 7 mg/l (Heinz 2008). Hundreds of such cooperative agreements are now in place in Germany alone.

Economic potential and competitive potential are typically expressed by marginal abatement costs (or in some cases, average abatement costs). Marginal abatement cost (MAC) curves are essentially the supply curve for GHG mitigation, forming a direct link between the GHG incentive and emissions reduction, sequestration, or both that can be supplied at that price. MAC curves can represent mitigation achieved from individual activities, or they can be expressed as the total abatement potential of several activities combined. MAC curves can be denoted at farm scale, by region, or nationally, facilitating assessment of project mitigation potential and offset market participation under different crediting scenarios and incentive schemes. Because adoption of new technology or activities tends to become less costly once equipment and knowledge are more common, MAC curves can change from year to year, with slopes of the curves declining over time.

28. For a more comprehensive review of economic studies on GHG mitigation potential in agriculture, see Bangsund et al. (2008) or Manley and van Kooten (2005).

Economic sector and equilibrium models

Sectoral (partial equilibrium) economic models estimate economic potential and competitive potential in the context of an integrated market system in which management activities implemented for GHG mitigation not only compete against one another for land and water, but also with conventional commodity and bioenergy demands. To allow for estimation of marginal abatement costs, structural equilibrium models typically account for

- a finite land resource base,
- competition among natural resources,
- endogenous commodity prices (such that production shifts affect market equilibrium prices),
- explicit production function relationships between inputs and productivity,
- production options that reduce emissions/sequester additional carbon (including cost components that recognize the increased/decreased expenditures associated with the activity), and
- detailed GHG accounting.

National economic estimates of GHG mitigation potential reveal an important role for agriculture, particularly at lower C prices. FASOMGHG is an example of a partial equilibrium model spanning multiple sectors (agriculture and forestry, with links to the energy sector through biofuels and bioelectricity).[29] FASOMGHG simulates land use competition and production practices in the presence of incentives (payments) for GHG emissions reduction or enhanced C sequestration from agriculture and forestry. Its estimates yield MAC curves by various practices for individual GHG accounts (composed of the sum of several GHG fluxes), or total abatement potential for the two sectors. In addition to numerous academic publications, this report used FASOMGHG to estimate mitigation supply across multiple activities, scales, and price levels in an EPA-sponsored report on aggregate national economic mitigation potential for agriculture and forestry (Murray et al. 2005). This study found significant potential for the agricultural sector, including 168 Mt CO_2e yr^{-1} for soil C sequestration from altered tillage practices at an incentive price of \$15 t CO_2e^{-1} (though as previously stated, competitive potential for soil C sequestration decreases with the CO_2 price). The study also found potential for non-CO_2 reductions for livestock and fertilizer N management activities (ranging from 32–110 Mt CO_2e yr^{-1} for prices ranging from \$15–\$50 t CO_2e^{-1}). The bulk of the mitigation potential found in Murray et al. (2005) came from afforestation of crop and grazing lands, forest management, and biofuels.

Baker et al. (2010) also apply FASOMGHG—with different results. Although afforestation and forest management remain the dominant mitigation strategies, their potential for agricultural soil C sequestration is lower (ranging from 4 to 23 MtCO_2e yr^{-1} at \$15–\$50 tCO_2e) than that found by Murray et al. (2005). Part of this difference can be attributed to an evolving baseline. For example, the no-till cropland area for four major crops (corn, cotton, soybeans, and rice) increased at a median rate of 1.5% per year from 2000 to 2007 (Horowitz et al. 2010), making additional mitigation potential more difficult to generate. Furthermore, expanded export demand parameters, reflecting higher demand for U.S. agricultural grain and meat exports, have raised the opportunity costs of any GHG mitigation that lowers productivity. Probably the most important difference between the Baker et al. and Murray et al. studies, however, is that the former reflects the existence of renewable fuels standard (RFS2) legislation. Simulation results reveal that the RFS2 legislation considerably affects the projected land resource base (both in terms of land requirements and production intensity), decreasing opportunities for further GHG mitigation from a GHG payments program.

Other national-based partial equilibrium modeling efforts have produced estimates of agricultural mitigation potential. The USDA's Economic Research Service (Lewandrowski et al. 2004) estimated that at a low incentive value of approximately \$3 t CO_2e^{-1}, changes in land use (e.g., converting cropland to forest or grassland) and cropland management (e.g., introducing conservation tillage or changing crop rotations) could sequester from 1.5–36.7 Mt CO_2e. At a higher incentive price of \$34 t CO_2e^{-1}, the study estimated that these activities could sequester 264–587 Mt CO_2e yr^{-1}. The USDA study showed that producers nationwide would adopt cropland management practices such as conservation tillage at the lowest value of \$3 t CO_2e^{-1}; afforestation adoption would begin at \$7 t CO_2e^{-1}. Afforestation would outcompete cropland to grassland conversion due to higher C sequestration rates on forestland.

A study published by the University of Tennessee and 25 X '25 (using POLYSYS, an agricultural policy simulation model) shows that coupling GHG payments with the renewable fuels standard and regional renewable electricity

29. The FASOMGHG model uses the DAYCent biogeochemical model to generate the GHG impact of various agricultural management practices in the relevant ranges of cropping practices and regions of the United States.

standards could produce 76 $MtCO_2e$ yr^{-1} in GHG benefits (English et al. 2010). Unlike the FASOMGHG studies, the English et al. (2010) analysis accounts for the additional soil carbon sequestered by perennial energy crops such as switchgrass. The policy implication is that landowners could potentially be awarded offset payments by switching to a perennial energy crop that stores additional carbon in lieu of an annual feedstock that competes with food production. This study also showed very little potential for additional afforestation or for shifts in conventional commodity production. Note, however, that POLYSYS does not include a forest land use sector, so the lack of land use transfer from agriculture to forestry is not unexpected.

In general, partial equilibrium modeling can provide national or global estimates of competitive potential in agriculture. Depicting the competition among activities in a comprehensive market setting provides a meaningful policy metric that could deviate from the EP estimates found through econometric and simulation modeling processes. Economic potential is typically estimated at a much finer spatial scale, with improved biophysical parameters, though EP measurements can be isolated in larger sectoral models. In fact, one section of Murray et al. (2005) presents estimates of the economic potential of individual management activities and finds them to be higher than the competitive potential estimates that make up the bulk of the report. In addition, EP estimates can produce confidence intervals around MAC curve parameters. However, such estimates ignore any residual market feedback from the mitigation activity. Economic potential and competitive potential should be considered in conjunction with BP estimates for the most robust assessment of an activity's potential for reducing or offsetting GHG emissions.

Econometric and simulation models

Equilibrium economic models can provide projections of the competitive potential of mitigation alternatives at a high level of spatial aggregation relative to an assumed business-as-usual future. However, they might exclude important biophysical variables—heterogeneity in crop management practices and land quality for particular regions—which can have important implications for the economic costs of GHG abatement. Alternative economic tools, such as simulations based on observed relationships estimated by econometric models, can provide a more detailed estimate of the site-specific marginal abatement costs of individual practices by directly including more precise biophysical parameters and heterogeneity in management choices and land characteristics. Previous studies have applied econometrics, simulation modeling, or both to estimate the economic potential of abatement activities at more spatially refined scales.

Simulations based on econometric models typically depict producer choices through detailed production functions and budgets that depict the economic choices facing producers and the costs of adopting a GHG abatement action. These simulations can be spatially explicit, with detailed representation of land productivity and environmental feedback. Feng et al. (2004a) apply simulation modeling techniques to estimate mitigation potential and the economic effectiveness of policy instruments that incentivize carbon sequestration on productive land and land set asides. This study finds conservation tillage to be more economically viable than retiring productive agricultural lands, at low carbon prices (see also van Kooten et al. 2002). A similar simulation technique was applied by Kurkalova et al. (2004b), who found sequestration rates comparable to those in Feng et al. (2004a). However, the latter study also highlighted some important environmental co-effects of reduced tillage adoption, including reduced nitrogen runoff, water erosion, and wind erosion. One of the distinct advantages of simulation modeling at fine scales is the capacity to include spatially explicit production functions and physical parameters which allows linking of the economic models with biogeochemical process models for estimation of environmental co-effects.

Simulation models have increased in complexity and functionality, offering an ever more comprehensive assessment of agricultural mitigation potential for a greater number of activities. For example, an examination of soil C sequestration from tillage changes by Choi and Sohngen (2010) accounts for two factors—residue management and crop rotation choice—never before explored in economic analyses. The addition of these production characteristics may raise the opportunity costs of carbon sequestration through altered tillage practices, suggesting that previous studies might have underestimated these costs (or overestimated economic mitigation potential).

Unlike simulation modeling, econometric techniques help establish behavioral trends using observed production, land use data, or both. These techniques illustrate how behavior and landowner preferences can influence participation in a GHG mitigation market, whereas the simulations based on those relationships typically focus on the expected benefits and costs of that participation. For example, Kurkalova et al. (2006) show that although the opportunity costs for adopting conservation tillage methods are quite low, landowners typically require a premium payment to participate in such efforts. Their behavior reflects some inherent perceived costs associated with adoption, similar to the response of more than one-quarter of farmers considering afforestation activity in a Canadian survey (van Kooten et al. 2002). Many studies directly combine econometrics with simulation modeling (Antle et al. 2007; Antle et al. 2001; Antle et al. 2003; Lubowski et al. 2006a). Such studies allow for heterogeneity in landowner preferences, production behavior, and sequestration rates (Antle et al. 2003), and they can target adoption rates for a variety of mitigation practices (in addition to conservation tillage, which has dominated the majority of numerical/econometric simulation studies). Lubowski et al. (2006) econometrically estimate a carbon sequestration supply function by isolating determinants of landowner preferences for alternative land uses. Parameters from the econometric estimation are used to simulate land use decisions with carbon price incentives for afforestation and avoided deforestation. The revealed preference approach used in Lubowski et al. (2006) yields higher estimates of carbon sequestration costs than produced by previous sectoral modeling and engineering estimates.

Other examples of coupled econometric and simulation analysis includes a series of papers led by Antle (2007, 2001, 2003). Antle et al. (2001) estimate that the marginal costs of converting cropland to permanent grassland range from \$14 t CO_2e^{-1} to more than \$136 t CO_2e^{-1} and of converting fallow or grassland to cropland ranges from \$3 t CO_2e^{-1} to \$38 t CO_2e^{-1}. This study applied a field-scale model to illustrate how the opportunity costs of these activities vary by region, cropping system, and soil characteristics. Antle et al. (2003) show that mitigation costs can be dramatically affected by policy (or contract) instrument; per-tonne payments increase efficiency up to five times relative to an area-based payment (due to spatial heterogeneity). Antle et al. (2007) estimate carbon sequestration potential in the Central United States for reducing fallow in a dryland grain system. Results from this study show that total economic potential for this activity could reach approximately half of estimated total biophysical potential at \$55 t CO_2e^{-1}.

Statistical meta-analyses have also been applied to provide a comprehensive assessment of soil C estimates from the economics literature. Meta-analysis can summarize results from numerous studies that use a variety of modeling techniques as one metric. For example, Manley et al. (2005) develop a metaregression analysis of 52 studies (536 observations) to determine the cost-effectiveness of achieving carbon sequestration by moving from conventional tillage to no-till. C storage costs were found to be lower in the southern United States than in the Great Plains and Corn Belt. As additional information on the economic and engineering costs of adopting mitigation activities emerges, meta-analysis presents a valuable tool for explaining variation in estimates of these costs—variation driven by regional physical and production characteristics and modeling assumptions that differ from study to study.

Estimates of costs for implementing GHG mitigation activities can also be drawn from research that may not directly calculate marginal abatement curves or estimate total mitigation potential. Burney et al. (2010) assessed investments in agricultural intensification from 1961 to 2005 to conclude that such investments reduced GHG emissions at a cost of \$4 t CO_2e^{-1}. The economic implications of eliminating summer fallow vary by choice of crop to replace the fallow period; yield responses in the subsequent winter wheat varied (all declined somewhat) and some systems proved economically competitive without a GHG incentive (Lyon et al. 2004). For grazing land in South Dakota, Dunn et al. (2010) determined that income from low-good range condition was \$4.60–\$6.40 ha^{-1} greater than that for range in excellent condition. This finding could provide a cost estimate for C sequestration gains of excellent versus poor range condition.

Table 7 summarizes a selection of regional and practice-specific studies that estimate economic or competitive potential.

Table 7. Estimates of economic potential (EP) and competitive potential (CP) from the literature

Citation	Mitigation practice, region	Economic or competitive potential	Carbon price or incentive value ($ t CO_2e^{-1})	Mitigation potential (Mt CO_2e yr^{-1})
Antle et al. (2007)	Reduced fallow, central United States (20 years)	EP	$55 $14	3.3 1.65
Antle et al. (2007)	Conservation tillage, central United States (20 years)	EP	$55 $14	22.75 11.38
Antle et al. (2001)	Reduced fallow/ continuous cropping, Montana (20 years)	EP	$27 $6	5.14–15.78 1.47–4.4
Antle et al. (2001)	Conversion to permanent grass, Montana (20 years)	EP	$55 $14	1.47–3.67 0.73–1.84
Baker et al. (2010)	Conservation tillage or no-till (United States)	CP	$50 $15	22.5 3.87
Baker et al. (2010)	N_2O reductions through decreased N use (United States)	CP	$50 $15	18.4 2.5
Baker et al. (2010)	Afforestation of cropland and pasture	CP	$50 $15	390 134
Choi and Sohngen (2010)	Soil C sequestration through altered tillage and residue management (Ohio, Indiana, and Illinois only)	CP	$7.3 $550	0.45 6.99
Feng et al. (2004b) from Antle et al. (2007)	No-till conversion, Iowa	EP	$22	3.67
Lewandrowski et al. (2004)†	Conventional to conservation tillage, national	CP	$34 $14 $3	100.93 31.2 4.04
Lewandrowski et al. (2004)†	Afforestation from cropland or pasture, national		$34 $14 $3	488.48 265.71 31.2
Lubowski et al. (2006a)	Afforestation from cropland, pasture; avoided deforestation (full MAC curve shown)	EP	$180	800
Murray et al. (2005)	Afforestation from cropland or pasture, national	CP	$50 $30 $15	137 435 823
Murray et al. (2005)	Agricultural soil C sequestration	CP	$50 $30 $15	131 162 168
Murray et al. (2005)	Agricultural N_2O and CH_4 reductions	CP	$50 $30 $15	110 67 32
Weersink et al. (2003)	Corn to forage, Ontario		$14	0.72 t CO_2e ha^{-1}
Stavins (1999)	Afforestation in 36 counties in U.S. Delta region	EP	$109 $55 $18	47.07 41.59 25.86

† This paper includes cost estimates for different policy scenarios. The estimates in this table are asset payment estimates with no cost-share component.

Limitations in economic modeling

The economic optimization models described above assume a well-functioning market and profit maximization by producers, who have complete access to all necessary information. But nonmarket factors may shift farmers' decisions in ways unexpected if maximum profit was the objective—one reason that observed payments for environmental services such as GHG mitigation may be higher than the economic optimum (Kurkalova et al. 2006). Nevertheless, optimization models can assess policy and implementation options at a broad level and over the long term, accounting for various market interactions and helping policy makers compare options. They can also "ballpark" economic impacts.

Factors such as transaction costs (e.g., measurement and verification costs), embedded discounts (e.g., uncertainty or leakage discounts) and farm-level barriers (e.g., new equipment needs and lack of familiarity) can significantly change the value of GHG projects to developers and producers, affecting their participation and cost thresholds and throwing off estimates of the economic optimum. Such factors tend to be considered rough adjustments to the broader modeled system (i.e., the cause of deviations from the economic optimum).

Co-benefits associated with agricultural GHG-mitigating activities that could help encourage adoption are rarely represented fully in the economic models. Producers may be motivated by stewardship values, or they may be motivated by the additional financial incentive provided by a co-product (e.g., biofuels) market or by environmental co-benefits that are valued in ecosystem services markets (Bangsund and Leistritz 2008; Kurkalova et al. 2004b). Increasing market demand for "green" or low-carbon products could also stimulate adoption (Paustian et al. 2004). Calculating the value of so many different co-effects may be difficult; Elbadidze and McCarl (2007) conclude that given the cost and effort associated with measuring this value, "it may be appropriate to exclude co-effects from GHG mitigation efforts from cost-benefit analysis." On the other hand, these co-benefits may encourage the participation of different interest groups or government bodies in these efforts (Pittel and Rubbelke 2008), and the relative economic worth of co-effects to GHG mitigation depends on shifts in societal values.

Other considerations for assessing economic potential

How a program for agricultural mitigation is designed will determine whether producers are willing and able to engage in it (in response to incentive price and other factors) and therefore will affect the potential scale of a program and its overall mitigation outcome. Design also affects the costs of a program relative to its impact. For instance, Faeth and Greenhalgh (2000) compared different policy and market scenarios and estimated that a soil C trading policy at a price of approximately \$6 t CO_2e^{-1} would net a 19% increase in soil C sequestration[30] compared with a 1% decrease with a conservation tillage subsidy of \$62 ha^{-1}, a 7% increase with an extension of the CRP program, and an 8% increase with a nutrient trading program.

Emerging agricultural mitigation programs and protocols appear to have one of two main objectives. The first is accelerating innovation in farming practices that would engage few producers and generate little mitigation directly but that could influence the design of incentive and education programs that support producers. The second objective is accelerating adoption of well-designed activities that can engage large numbers of producers and directly generate nationally significant levels of mitigation.

For a program to accelerate implementation of activities on a large scale, the benefits to producers must be clear, and they must outweigh costs and risks. It is therefore important to keep transaction costs—the design, measurement, verification, and other costs involved in setting up a project so it can receive compensation—as low as possible. These costs are significantly affected by policy, market, and program design. Agricultural management practices with complex quantification, significant uncertainties, and verification difficulties could entail transaction costs greater than potential program payments or benefits (Antle et al. 2007; Smith 2004). Most models of economic potential have not incorporated transaction costs because of uncertainty about program design (Bangsund and Leistritz 2008). Nevertheless, two programs and one market–the Alberta Voluntary Offsets Program, the U.S. Federal Conservation Reserve Program, and The Chicago Climate Exchange—provide examples for estimating transaction costs and economic impacts.

Measurement (i.e., quantification) costs can be significant. Costs of field measurement are lower when the change (GHG mitigation per unit area) is larger and when the land area is greater (Smith et al. 2007b). Use of modeling approaches for quantification, particularly standardized approaches, can reduce measurement costs substantially, and high levels of aggregation combined with modeling can reduce measurement costs to within 3% of the total value of carbon credits (Mooney et al. 2004). Antle et al. (2003) and Pautsch et al. (2001) compared the difference in economic efficiency of different soil C sequestration programs: price-discriminating programs in which farmers were paid for sequestered carbon and single-price programs in which a set payment per hectare was provided regardless of measured sequestration. The price-discriminating programs had the lower cost per unit of GHG mitigation: as much as five times (Antle et al. 2003) or four times (Pautsch et al. 2001) lower than single-price programs. These more farm-specific heterogeneous programs are therefore preferred, provided that transaction costs associated with finer-scale quantification and monitoring remain low. A challenge for mitigation programs is finding a middle ground between accuracy at a fine (farm/site) scale and transaction costs.

To bring agricultural GHG benefits to market, most producers, especially smallholders, will have to go through an aggregator. Hence, they will pay brokerage costs that will diminish their profits. By way of reference, the brokerage costs of

30. When soil C credits were permitted for trade, farmers chose to implement activities that were most economically efficient, so that a significant amount of cropland was set aside as CRP. Therefore, the results were different than they would have been if conservation tillage alone had been incentivized.

crop insurance, which has a similarly aggregated structure, are 25% of the insurance's market price (Smith et al. 2007b).

In a market system, transactions must include a minimum level of carbon credits in order to be worthwhile to the buyer. For example, Clean Development Mechanism (CDM) projects under the Kyoto protocol must reduce emissions by 5,000 to 15,000 t CO_2e yr^{-1} to justify transaction costs.[31] An example given by Smith et al. (2007b) assumed a contract of 50,000 t CO_2e yr^{-1}. Given a soil C sequestration rate of 0.75 t CO_2e ha^{-1} yr^{-1}, approximately 7,000 to 70,000 hectares would need to be contracted together. In agricultural systems, longer-term contracts with less frequent quantification (e.g., sampling every five years) might make smaller-area contracts feasible. Given that agricultural managers and land owners are concerned about making long-term commitments on their lands, longer-term contracts may not be palatable. However, aggregators with a large pool of producers might be able to give these participants some flexibility in their contracts.

The choice of measurement approach will also affect verification costs. Field measurement may require more costly verification than use of standardized practice-based approaches, which with their relatively low resolution work better on average and thus are better for large-scale programs. The optimum verification instrument to confirm adoption of a practice or to monitor a change in greenhouse gases depends on the relative scale of the program and the costs of increasingly refined quantification. A lower-resolution method can result in lower costs and more producer engagement and thus more total environmental benefit (Antle et al. 2003; Kurkalova et al. 2004a).

Uncertainty, leakage, or potential for reversals may necessitate discounting of the carbon value, in order to maintain a high-quality GHG credit and account for the investment risk. This discounting lowers the price incentive offered to land managers for GHG mitigation efforts, and it may be worth the corresponding reduction in monitoring and measurement costs and the reduction in perceived risks. Multiregion/multiyear contracts can be used to reduce uncertainty caused by variability within the physical system and in the measurement of the GHG fluxes in the system (McCarl et al. 2007; Smith et al. 2007b).

Other critical accounting adjustments have been embedded in many programs targeting mitigation to help ensure that GHG objectives are achieved and at least cost, but some of these have an impact on costs. One such adjustment is additionality, which ensures that emissions decrease (sequestration increases) more than they would without the incentives. Requiring additionality tends to reduce program costs by ensuring that non-additional mitigation is not included in the payment program. Kurkalova et al. (2004a) determined that inclusion of all existing (non-additional) conservation tillage area in the program increased costs from \$8 to \$27 t CO_2e^{-1}.

Adoption of new practices: Nonmarket factors

In a review of economics and policy literature related to agricultural GHG mitigation in the United States, Bangsund and Leistritz (2008) conclude that adoption rates and behavior are major uncertainties. Producers often need an incentive exceeding opportunity costs alone to adopt a new practice (Kurkalova et al. 2006; van Kooten et al. 2002). Therefore, an understanding of the barriers to adoption is critical for successful implementation.

When an activity is incentivized, the gap between opportunity costs and the payments required for adoption may reflect perceived problems with the mitigation activity or economic factors that differ in the short run versus the long run (see Table 8). Predicting behavior on the basis of producer characteristics is complex and has produced conflicting results in the adoption literature (Knowler and Bradshaw 2007). Considering how the characteristics of mitigation activities might influence adoption may be a more fruitful approach. Capital constraints, risk perception, ease of compliance, and the availability of knowledge have the most significant short-run impacts on adoption (Smith et al. 2007b). Nonmarket factors that are less affected by time include social pressures, consistency with traditional practices, and interest/disinterest in the mitigation activity on aesthetic grounds. Although highly variable across unique cultural, social, and biophysical contexts, the economic impact of these factors[32] as well as capital and knowledge constraints can be estimated with nonmarket valuation studies—for example, surveys that estimate willingness to accept payment for a change in practice.

Many producers do not have the financial flexibility to adopt new activities (Miller 2009). Much agricultural capital is

31. Estimates are taken from various market sources. See the following websites: http://caaltd.org/Carbon/Market.aspx, http://www.riaed.net/IMG/pdf/GTZ_Energy_News_March_08.pdf, http://www.ecn.nl/docs/library/report/2005/b05002.pdf, and http://www.climatefinanceoptions.org/cfo/node/210.
32. They have nonmarket value (are not a commodity that can be traded) but still affect market price.

invested in fixed farm equipment, and a change in equipment requires liquidity, access to capital, or the opportunity to sell existing (and perhaps now less desirable) equipment. The most significant barrier in this regard is not capital investments, the costs of which are averaged over time, but immediate financing issues. GHG mitigation activities that affect cropping systems can also alter the timing of the income stream (as when different crops or perennials replace annuals). This timing change can result in liquidity constraints, increasing the near-term costs of adoption. As a specific capital constraint, property rights may also pose a barrier (Smith et al. 2007b). Because the majority of U.S. producers lease at least a portion of the land that they farm, they may be inhibited from making management changes or entering into long-term contracts.

Table 8. Perceptions of agricultural practices that influence adoption and implementation

Characteristic	Description
Relative advantage	Relative advantage for an innovation is a ratio of benefits to costs, not only in economic terms. Some possible benefits include an increase in social prestige, time savings, reduced discomfort, low initial costs, and immediacy of the rewards from the innovation.
Compatibility	Research has shown that even if an innovation provides relative advantage, it will often be rejected if it is incompatible with the community's sociocultural values and beliefs.
Complexity	Studies have shown that the greater the complexity of an innovation, the slower the rate of adoption.
Trialability	Trialability is the degree to which an innovation may be experimented with on a limited basis, thereby reducing uncertainty. Producers are more willing to invest in a technology that could be easily rejected (without great economic losses) if it does not provide the expected benefits or proves too difficult to maintain.
Observability	The more observable the results of an innovation, the more likely the innovation will be adopted.

Note: Based on Rogers (2003).

In a case study published by the USDA NRCS, Brant (2003) describes barriers to producer adoption of nutrient management practices. In the area of *relative* advantage, producers were limited by issues such as the costs of manure hauling and disposal and of soil testing and the time required to calibrate equipment and keep records. Economic models account for the direct financial aspects of relative advantage, but they may not reflect management effort and other issues affecting adoption (see Table 8). With positive relative advantages, activities that increase soil C levels offer the added benefits of increased productivity, reduced erosion, and improved soil structure (Smith et al. 2007b), even though these benefits may be realized only years after the activities' implementation. Other activities may diversify production in the farm system (e.g., strip intercropping [Exner et al. 1999]), reducing financial risk and providing stability.

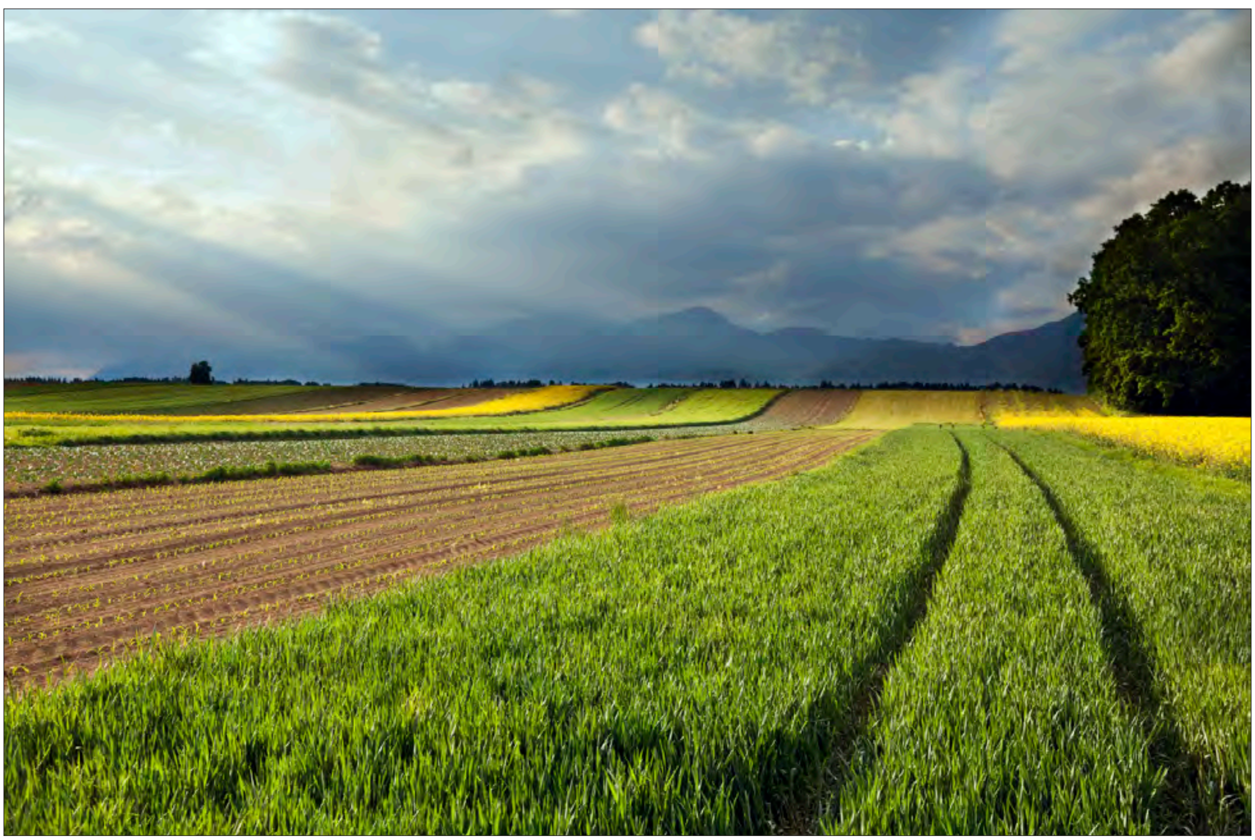
In the area of *compatibility*, decision-making control, cost considerations, and aesthetic and stewardship concerns come into play. Highly prescriptive programs remove more decision-making control from producers, making these programs less attractive to producers, who value the opportunity to judge nutrient and other management needs for their own land. Costs tend to be lower for GHG mitigation activities that do not require significant land use or cropping changes (van Kooten et al. 2002), suggesting that capital constraints or the uncertainties associated with unknown situations can be a significant barrier to voluntarily changing management practices for a given payment. Many farmers have a landscape and stewardship aesthetic that is handed down from previous generations (Nassauer 1995). For example, for largely aesthetic reasons, conservation buffers that are designed with well-defined, managed edges are more highly regarded by farmers and other residents (Lovell and Sullivan 2006).

In the area of *complexity*, issues of concern are confusion about the roles and responsibilities of different agencies; mixed messages from local, state, federal, and university representatives about the amount, source, placement, timing, and application of nutrients; and the difficulty of calibrating equipment and taking soil samples (Brant 2003). Complexity in the market mechanism can also be a barrier to participation in mitigation activities. A recent survey found that of the 88% of Ohio farmers who knew they were eligible to receive no-till carbon credits from the Chicago Climate Exchange, only 3% chose to participate in the CCX (Miller 2009). The study concluded that farmers misperceived the burdensomeness of the required paperwork, lacked knowledge about how and where to apply for credits, and believed the program provided low capital gains. According to the study, some farmers chose not to participate because of land tenure arrangements and the desire to not enter into a long-term contract.

Uncertainty and risk also play a role in farmers' decision making. Uncertainty about commodity and carbon prices, yield impacts, GHG mitigation amount, and production costs complicate decision making (Antle et al. 2007).[33] Because activities new to land managers entail uncertainty, they are associated with greater risk and lesser value than activities familiar to these managers. Insurance, warranties, and other measures can be taken to manage risk (Paustian et al. 2004).

Other important factors for farmers are *observability*—the ability to see a practice on the ground—and *trialability*—the ability to implement on a trial basis. A practice that saves a producer time and labor, reduces complexity, and for which the benefits are easily observable should be quickly adopted once the right incentives are in place. No-till is an example of this; even with capital constraints, it reduces labor and fuel by requiring fewer equipment passes over the field, is easily understood, and soil quality improvements are readily observed. Extension agents can also play a significant role in the adoption and diffusion of agricultural innovations, providing opportunities to observe and understand new GHG technologies in practice. Field research trials, market analysis, and the testimony of early adopters can mitigate uncertainty.

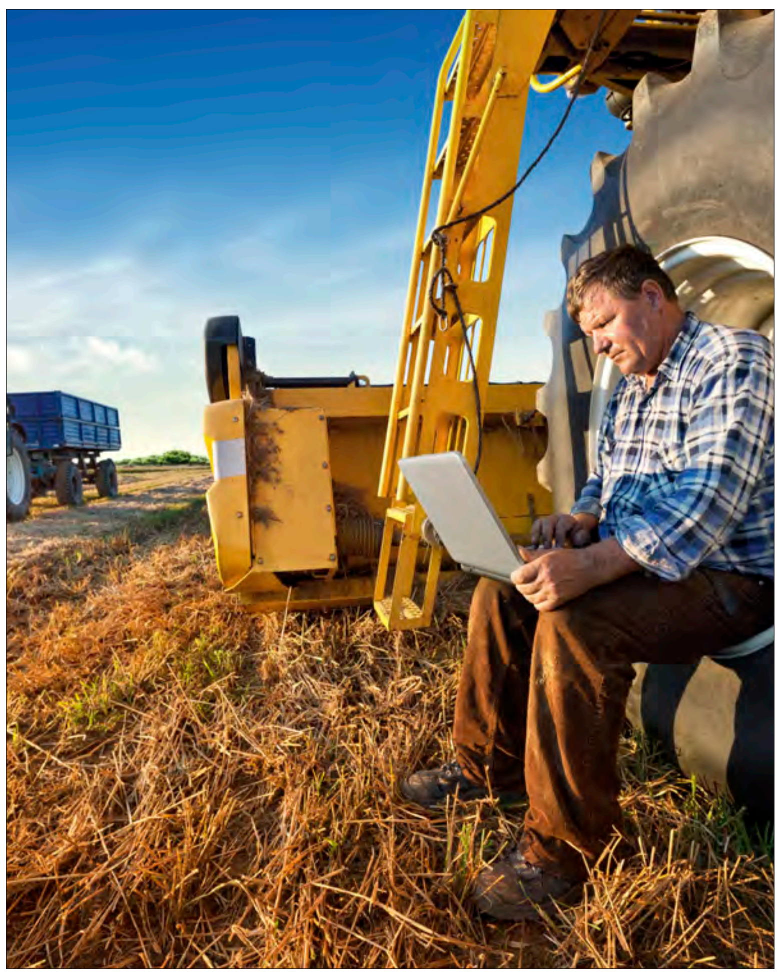
In contrast with more extensive field crops, specialty crops tend to generate higher farm-gate receipts, which may make GHG mitigation payments through offsets or other programs a much smaller portion of total farm revenue. Therefore, there may be less incentive for specialty crop producers to become involved in such programs. Instead, the increasing prevalence of supply-chain and market efforts to increase sustainability could play a much larger role in these cropping systems, in response to consumer and retailer demand for "low carbon" impact.



33. Because such uncertainty is difficult to model, empirical analyses such as that of Antle et al. (2007) have used assumed that expected costs and benefits are constant over time.

IMPLEMENTATION CONSIDERATIONS

This section reviews issues that can affect implementation of agricultural GHG-mitigation programs and projects. It assesses how approaches for quantifying GHG changes meet criteria for certainty, consistency, ease of use, and cost and how these approaches vary for different types of management practices. In addition, the section discusses accounting procedures—determining additionality, setting standardized baselines, estimating leakage, verifying performance, and assessing reversal risk—and considers the data and tools necessary to undertake these procedures for different management practices.



Quantifying Greenhouse Gases

Accurate quantification of soil C sequestration or changes in GHG flux (t CO_2e yr^{-1}) is necessary for determining net GHG outcome of shifts in management practice. For some programs, the credit a farmer or project developer would receive for implementing a new practice is based on this quantification. The choice of quantification approach affects transaction costs, risks to both project developers and programs, and the ease and cost of verification.

Quantifying the GHG reduction or carbon storage that results from a specific change in management requires assessing a change from baseline conditions. Baseline conditions are, in theory, the conditions that would exist without the policy or project in place. Approaches for quantifying changes in GHG emissions and carbon storage range from onsite field sampling to local flux towers, airplane or satellite remote sensing, and biogeochemical models. The models integrate various sources of field data.

Viability of the different quantification approaches will depend on a number of factors. When evaluating these approaches, protocol developers and program administrators should consider the following criteria:

- ***Accuracy and precision***—Does the method reflect the real impact of the change in management practices? Are the results repeatable? Does it hit the bull's eye or close to it?
- ***Certainty***—What are the chances that the method is wrong? How often will it miss the bull's eye?
- ***Comprehensiveness***—Does the method include all greenhouse gases affected by the management shift? Can it integrate multiple land uses or practices?
- ***Cost***—How much will it cost a land manager to use the method? Are the costs greater than the potential profits?
- ***Risk***—Could quantification uncertainty cause the land manager to fail to profit from the project?
- ***Fairness***—Will land managers get credited if they take additional actions or achieve additional benefits?
- ***Promotion of innovation***—Can the selected method allow for new mitigation practices or improvements in existing practices to count toward a land owner's credit/profit?
- ***Scalability***—Does the method work for both small and large projects? Will it also work for large-scale programs (e.g., programs in which two-thirds of all U.S. producers participate)?
- ***Alignment with national accounting***—Does the approach ensure that the results of individual projects add up to equal the national impact of those projects? Can such reconciliation be built into the system or achieved after the fact?

Box 4. Accuracy versus precision, error versus uncertainty
—an excerpt from *Carbon and Agriculture: Getting Measureable Results* (C-AGG 2010)

Accuracy refers to the agreement between a measurement and the true or correct value. If a clock strikes 12 when the sun is exactly overhead, the clock is said to be accurate. The measurement of the clock (12) and the phenomena it is meant to measure (the sun located at zenith) are in agreement. Accuracy cannot be discussed meaningfully unless the true value is known or is knowable.

Precision refers to the repeatability of measurement. It does not require knowledge of the correct or true value. If each day for several years a clock reads exactly 10:17 a.m. when the sun is at the zenith, the clock is precise (but not accurate). Note that the complications of edges of time zones do not need to be considered in order to decide whether the clock is good. The true meaning of noon is not important; what is important is that the clock is giving a repeatable result.

Error refers to the disagreement between a measurement and the true or accepted value.

Uncertainty of a measured value is an interval around that value such that any repetition of the measurement will produce a new result that lies within this interval. This uncertainty interval is assigned by the experimenter following established principles of uncertainty estimation. One of the goals of this report is proficiency at assigning and working with uncertainty intervals. Uncertainty, rather than error, is the important term to the working scientist.

Field measurement

Direct field measurement is appealing because it allows land managers to see the results of their efforts and to integrate the outcomes of all of their management choices, which can facilitate crediting for combinations of practices. It is also the only way to quantify outcomes of innovative practices that have not yet been studied and quantified. Such measurement must be performed initially to establish baseline conditions and subsequently to assess project performance. Carbon sequestration and reduced emissions resulting from a change in practice can be estimated in two ways: (1) by measuring the change that occurs after a new practice is implemented (e.g., repeat sampling, measurement before and after new practice is in place), or by (2) by comparing the new practice to the old practice on control sites using concurrent paired sampling (paired measurement).[1] The first approach accurately measures the impact of the management action if the system is in steady state. For example, if the soil organic carbon is at steady state, the soil is not a net source or sink due to factors other than the change in practice. On the other hand, if the soil had already been gaining carbon, the change in C stocks from time zero to time *t* would overestimate the C sequestration rates arising from the new practice. And finally, if the soil was already losing carbon, the C sequestration rates would not have detected the avoided soil loss of soil carbon from the practice. This before-and-after measurement approach integrates the impact of climate change and weather. So a good weather year or bad weather year could alter carbon gains or N_2O emissions and yet have nothing to do with management practice.

In the paired sampling approach, the control and project sites are experiencing the same outside forces, like weather and climate, and thus these will not affect the difference between project and control outcomes and will not count in the project's crediting. This method also provides estimates not only of sequestration gains, but also of avoided losses, for example, avoided loss of soil C in a no-till system (Izaurralde et al. 2001). The paired measurement approach helps account for nonsteady states of soil C equlibria, and year-to-year variability from factors such as weather and climate change.[2] However, farmers may not be able to afford to set aside representative portions of their land to remain under the original management scheme as control sites. At this time, control sites are most commonly used to recalculate the baseline when the original baseline expires. If individual project controls are not possible, repeat sampling on project sites combined with broader-based reference sites to follow broader trends may work, but care must be taken to ensure that variability in the baseline reference site is comparable to that in the project sites or that adjustments can be rationalized. Therefore, the project sites and the reference sites must experience similar weather and climate trends and have similar historical land uses. Given these challenges, reference sites are not commonly used by project developers at this time. Farmers may need to use existing research sites or permanent plots as reference sites. Federal or state agencies could assist in this task by expanding networks of permanent plots.

Measuring soil carbon pools

Soil sampling can be used at the beginning of a project to establish a baseline and then to monitor changes periodically (e.g., every five years). Because soil C change is relatively slow, a change may take many years to be detectable with a reasonable sampling effort (longer time = more change = fewer samples needed). Many sampling or in-field methods can be used to assess how organic and inorganic carbon pools change under different management practices. Well-tested and accepted methods include those published by the Soil Science Society of America and the GraceNET research group of the USDA (Liebig et al. 2010b; Sparks 1996). Most involve the collection of cores of soil at specified depth intervals and subsequent analysis of that soil. Traditional methods, like dry combustion or wet chemistry techniques, are then used to determine soil C within a lab setting, and new methods like near-infrared spectroscopy can be performed in the field or at the lab (Table 9). Although the new methods can make analysis of carbon in soils faster and cheaper, most still require concurrent field sampling to determine soil bulk density,[3] which is essential for quantifying soil carbon. Most of these methods also require soil preparation, including homogenization of soils and the removal and measurement of rock component, before analysis. Inelastic neutron scattering is an exception, but this technology is not commercially available and still requires the independent estimation of soil carbon with depth.

At this point, the new methods are all relatively time consuming and expensive, and even research protocols, such as that adopted by the USDA's GraceNET, use only dry combustion and MIR/NIR (see Table 9) (Liebig et al. 2010b). This review covers only measurement of soil C pools, as C flux techniques are not ready for widespread application and use. Micrometeorlogical assessments and the Bowen ratio energy balance technique are used for research (Johnson et al. 2010; Liebig et al. 2010b; Svejcar et al. 2008), but they require significant expertise and are too costly to be practical (Parkin and Venterea 2010).

1. In statistics, repeat sampling of an individual object (e.g., a single field) of study is differentiated from the matched-pair method, which uses common characteristics (e.g., between fields) to group subjects together. Therefore, we denote these two methods as "repeat sampling" and "paired sampling," respectively.

2. Recent studies have found that increased air temperatures are causing higher soil microbial respiration rates, so that even well-managed sites might experience decreases in soil organic carbon (Senthilkumar et al. 2009). Therefore, a reference site might be necessary to prove that the activity actually had a positive impact when compared to a business-as-usual scenario. If climate changes are expected to affect large regions, establishment of regional sites for baseline adjustment may be sufficient.

3. Bulk density is the mass of a given volume of soil, generally expressed in the units "g/cm^3" or "kg/m^3." Bulk density is needed to calculate soil carbon on a mass per unit volume basis for accurate measurements of soil organic C stock changes as a result of a soil-sequestering practice. Application of the method described by Ellert et al. (2002) requires that both soil C concentration and soil bulk density be measured for each depth increment, preferably from the same core. The mass of any rocks or gravel in the samples should be subtracted.

Table 9. Methods for measuring soil carbon

Method	Description	References
Dry combustion for soil C	Standard method against which others are compared Consumptive process of soil samples Soil samples collected in field but processed[a] and analyzed in laboratory Separate procedures for measuring organic C and inorganic C	Kimble et al. 2001; Nelson and Sommers 1996
Inelastic neutron scattering (INS)	Rapid and nondestructive[b] No soil samples needed (so relatively inexpensive) – in-situ and non-invasive Can provide continuous rather than discrete data Measures total carbon (organic C plus inorganic C) Still in development and testing stage, so accuracy and complexity are difficult to assess	Gehl and Rice 2007; Wielopolski et al. 2000; Wielopolski et al. 2008
Laser-induced breakdown spectroscopy (LIBS)	Rapid and nondestructive Soil samples collected in field, but soil core remains intact, and analysis could be in-field, reducing cost Measures total carbon (organic C plus inorganic C) Not widely used; difficult to assess accuracy and complexity	Cremers et al. 2001; Ebinger et al. 2003
Near-infrared spectroscopy (NIR)	Rapid and nondestructive Relatively simple technique Soil samples collected in field and most often processed, but mobile in-situ testing with minimal processing has been explored Can measure organic C and inorganic C separately Can be used for moist soils with only slightly less accuracy	Brown et al. 2006; Chang and Laird 2002; Chang et al. 2005; Gehl and Rice 2007; McCarty et al. 2002; Reeves et al. 2006
Mid-infrared spectroscopy (MIR)	Rapid and nondestructive Relatively simple technique Soil samples collected in field but processed and analyzed in laboratory Can measure organic C and inorganic C separately Provides more accurate results than NIR yet not as fast	McCarty et al. 2002; Reeves et al. 2006
Pyrolytic molecular beam mass spectroscopy (py-MBMS)	Rapid and consumptive process of soil samples Soil samples collected in field, requires some processing Can detect old and new C, and can quantify different types of SOM No cost advantage over dry combustion Still under development, so accuracy and complexity are unknown	Hoover et al. 2002; Magrini et al. 2002; Plante et al. 2009
Remote sensing	Aerial photography of bare surface soil used to correlate color with SOC concentration In experiments, surface SOC concentrations were predicted correctly 74%–77% of the time Lower accuracy but good for large scale estimates	Chen et al. 2000; Gehl and Rice 2007

[a] The processing of soil samples includes removal of rocks, drying, and homogenization.
[b] These nondestructive techniques allow the sample to be kept for future analyses.

There is substantial horizontal (across a field) and vertical (with depth of soil) variability in soil C pools, as well as temporal variation as management and climate changes. The variability of soil C and the implications for quantification of soil C sequestration are an ongoing focus of scientific effort (Franzluebbers 2010; Kravchenko et al. 2006; VandenBygaart 2006; VandenBygaart and Angers 2006). Capturing this variability and detecting changes in soil carbon requires sampling at multiple depths in the soil horizon and at multiple points across the fields and farms of interest. VandenBygaart et al. (2007) show that sampling at two depths in the soil profile is often sufficient.

In many cases, mitigation projects are trying to measure a relatively small change in C pools against what can be a large background pool of soil C. Soil C change associated with management change, typically about 1 t CO_2e ha–1 yr–1, is difficult to detect compared with total soil C, which can be as high as 370 t CO_2e ha–1 (Bolinder et al. 2006). For example, directly measuring changes in SOC stocks through sampling and analysis in rangelands can be particularly difficult, given the larger inherent background levels of SOC in rangelands and pastures.

Careful design of a sampling approach can increase detection of soil C changes and address some of the systematic variability in soil carbon across a field, while minimizing the required sample numbers. For example, stratified sampling is based on site characteristics such as hill-slope position (e.g., upland and lowland) or soil texture (Smith 2001; Willey and Chameides 2007). Repeated measurements at fixed locations can help farmers detect soil organic C changes by eliminating the impact of spatial variability across a site (Ellert et al. 2002; Lark 2009). In a simulation model example, Lark (2009) determined that repeat sampling (as compared with independent random sampling) reduced the total number of soil samples needed to determine a 18.3 t CO_2e ha^{-1} change by at least 7.5 times.

Mass equivalent, rather than fixed volume, soil sampling can significantly reduce vertical variability. It can also reduce incorrect determination of soil C change when bulk density is affected by a change in practice (Ellert et al. 2002; Lee et al. 2009a), a problem for traditional methods, which measure by depth. Mass equivalent soil sampling corrects the depth of sampling by the change in bulk density, ensuring that the same mass of soil is sampled with and without the practice change. This technique allows for measurement of the change in soil C associated with the same "population" of soil particles, especially if changes in bulk density have increased or decreased the volume occupied by those particles.

An example of sample analysis comes from a report by Paragon Soil and Environmental Consulting. According to the report, analysis of one organic carbon sample is approximately $85, and analysis of one organic and inorganic carbon sample is $100. (Table 10). Travel, equipment, and labor costs for sample collection depend on the required number of samples per unit area, distance from a central location, and field or soil characteristics. For large projects or those with substantial carbon sequestration in an area with a low background-carbon level, sampling may be viable. However, for other projects with small changes in carbon or high variability, the number of samples needed to detect a change with statistical certainty increases significantly (see section below on sampling approach and number of samples), and the costs of sampling and analysis could outweigh the financial benefits from carbon markets or mitigation program payments. An estimate from Paragon suggests that for a representative farm or project with typical natural variability in soil carbon, sampling costs can range from $10,000 with low background SOC to as much as $600,000 when SOC is high[4] (Paragon Soil and Environmental Consulting 2006).

Table 10. Costs of sampling and traditional analysis of soil carbon based on quotes from four to five commercial laboratories

Parameter	Unit cost
Sample preparation	$6.00
Total organic carbon (SOC)	$18.00
Total inorganic carbon (SIC)	$15.00
Bulk density	$8.50
Field sampling/reporting	$52.50
Total for SOC	$85.00
Total for SOC and SIC	$100.00
Sample archiving	$10/month

Source: Data from Paragon Soil and Environmental Consulting (2006).

Depth of sampling is an important question and a source of ongoing debate, especially when implementing changes in tillage regime. Even though soil C is concentrated in the top layers, the amount present in depths below one meter can be significant; in some soils, more than 40% of total carbon is in the second and third meters of depth (Jobbágy and Jackson 2000). This finding leads to the question of whether agricultural GHG mitigation activity sufficiently affects soil C at depths below one meter to warrant intensive sampling to depths much below the typical 20–30 centimeters, that is, the "plow layer." Deeper sampling is expensive and sometimes difficult.

A number of recent studies have raised questions about whether no-till management truly sequesters soil C, since consideration of deeper soil layers in some locations has resulted in lower or even negative estimates of soil C change (Angers and Eriksen-Hamel 2008; Baker et al. 2007; Luo et al. 2010). However, naturally high variability in soil C concentrations at greater depth (and less soil C change as a proportion of background levels) can make it difficult to detect differences (Franzluebbers 2010; Johnson et al. 2007), and a lack of detection should not necessarily be interpreted as a lack of change, as it may be due to an insufficient number of samples (Kravchenko and Robertson 2011).

Although sampling for soil C below 30 cm to include more of the root zone has been recommended by some researchers (Gál et al. 2007; Johnson et al. 2007), there is not sufficient evidence that such sampling will be conclusive and worth the costs. At least for certain regions, experts suggest that changes at depth are likely a small component of total C changes over shorter time scales (VandenBygaart et al. 2011). Thus, given conservative accounting principles and financial limitations, it may not be necessary to collect samples at depths below 30 cm (Syswerda et al. 2011). Further research is

4. This range assumes that the CV of the samples is 20% (a reasonable expectation) and that there is a change of 1.84 t ha^{-1} CO_2e (0.5 t C). A decrease in sample variability to 10% CV reduces the number of samples required, and the cost, to about 30% of that mentioned above. Low and high background C levels in this example are 10 and 100 t C ha^{-1} (3.7 and 36.7 t CO_2e ha^{-1}).

needed in this area, and recommendations may change. New meta-analyses forthcoming from Ogle et al. and Rice et al. (personal communication) may provide additional guidance on where carbon in deeper soils is most likely to matter.

Directly measuring changes in soil organic C stocks through sampling and analysis in rangelands will be particularly difficult given these lands' comparatively large inherent background levels of soil organic carbon.

Sampling Approach and Number of Samples

The number of samples needed to determine soil C quantity and change will have a significant impact on the quantification cost with sampling and thus its feasibility in comparison with modeling (or a sampling/modeling combination). In designing protocols and programs it will be important to set an appropriate level of certainty (acceptable levels of Type I and Type II errors), so that the target is achievable and maintains the objectives of the program. This level of certainty will impact the number of samples needed and the cost of the project; higher certainty means more samples will be needed. Type I error level is set on the basis of the desired level of confidence level the program wants in determining that a detected difference is real. Type II error level is set on the basis of confidence the program needs in determining that a change that has really occurred is detected. In deciding whether to develop a project, producers and project developers would consider the variance of critical variables (soil carbon, N_2O emissions, rainfall) in deciding the size of the difference in GHG flux or carbon sequestered the project would like to be able to detect. This size will affect the number of samples needed and project cost.

The statistical methods for determining sample size can be set for any confidence level used by the program or protocol but will depend on the sampling approach:

1. In repeat sampling, a single field that has been sampled at *r* locations at time zero is resampled using those same locations using a t-test for paired samples.

2. In paired sampling, two fields—one on which the management regime has changed and one on which it has not—with *r* randomly selected samples are compared using a t-test for independent samples. One concern with this method is that misleading results might be produced if only one subjectively picked control field is used. Ideally, multiple randomly selected fields would be used.

3. In multiple paired sampling, *r* randomly selected paired sites (paired fields or sub-areas within fields)—where one member of each pair is in a conventional management regime and one under the new management regime—uses a t-test for paired samples.

Appendix B details each of these methods and how to assess samples size for each.

Measuring GHG fluxes

Many methods can be used to measure GHG fluxes. Chamber methods work well for measuring gas fluxes. The chambers are relatively inexpensive, but their small size means that many of them are required to account for the high spatial variability in GHG flux at the field or landscape level. Even then, absolute measurements may not capture spatial variability well. Highly labor intensive, chamber methods require continuous sampling and sophisticated gas detection devices. Further, they sometimes disturb the soil surface and can't be used on fields under water, snow, or high-growing vegetation (Janzen et al. 2008; Paustian et al. 2004). They are probably not viable for GHG quantification of projects.

Measurement of N_2O fluxes in fields is complicated by significant offsite (indirect) emissions that can be caused by field activities. As water flows underground, it leaches nitrate and then produces additional N_2O fluxes as it enters streams bordering agricultural fields. These offsite emissions can be quite significant; studies show they can make up 25%–40% of total emissions (Minamikawa et al. 2010; Reay et al. 2009; Del Grosso et al. 2006). Similarly, indirect emissions can arise from NH3 volatilization from open manure storage and land application of manure whereby a portion of the redeposited NH3 on downstream soils is converted to N_2O.

Flux towers and aircraft measurements, unlike field-based chambers, can capture and quantify indirect nitrous oxide and other GHG emissions (Desjardins et al. 2010). By measuring vertical wind speed and gas concentrations at a point above a field, scientists can calculate how much gas is released or absorbed by the field. Flux towers are relatively new, are very expensive, and require user sophistication. Although becoming commercially available, they are still largely applied in the research realm (Janzen et al. 2008; Paustian et al. 2004). Overall flux measurements can be valuable for monitoring reference sites, validating process-based model estimates and associated scaling procedures, and running and calibrating process models, but they are not yet cost-effective for project-level use in large-scale GHG mitigation programs.

Field sampling summary

Field measurement of soil carbon pools for typical agricultural projects may be cost-prohibitive, working best for large-scale projects or those in which large changes in carbon relative to background carbon pools are expected (Table 11). Project operators may be unwilling or unable to assume the risk of finding unexpected results.[5] Waiting a year or more to determine whether mitigation is occurring may also not be feasible for small agricultural operations. Field measurement of CH_4 and N_2O fluxes is not yet viable. At the T-AGG experts meeting in April 2010, academic and government researchers suggested that in most cases field or project measurement would not be a cost-effective approach for land managers. They recommended using well-calibrated, scaled models to avoid the complexity of measuring GHG changes on individual projects or fields, where spatial variability of soil organic carbon and N_2O dynamics can create unmanageable uncertainty. These models would also overcome problems with high sampling costs and lag times for offset rewards and would be easier to scale to a large national-level program. In summary, under some circumstances, measurement may be a good option, particularly where programs want to encourage innovative new practices or systems. However, if a standardized approach that works for the majority of farms is the objective, modeling with some site-specific calibration data (perhaps from field measurement) may be the preferred approach.

Table 11. Assessment of field measurement to quantify changes in soil carbon

Criteria	Field measurement of soil carbon pools
Accuracy and precision	Can be good
Certainty	Can reach acceptable levels of certainty but can require a lot of sampling if soil variability or SOC are high
Comprehensiveness	Only measures soil carbon pools, which may be sufficient for some practices Must be combined with modeling to capture impacts on N_2O emissions and calculations of upstream or on-site fuel and energy shifts Accounts for all land management actions in the agricultural system
Cost	Sampling design and analyses require standardization and expertise to implement At $100/sample, costs can be high if detecting small changes in carbon against large C background or highly variable C pools, which may be common for production agricultural land and rangelands in the United States
Risk	Must wait one or more years to measure changes if lag times (2–5 yrs) are needed as carbon builds in soils Risk of high uncertainty from sampling or analysis, making required materiality threshold for verification difficult to achieve
Fairness	Accounts for all actions of land owner and possibly the impacts of climate change and weather irregularities.
Innovation	Would integrate all actions of land managers, not just common practices
Scalability	Cost, risk, and complexity may reduce farmer engagement Better for large farms and could be scalable if small farm projects are aggregated
Alignment with national accounting	Less likelihood of alignment with national inventory

Modeling GHG fluxes and carbon pools

In 2000 and 2003, the Intergovernmental Panel on Climate Change (IPCC) published *IPCC Good Practice Guidance for Land Use, Land Use Change, and Forestry* (IPCC 2003), and in 2006, *IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC 2006) to provide methods for estimating GHG emissions and removals. These guidelines were to inform the development of national GHG inventories, but they have been widely used for development of protocols and programs for voluntary carbon markets and for the international offsets program associated with the United Nations Framework Convention on Climate Change, the Clean Development Mechanism.

The IPCC methodology is divided into three tiers: from least to most precise, complex, and data-demanding. Tier 1 consists of simple equations and default emissions factors for broad geographical regions of the world. Broad country-level activity data, such as animal population numbers for livestock-related emissions, are used in the calculations.

5. Projects are moving away from soil sampling and utilizing CDM default sequestration rates because of the risks in direct sampling (G. Smith, personal communication, July 2011)

Accounting for spatial heterogeneity due to factors such as geographic variations in climate or soils tends to be less detailed in the Tier 1 approaches than in the Tier 2 and Tier 3 approaches. For that reason, Tier 1 methodologies are intended for national-scale inventories for which activity (land use and management practices) data are limited, region-specific emissions factors are lacking, or both. Tier 2 methodologies use IPCC equations and GHG emission categories similar to those in Tier 1, but they require country-specific or region-specific parameters that better account for geographic differences in climate, soil, management, and other conditions. The required activity data are more complex: for example, cattle populations are divided into sex and age classes (heifers, bulls, steers, dairy, calves), which modify emissions factors, leading to more accurate and precise estimates. Tier 3 methodologies typically employ more complex, process-oriented models to develop customized emissions estimates, along with fine-scale, well-distributed field measurement data that better capture spatial and temporal variability (Lokupitiya and Paustian 2006).

IPCC Good Practice Guidance for Land Use, Land Use Change, and Forestry identifies the major emissions categories for agriculture as follows:

- CH_4 emissions from enteric fermentation in domestic livestock
- CH_4 emissions from manure management
- N_2O emissions from manure management
- CH_4 and N_2O emissions from savannah burning
- CH_4 and N_2O emissions from residue burning
- Direct N_2O emissions from agricultural soils (accounting for all N inputs), with emissions occurring on the site where the nitrogen was applied
- Indirect N_2O emissions (offsite) (due to leaching, volatilization, and redeposition of N sources) from nitrogen used in agriculture
- CH_4 emissions from flooded rice production

For cropland systems, the 2006 *IPCC Guidelines for National GHG Inventories* considers non-CO_2 GHG emissions from biomass burning and CH_4 emissions from rice cultivation as well as emissions and removals from the following carbon pools:

- Aboveground and belowground biomass
- Deadwood and litter organic matter
- Soils (soil organic matter)

For each of these major emission and removal categories, the report details subcategories, decision trees for tier method selection, equations, and guidelines for obtaining parameters, emissions factors, and uncertainty estimates. The equations are the basis for both Tier 1 and Tier 2 methods; Tier 3 involves more complex models.

This report uses a typology similar to that in the IPCC guidelines to categorize approaches for quantifying greenhouse gases for mitigation projects or programs (Table 12). This typology is denoted as “pTiers” to designate project- or program-scale approaches rather than national inventories.

Table 12. Relative complexity of different program or project quantification approaches

Complexity	Quantification approach	Data requirements	Aggregation level/ uncertainty	Notes
pTier 1	IPCC Tier 1 default factors	Limited land use and management activity data (e.g., N application rates, no-till area); coarse delineation of soils; animal populations	Typically large spatial units; national scale; annual resolution; highest uncertainty when applied at project scale	Suitable for rough overviews and areas of limited data (e.g., indirect N_2O emission factor from leaching)
pTier 2	Hybrid approaches – using process or empirical models to develop region-specific empirical equations with emissions factors	Intermediate spatial/temporal scale input data; land use and activity data scaled to the spatial unit of analysis (tillage types, animal classes, fertilizer N type, crop type); requires longer-term scientific data to develop empirical models or calibrate process models	Finer spatial and temporal resolution than above; can achieve reasonable uncertainty due to "averaging" of modeled results	Can be suitable for project-based accounting and inventory roll-ups to national scale; application will depend on available scientific and management data
pTier 3	Process-based models	Spatially explicit fine-scale data for model variables; detailed land use and management histories; fine-scale soil maps and daily/weekly climate data; extensive scientific information required to calibrate models at this scale; field-measured data for estimating uncertainty are often a limiting factor	Finest spatial scale with representation of environmental and management variables at the individual farm level	Suitable for small-scale applications where local variability can be managed; model parameterization and testing can be performed; collection of land use and verified activity data obtained; systems will be needed to make advanced modeling approaches accessible to project developers
	Field sampling and measurement	Highest data requirements; costly to measure and variability high; long sampling intervals and crediting periods for soil carbon; can have highest precision	Site scale; may be subdaily if micrometerological techniques are used to estimate near-continuous gas emission rates, or every few years with soil carbon stock change; uncertainty can be high if not applied correctly	Level of errors may become overwhelming in sites/projects with high variability without good sampling and statistical design; can be most costly to implement

Source: Modified from a presentation by Jon Hillier, Aberdeen University.

pTier 1: Global default factors

All default emissions and removal factors can be found in the IPCC *Good Practice Guidance and National Greenhouse Gas Inventory Guidelines* (IPCC 2001, 2003, 2006). These and additional emissions factors submitted by countries are also posted on the IPCC Emission Factor Database (EFDB) (http://www.ipcc-nggip.iges.or.jp/EFDB/main.php). The website provides default emissions factors pertaining to source categories, subcategories, and greenhouse gases. Tier 1 methodologies for determining agricultural source emissions can be more difficult to execute than Tier 1 methodology for other categories or sectors. For example, the Tier 1 methodology for estimating CO_2 emissions from mineral soil requires some level of land area stratification according to climate and soil type as well as additional information on land use and management changes over time (Lokupitiya and Paustian 2006). These Tier 1 methodologies are sometimes applied for quantification in project protocols, resulting this report calls a "pTier1 approach."

Studies have suggested that the lack of specificity in Tier 1 default emissions factors may result in drastic over- or under-estimations of emissions when the factors are applied at smaller, local scales due to local environmental factors such as climatic and soil conditions (Flynn et al. 2005; Snyder et al. 2009). Accurate and specific default emissions factors for agricultural N_2O emissions are of particular concern in the literature, and many researchers have sought to revise the IPCC default factors for many N_2O sources, favoring Tier 2 approaches (Kuikman et al. 2006; Minamikawa et al. 2010; Reay et al. 2005). This concern becomes particularly important when incorporating Tier 1 approaches into protocols for projects to be implemented at a scale finer than the variability captured in Tier 1 methodologies.

The critical issue to consider about default factors when designing mitigation programs or projects is scale. The IPCC Tier 1 emissions factors were designed for national inventories at scales so large that countries do not have the information, the resources, or both to determine regional-specific emissions factors. Studies have shown that the IPCC emissions factors often do not work well at fine scales. At these scales, the confidence interval around the estimates increases (Del Grosso et al. 2010).

Box 5. Example of Tier 1 model: Indirect N_2O emissions

In 2006, the IPCC revised its methods of accounting for indirect N_2O emissions due to leaching and runoff from agriculture. The default emission factor (EF5) of 0.025 from 2002 was revised to 0.0075 kg N_2O kg-1 N from leaching runoff in 2006. This change was based on published data, including Sawamoto et al. (2005) and Hiscock et al. (2003). The order of magnitude difference between these values is indicative of the high levels of uncertainty that exist in determining indirect N_2O emissions (Mosier et al. 1998). The range of uncertainty for the current default EF5 value is 0.0005 to 0.025. This range reflects both the natural variability that exists in groundwater, rivers, and estuaries as well as the limited availability of data. The equation below is taken directly from the IPCC guidelines for national GHG inventories (IPCC 2006). It shows that N_2O emissions from leaching and runoff are dependent on the amount of nitrogen made available from fertilizer application rates (synthetic $_{(SN)}$ and organic $_{(ON)}$), change in soil organic matter $_{(SOM)}$, animal inputs $_{(PRP)}$, and crop residues $_{(CR)}$. The fraction of nitrogen leached ($Frac_{Leach\text{-}(H)}$) from agricultural fields is also highly variable. The default value here is 0.3, and the uncertainty range is 0.1 to 0.8. Nevison (2002) reviewed six studies from the midwestern United States that tracked agricultural N inputs and outputs in order to construct a mass balance. The variability in N leaching was found to be most highly related to precipitation levels; effects of different tillage systems and crop types were minimal.

Equation 11.10. N_2O from N leaching/runoff from managed soils in regions where leaching/runoff occurs (Tier 1)

$$N_2O_{(L)} - N = (F_{SN} + F_{ON} + F_{PRP} + F_{CR} + F_{SOM}) \cdot Frac_{LEACH\text{-}(H)} \cdot EF_5$$

The Field to Market Field Print Calculator (http://www.fieldtomarket.org/fieldprint-calculator/), covers a range of sustainability metrics, including water use and soil loss, and has a climate impact calculator that estimates changes in carbon and nitrous oxide on the basis of tillage and fertilizer application. It focuses on major commodity crops: corn, cotton, soybeans, and wheat. The version available in early 2011 uses methods and data from West and Marland (2002). It can calculate crop-specific estimates of kilograms of carbon per hectare stored as a result of continuous no-till management of corn, soybean, and wheat production, accounting for upstream emissions from energy and inputs. Using IPCC factors and national data on fertilizer and manure application rates, it estimates N_2O emissions.[6] Although the calculator primarily relies on Tier 1 approaches, Tier 2 or 3 approaches can be substituted, assuming the required data and models are available.

pTier 2: Regional emissions factors supported by regional inventory data

Given significant data collection, from inventories and research, in the United States, national or regional default factors should be obtainable for most variables in the IPCC equations. Tier 2 models refine Tier 1 methodologies through more regionally based activity data and customized emissions factors based on country-specific science (empirical or model-based). For example, pTier 2 methods to estimate soil N_2O emissions could potentially include factors such as soil texture, pH, carbon content, fertilizer type, crop type, tillage practice, and application methods—none of which are required inputs in Tier 1 methodologies.

Development of a pTier 2 or regional quantification can take two approaches. The first is an empirically based regression analysis to develop a regionally specific emissions factor. This analysis will likely incorporate results from regional studies as well as details of regional characteristics of soils and management practices. The second approach employs process-based biogeochemical models, which use mechanistic approaches for quantifying the impacts of changing land use practices. These models can be used at very local (farm or parcel) scales and also can be scaled up to develop regional estimates for similar soil and climatic zones.

The crediting programs developed by the Chicago Climate Exchange used a regional practice-specific crediting approach based on regionally determined emissions factors (see Box 6 for an example). The resolution of the approach was fairly low.[7] Increasing the resolution to smaller regional units, more specific to critical differences in soil type, should be possible given the precision of data in North America. The Alberta Offsets System uses a higher-resolution regional approach for its tillage program. The system has emissions factors for each ecozone, which are similar to major land resource areas in the United States.

An under-development open-source calculator, the Cool Farm Tool,[8] uses a combination of Tier 1 and Tier 2 approaches. This decision-support tool is designed for use at the farm scale and covers a stratified set of cropping and livestock

6. Details on the calculator's methods can be found in The Keystone Center report on the Field to Market project (2009). http://www.fieldtomarket.org/fieldprint-calculator.

7. CCX crediting is based in part on review of existing research on biophysical potential and expert input on how to extrapolate these values over the landscape. Programs could use other approaches, such as Tier 2 or Tier 3 modeling.

8. http://www.growingforthefuture.com/content/Cool+Farm+Tool.

Box 6. A regional crediting approach: CCX rangeland soil carbon sequestration

The Chicago Climate Exchange (CCX) is no longer accepting projects, but its Sustainably Managed Rangeland Soil Carbon Sequestration Project is a helpful example of a regional crediting approach. It adheres to simple, standardized rules for issuing contracts for agricultural carbon emissions reductions for improved rangeland management. Eligible and participating rangelands are measured against a baseline scenario of business-as-usual rangeland management practices that result in a loss of or no net gain in soil carbon. CCX quantifies GHG emissions reductions by multiplying qualifying acres by crediting rate-less project emissions. Recognizing that actual soil carbon sequestration on rangeland is continuously changing, CCX established offset issuance rates as a conservative approximation of average soil carbon uptake under specific management practices, regions, and climate. The range of soil carbon sequestration rates were compiled by a group of rangeland experts on the basis of a detailed assessment of peer-reviewed scientific literature and actual soil sampling at Natural Resources Conservation Service plots. The resulting approximations are conservative because they are discounted from the mean of range soil carbon sequestration values.

practices around the world. The tool provides a single overall estimate based on previous and current farming practices, allowing producers to compare management choices. While perhaps not suitable for project-level GHG accounting in North America, where more refined approaches are available and have been tested, the tool has several practical applications in the developing world, where countries and producers want to explore GHG mitigation options (Hillier et al. 2011).

pTier 3: Complex models using spatially explicit local and regional inputs and inventory data with refined time steps

The United States is well positioned to use pTier 3 or project/farm-level approaches to measure and monitor GHG fluxes from agricultural land management for mitigation policies. It has substantial crop and land use inventory data, fine-scale climate data, long-term research plots with field measurements of C stock changes and GHG emissions, and a substantial peer-reviewed literature from which to develop locally relevant emissions factors, empirical estimates (e.g., regression equations from research results), or process-based models to describe emissions and removals of greenhouse gases. The U.S. national GHG inventory, reported annually to the UN Framework Convention on Climate Change, employs a Tier 3 approach for soil CO_2 and N_2O emissions/removals using dynamic simulation models, the USDA National Resources Inventory (NRI), and other databases on land use and management practices as well as more than 50 long-term field experiments for uncertainty estimation (EPA 2010, Del Grosso et al. 2010, Ogle et al. 2007). In addition, four biogeochemical process models are calibrated and parameterized for use in the United States at the national level. These models have compiled the appropriate spatially and temporally relevant input databases for soil types, climate parameters, major crops, and dominant management practices embedded in U.S. agriculture. It also means that data from U.S. ground-based inventories, long-term research sites, and management-based field research have been integrated and used to test and calibrate the models. These four models—DAYCENT, DNDC, EPIC/APEX, and NASA-CASA (Table 13)—can quantify average GHG fluxes from soils from most agricultural management practices on a wide variety of crops with a relatively high level of accuracy (Del Grosso et al. 2002; Feng and Li 2001; Grant et al. 2001; Izaurralde et al. 2006; McGill 1996; Petersen et al. 2002; Ranatunga et al. 2001; Smith et al. 1997). The accuracy of these models improves significantly with greater site-specific data and finer temporal agro-climatic data input. Fine-scale national data on management practices, the last critical input, are not fully available and therefore are best entered at the farm or project level.

Management input variables include cropping systems (crop type, rotation sequence, field size, yield data), farm operations (seeding, tillage, harvest, residue management, spraying, irrigation data), fertilizer and manure N content, source, rate, placement, and timing data. These data can be gathered at a regional scale (observational datasets) or at the farm level.

Environmental input variables – fine-scale soil map data with soil properties (soil type, texture, organic carbon and nitrogen levels, soil profile data, pH, etc); topographical information; daily or weekly climate data (precipitation, temperature), hydrography etc. Information can be found in national databases or gathered at the site level.

The choice between a pTier 2 regional and pTier 3 project-level modeling approach will be a balancing act between better precision and greater data input needs. A regional approach might be used where research is lacking and confidence in the process models is relatively low or where sufficient site-level data cannot be acquired; otherwise a site-level approach appears preferable for quantification. However, complexity in implementing programs, particularly in developing simple and sufficient verification indicators and aligning the verification method with the modeling approach (aligning definitions of practices), may make application at the farm or project scale difficult. This consideration can make the regional (pTier2) scale preferable.

Table 13. Description of the major biogeochemical process models capable of quantifying GHG fluxes for the U.S. agricultural sector

Model	Description	Activities (1)/GHGs (2)
DAYCENT[*]	DAYCENT simulates exchanges of carbon, nutrients, and trace gases among the atmosphere, soil, and plants. Carbon and nutrient flows are controlled by the amount of carbon in the various pools, the N concentrations of the pools, abiotic temperature/soil water factors, and soil physical properties related to texture. Since 2005, DAYCENT has been used to estimate N_2O emissions from cropped and grazed soils for the U.S. National GHG Inventory. The model is also used to investigate how land use and climate change affect plant growth and soil C and N fluxes. http://www.nrel.colostate.edu/projects/daycent/index.html Contact: Stephen Del Grosso	(1) Events and management practices such as fire, grazing, cultivation, residue management, and organic matter or fertilizer additions are modeled. A wide variety of crop, grass, and forest types are supported by the model. Primary model inputs are soil texture, current and historical land use, and daily maximum/minimum temperature and precipitation; (2) CO_2, N_2O, NO_x, and NH_3 emissions; CH_4 uptake; leached NO_3; crop/biomass yields
DNDC[†] **DeNitrification-DeComposition Model**	DNDC is a family of models for predicting plant growth, soil C sequestration, trace gas emissions, and nitrate leaching for cropland, pasture, forest, wetland, and livestock operation systems. The core of DNDC is a soil biogeochemistry model simulating thermodynamic and reaction kinetic processes of carbon, nitrogen, and water driven by plant and microbial activities in the ecosystems. DNDC can be applied at various scales, ranging from site-specific applications to quantify within-field variability to county and regional scales to account for differences in environmental conditions and management practices. Soil organic carbon is divided into four compartments – litter, microbial biomass, active humus, and passive humus. The first three are subdivided into pools that vary by their resistance to decomposition. Soil rate constants vary by abiotic factors of soil moisture, temperature, and texture. To relate C and N cycles, the output of soluble C drives denitrification. Carbon dynamics are computed on a daily time step, but N_2O is based on an hourly time step. http://www.dndc.sr.unh.edu/ Contact: William Salas or Changsheng Li	(1) A relatively complete set of farming management practices such as crop rotation, tillage, residue management, fertilization, manure amendment, irrigation, flooding, and grazing have been parameterized to regulate their impacts on soil environmental factors (e.g., temperature, moisture, pH, redox potential, and substrate concentration gradients) (2) N_2O, NO_x, CH_4, and CO_2; from cropping systems (including rice CH_4), grazing systems and manure application/management; nitrate leaching loss (NO_3); soil C sequestration, crop development, and biomass yields
EPIC[‡] **(Erosion Productivity Impact Calculator)**	EPIC (Environmental Policy Integrated Climate) is a comprehensive terrestrial ecosystem model capable of simulating many biophysical processes as influenced by climate, landscape, soil, and management conditions. Salient processes modeled include growth and yield of numerous crops as well as herbaceous and woody vegetation; water and wind erosion; and the cycling of water, heat, carbon, and nitrogen. The carbon algorithms in EPIC are based on concepts used in the Century model and are applied to entire soil profiles. In addition to soil respiration, EPIC calculates carbon losses in eroded soil sediments, runoff water, and percolating waters; carbon lost during vegetation burning; and carbon emissions due to management and inputs (e.g., tillage, fertilization). EPIC also uses a process-based algorithm to estimate N_2O flux during denitrification and N_2O and NO fluxes during nitrification. http://www.brc.tamus.edu/simulation-models/epic-and-apex Contact: Cesar Izaurralde	(1) A relatively complete set of farming management practices, including soil management, crop management, nitrogen management, land use management, and livestock management (2) Soil nutrient (C and N) stocks, CO_2 and N volatilization, N_2O flux from denitrification
APEX[‡] **Agricultural Policy Extender**	APEX is the watershed version of EPIC. It contains all of the algorithms in EPIC plus algorithms to quantify the hydrological balance at different spatial resolutions (farms to large watersheds) under different land covers and uses. The fate of eroded carbon and nitrate can be traced through the entire watershed. http://www.brc.tamus.edu/apex.aspx http://www.brc.tamus.edu/simulation-models/epic-and-apex	

Model	Description	Activities (1)/GHGs (2)
NASA-CASA (Carnegie-Ames-Stanford Approach)	The model simulates net primary production (NPP) and soil heterotrophic respiration (Rh) at regional to global scales. Calculation of monthly terrestrial NPP is based on the concept of light-use efficiency, modified by temperature and moisture stress scalars. Soil C cycling and Rh flux components of the CQuest model are based on a compartmental pool structure, with first-order equations to simulate loss of CO_2 from decomposing plant residue and surface soil organic matter pools. Model outputs include the response of net CO_2 exchange and other major trace gases in terrestrial ecosystems to interannual climate variability in a transient simulation mode. CASA EXPRESS CQUEST http://geo.arc.nasa.gov/sge/casa/index.html (currently only includes forests) Contact: Chris Potter	(1) A relatively complete set of farming management practices, including soil management, crop management, nitrogen management, land use management, and livestock management (as it pertains to grazing)

* Description edited by Steven Del Grosso.
† Description edited by Bill Salas.
‡ Description edited by Cesar Izaurralde. Note that the acronym EPIC stands for "Erosion Productivity Impact Calculator" *and* "Environmental Policy Integrated Climate."

Applying a modeling approach

Detailed guidance will be required for any quantification approach. Field measurements require specific sampling design and standardized analytical techniques to be used and data to be reported in a specific manner, but pTier 1 and pTier 2 modeling approaches use decision trees to walk users through the appropriate equations to determine the input values for those equations. The input values can come from published default tables, databases, or farm or site data. These often online documents are designed for use by project developers or others providing technical assistance to producers or land managers. Given the complexity of biogeochemical process models, development of appropriate guidance and even better, user interfaces (e.g., decision support tools) is needed. One of the primary challenges in using sophisticated process models and models for integrated GHG accounting at farm or project scales is to make the technology available to non-expert users such as project developers, consultants, and verifiers. A single model needs to be selected, set up, and calibrated for a protocol so that it can be consistently, systematically, and accurately run using consistent data inputs for project quantification.

The United States has high-quality, spatial, georeferenced, and fine-resolution data on environmental characteristics (climate, soil, topography) sufficient for modeling at any resolution, but it lacks management data. A few databases include management data at regional scales and can be used for regional-scale modeling. At finer resolutions, however, producers or project developers will have to provide detailed information (Paustian et al. 2006).

Running a full model may be too complex and costly for project developers and for verification, and it may allow gaming. Developing some type of decision support interface that locks down (predefines) critical parameters may therefore be preferable (Table 14). Such interfaces must inspire confidence in their accuracy, include the right crops and practices, and have the capacity to estimate uncertainties. In addition, they need to be integrated into the program or protocol with the appropriate baseline scenario and accounting framework.[9] Tradeoffs between flexibility, which may allow gaming, and simplicity will need to be considered. The COMET-VR (CarbOn Management and Evaluation Tool for Voluntary Reporting) and COMET-Farm online estimation tools are examples of such interfaces. They are designed for use by nonspecialist with no prior training. Attributes of COMET-VR and COMET-Farm are described in Box 7.

Biogeochemical process models assume a static climate, whereby regional data on climatic factors such as rainfall, temperature, and season length are based on historic patterns. Another important qualification is that the studies that are used to calibrate the models were also conducted under the current or past climate regimes. This type of assumption is likely reasonable for time horizons of 10 or fewer years, which may be a typical time frame for many programs or protocols focused on agricultural production. Many studies have used these models to examine how climate change is likely to affect yields and interact with mitigation (e.g., Li et al. 2005a; Thomson et al. 2006), but fine-scale regional predictions of climate change are highly uncertain. Thus these predictions, which would be necessary for a project/farm or perhaps even a regional-scale approach to quantifying GHG changes from agricultural management, are not feasible at this time. Exploring whether the impacts of climate change will negate or enhance the benefits of management actions to mitigate climate change is an important research question outside the scope of this assessment.

9. More detail on these issues can be found in the T-AGG companion modeling report at http://nicholasinstitute.duke.edu/ecosystem/t-agg.

Table 14. Web-based user-friendly decision support versions of selected biogeochemical models

Base model	Decision support versions	Notes
CENTURY/ DAYCENT	COMET-VR http://www.cometvr.colostate.edu/ COMET-VR 2.0 http://www.comet2.colostate.edu/ COMET Farm (Beta available soon)	COMET-VR and COMET-VR 2.0 were developed with support from USDA-NRCS and are being updated and integrated into COMET-FARM, a whole farm/ranch GHG emission estimation tool that uses DAYCENT for estimating soil emissions and CO_2 and N_2O uptake (and other models for estimating livestock and other on-farm emissions). References: Paustian et al. (2010); Paustian et al. (2009)
APEX/EPIC	Nutrient Trading Tool http://ntt.tarleton.edu/nttwebars/ ARCGIS APEX	This tool was developed with support from USDA NRCS. It tracks the nitrogen impacts of agricultural practices on water quality but can also but used to quantify GHG impacts. It is being linked to the DAYCENT model. A second decision support system for EPIC is under development by USDA and other researchers with support from NASA.
DNDC	U.S. Cropland Greenhouse Gas Calculator http://www.dndc.sr.unh.edu/	This tool is the ARCGIS version of the DNDC model for U.S. croplands.
NASA/CASA	CASA EXPRESS CQUEST http://geo.arc.nasa.gov/sge/casa/index.html CQUEST online tool (slightly more limited in scope and customizability): http://sgeaims.arc.nasa.gov/website/cquest/viewer.htm	This observational tool assesses climate and land management trends and landscape impacts. It does not specifically model scenarios, but it can be run with prepopulated, externally created scenario models. This tool is especially effective in identifying current problems and sources of emissions. It uses remote sensing with an ARCGIS interface and background calculations based on user-provided data, satellite imagery and remote sensing data, and IPCC baseline information. The tool is scalable to the quarter acre (0.10 ha) and to a region and nation. It is usable worldwide.

Modeling summary

Models present a viable alternative to field sampling. Some of the criteria by which use of models may be considered by those developing programs and protocols for GHG mitigation are presented below. Tables 16 and 17 compare pTier 1 and pTier 3 as ends of a modeling approach spectrum.

Cost

The cost of running models to quantify greenhouse gases is quite low from the user, project developer, and producer perspective as long as the data inputs are known or are easily accessible by the landowner. If highly detailed or field-measured inputs are required, the costs for users may become prohibitive. but However, they are likely to be less than field-sampling costs in many cases. The burden of cost is shifted to those who are developing the program and the infrastructure for the program. Using a standardized decision support interface with models will reduce user costs and streamline verification costs. Calibrating models for new crops and practices will entail additional costs. Modelers estimate that calibrating a model for one new crop will range from $10,000 to $50,000. Economies of scale can be achieved if data on multiple crops can be combined.

The accuracy and value of these modeling tools could be enhanced by a national reference-site network. Such a network could leverage existing activities and expertise, such as the USDA Natural Resources Inventory system. A national system of approximately 5,000 monitoring locations could be established and maintained at a cost of $2–$3 million per year.[10]

Risk

If project developers or land managers are guaranteed credit for a model's GHG mitigation estimate, their risk is low. If a process model is used in a standardized way to quantify and monitor the GHG impact of management changes and if it uses conservative assumptions, risk of program failure is also low. Programs will want to ensure that their model is accurate or conservative. If programs allow significant flexibility in which models are used and how the models are used, significant risks of error or bias arise, particularly with complex process models with numerous variables and parameters. Given flexibility, project developers can shop for the best default factor or model and can bias the outcome toward over crediting (Pfaff et al. 2010). Allowing such "default shopping" is against principles of conservatism in program design.

Inherent uncertainty in the model or the science behind the model may also pose risks regarding the program performance or project viability. If crediting based on model outcomes is not guaranteed by policy, and model outcomes are revealed, the market value of credits could change, affecting the land or project owner. For example, if a model is found to overestimate the N_2O flux reduction from nitrification inhibitors by 50% for most soils, the program relying

10. K. Paustian, personal communication, 2010.

on such a model would be overcrediting, which means that the achieved environmental benefit is less than that anticipated. If the program readjusts the value of the N_2O reduction strategy on the basis of this revelation, the value of nitrification inhibitor-based credits could drop 50%. This type of risk can be managed in various ways in the program or policy design, which shifts risk between the program and projects (Olander et al. 2008). If errors in model estimates are unbiased (equally negative and positive), the overall impact on environmental outcomes would be balanced, and the program should, on average, achieve expected outcomes, even if individual producers are over- or undercredited.

Fairness

Use of one model by everyone in a program ensures fairness across projects. However, if generalized or regional averaging approaches (pTier1 or pTier2) are used, outcomes across regions will be averaged. This averaging may over- and underestimate outcomes for particular parcels of land.

Innovation

Because models are parameterized using results from research, they work best for programs encouraging expansion of studied practices. Integration of field data in models is important. If models are used as the primary quantification tool for an agricultural GHG mitigation program, they must be adaptive to new knowledge about different crops and practices. This capacity allows programs to expand and quantification to improve. In a federal program, modeling efforts could be tied to existing USDA (NRCS) research programs to ensure rapid transfer of knowledge. For truly innovative practices that have not been studied, direct measurement will likely be needed.

Box 7. COMET-VR and COMET-Farm

COMET-VR (http://www.cometvr.colostate.edu) was developed for the U.S. 1605B program for self-reporting of changes in soil C stocks and CO_2 due to fuel use in implementation of conservation practices on U.S. croplands and grazing lands. To estimate soil C stock changes, users run the Century ecosystem model, inputting location, soil attributes, past and current crop rotation, tillage practices, and other information (Paustian et al. 2009). The system is supported by a large database of management choices based on the USDA Natural Resources Inventory (NRI) as well as by other databases of environmental and management factors. The system includes an empirical uncertainty estimator that computes 95% confidence intervals around mean estimates.

A new version, COMET-VR 2.0, includes specialty crops as well as perennial crops and agroforestry practices (Paustian et al. 2010). For agroforestry practices, users can employ an empirical model to estimate future changes in woody biomass C stocks if they have current measurements of species, stocking rates, and mean tree diameters. Soil N_2O emissions are provided on the basis of a meta-model derived from the DAYCENT simulation model described above.

The COMET-Farm system is a web-based, full GHG accounting system designed for comprehensive farm-level analyses. The system has a fully spatial user interface to specify individual field boundaries and management systems. The interface uses Web Soil Survey to overlay soil maps and delineate soil $\times$ topographic position $\times$ management subunits to estimate biomass and soil CO_2 and N_2O emissions and removals using the DAYCENT model, which runs in real time on the system. The empirical model option for estimating woody biomass C stock changes in COMET-VR 2.0 is also included. Emissions from livestock (enteric CH_4) and manure management (CH_4 and N_2O) are estimated using empirical functions derived from 2006 IPCC methods. Livestock emissions estimates include options for enhanced characterizations based on user-supplied feed and livestock attributes designed for CH_4 emission abatement. The system also provides for monitoring of energy use, fossil-derived CO_2 emissions, and on-farm energy production.

—Contributed by Keith Paustian

Scalability

Process models are easily scalable. They can be used for farm-scale estimates or program-scale assessments. The quality of the estimates at the farm scale will depend on the availability of site-specific input data and data for calibrating a model for a specific region. The accuracy of the model, particularly at small scales, will vary geographically and with management practices, depending on the quality of the information available for calibrating the model. For example, the abundance of information on corn and tillage practices in the Corn Belt should make the accuracy of the model's estimates for this region fairly high. In contrast, estimates of yield and greenhouse gases from corn production and tillage practices in the Southeast will likely less accurate due to fewer data points for model calibration.

Alignment with National Accounting

Because the U.S. national emissions inventory for soil carbon and N_2O flux use a Tier 3 approach, using one of the primary biogeochemical models (CENTURY/DAYCENT), programs or projects that use similar models and datasets will typically align well with national accounting. This compatibility is desirable because annual national accounting can be compared to program reporting or verified offset totals to perform a true-up and a process of checks and balances on the program, offset markets, or both. This measure also can indicate leakage effects that may not have been accounted for in protocol development and can promote continuous improvement and innovation in accounting frameworks.

Table 15. Performance assessment of pTier 1 approaches to GHG quantification

Criteria	Modeling of GHG fluxes and soil carbon pools (pTier 1)
Accuracy and precision	Moderate to good at national scale, low at farm/field scale
Certainty	Moderate to good at national scale, low at farm/field scale
Comprehensiveness	Equations and default factors cover all major GHG fluxes and C storage pathways Accounts for a limited set of agricultural land management actions but does not address finer controls on GHG processes like soil texture, pH, and drainage or on fertilizer type, crop type, climate fluctuations, and animal classes like Tier 2 Does not address tradeoffs in outcomes (methods would add up impacts of various practices but not account for marginal benefits of employing additional practices)
Cost	Costs for programs are low; IPCC updates equations and default values Costs for project developers to wade through protocol equations are low to moderate
Risk	Low risk to producers/project developers if policy guarantees them model values; producers can estimate GHG outcomes using program-selected models and defaults If variability in model estimates is unbiased (equally negative and positive), the overall impact on environmental outcomes would be balanced and the program should, on average, achieve expected outcomes, even if individual producers are over or under credited Can be significant risk to the environment if variability is biased, because the program could significantly over or under credit mitigation
Fairness	May greatly under or over value field/farm outcomes May not include all management options Producers may not be comfortable with models for quantification but results should be predictable and comparable across farms/projects
Innovation	Equations and default factors would need to evolve with new practices Will not work for unstudied practices
Scalability	Better accuracy at large scales (nations) than small scales (farms, projects)
Alignment with national accounting	High for overall net impact for a major source of GHG emissions across the United States; regional differences not captured well because United States is using Tier 3

Tier 2 models are appropriate for regional or large farm or ranch estimates. They are a hybrid with lower uncertainties than Tier 1 approaches and lower data needs than Tier 3, and as a result have been used in many programs and protocols. Programs will bear a cost to develop acceptable Tier 2 defaults for the regions of interest.

Table 16. Performance assessment of pTier 3 approaches to GHG quantification

Criteria	Modeling of GHG fluxes and soil carbon pools (pTier 3)
Accuracy and precision	Good at national scale, good to moderate at farm/field scale, depending on availability and spatial/temporal scale of input data
Certainty	Good to moderate at all scales, depending on data availability
Comprehensiveness	Models include all major GHG fluxes and C storage pathways Regional and time-scale data for most agricultural land management actions and crop/livestock practices are captured; major GHG soil and livestock drivers are captured through refinements in emission estimates
Cost	Low to moderate cost for programs; addition of specialty crops increases costs, as does updating or adjustment of decision support tools for specific uses Low cost for producers and project developers to enter data into decision support tools Verification can be more costly when project-scale methods are used
Risk	Low risk to producers and project developers if policy guarantees them model values; producers can estimate GHG outcomes using the program-selected model Model results are relatively accurate across scales and should reduce environmental risk if programs have good quality control of verification
Fairness	Results should more closely represent management activities and site differences
Innovation	Models must evolve with new practices Will not work for unstudied practices
Scalability	Can scale up from farm to nation; accuracy at small scales is dependent on availability of input data
Alignment with national accounting	Use of similar input and activity data should keep alignment high

Quantifying upstream and process emissions

Like the soil C changes and direct flux impacts of nitrous oxide and methane, process and upstream GHG impacts can also contribute significantly to net changes in greenhouse gases. Process emissions are emissions from the use of fuel in tractors, irrigation pumps, grain dryers, and other equipment, which might be affected by implementation of alternative management activities. Upstream emissions are the GHG emissions from energy used in production of agricultural inputs such as fertilizer. These emissions can be thought of as embedded energy or embedded greenhouse gases.

Table 17. Emissions from fuel use

Fuel type	Total C	t CO_2e/gallon	t CO_2e/liter
1 U.S. gallon of gas	2.42 kg C	0.00887	0.002341
1 U.S. gallon of diesel	2.77 kg C	0.01015	0.002679

The primary emission sources for fuel use are diesel and gasoline for field operations (process emissions, Table 17), which, when calculated from Miranowski (2005), result in average national emissions of 0.36 t CO_2e ha^{-1} yr^{-1}.[11] Crop production data from state extension services reveal significant variation among crops and among regions. For example, California data indicate a range in fuel use that results in GHG emissions of 0.13–0.71 t CO_2e ha^{-1} yr^{-1}, depending on crop type (corn > hay > wheat).[12] The various agricultural activities assessed for GHG mitigation potential in this report have varying effects on field operations and associated fuel use. Although assumptions can be made—and may be appropriate—in the planning process, monitoring farm fuel consumption is relatively simple, and fuel records may be appropriate data for project or program calculations and verification.

Upstream emissions from the use of fertilizer N (for manufacture, distribution, and transportation) amount to approximately 3.2–4.5 t CO_2 t^{-1} of fertilizer N manufactured (Izaurralde et al. 1998; West and Marland 2002). For all U.S. fertilizer N consumption (11.4–14.5 Mt N yr^{-1}), total upstream emissions are approximately 56 Mt CO_2e, or an average of 0.45 t CO_2e ha^{-1} yr^{-1}. As with fuel consumption, fertilizer N application can vary significantly among regions, and especially among crops, so the associated upstream emissions also vary. State extension sources can provide crop-specific estimates of fertilizer use, although project or program calculations would most appropriately input site-specific data from farm application or regional sales records.

Summary of quantification options

Changes in soil carbon can be quantified with field sampling using various sampling methods that reduce numbers of samples required and thus costs. However, for most productive U.S. agricultural and grazing lands, projects will be attempting to detect relatively small annual changes in carbon, often in soils with high C background levels. To increase the viability of direct sampling, fields can be aggregated to generate large-scale projects and measurement can be delayed multiple years to allow soil carbon accumulation. Unlike models, direct sampling can integrate GHG outcomes across management practices and crops and can include innovative techniques not yet thoroughly studied by researchers. If the objective of a program is to foster innovation in agricultural management approaches for soil carbon mitigation, field sampling is likely the preferred quantification approach. But field measurement of N_2O and CH_4 fluxes will not be viable for farmers or project developers for the foreseeable future. Therefore, models will likely be required to quantify changes in these fluxes even where field sampling is used for carbon.

If the objective of a program is scaling up the use of known management practices, rather than innovation, models may be a more cost-effective option, particularly for a large national or regional program for which encouraging high levels of participation is critical. Landscape-scale or farm-scale modeling approaches can provide relatively high

11. Miranowski (2005) indicated a total U.S. agricultural energy use in 2002 of 1.7 quadrillion BTUs, of which 8.5% was gasoline and 27.3% was diesel. Total U.S. cropland area is 124 Mha (USDA NASS 2007). Conversions for diesel and gasoline CO_2e coefficients were obtained from the U.S. Energy Information Administration at http://www.eia.doe.gov/oiaf/1605/coefficients.html.

12. This figure is calculated from the carbon content of fuel and a series of California crop production cost reports published by University of California Cooperative Extension (http://coststudies.ucdavis.edu).

accuracy and, given existing data, are likely to be viable for national programs in the United States. Landscape-scale approaches, which can use either up-scaled process models or empirical extrapolations from regional data, are easy to use and entail low costs, embed standardization and thus reduce error, have low data-input requirements, and require minimal expert input. But these approaches are often designed around specific practices and thus may not as easily integrate the interaction effects that result from multiple practices in combination. In contrast, farm-scale approaches, which use process-based models, can integrate the interaction effects of multiple practices, in part by requiring more site-specific input data. Farm-scale approaches require input data from producers and may be more complex to verify. Incorporation of a decision support tool like COMETFarm may be the best option for a farm-level approach. Without such a tool, farm-scales approaches can be more complicated and expensive to apply and can require greater expertise to implement without introducing errors and bias. Conceivably, process-based models could be used at farm scale, but emissions factors from pTier 2 or pTier 1 sources would be needed to fill gaps. As research advances, the process models used in these approaches must add coverage and improve certainty for different regions, cropping systems, and management practices.

Table 18. Viability of methods for quantifying GHG change for new types of management

Management type	Field-based (C only)	Model-based (C, N_2O, and CH_4)		
		*pTier1**	*pTier2*	*pTier3*
Land use change	Yes-d		Yes	Yes
Manage soil carbon on crop land	Yes-d		Yes	Yes
Manage N use for N_2O reduction		Yes		Yes†
Manage CH_4 through crop management		Yes	Yes	Maybe
Manage rangeland C by amendment	Yes-d		Maybe‡	Maybe‡
Manage rangeland C by animal management	Yes-d		Maybe‡	Maybe‡

Yes-d—The viability of this method depends on SOC and spatial variability. High SOC and spatial variability make field sampling difficult and expensive, especially if the annual changes in soil carbon are small relative to background carbon.

** Only use Tier 1 if no other more accurate method is available. Tier 1 likely will provide insufficient certainty for many protocols and programs in the United States.*

† Tier 1 is likely needed for offsite N_2O (from leached and volatilized N sources) and may require several measured field data inputs.

‡ Process-based models that integrate pasture/range productivity and soil carbon dynamics with livestock-based emissions of nitrous oxide and methane are still in development.

Accounting Procedures

GHG-accounting protocols or programs normally include detailed guidance for setting project boundaries; determining which GHG sources, sinks and reservoirs are included/excluded from accounting; quantifying project and baseline GHGs; monitoring and verifying GHG projects; determining additionality; estimating leakage; and assessing reversal risk. These accounting procedures are described below. In addition, the availability of data and methods needed for accounting for the range of agricultural practices covered in this report is assessed.

Setting project boundaries, including GHG assessment boundary

GHG accounting protocols generally begin by requiring the project proponent applying the protocol to define the boundaries of the project activity, including the physical boundary (lands included in the project, geographically delineated, which in the case of aggregated projects may include many farms), the temporal boundary (project start date and crediting period), and the GHG assessment boundary. The GHG assessment boundary identifies the GHG sources, sinks, and reservoirs (SSRs) included in, and excluded from, project accounting. Some SSRs will be included as mandatory because they represent the major C pool or emission source targeted by the project activity, or because the SSR may be affected significantly by the project activity and could result in emissions whose exclusion from accounting would not be conservative. Other SSRs may be excluded (or left optional to include), either because their exclusion is conservative—excluding this SSR will tend to underestimate net GHG reductions and thus credits to the project—or the SSR can be demonstrated to be insignificant, that is, it falls below a defined *de minimis* threshold.

Additionality and baseline

Additionality is a criterion that is often required for GHG mitigation projects, particularly if those tied to a regulatory program. An additional project is one that would not have been implemented without the policy or program and thus it produces GHG mitigation that would not have been generated otherwise.[13] In a regulatory setting, these additional tons can be used to offset emissions by a regulated entity, and in other nonregulatory incentive systems, additionality gives buyers assurance that their investment achieved environmental gain. A variety of rules are used to assess additionality (Olander et al. 2008; Trexler et al. 2006; World Resources Institute 2005). These tend to be yes/no tests:

- ***Regulatory test***—Is the action required by law or regulation? If so, it cannot be considered additional.
- ***Start date***—Did the action or shift in management take place after the date set by the policy or program?
- ***Financial test***—Did the action already receive government program funding? Would the project have been financially viable without the financing from the mitigation program?
- ***Barriers test***—Do other barriers (e.g., technological, institutional) stand in the way of business-as-usual adoption of the action?
- ***Common practice***—Are the actions taken common practice in the region or for the industry, that is, already implemented by enough producers that the actions might be considered business-as-usual? If so, the actions are not additional. Evaluation of common practice adoption rates can be the basis for some types of performance standards (see Box 8).

Box 8. Performance standard

The term performance standard is used in a few different ways, but its most common usage for GHG offsets protocols is as a type of threshold additionality test based on common practice, technology, or baseline emissions or sequestration rates. The Western Climate Initiative defines a performance standard approach as follows:

A performance standard approach seeks to determine through initial study of a sector or project type what level of performance is necessary to provide confidence that projects meeting or exceeding the standard are additional. The standard may be the identification of a particular technology (such as a methane digester) that is nearly always additional to common practice or the establishment of a set performance baseline emission rate that project reductions are measured against.

http://www.westernclimateinitiative.org/component/remository/func-startdown/124/

Other important references:

Climate Action Reserve Program Manual (section 2.4.1). http://www.climateactionreserve.org/wp-content/uploads/2009/04/Climate_Action_Reserve_Program_Manual_031610.pdf

U.S. EPA Climate Leaders Program Protocol Guidance http://www.epa.gov/climateleaders/documents/resources/OffsetProgramOverview.pdf

Of these yes/no threshold additionality tests, only the common practice or performance standard approach (see Box 8), which uses a threshold to be set on the basis of business-as-usual or standard practices, requires the program, rather than the project developer, to assemble data and technical documentation. The data needs and assessment are the same as those described below in the baseline section.

An alternative approach for addressing additionality, which applies a discount to adjust for non-additional projects rather than a yes/no test, is called proportional additionality (Willey and Chameides 2007).[14] Proportional additionality is set relative to the industry uptake of a practice, or the common practice in a region, which may be the same as the baseline (discussed in next section). Thus an individual project is compared against this regional industry standard and discounted on the basis of the practices used on or GHG emissions from surrounding farms. If neighboring farms have already instituted many mitigation practices, the assumption is that barriers to these practices are relatively few and that the project is less likely to be additional, so it would receive a high discount. A proportional additionality approach can be used in combination with other additionality approaches. The proportional additionality approach is discussed further below because it also requires information on the industry or sector as a whole—the same data needed for establishing baselines.

13. If pre-project management is atypical and gives less than normal financial return, projects may be able to use common practice as the baseline to show additionality, rather than using historical practice.

14. Proportional additionality allows incorporation and crediting of early actors by discounting credits so that the total number of credits add up to only the new (additional) mitigation achieved (Western Climate Initiative 2010; Willey and Chameides 2007). In a market or incentive program, this strategy tends to reduce the price paid for each credit registered or for each new unit of GHG mitigation achieved.

Baseline determination

A baseline represents the likely activities—and associated GHG emissions and removals—that would happen at a particular project location or region without a policy, program, or project in place. Baselines can be set at the outset of a crediting or contract period and remain set for that period (static), or they can be allowed to fluctuate over time as various input factors, such as weather conditions, number of livestock per hectare, or regional climate trends, change (dynamic). This decision is a policy decision, but it has implications for producer participation and compensation. Baselines can be established in either of two general ways, each of which lends itself to a different use.

- A ***project-specific*** approach at the project or farm scale is based on current activities and trajectories of change. This type of baseline would be used to quantify project outcome and determine number of credits or level of incentive that a project deserves.
- A ***standardized*** approach is based on current sectoral or industry trends for each management practice, crop, and region. For agriculture, this type of baseline may be embedded in standardized crediting approaches and could be used for proportional additionality approaches but is not likely useful for determining crediting or incentive level.

Project-specific baseline

A project-specific baseline can be determined at a field or a farm level with field sampling or modeling. Used to determine project achievement, the baseline is compared to after-project net greenhouse gases to assess level of incentive or credits deserved (project net greenhouse gas – baseline net greenhouse gas = crediting level). The Clean Development Mechanism, the world's largest offset program, uses a project-specific baseline method. Some of the voluntary registries like the American Carbon Registry and Verified Carbon Standard have also used this approach in some of their protocols.

If using a project-specific baseline, each project estimates what its GHG fluxes would have been without the project. The project developers have two ways to use field-based measurements for their baseline. First, they can take a full set of field samples and measurements before instituting a management change and use these data to determine the static baseline. The concern with this approach is that background levels of soil sequestration and natural annual variation in GHG emissions may change year to year even without management changes. An unaccounted-for background trend of soil carbon loss under the original practices would mean that a project that shifted the system to carbon accumulation would be undercredited for the avoided carbon losses. Alternatively, overcrediting could occur if the system was already accumulating carbon. Annual variation in nitrous oxide and methane fluxes can be significant, meaning that the time zero data point may not be an average year and thus may not result in accurate crediting.

A second field approach is to use a reference site or multiple reference sites on the project lands. These sites are left in the original management state and are measured each time the project sites are measured; the reference samples are used to determine the (dynamic) project baseline. This approach addresses all of the concerns regarding background trends in carbon and variability in other greenhouse gases. However, it may raise other issues: the costs of maintaining land in original practices and concerns about how surrounding land management may affect the reference sites. Because nitrogen is mobile (e.g., nitrate leaching and deposition), an emission reduction at a project site could also result in a reduction at the reference site if it is embedded in project lands. Changes in moisture or tillage-induced translocation of soil materials on surrounding lands may also affect GHG fluxes on an adjacent reference site. Therefore, the choice and design of reference sites is critical to the quantification of greenhouse gases. Verification of field measurements used to set a baseline may also be complex and can add a risk of high uncertainty to a project (see verification section).

The primary concerns about project-level field sampling approaches are the costs and risks for the producer or project developer. The use of process-based models with standardized user interfaces (pTier 3) or standard equations with predefined emissions factors (pTier 2) may help reduce costs and risks inherent in field measurement, while still providing transparent and verifiable farm-level baselines. If models are being used for quantification, they can of course also be used to develop a baseline. Although a less common approach, models can also be used to develop a project baseline when field measurements will be used for quantification. Models can be used to help develop more robust baselines when a point sample (one time point) is insufficient—perhaps using data from this point sample as input into a model that can capture likely annual variability and background trends. Project developers have two fundamentally different options for developing a modeled baseline scenario: (1) simple forward extrapolation of historic trends (both in management and in physical processes) and (2) forward predictions that use historic trends as a starting point but also consider other likely drivers of change. These drivers could include new policy mandates that may shift prices for certain crops, the impacts of climate change on growing seasons or rainfall in a region, and diffusion of new technology. The primary process models described above are parameterized on the basis of historic trends and do not normally incorporate other major drivers; however, adjustments to the models could be made to consider alternative baseline scenarios that could incorporate predicted trends.

Predicted trends add uncertainty and would most likely be estimated across large regional scales, with farm- and project-scale effects. Most of the data available for determining baseline conditions for process models are current or historical, not projections of potentially significant new drivers (see the management data section below). USDA Outlook reports are one source that would incorporate the likely impacts of existing policy on total yields and crop area. Economic optimization models such as the FASOMGHG model are another option for producing scenarios of crop and management shifts driven by various policies. Although not available to the public, this tool is being updated with input from the USDA and EPA and could possibly be used to generate projections for use in baseline determination. Given the difficulties, uncertainties, complexity, and cost of modeling most of these additional drivers of agricultural management choices, modeling predicted trends for baselines may not be a viable option. However, adjusting extrapolated historical trends to account for significant new drivers through rough adjustments might be feasible and would ensure conservative crediting. Linear extrapolation of historical trends is relatively straightforward when data are available. Linear extrapolation uses national or regional data on the current (pre-program) crop distribution and agricultural practices by region, which are then scaled by the practices' average GHG performance. Data can come from standardized sources but have limitations that diminish confidence in how they will estimate trends.

If a model is used for quantification and crediting of a project, it should be used to determine the baseline. If field measurement is used for quantification and crediting, but a model is used to determine the baseline, the model scale should align with the project scale for highest accuracy. Using a model to determine a baseline scenario at the farm scale, as opposed to a regional scale, will require greater site-level specificity; farm-level input data and site-level sampling, as opposed to default data. Process-based models will be needed when site-level specificity is desired, and like field measurement, should be used in a consistent and transparent manner, which may require a user-friendly decision support interface. As noted above, models will be limited in their capacity to incorporate innovation and less common practices and crops until data are available. Thus, in some cases, particularly those focused on innovation, field sampling may be the best option. However, with increasing levels of understanding of the biophysical processes affecting GHG fluxes, models have the further advantage of aggregating multiple activities that may be limited in actual field data.

Standardized or regional baseline

A standardized approach for setting baselines would account for industry trends, such as the current use of permanent no-till or irrigation management for specific cropping systems, at a regional or national level. This type of regional baseline may be used as a test of additionality for common practice or proportional additionality. A common practice test is industry technology standards that projects must meet to qualify. For example, where the use of a technology or practice like nitrification inhibitors is below some threshold of use across the industry/sector, its use would be considered additional. The threshold could also be based on a net background emissions or sequestration rate rather than specific activities. Although standardized approaches can greatly simplify project accounting, increase transparency, and reduce project or program costs, they can also decrease participation. For example, if a standard baseline is used to show that more than 50% of producers are using a particular technique and thus determines that the technique is non-additional and therefore ineligible, those who likely have the most mitigation potential will have no incentive to participate and contribute.[15] If instead an additionality threshold is based on current rates of carbon sequestration, it may bias the program against producers with poor-quality soils.

Standardized baselines are often designed by practice, which works well for clearly separable activities (e.g., new industrial process). But in agriculture, numerous management practices can be used in combination and can have interactions that affect overall performance (net greenhouse gases). Thus in some cases, rather than developing a baseline for carbon sequestration due to changes in tillage practice alone, it may be worth considering how to integrate soil C management practices. The same can be said for N_2O or CH_4 management. The method for accomplishing this task will likely depend

15. The other line of argument here is that if 50% of farmers are already using it, the technique is likely cost-effective and will not need financing to spread further, and therefore it should be part of the baseline and considered non-additional. If uptake of a practice has been increasing, this assumption may be reasonable; however, uptake of many practices will plateau, suggesting that incentives may help expand uptake to regions where it is less cost effective.

on the approach used to determine the baseline. If historical trends were calculated and averaged by practice and region, practice-specific baselines would be produced. If instead these trend data were input in process models, they could be used to create baselines for management systems that include a GHG estimate for multiple practices. For agriculture it will be important that standardized baselines be set at a fine enough scale that baseline emissions or sequestration will be relatively uniform to prevent a selection bias whereby enrollment consists primarily of lands where the baseline is favorable.

A standardized baseline that uses industry and regional data trends can also be incorporated into regional estimates of net GHG outcomes from various region-crop-practice combinations. This task is described above as a pTier2 approach. In this approach, empirical or process models are used to come up with default emissions factors for various management shifts. The modeling can take into account the background industry/regional trends on the basis of historic trends alone or with some estimate of other critical drivers. Thus, the emissions factors can be developed with a baseline that assumes no uptake of a practice on a regional level, which may be appropriate for a program that has strict additionality rules to keep out early actors. On the other hand, where a proportional additionality approach is used to include early actors, emissions factors can account for the current level of uptake of various practices, incorporating discounting in regions where uptake is already significant. So, standardized baselines become embedded in these standardized quantification approaches.

Management data for baseline formation

Data on regional or industry trends in management can be used to develop baseline scenarios for (1) process models that can be used at the farm scale for direct project quantification or at the regional scale to develop emissions factors and (2) empirical models to determine emissions factors. These data can also be used to develop baseline scenarios for establishing a common practice or performance standard test for additionality. Data on existing levels and types of management could be incorporated into models to establish a realistic baseline scenario (e.g., level of fertilizer use) and, if desired, to develop proportional additionality discounting (e.g., regional update of no-till practices) approaches to account for early actors. Where farm- or project-scale approaches are used, many of these management data can come from site-specific input from producers or project developers.

A realistic baseline requires knowledge of, and data pertaining to, agricultural practices under business-as-usual conditions—that is, current (pre-program) crop distribution and agricultural practices by region and scaled by the practices' average GHG performance, management trends, and projections for baseline practices. However, agricultural management data representing observed farming practices tend to be incomplete. Moreover, they often require use of multiple data sources to create a realistic portrayal of a representative farm.

Major agricultural management data sources in the United States are discussed below. Particular attention is paid to the ways that the data can be used to create a baseline for various mitigation activities. It appears that existing management data have major limitations with regard to this task. For several of the activities focused on in this report, these data provide little to no information necessary for baseline formulation. In these cases, literature or public reports on regional pilot projects will be needed to develop accurate performance measurement. Accordingly, use of farm-scale modeling tools would be preferable for such activities.

USDA Agricultural Census and National Agricultural Statistical Service (NASS)

The wealth of data reported by USDA-NASS could be used in national or regional baseline formulations. USDA Agricultural Census data provide the most comprehensive set of information regarding cropping trends, productivity, and agricultural land use. The major weakness of these data is that they offer little insight into specific production technologies associated with each crop.

Nevertheless, protocol developers who require estimates of crop area totals and yield can calibrate baselines with USDA-NASS or Agricultural Census data. USDA-NASS provides statistics on county-level planted area, harvested area, and yield. These data are compiled for most major crop and livestock commodities, on an annual basis, and include geographic mapping information for individual counties. These data can provide useful information regarding use of winter cover crops as well. Where cover crops such as rye, wheat, or hay are planted but not harvested, NASS data will still provide an area estimate of baseline cover-crop adoption. NASS also reports on farm-related expenses for most major inputs (including fertilizer), offering an avenue to assess farm-level input use by region (though such a methodology would require additional effort to disaggregate input use by crop).

The USDA Agricultural Census provides extensive details on U.S. farming trends but is available only every five years; the latest census release was 2007. It provides a bit more detail on management variables relevant to environmental quality. Specifically, the census reports on the area receiving synthetic (nitrogen) or organic (manure) treatments in a cropping system, by county. Additional data sources on fertilizer sales by state and county could be coupled with these USDA data to estimate application rates per unit area. The agricultural census also provides a detailed look at agricultural land use through the Major Land Use Database (which defines crop and grazing land uses differently than the National Resources Inventory—described below). This information is especially important for mitigation on grazing lands, where land use data can be coupled with annual production histories and USDA agricultural census data to form baseline grazing rates per unit area.

In addition to the Agricultural Census and annual NASS data, USDA compiles and manages other data that are useful for baseline development, including the Economic Research Service's Long-Term Agricultural Projection Tables. These tables provide a forward-looking baseline of expected cropping patterns, productivity, and agricultural land use. Such projections can be used to establish regional crop mix baselines, or in the case of emissions intensity-based mitigation protocols, parameters representing baseline yield growth over time. A multiyear record of tillage practices (2003–2006) has also been collected by the USDA-NRCS through from the National Resources Inventory (NRI-CEAP), but the record is limited to the upper Mississippi River basin (USDA NRCS 2010). These data may be a useful tool for (1) determining the baseline occurrence of continuous no-till management, as opposed to biennial or intermittent tillage, and (2) comparing with other data sources for this region.

USDA Agricultural Resource Management Survey

Perhaps the most comprehensive crop management dataset available is the USDA Agricultural Resource Management Survey (ARMS). ARMS data are crop- and state-specific and rely on surveys administered to real producers to obtain information about on-farm management practices. Although the survey does not cover the entire productive land base for each crop, it includes enough U.S. states to represent 90% or more of each surveyed crop. ARMS focuses on most major crop commodities and on some fruit and vegetable systems. The USDA uses an involved survey methodology carried out in three phases to create a comprehensive dataset of landowner responses to questions regarding production decisions and economic returns to production. Once these data are compiled and processed by USDA, they are made publicly available at the state and national levels through an online dissemination tool available at http://www.ers.usda.gov/Data/ARMS/CropOverview.htm.

As mentioned, the ARMS survey collects data on a multitude of management activities relevant to GHG mitigation and allows users to explore the interactions of different activities to illustrate how a baseline for one activity (say, N application rates) can vary by other management choices (such as tillage practice or irrigation system). State-level results can provide great insight into production trends that can be used to form a baseline representing contemporary farming practices. Greater public access to the raw ARMS data would allow protocol developers additional insight into the variation within and among management activities (e.g., fertilizer N application rates as a function of timing and application method), leading to a more robust regional baseline. However, obtaining raw ARMS data with a greater spatial resolution requires USDA approval and collaboration.

The ARMS database contains regional area totals for several tillage practices, including conventional tillage, no tillage, ridge tillage, mulch tillage, reduced tillage with <15%–30% residue harvest, and conventional tillage with <15% residue harvest. Area totals can be evaluated by crop and several points in time to show how regional tillage practices have evolved over time, and, if a reasonable trend emerges, can be extrapolated into the future to form expected baseline area totals by tillage system. Another advantage of the ARMS data is that project managers can show how other management practices (for instance, fertilizer N use and application methods) vary by tillage system, which allows for more comprehensive energy and GHG accounting. However, ARMS does not distinguish between permanent no-till management or conservation tillage and rotational or intermittent implementation of these activities.

For fertilizer use, ARMS distinguishes between multiple synthetic fertilizers (N, P_2O_5, and K_2O) and reports the percentage of total area to which manure is applied. For fertilizer N, publicly available ARMS data report the total proportion of crop area treated with nitrogen and the average per-hectare use (by crop and state). Additionally, application timing is reported as the percentage of total area receiving nitrogen in the spring before planting or in the fall (this reporting does not allow split applications to be separated out). Alternative N application methods reported include direct injection and broadcast with or without incorporation as well as the percent of total area where nitrification inhibitors are applied to the field. Nutrient application rates vary by alternative timing or application methods, but the publicly available data cannot be used to filter out application rates by timing *and* application method. Even with these shortcomings, the ARMS data on N use provide a detailed look at N management trends as solicited from actual producers.

With regard to other activities assessed for GHG mitigation potential, ARMS includes data on the number of tillage operations in a year, irrigation technology and water use, use of conservation buffers, pest management practices, and herbicide application. These data are helpful in capturing the full upstream CO_2 emissions associated with baseline practices. Other activities reported (amount of residue left on the field, previous crop) are not specifically considered GHG mitigation activities but may be useful model inputs.

In general, ARMS represents the most comprehensive set of management data for evaluating trends in tillage and N management at the state and national level. One glaring weakness of the ARMS data is relatively coarse spatial aggregation; within-state variability in management practices by county (or smaller areas) cannot be assessed. Another weakness is the mismatch between approaches for data aggregation with county management data and emissions estimates based on management areas with similar soil and climate regimes (MLRA), but this weakness can be addressed with data-weighting approaches. In addition, data are not collected for every crop, every year, possibly leading to inconsistent time series trends. Nevertheless, the ARMS data represent perhaps the best opportunity to create a baseline with interactions among practices. For this reason, ARMS data have been used to calibrate production functions and crop budgets in models such as FASOMGHG.

National Resources Inventory

The National Resources Inventory (NRI) is a spatially explicit land use/cover dataset compiled by the USDA Natural Resource Conservation Service (NRCS). This dataset provides a detailed look at past and contemporary land use trends on nonfederal lands. NRCS collects data by dividing the U.S. nonfederal land base into approximately 300,000 sample segments and by surveying specific locations within those segments. It surveys individual points within those parcels at various points in time. The NRI categorizes land resources (including cultivated and noncultivated cropland, forestland, pasture, rangeland, and CRP) and explicitly tracks the evolution of individual survey points, thus cataloguing changes in land use and environmental conditions over time.

These data may not offer a clear look at most of the mitigation practices discussed in this report, but land use and land cover changes could be an important component of regional or national baseline development, particularly when the focus is on changes in C sequestration and storage. For example, management of private grazing lands for mitigation or conversion of marginal cropland to pasture for enhanced soil C accumulation requires some knowledge of how these land use changes have occurred in the past and are expected to continue under baseline conditions. Additionally, land use transitions into crop production from alternative uses (forestland, pasture, or conservation set-asides) can alter the soil C stock, affecting the mitigation potential of various practices. Hence, fully establishing mitigation potential at the farm or local level could require information on previous land use changes at the parcel level.

NRI data might also be useful whenever terrestrial mitigation practices overlap to an extent with traditional land conservation or sustainable farming practices that are assessed by the NRCS through the NRI. Land movement into or out of the Conservation Reserve Program (CRP) or observed transitions from cultivated crop production to grazing lands can be evaluated using parcel-level NRI data, ultimately forming a region-specific or national baseline of expected land use transitions. Alternatively, conservation practices that reduce soil erosion are evaluated by the NRI, and these practices could overlap with mitigation efforts (such as wetland preservation).

Conservation Technology Information Center's National Crop Residue Management Survey

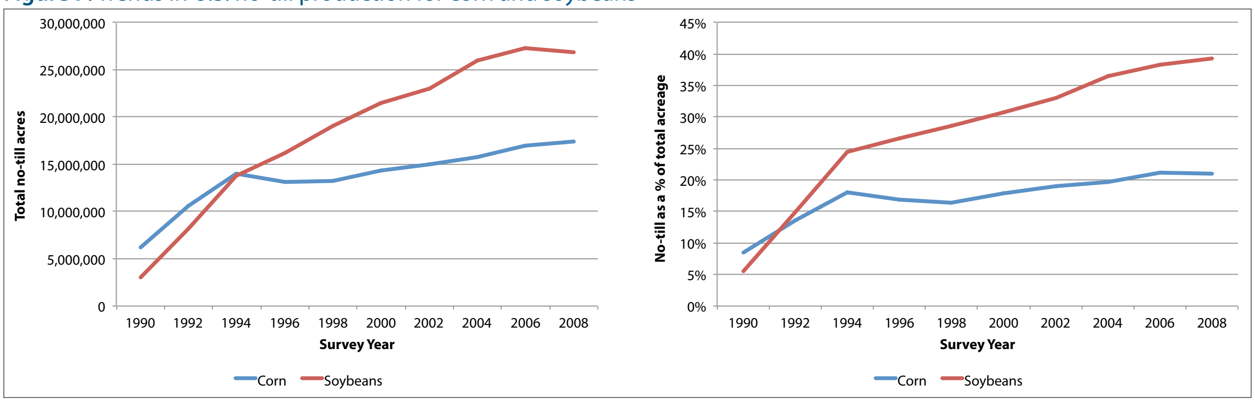
The Conservation Technology Information Center (CTIC) conducts a National Crop Residue Management Survey (NCRM) to obtain county-level information about tillage practices by crop type and collects yearly data from voluntary reporting sources on county-level tillage practices. This database represents the most spatially disaggregated national database of tillage practices in existence. The CTIC defines multiple tillage categories, including conventional tillage, no tillage, ridge tillage, mulch tillage, reduced tillage with 15%–30% residue harvest, and conventional tillage with 0%–15% residue harvest. Although these data are collected in a different manners (see below), the definitions of tillage categories are the same as those for the ARMS survey of farm operators. Data are available from 1989 to 2008 in summary form and in more detail with a CTIC membership.

CTIC estimates come from one of two data collection procedures conducted by the local Extension Service and Soil and

Water Conservation District, local producers, other local partners, and the NRCS (CTIC 2008). Local partners select one of the two data collection procedures, either roadside transects or expert estimates based on local knowledge. The collection procedure also considers cropland area in the county, cropland density, road system type, and tillage adoption history. CTIC estimates are used in many peer-reviewed publications (Follett 2001; Lal 1997; Sperow et al. 2003; Eve and Sperow et al. 2002) to estimate national tillage trends.

CTIC data can be used to examine trends in no-till or conservation tillage areas by region and commodity. Figure 7 illustrates national trends in the total no-till area and the proportion of no-till to total harvested land area for corn and soybeans. A baseline projection for no-till land area, regardless of whether the baseline is formed at the national, state, or county level, could use CTIC data and extrapolate from observed trends to form expected total areas—an important baseline parameter for ensuring that credits from tillage practices are additional to business-as-usual (BAU) projections. Figure 7 displays how adoption of no-till practices for corn and soybean production has grown over time, though that growth has perhaps tapered off in recent years. Horowitz et al. (2010) use ARMS data to calculate that no-till area increased at a median rate of 1.5% per year (from 2000–2007) for four major U.S. crops: corn, soybeans, rice, and cotton. This trend is not consistent in all states.

Figure 7. Trends in U.S. no-till production for corn and soybeans



However, CTIC data do not document continuous implementation of alternative tillage practices, because national surveys are not conducted yearly and are not necessarily attributed to the same farmland each survey year. Horowitz et al. (2010) use the NRI-CEAP Cropland Survey (a component of the previously mentioned NRI data but with coverage limited to the upper Mississippi River basin) to calculate how much no-till land is continuously versus intermittently under no-till management. They found that approximately 50% of all land in no-till management at some point during the three-year survey period was tilled during another year of that period. If this proportion is consistent throughout all U.S. no-till area, the total amount of continuous no-till area could be approximately half of that reported by the CTIC. The CTIC survey does not cover the entire national land base. Approximately 112 Mha of farmland were included in the 2004 NCRM, which is approximately 90% of the U.S. cropland base, but in 2008 a much smaller land base was surveyed due to lack of funding. Finally, these data are not publicly available and cost money to acquire, which could limit their use by stakeholders, protocol developers, or academics.

Nutrient Use Geographic Information System

The Nutrient Use Geographic Information System (NUGIS) provides spatially explicit information on nutrient use across crop and livestock systems, using data compiled by the International Plant Nutrition Institute (IPNI). NUGIS incorporates multiple data layers into a geographic information system (GIS) to estimate fertilizer application, crop nutrient removal, and excreted and recoverable manure nutrients at the county level and the 8-digit hydrologic unit, or watershed level. The nutrient balance data for NUGIS are reported for the five census years from 1987 until 2007. To estimate the nutrients applied at the county level, NUGIS uses commercial fertilizer sales data from the Association of American Plant Food Controls (AAPFC). These data are reported in tons of N, P2O5, and K2O contained in fertilizers sold in a given census year. Fertilizer control offices for most counties in each state (more than 70% of all counties in the lower 48 states) report sales data annually.

NUGIS also estimated excreted and recoverable manure nutrients for each county. It calculated state-level excreted manure volume and manure nutrients for each state and apportioned them to counties on the basis of Kellogg et al.'s (2000) procedure that compiles agricultural census data to estimate animal units, excreted manure, manure nutrient content, and manure recoverability for livestock, poultry, and swine. County estimates were apportioned using Kellogg et al. (2000) reported values for 1985–1997 and calculated using Agricultural Census-derived coefficients for 2002–2007.

NUGIS determined nutrient removal and N fixation for crops using planted and harvested area, average yield, and production data at the county level from the USDA-NASS, the USDA Agricultural Census, and USDA Economic Research Service. It multiplied these data by generalized crop-nutrient removal coefficients (IPNI 2010). The methodology accounts for removals by specialty crops not included in the 21 NUGIS crops using a state adjustment factor.

The IPNI report (IPNI 2010) explains the several caveats to applying NUGIS data. For fertilizer use, the caveat is uncertainty about whether sales data can be fully attributed to use in a particular county, and sales data do not imply crop-specific application rates. For the 30% of counties without AAPFCO data, census data is used, and the same N:P:K ratio is attributed statewide (because census data report total fertilizer expenditures and do not delineate by nutrient). While using a consistent ratio may be reasonable in states with consistent soil types, it could be more problematic in places where soils and cropping systems are more variable. Farm to nonfarm coefficients of fertilizer use are based on 1987–2001 data and may not reflect changes in use. For nutrients from manure systems, lack of animal unit data at a county level is problematic, because temporal changes in feeding and management systems could be better reflected with this kind of detail.

State extension reports and miscellaneous

Each national data set has potential shortcomings for baseline development, but region-specific information could help fill the void. For example, most state extension services will publish bulletins or reports on best management practices by crop as well as data on typical crop budgets by crop and management practice. For example, extension services in Texas, California, and Iowa provide easily accessible information on returns and expected input use across a wide range of commodities and management practices.[16] This information can help producers to maximize returns. It also can be used to form a regional performance-standard baseline that is well documented, considers most major sources of input use and yield responses of different practices, and comes directly from regional experts familiar with producers and farming practices typical for the area.

Additionally, reports from the USDA Economic Research Service, or publications from field experiments and data collection on the ground, can supplement national or state-level data where needed, especially for practices that are just beginning to gain attention in policy circles for mitigation potential. For example, to develop a baseline for alternative rice management practices, project managers might draw information from Garnache et al. (2011). For nutrient use practices, such as the use of inhibitors, a wealth of relevant scientific articles and extension reports specific to N management are available. For example, see Nelson et al. (1992) for information on N inhibitors in Iowa.

Table 19 attempts to match specific management activities with the data needed for GHG-flux baseline formulation. Where existing national data might not be available—or may have significant limitations—local information can be complimentary in baseline development. Information on activities with a "No" entry in the data availability column may come from other sources.

16. See http://agecoext.tamu.edu/resources/crop-livestock-budgets/by-commodity.html; http://coststudies.ucdavis.edu/; and http://www.extension.iastate.edu/agdm/crops/html/a1-20.html.

Table 19. Data sources for developing performance standards and baselines for U.S. agricultural mitigation practices

Activity	Data requirement	Data available	Data source(s)	Caveats
Switch to conservation tillage	Adoption rates for conservation tillage	Yes	ARMS, CTIC	Cannot distinguish between permanent and rotational conservation tillage ARMS has state-level data only
Switch to no-till	Adoption rates for No Till	Yes	ARMS, CTIC	Cannot distinguish between permanent and rotational no till. ARMS is state-level data only
Use winter cover crops	Cover crops used (species, land area) at local or region scale	Yes	USDA-NASS Ag. Census	Only reports land area and production totals Not consistently mapped with primary production systems
Diversify annual crop rotations	Baseline crop mix at farm or local scale	Yes	USDA-NASS	Might exclude some crops "Diversification" might introduce crops for which data are lacking
Incorporate perennials into crop rotations	BAU perennial crop area and yield	No		
Replace annuals with perennial crops	BAU perennial crop area and yield	No		
Switch to short-rotation woody crops (SRWCs)	BAU SRWC area and yield	No		
Introduce agroforestry (windbreaks, alley cropping, etc.)	Baseline adoption by region	No		
Apply organic material	Manure use by cropping system and region	Yes	ARMS NUGIS	ARMS only reports % of cropland area to which organic material is applied, not total use per unit area Yield response to organics as a replacement to synthetic N requires local field tests
Apply biochar	Baseline application rates	No		Still experimental
Set aside cropland or plant herbaceous buffers	Observed transitions out of cropland (especially for marginal cropland); Baseline use of herbaceous buffers	Yes	NRI Agriculture Census (Major Land Use Database)	Only the NRI allows users to track land use changes over time; other sources report aggregate trends The NRI only reports total area in land use categories; it does not tie that land area to specific production activities, Only NRI identifies land use and landowner characteristics, along with land uses County-level USDA land use data only available in Agriculture Census years
Reduce fertilizer N application rate	Application by crop and spatial scale	Yes	ARMS NUGIS Agriculture Census + other sources	ARMS is only available at the state level Other data sources are not crop-specific NUGIS infers application rates for different crops and cannot isolate variation in N use across different management regimes
Switch to slow-release fertilizer N source	Baseline use of slow-release technology	No		
Change fertilizer N application timing	BAU application timing	Yes	ARMS	Does not report split applications (only fall versus spring) Cannot isolate difference in application rates for split applications No reported yield response to alternative application timing
Change fertilizer N placement	Baseline use of fertilizer N banding or depth of injection	No		
Use nitrification inhibitors	Observed inhibitor use totals by crop and spatial scale	Yes	ARMS	No reported yield response to inhibitors No reported change in N application rates if N inhibitors are used
Improve irrigation management (e.g., drip,)	Baseline irrigation practices	Yes	ARMS USDA-NASS NRI	Most data report on irrigated versus dryland production, not irrigation intensity Data on irrigation systems might require locally published documents and reports (extension reports, USGS, etc.)
Manage farmed histosols	Baseline histosol land area, management information	No		

Activity	Data requirement	Data available	Data source(s)	Caveats
Set aside histosol cropland	See above	No		
Adjust rice water management to reduce CH_4	Baseline use of midseason drainage			No national data Local data available from observed rice systems in Californiamight not apply to rice systems in the southern United States
Plant rice cultivars that produce less CH_4				
Restore wetlands	Baseline restoration rates, previous land use	Yes	NRI	
Convert cropland to pasture	Observed crop-to-pasture transitions	Yes	NRI Ag. Census (USDA-MLU)	MLU only reports land use totals NRI distinction among cropland, noncultivated cropland, and intensively managed pasture only vaguely defined
Improve grazing management on rangeland	Grazing-land use data, stocking rates	Yes	NRI USDA-MLU Ag. Census USDA-NASS	
Improve grazing management on pasture	Land management trends, input use (e.g. fertilizer, etc.)	No		
Manage species composition on grazing land	Baseline species composition	No		
Introduce rotational grazing on pasture	Data on rotation patterns, including frequency with which parcels are grazed	No		

Monitoring and verification

Program administrators need to decide the level and type of monitoring and verification necessary for their program. Many government incentive programs have reporting requirements with periodic auditing, whereas most private offset market programs and proposed regulatory offset programs have required third-party verification of GHG performance for every project for every reporting period. Monitoring and verification can be used for a range of purposes that should be explicit in program design. These purposes could include

- checking for reversals of C sequestration activities and accounting for such reversal events;
- verifying maintenance of intended management practices;
- ascertaining whether quantification procedures and calculations are correct;
- ensuring data integrity and consistency with project plans and quantification protocols; and
- determining whether expected outcomes are being achieved and, if not, adjusting crediting.

If all of these purposes are included in program design, monitoring and verification may require paper reviews, site visits, and sampling.

Monitoring

To check for reversals, a visual inspection through a site visit may be sufficient. Reversals require releases of stored carbon, which are visibly traceable through loss of aboveground woody biomass or shifts in tillage practices. Alternatively, remote sensing products may be available to track tillage and cover crop practices (Brown et al. 2010; West et al. 2008). Regular monitoring for maintenance of management practices is relatively simple through visual checks for carbon but not for nitrogen management and methane management, which may involve variability in fertilizer input and drainage events. Thus, farm records of fertilizer purchase and application, fuel purchase and equipment use, and, possibly, yield data may be necessary. Some states are moving toward nutrient management plans to reduce major sources of nitrogen loading into waterways. For example, California certifies manure application and tracks other fertilizer use on agricultural lands. These processes could perhaps be used for verification.[17]

17. https://www.certifiedcropadviser.org/about-the-program; http://www.cacca.org.

Monitoring achievement of outcomes will in some cases be an unrealistic expectation. If a project uses field-based quantification, selective resampling of soil C or GHG fluxes in fields would help managers track outcomes. This sampling is probably too costly for individual projects to bear. However, sampling on a programmatic scale with auditing by programs that spot check may be possible. When models have been used for quantification, programs could use reference sites or periodic project site visits to run field sampling, compare outcomes to model results, and determine whether the selected model is fairly representing GHG outcomes.

Monitoring of practices could be relatively simple—visual checks and regularly kept records—keeping costs relatively low. Monitoring and verifying GHG outcomes requires greater effort and cost but may be necessary to maintain confidence in a program. Thus the frequency of different kinds of monitoring and the bearer of costs must be considered in program design.

Box 9. Materiality or error threshold for verification

Materiality is a measure of the estimated effect that an error, omission, or misrepresentation of project data and information may have on the accuracy or validity of a project's GHG assertion or reduction claim. Most regulatory and voluntary market programs require a reasonable level of assurance on verification of GHG assertions for offset projects, and the materiality or error threshold applied has typically been 5% or less (see list below). Therefore, the cumulative certainty or accuracy of the GHG reduction or removal estimates must be in the 5% or less range.

This threshold has been used for a wide range of activities, most of which are not in the agricultural and forestry sectors. Some programs in these sectors have achieved the 5% threshold, but programs in as-yet-untried activities, regions, or scenarios may find that threshold difficult to meet. Therefore, it may be necessary to revisit this assumed threshold for these sectors, particularly for nonmarket, nonregulatory, or developing country programs.

Program rules that have adopted a 5% or lower materiality threshold include Environment Canada, UNFCCC (http://ji.unfccc.int/Sup_Committee/Meetings/022/Reports/Annex1.pdf), EU ETS, Alberta Regulatory System, BC Regulatory System, The Climate Registry (http://www.theclimateregistry.org/downloads/GVP.pdf), USEPA Climate Leaders (http://www.epa.gov/climateleaders/documents/resources/design-principles.pdf), and Western Climate Initiative (http://www.westernclimateinitiative.org/component/remository/Offsets-Committee-Documents/Offsets-System-Essential-Elements-Final-Recommendations).

Verification

In most GHG programs, a qualified, objective, and usually independently accredited[18] third-party examines a GHG assertion (i.e., a claim of reduction in greenhouse gases arising from an offset project) and provides an opinion or conclusion about that claim. The goal of the verification is to ensure that offset credits are credible and of sufficient quality to meet system requirements. The actual act of verification, depending on the program, entails a combination of field visits to project sites and desk audits conducted by the verifier. A general framework for the process of verifying GHG emissions is articulated in the ISO 14064-3 standards (ISO 14064-3 2006), which focuses predominantly on evaluating data management systems and reports. Verifiers compare the implementation of projects with the criteria set out in project-specific methodologies. For example, for its forest project protocol, CAR provides a document solely addressing verification of forest carbon projects. For most types of offset projects, a verification event involving a site visit is required for every year of credits. Forest-based offset programs require less frequent site visits; for such projects ACR and CAR require yearly desk audits but field verification only every five to six years.

Each of the above-described techniques for determining baselines and quantifying changes in GHG flux due to management practice changes has different verification requirements and complexities. In deciding between field- or modeling-based quantification approaches, program managers must consider the expected materiality or error threshold that projects will need to meet on verification (see Box 9). The convention for most programs is a 5% materiality threshold. For measurement-based sampling schemes, this threshold means that the statistical sampling protocols, baseline selection and control, baseline and project measurements, laboratory analyses, and calculation approaches must fall within a 5% margin of error to satisfy the required level of assurance. The risks associated with multiplicative errors in project execution for field-based measurement systems can be quite high. These risks are coupled with the likelihood that the project will also need to run models in tandem to derive estimates of N_2O and CH_4 changes.

In model-based options, if the program developer decides to implement standardized quantification approaches and "locks down" many of the variables for baseline and project practices, implementation and data collection for verification would be relatively streamlined. This strategy increases the likelihood that the project will meet a reasonable level of assurance and reduces risk for the project developer.

When models are applied by programs at a regional scale—using empirical extrapolation of research data or up-scaling

18. For example ACR, CAR, CCX and VCS all require independent accreditation of Validation/Verification Bodies by the American National Standards Institute against ISO 14065 criteria.

of biogeochemical process models—to produce regionally specific emissions factors for protocols or accounting frameworks, verification of the baseline determination and quantification process is significantly simplified; verifiers need only check the math. Questions about the quality of the data embedded in standardized approaches, which can include aggregated self-reported data, can be taken into account by using conservative assumptions about the data. However, verifiers must still verify whether baseline management was as stated and that promised management changes occurred—whether through site visits, use of remote sensing, or recovery of a clear paper trail (receipts and so on). Use of biogeochemical models at a farm or project scale requires that site-level management be highly specific to increase model accuracy.

A critical issue for verification will be alignment of definitions used for various management practices so that models, databases, producers, and verifiers are consistent with one another. For example, *conservation tillage* is a general term for many different kinds and intensities of tillage. It should be defined on the basis of the quantity of residue left on fields. For more details, see the T-AGG companion report on using biogeochemical process models, which includes two specific examples in Appendix A.

Leakage

While U.S. federal-level cap-and-trade legislation is not likely in the near term, if one develops, any participation by agriculture will likely be through offsets markets, rather than as a capped sector. Without a cap-and-trade offset program, agriculture's path to GHG mitigation is likely to be through a voluntary program involving government payments, through the sale of offsets to voluntary/corporate sustainability buyers, or through efforts to market low-emission agricultural products to commodity buyers, retailers, etc. concerned with their supply-chain emissions or product labeling. In any of these cases, agricultural GHG reductions will remain voluntary, and some parties will opt in and others will opt out. Therefore, the policy will have incomplete coverage.

An important consequence of incomplete policy coverage is leakage. The potential for leakage arises when rules, regulations, and incentives for action affect only part of the potential pool of participants or emissions sources. Largely an economic phenomenon, leakage is driven by unmet demand for goods previously produced in the policy or project area. It occurs when efforts targeted to reduce emissions in one place simply shift emissions to another location, where they remain uncontrolled or uncounted. For instance, with agriculture uncapped and not under a voluntary program, agricultural producers have no binding obligation to cut emissions. They can choose to do so through mitigation projects, but they can also opt not to and face no emission penalty.[19] Those who opt in to the offset program have a financial incentive to reduce emissions; those who opt out do not. But the actions taken by those who opt in can affect market signals. Some GHG mitigation actions (e.g., reduced fertilizer use) could reduce commodity yield and put pressure on the rest of the market to replace that output, perhaps by raising fertilizer applications elsewhere or by tilling new land, both of which can increase emissions. Activities that generate land use or significant crop mix change (e.g., introduction of short-rotation woody crops, set aside of cropland, and conversion of cropland to pasture) can have even larger leakage effects. The displacement of emitting activity and emissions undermines the environmental integrity of GHG abatement realized through offsets.

The primary emphasis on the leakage issue is on negative (or "bad") leakage, whereby a project activity induces emissions outside the project area. But leakage can also be positive ("good") if the actions taken inside the project boundaries lead to emissions reductions outside those boundaries. An example is actions that mitigate GHGs and also result in crop or livestock production gains, potentially reducing production pressure—and emissions—on other lands. This issue is addressed by Murray and Baker (2011).

Options for addressing leakage

Project developers have several approaches to mitigate the impact of leakage. Because leakage is essentially an accounting problem, addressing it often involves making accounting more comprehensive through discounting, better emissions monitoring, or expanding policy coverage. Jenkins et al. (2009) describe a range of policy approaches to address leakage. Table 20 reorders the options from Jenkins et al., describes them, and assesses them in terms of their applicability for U.S. agriculture.

19. To be clear, producers may have obligations to maintain emission levels once they opt into a project, but they are not obligated to opt in to the project in the first place.

The first of these options (accept imperfection without any adjustment) is certainly the easiest path, but it essentially ignores the problem. Moreover, it involves policy decisions and negotiations well beyond the agricultural sector. Similarly, although the second option does not ignore the problem—it suggests addressing the problem simply by expanding the compliance cap in recognition of leakage—it too involves policy decisions well outside of agriculture.[20]

The third approach, decoupling, is also an issue involving broader considerations about the role of offsets in an economy-wide mitigation policy. All recent legislative proposals in the United States (Kerry-Lieberman in the Senate, Waxman-Markey in the House of Representatives) have contained robust provisions for offsets, which are viewed as an effective compliance option that can significantly reduce the overall cost of achieving the aggregate emissions cap (U.S. EPA 2010a). Decoupling would not necessarily exclude agricultural mitigation but would move it to policy platforms operating outside of the compliance regime, such as Farm Bill provisions targeting GHG reductions. By allowing agricultural GHG mitigation to qualify for payments but not provide offsets for capped entity compliance, the policy might achieve significant GHG reductions in agriculture, but would not help contain costs of the cap-and-trade program covering other economic sectors, since capped entities would compete for a smaller overall offset supply.

Table 20. Applicability of leakage adjustment options for U.S. agriculture

Option	Description	Applicability to U.S. agriculture
Accept the imperfection	Accept leakage as an artifact of incomplete policy coverage and make no further policy or accounting adjustment for offsets.	This policy decision goes beyond agriculture. But agriculture has at least the potential for large leakage effects, given the wide geographic scope of commodity markets.
Adjust the economy-wide cap	Either tighten the cap in recognition of leakage deficiencies or expand its scope to encompass more activities so that fewer of them can cause leakage.	In principle, this strategy could mean including agriculture under the cap and excluding it from an offset system. The strategy is likely infeasible at this time for a variety of technical, economic, and political reasons.
Decouple agriculture from compliance regime	Incentivize reductions in uncapped sectors but not through compliance offsets for capped-sector entities.	Incentives could occur through traditional government payment programs, such as through the Farm Bill. However, even if agriculture is decoupled, it might still want to adjust for leakage effects.
Introduce system-wide accounting and reconciliation	Keep the offset structure but deal with it at a more complete level (sectoral, national). Measure net emission effects, which can be used to directly reconcile any intranational leakage effects.	This strategy would require national accounting of all major agricultural GHG sources and sinks (accounting that already exists to some extent through national reporting requirements) as well as a process to reconcile offset accounts from national accounts (a process that does not yet exist but that is being explored in other contexts of "nested" projects).*
Introduce project-level local monitoring/design efforts	Directly monitor leakage just outside the boundaries of the project (e.g., by monitoring a "leakage belt").	Low applicability, as most leakage in highly developed agriculture is driven by commodity markets not necessarily local in nature and therefore not locally monitorable.
Estimate system-wide leakage and discount offset credits	Estimate leakage using models of affected commodity markets and translate estimates to proportional "discount" factors to adjust for leakage	This strategy is possible in principle, but currently only a few such system-wide estimates are available for U.S. agricultural leakage.

* The state of California, for example, has allowed the use of "sector-based" offsets to provide compliance offset credits into California's cap-and-trade system. This could include "nested" projects whose reductions would be accounted and reconciled against a jurisdiction-level (national or subnational) baseline for a particular sector in a particular jurisdiction. Currently only reducing emissions from deforestation and degradation (REDD) qualifies, but other sectors could qualify in the future.

The other options in Table 20 accept offsets while correcting for leakage. System-wide accounting would take a sectoral approach that seeks to measure all significant GHG emissions from U.S. agriculture—an approach currently being implemented for national accounting purposes under the UN Framework Convention on Climate Change. Then, any subnational crediting through, for example, projects, would need to be reconciled with the sectoral accounts to ensure that any subnational/subsectoral leakage is covered. This task requires a nesting structure similar to the type being explored by international bodies implementing national or provincial REDD (reduced emissions from deforestation and degradation) activity. This structure has not been significantly examined for U.S. agriculture.

20. Murray and Jenkins (2010) examine the second option in Table 20 (expand the cap) and find that, although it can in theory make efficient adjustments to deal with offset imperfections such as leakage and additionality, it creates potentially large distributional effects that could undermine its viability.

On the other end of the spectrum, project protocols, such as those developed for the Clean Development Mechanism and some of the voluntary standards such as the Verified Carbon Standard and American Carbon Registry seek to address leakage in some sectors through local monitoring of displaced activity (emissions). This approach is sometimes referred to as *leakage belt monitoring*. It may have merit for activities that are likely to shift locally, but it is unlikely to deal effectively with leakage that occurs in the geographically broad markets that most U.S. agricultural producers serve. Therefore, more attention on market-level leakage is warranted.

The last option in Table 20 is to acknowledge leakage as a market-wide concept, attempt to estimate its extent, and assign "discounts" to adjust the issued credits for its estimated magnitude. This approach combines a system-wide view of the leakage problem with project-level crediting. It may be the best way to address the problem holistically (Murray and Jenkins 2010). However, leakage discounts developed from large-scale analyses (the most common approach) are very broad measures that will overstate leakage potential in some cases and understate it in other cases, causing distortions in the economic incentives provided to individual projects with different leakage potential. For example, imposing leakage discounts higher than the "actual" leakage impact for a particular project in a particular location could disincentivize producer participation.

Estimation/adjustment options for leakage in U.S. agriculture

Murray and Baker (2011) suggest that an output-based intensity approach for mitigation projects is one way to capture leakage by indirectly accounting for the effect of productivity changes on market outcomes. However, it is far from certain that the output-based offset (OBO) approach—in which credits are given for reducing the emissions intensity of output—will be deployed in an agricultural GHG policy. The OBO approach has only recently been included in legislative proposals—and if allowed, it seems likely that it would be combined with other approaches that do not directly capture productivity effects. Therefore, a separate treatment of leakage may be in order.

Empirical estimates for leakage in U.S. agriculture in the literature

The empirical literature on GHG leakage in U.S. agriculture is somewhat thin. Lee et al. (2007) evaluate international leakage from those agricultural producers participating in a global mitigation program such as the Kyoto Protocol and those who do not and found, unsurprisingly, that production from the former group does shift to the latter group. The study does not explicitly calculate the emissions leakage in absolute or percentage terms as a result of the mitigation actions, but it does show that the implied leakage from a unilateral U.S. policy is higher than the leakage that would occur if the United States acted in concert with all other major producing countries.

Some assessment has been made of leakage potential within the United States from various agricultural practices, such as land retirement and conservation tillage. Wu (2000) finds that about 15%–20% of the direct benefits of the Conservation Reserve Program are offset by *slippage,* a term analogous to leakage, except that it measures land use displacement and not carbon. Murray et al. (2005) estimate a small amount of emissions leakage (0%–5%) from agricultural activities. That study and another study (Murray et al. 2004) found substantially more leakage potential from afforestation and avoided deforestation programs within the United States than from agriculture.

Comprehensive modeling approach

To more carefully examine the leakage impacts of specific agricultural mitigation practices in the United States, researchers need a comprehensive modeling study that can capture the impacts of production shifts, emissions, and sequestration within and across land uses and agricultural accounts in the agriculture of the United States and other countries. Some models, such as FASOMGHG (discussed in this report), POLYSIS, or FAPRI, are, in principle, capable of producing such estimates, but no comprehensive study has gauged these impacts across practice types.[21] Such a study would require specificity about practices are allowed and disallowed in a national mitigation program (offsets or otherwise) and about possible model refinements to incorporate the allowed practices and quantification of international emissions.

Formulaic approach

Absent a comprehensive study to assess leakage across a range of agricultural practices, a more formulaic to practice-level leakage can be used to develop first-order estimates using the general leakage equation in Murray et al. (2004). The

21. FASOMGHG has been used to examine international production shifting at the sectoral level (Lee et al. 2007) and for some specific practices within the United States, such as agricultural soil carbon management and afforestation (Murray et al. 2005), but it has not been used to comprehensively assess emissions leakage across a range of specific practices within the United States.

equation can be expressed as follows for commodity *i* in response to mitigation activity *j*:

$$L_{ij} = [e_i/(e_i - E_i(1+\Phi_{ij})]*c_{ij}$$

Where

- e_i = elasticity of supply for commodity *i*
- E_i = elasticity of demand for commodity *i*
- $\Phi_{ij} = s_{ij}/(1-s_{ij})$, where s is the size of the supply shock to commodity *i* caused by the mitigation activity *j* project or program (i.e., the share of commodity production withdrawn (or in the case of positive leakage, the supply enhancement) by the mitigation action
- $c_{ij} = C^N_{ij}/C^R_{ij}$ = Carbon (GHG) ratio of the leakage "receiving area" to the area targeted by the projects for activity j

This equation captures the notion of activity *j* as a supply shock to commodity market *i*.[22] For example, suppose the adoption of a fertilizer management activity in commodity *i* leads to a negative supply shock of 1%. Assume that the elasticity of supply for this commodity $e_i = +1.0$ and the elasticity of demand, $E_i = -1.0$. Further assume that the GHG intensity of commodity production outside the project areas is exactly the same as the baseline (nonproject) intensity within the project areas (implying that $c_{ij} = 1.0$). Taken together, these assumptions lead to a leakage estimate of 0.5, meaning that half of the GHG benefits of the project/program are offset by emissions increases elsewhere. This equation is relatively straightforward way to estimate leakage estimates, but it requires data and parameters identified in Table 21.

Table 21. Information needed to estimate leakage using a formulaic approach for each practice

Information/data needs	Sources	Comment
Determine whether productivity rises or falls as a result of the practice change and the relative magnitude of the supply shift (Φij)	Field studies associated with given practices	Can/will depend significantly on the type of practice, commodity, region, and possibly time period Independent field study evidence may be fairly limited in some cases
Elasticity of supply (ei) and demand (Ei)	Agricultural economics literature – e.g., FAPRI data base (FAPRI 2010)	Theoretical expectation is that ei > 0 and Ei < 0. Use long-run supply elasticities as they reflect changes in land use and practices, which is more germane to GHG mitigation than short-run shocks
Carbon (GHG) ratio (cij)—an estimate of GHG emissions from production outside the project or program area relative to that within the project/program area	Field studies or models	Requires assumption/estimate of the regional and relative land quality differences between land in and out of the project/program
Is commodity a global good? If so, how much of the good does the United States supply to the global market?	FAO stat for numerous commodities (FAO 2010)	This information is relevant if the commodity of interest is traded and domestic leakage must be separated from international leakage

Although straightforward, the formulaic approach may not capture some important feedback effects in land use, changes in agricultural practices, and regional shifting of production that the more complex models capture. Therefore, the approach should be viewed, perhaps, as a stopgap measure until a comprehensive agricultural-leakage estimate exercise is undertaken.

Reversals

Reversals are the release of previously sequestered carbon, which negates some or all of the benefits paid for in previous years. This issue is sometimes referred to as "permanence": certain types of offsets (e.g., landfill gas capture, avoided N_2O from fertilizer) are effectively permanent since the emissions, once avoided, cannot be re-emitted, while other types (e.g., forestry, agricultural soil C) have an inherent risk of future reversals of sequestered C that must be mitigated through some mechanism (e.g., buffer pool, insurance) to compensate for reversals that occur. Most activities that reverse carbon sequestration are relatively easy to track visually: a plowed field with residue removed, the removal of a forested buffer, and so on. What cannot be seen is how much carbon is lost when reversals occur. For example, as herbicide (glyphosate)-resistant weeds become a problem, some farmers may manage them by periodically plowing a normally no-till farm (see Box 10). If such a change occurs, is all or just a portion of previously stored carbon lost? How does the loss vary by region?

22. See the original article by Murray et al. (2004) for more detail on the derivation of the equation. The article is focused on forest carbon projects and timber markets, but the equation can be generalized to agricultural commodities (i) and mitigation activities (j).

Box 10. Superweed-driven reversals: Using periodic tillage to manage weeds

Weeds resistant to herbicides (superweeds) are a concern for many farmers in the United States, and threaten to alter agricultural management practices. Such management changes typically taken by farmers include herbicide rotation, herbicide application sequence changes, the use of different herbicides, or the use of tillage to control weeds (National Research Council 2010). The management decision may depend on several factors, such as topography or production methodology (Kim and Dale 2005; Mueller et al. 2005). Glyphosate use for weed control has increased with adoption of genetically modified glyphosate-resistant (GR) corn, soybean, and cotton in the United States; and is associated with tremendous growth in resistance to the herbicide. The number of weeds resistant to glyphosate has grown from 2 to 18 globally (0 to 10 in the U.S.) between 1997 and 2010 (Heap 2011).

Scott and VanGessel (2007) examined glyphosate resistance of horseweed in GR soybean crops in Delaware. Before the introduction of GR soybean, horseweed was not difficult to control with glyphosate in no-till systems. In a survey, 98% of growers reported planning GR soybean and 76% reported using no-till or conservation tillage. Thirty eight percent of respondents had experienced GR horseweed on their land, and 31% of those reported that their response would be to implement tillage prior to planting the GR soybean.

Similar stories of such management responses to glyphosate resistance can be told in many areas of the United States; for example, horseweed in Tennessee, tropical spiderwort in Georgia, and common waterhemp in Illinois (Mueller et al. 2005). These management changes will introduce additional costs to growers. Mueller et al. (2005) estimated that the cost of new resistant-management practices for horseweed was $12.33/acre in western Tennessee, largely due to a shift from no-till to conventional tillage and the need for new pre-plant herbicides. With the prevalence of glyphosate resistance in weeds expected to grow in the future, these management changes may also become more prevalent, reversing the benefits of GHG mitigating agricultural practices such as conservation tillage.

The expected management responses to herbicide resistance can be speculated; however, there are still few studies that have examined either the responses to actual instances of GR weeds or the reasons underlying specific management responses. In order to predict specific management responses through time, such as conversion back to conventional tillage, future research should examine the rationale and circumstances underlying such management decisions, as well as alternative management choices.

—Contributed by Andrea Martin

Often a key question is whether the carbon reversal was intentional—caused by the land owner due to shifts in profitability or management needs—or unintentional—caused by events such as wildfire, flood, or pest infestation. If this distinction results in different penalties or costs for the project, the distinction should be verifiable. For example, if the farm falls into a region that has recently experienced a flood, drought, wildfire, or pest problem, it could qualify for unintentional status. Many GHG mitigation protocols and projects consider the use of discounts or buffers to manage for unintentional reversal risk. Intentional reversals require repayment by project owners.

Intentional reversals can include shifting management from conservation tillage or no-till back to conventional tillage. Conventional tillage of lands currently under conservation or no-till management is known to release some of their stored carbon. Land use changes such as restoration of grasslands or conversion of agroforestry, windbreaks, or perennials back into annual crops would immediately release the carbon stored above ground in the trees, and if soil is disturbed, it may over time release belowground carbon as well.

Intentional reversals driven by changes in perceived risks and profits have been observed in other agricultural programs. For example, Secchi et al. (2008) found that of the 9.4 Mha reenrolling in the USDA Conservation Reserve Program (CRP) in 2007 only 15% are reenrolling for more than five years. Hence, it appears that approximately 1.6 Mha of CRP land will be going back into agricultural production each year for the five following years, possibly releasing much of the carbon that has been stored. The driver for intentional reversal is primarily financial: as crop prices rise, movement out of conservation programs rises. Carbon markets may be similar. Other reasons for intentional reversal could include the invasion of superweeds that are best managed by periodic tillage (see Box 10). Given that the superweeds are not caused by an intentional action, tillage to manage for them could potentially fall into the unintentional reversal category. And if periodic tillage is a baseline management tool, it could be incorporated into the baseline.

Unintentional reversals are usually caused by natural events. The natural events that affect yields (e.g., frost damage, pest infestation) will affect the annual increment of C sequestration or N_2O flux, but the resulting change is not a reversal. A reversal for crops requires release of previously stored carbon. Tillage practices are unlikely to change unintentionally, unless management for superweeds is considered unintentional. With respect to annual crops, wildfire would only affect the current year's carbon storage, unless it burns into the organic soil layer. However, wildfire in systems with tree or shrub crops or windbreaks could see substantial loss of aboveground stored carbon.

The potential for reversals in cropping systems appears to present less risk than the reversals expected for forest systems.

Forest carbon protocols are addressing wildfire and other risks for aboveground biomass (e.g., hurricanes, floods, and insect infestations) and may provide guidance for agricultural programs focused on agroforestry, buffers, or windbreaks from which loss of aboveground carbon is the greatest risk. For belowground carbon on crop and range lands, reversion to conventional tillage appears to be the primary risk. Cessation of other activities (e.g., reduction of fallow) that store soil carbon may result in slow releases of carbon if the activity is not reinstated. This slow loss, if considered significant, can be managed with tools similar to those used for forests.

Table 22. Reversal events and potential impact on greenhouse gases

	Event	GHG reversal impact
Intentional	Shift back to conventional tillage	Significant soil carbon release
	Removal of tree crop, windbreak, or other shrub crop	Significant removal of aboveground carbon
	Cessation of other carbon-storing activity (e.g., reduced fallow)	Possible slow release of stored carbon
Undefined	Tillage due to superweeds	Significant soil carbon release
Unintentional	Wildfire	For annual crops, release of carbon from organic matter if the fire burns the soil layer For tree and shrub crops, loss of aboveground stored carbon
	Impacts of climate change (droughts, rainfall patterns)	Changes in temperature and the timing and magnitude of rainfall can alter rates of decomposition and capacity for carbon storage*

*Uncertainty about the site-specific impacts of climate change make such risks difficult to incorporate into programs at this time.

The following are steps for addressing reversal risks in program or protocol design:

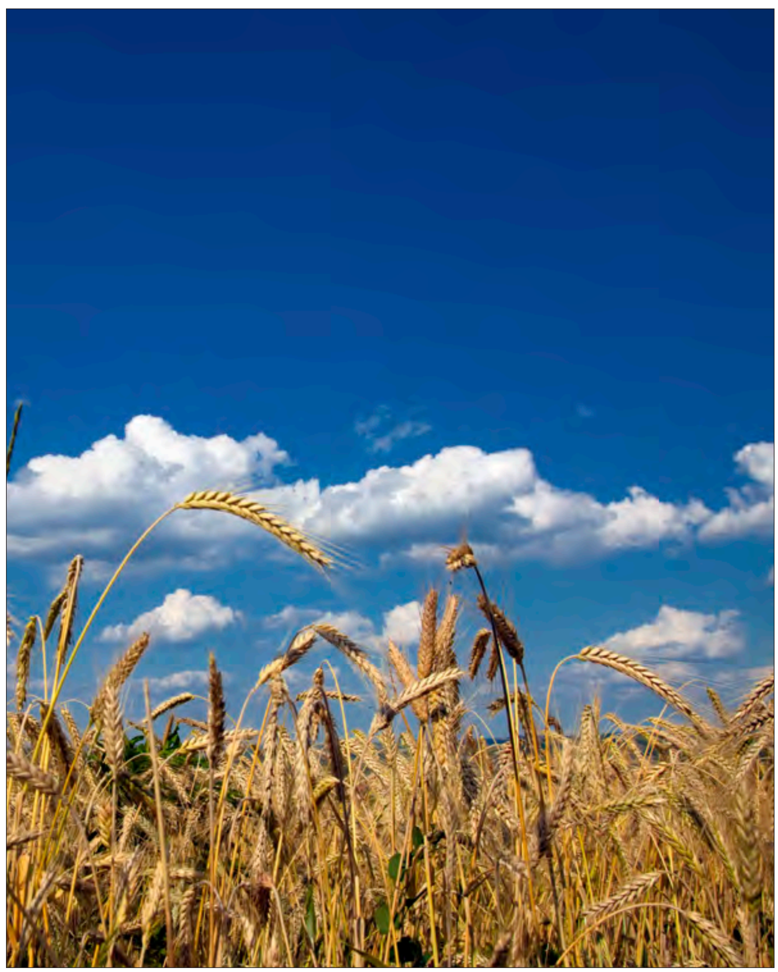
1. **Assess the risk of a reversal.** Determine the reversal incidence—the probable or possible level of risk for different regions and projects. This risk level can be based on historical data, future projections of loss, and expert judgment.

2. **Manage the reversal.** Implement measures to reduce the reversal risk—e.g., buffers or insurance schemes. For example, the project proponent may be required to deposit into a shared buffer account a certain number or percentage of offsets based on the project-specific risk assessment (Murray and Olander 2008).

3. **Verify the reversal.** Quantify any significant reversal. Procedures for doing so must be outlined in all biosequestration protocols and the verifier must review them to ensure their correct implementation.

4. **Mitigate actual reversals.** In the case of an actual reversal, once it is measured and verified, the program administrator may retire from the shared buffer pool an equivalent number of offsets to compensate for the reversal; require the project proponent to replace lost offsets through an approved insurance product; etc.

CONCLUSIONS



To achieve a balance of increased agricultural production and reduced environmental impacts, government programs and corporate supply-chain initiatives seek to motivate the use of increasingly efficient, intensive, and sustainable agricultural practices. Many agricultural practices that have mitigation potential can have direct impacts on efficiency; often these impacts are synergistic. The most promising management systems are those that combine efficiency gains with GHG reductions.

This report assessed the potential for a broad range of agricultural land management practices, providing scientific and technical information that will be needed by programs that aim to incentivize management changes that will reduce greenhouse gases. Of the 42 management practices reviewed, 20 of them appear promising for early action because they have positive net GHG mitigation potential and sufficient research evidence to support this conclusion (Table 4). Many of these more promising management options enhance soil C sequestration—including tillage reductions, fallow period reductions, incorporating more perennial crops, switching to short-rotation woody crops, conversion of cropland to pasture, and setting aside cropland. Others reduce N_2O emissions by reducing fertilizer N rates; changing the timing, placement, or source of fertilizer; and using nitrification inhibitors. The remaining activities—rice water management and variety development—reduce CH_4 emissions. A few of these promising management practices—introduction of winter cover crops, many of the N management activities, conservation tillage, and crop rotation diversification— are recommended as research priorities to assess potential for mitigation in understudied regions and to reduce uncertainty in quantification.

Eight additional practices are likely to have positive GHG mitigation potential, but the existing research is insufficient to warrant early action. These activities—including histosol management, crop rotation adjustments, irrigation management, agroforestry, and rotational grazing on pasture—would benefit from focused research to clarify GHG outcomes and other implications. Rotational grazing on pasture is particularly interesting. While its mitigation potential from land management alone seems promising, its broader impact on the efficiency of livestock production and the potential for positive leakage effects is even more promising.

Biochar application also seems to have very high mitigation potential, but research on the magnitude of the potential and on life cycle implications is needed. Researchers are still uncertain about the net GHG impacts once life cycle impacts are fully evaluated. The remaining activities do not appear worth pursuing at this time. They have significant data limitations, low or negative mitigation potential, or their life-cycle GHG effects appear likely to negate any potential mitigation.

An assessment of the relative costs of implementing the practices covered in this report would be quite helpful. Because the existing studies and models have insufficient details or coverage for this task, the report instead examined general cost and trends. It also discussed other factors that can affect costs and that may not be well represented—for example, transaction costs, social barriers, and measurement costs.

With respect to quantification, this report finds that direct field measurement is viable, although at times expensive, for assessing C sequestration; field measurement of CH_4 and N_2O is not yet ready for wide implementation. Direct measurement appears best suited for programs focused on innovative new practices for which research is lacking. In contrast, modeling will likely be most efficient for scaling up known management practices well supported by research and modeling capacity. Important data gaps remain for program or project implementation particularly management data for establishing baseline conditions. Additional work is needed to assess potential reversal rates for the subset of management practices for which this could be a problem. Leakage estimates, for both positive and negative leakage, would help assess the mitigation potential of a number of practices for which leakage may be a major contributor.

This report brings together information on a diverse set of agricultural land use practices, highlights key data and research gaps, and presents information on critical issues for implementation of GHG mitigation through these activities.

APPENDIX A: RESEARCH AND DATA GAPS FOR BIOPHYSICAL MITIGATION POTENTIAL

This appendix summarizes research gaps apparent through a literature review and conversations with scientists. Additional field studies[1]—especially studies specific to regions that are poorly represented in the existing data—are likely to add scientific certainty to the GHG mitigation potential of agricultural land management activities, but a targeted approach is needed to optimize research resource allocation. In many cases, specific unresolved issues and data gaps contribute to high levels of uncertainty in estimates of biophysical mitigation potential. Focusing efforts on these issues for practices that are expected to have high mitigation potential may have the greatest impact on developing mitigation programs and policies. Table A1 identifies key data gaps and technical issues.

Table A1. Data gaps and technical issues affecting GHG mitigation assessment for agricultural land management activities in the United States

Management activity	Data gap or technical issue
Tillage changes	Impact of varying tillage intensity: Baseline conventional tillage can be quite different among regions. How does the intensity of the baseline AND the improved tillage system affect soil C change? Impact of infrequent tillage events: Most research is on continuous no-till management. How does one-time or every-three-years tillage change soil C storage in different systems? Carbon storage in deep soils: Most research considers storage only in the top 30 centimeters. Current research is examining how tillage changes carbon storage at greater depths and assessing the potential for losses in soil C at depth. Tillage impacts on N_2O emissions: Increased emissions are related to climatic and soil conditions.[a] At what point (rainfall amount, soil texture) do increased N_2O emissions become problematic? (Six et al. 2004)
Summer fallow reduction and use of winter cover crops	Interactions between tillage regime and summer fallow elimination: How does soil C respond to summer fallow elimination if no-till, chisel plow, other? What are the GHG implications of changes in field operations (e.g., grain drying, more irrigation or tractor passes on the field) with respect to the main crop when cover crops are introduced?
Crop rotation changes	Rotation impacts on CH_4 and N_2O emissions: These impacts could be significant (Mackenzie et al. 1998; Omonode et al. 2011). Impacts on decomposition rates: Not well known is how crop choice affects decomposition rates and the potential for soil C storage. By diversifying a crop rotation, the total production of a region's primary grain crop may decrease (fewer cropping seasons). However, this impact has not been examined in terms of market or other influences.
All field buffers, agroforestry, windbreaks etc.	Effect of buffers and agroforestry on cropped area: Data exist for GHG effects under trees and other buffers, but few data are available on effects on crops and the soil under them. Might erosion or yield be affected and thus influence both soil C and other GHG emissions?
Manure (and other organic material) application	Net impacts of N_2O and CH_4 seem to be unclear, partly because of differences in the baselines used for comparison. How do manure application rates affect C sequestration? Existing data tend to come from studies on excessive application (nutrient loading above crop need). What is the C sequestration potential at appropriate application rates? How much distance can manure be transported before the GHG emissions associated with transportation exceed any GHG emission reduction related to improved manure application? If fertilizer N application is reduced as a result of improved manure application, what GHG emissions reductions could be achieved?
Fertilizer N management	The effect of fertilizer N application timing (e.g., spring vs. fall) on N_2O emissions needs to be studied in more detail (Millar et al. 2010) and in more regions. Data on fertilizer N placement are lacking (Millar et al. 2010). Data on N_2O emission differences between ammonium-based and urea fertilizer sources are extremely limited in terms of regions or cropping systems.
Organic soils	Most of the research into the GHG effects of organic soil management is from Europe rather than North America. More data on net greenhouse gases are needed.
Rice water management	The response of N_2O flux to changes in water management needs to be clarified in U.S. regions.
Pasture	Few baseline data and very little information about the GHG impacts of improved management are available.
All activities affecting soil C	Soil C saturation: Soil C sequestration rates decrease over time, appearing to reach a new equilibrium, but the time frame (and regional, soil, climate effects) is not well understood.

a No-till management can prompt changes in soil aggregates and improve drainage, reducing N_2O emissions (D'Haene et al. 2008), but in other cases it can increase soil carbon and nitrogen as well as bulk density and H_2O content, thus increasing N_2O emissions (Rochette et al. 2008a).

1. In some cases, it may not be necessary to initiate additional field studies but rather to obtain research results from studies that have not been published in the scientific literature. Therefore, targeted consultation with researchers at universities or research stations may reveal data more quickly than waiting for results from multiple years of new field studies.

APPENDIX B: STATISTICAL METHODS: DETERMINING SAMPLE SIZE

The key information necessary to calculate the number of samples required includes the following:

- ***Variance(s) or estimate(s) of variability for the random components of the project***—In a completely randomized design, only a variance for the residuals is needed. In a more complex design with multiple fields, information on the variances characterizing variability among fields might be beneficial. The sources of information about variances could be data from previous sampling or from experiments reported in literature.
- ***Size of the difference between the two means (δ) that the project developer wishes to detect***—This difference is the difference necessary for crediting and project profitability.
- ***Probability of Type I error (α), which is based on the α level required by the protocol or program***—This level is often 0.05, which provides a 95% confidence level that the difference detected is real.
- ***Probability of Type II error (β) or Power (1-β), which is likely set by the program or protocol using a typical power value (0.85, 0.90, 0.95)***—Type II error assesses the probability that a change that has occurred is actually detected by the statistical analysis.

The specific calculations will differ depending on the project design and on the type of statistical test that the researchers plan to conduct once they collect the data.

t-test for Paired Samples

A t-test is used when a single field has been sampled at *r* locations at time zero (initial) and resampled at the same locations in the future (final).

The number of sampling locations *r* can be calculated as following:

$$r \geq \frac{\sigma_d^2 (z_{\alpha/2} + z_\beta)^2}{\delta^2}$$

Here:

- σ_d^2 is the variance for the difference between the initial and final measurements.
- δ^2 is the hypothesized difference between the initial and final measurements (e.g., the size of a difference that would be necessary for crediting and project profitability).
- $z_{\alpha/2}$ is the critical value from a standard normal distribution corresponding to the probability of Type I error (α) desired by the researchers.

[Here are the $z_{\alpha/2}$ values for the three most commonly used α values of 0.01 (99% confidence), 0.05 (95%), and 0.10 (90%): α=0.01, $z_{\alpha/2}$= 2.54; α=0.05, $z_{\alpha/2}$= 1.96; α=0.10, $z_{\alpha/2}$= 1.64.]

- z_β is the critical value from a standard normal distribution corresponding to the probability of Type II error (β) desired by the researchers.

[Here are the z_β values for the three most commonly used β values of 0.05, 0.10, and 0.15, corresponding to power values of 95%, 90%, and 85%: β=0.05, z_β= 1.64; β=0.10, z_β = 1.28; β=0.15, z_β = 1.04.]

Example of calculations:

Consider an agricultural field under conventional management that will shift to conservation management, under which soil C content is believed to increase. The plan is to collect samples from several georeferenced locations within the field and to resample these locations after conservation management is implemented. The question is how many locations should be sampled in order to detect a 0.5 t CO_2e ha^{-1} increase, if it occurs?

Some preliminary assessments and the literature suggest that the standard deviation for the change in carbon is approximately 1 t CO_2e ha^{-1}. To detect the difference, a paired t-test with probability of Type I error, a, of 0.05 is used. The goal is to be 90% confident that any change can be detected. That is, the test should allow the tester to conclude that any difference between before and after C levels is statistically significant. That is, power is 90%, and Type II error, b, is 10%. Accordingly, the number of sampling locations is given by the following formula:

$$r \geq \frac{\sigma_d^2 (z_{\alpha/2} + z_\beta)^2}{\delta^2} = \frac{(1.0)^2 (1.96 + 1.28)^2}{(0.5)^2} = 44$$

The results are as follows:

- If the variance for the C change is hypothesized (estimated) correctly *and*
- If the difference of 0.5 t ha^{-1} between initial and final sampling dates indeed have occurred *and*
- If all necessary sampling and measurement procedures have been carried out correctly
- then before and after data should be collected from the 44 sampling locations and a paired t-test should be run to compare initial and final values.
- The chance that a statistically significant difference between initial and final samples is detected is 90%, and the tester will be 95% confident (with α=0.05) that the difference is real.

Calculating the number of samples using this formula is suitable if the numbers of samples are relatively large (e.g., $r >$ 30). But the formula will underestimate the numbers of samples when the numbers are small. The reason is that a t-test will be used to compare the initial and final values. The shape of a t-distribution depends on the number of samples. When the number of samples is large, the t-distribution resembles the shape of the standard normal distribution, and the critical values from a t-distribution must be used—that is, $t_{\alpha/2, f(r)}$ and $t_{\beta, f(r)}$ will be very close to the critical values $z_{\alpha/2}$ and z_β from the standard normal distribution. However, when the number of samples is small, $t_{\alpha/2, f(r)}$ and $t_{\beta, f(r)}$ will be higher than $z_{\alpha/2}$ and z_β. In this case, using a simple formula for the number of samples calculation becomes problematic: to calculate r, the tester must know $t_{\alpha/2, f(r)}$ and $t_{\beta, f(r)}$, but to find $t_{\alpha/2, f(r)}$ and $t_{\beta, f(r)}$, the tester must specify r. Because the shape of the sampling distribution depends on the number of samples, tables are constructed for different specific cases. These tables can be found in statistics textbooks, on the Internet, and in statistical software packages.

t-test for Independent Samples

A t-test for independent samples is used when r randomly selected samples are taken from two fields: one under original management and one under changed management.

The number of samples in each field r can be calculated as follows:

$$r \geq \frac{2(z_{\alpha/2} + z_\beta)^2 \sigma^2}{\delta^2}$$

Here:

- σ^2 is the variance for the residuals from the measurements of the two treatments.

The other terms are as defined above. As discussed above, this experimental setup can impose bias on the outcome if the selected field is not representative.

t-test for Multiple Paired Samples

Consider a scenario with a certain number, say r, of paired sites (e.g., paired fields or sub-areas within fields) one of which is under conventional management and one of which is under new management. For each site, a certain number of soil samples, say n, will be taken twice, that is, initially and then after a certain period of time, to determine changes in soil C (used here in illustration).[2] The question is what r (number of paired sites) and n (number of soil samples taken) should be used to ensure that any change in soil C of a certain size will be detected by the statistical analysis. For this calculation, the tester will need all the above-noted items necessary for power analysis. In addition, the tester must account for two sources of variability: variability due to sites and variability due to multiple soil samples within the sites.

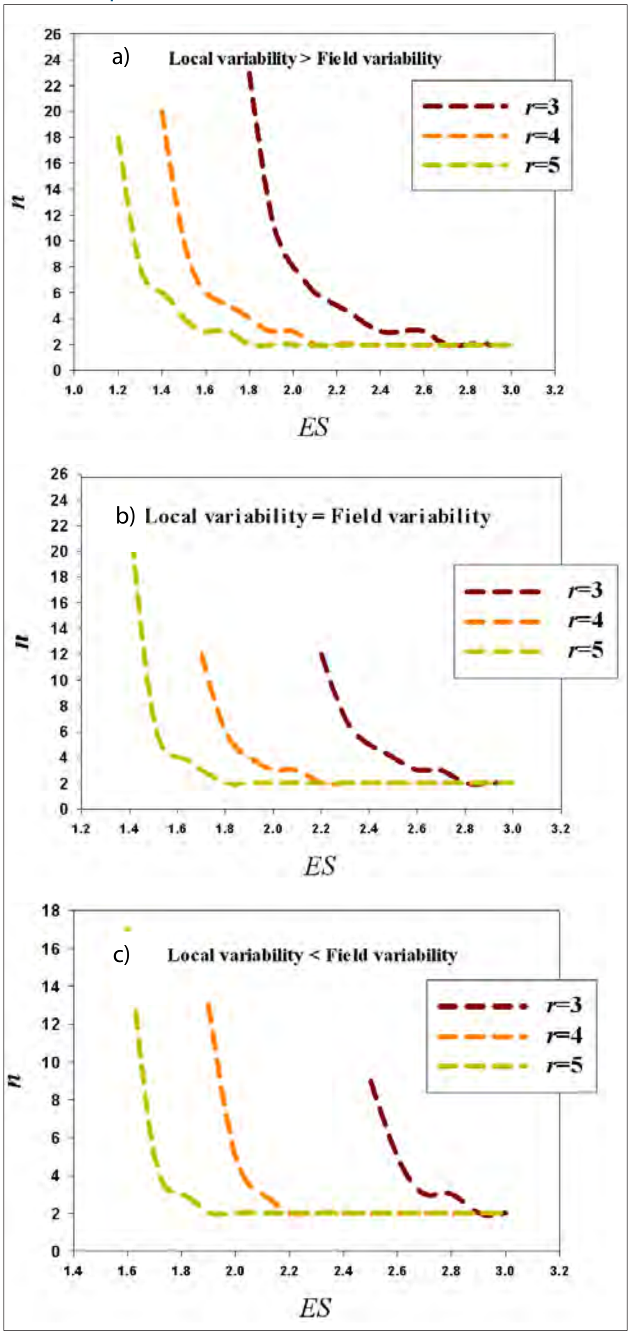
2. Similar statistical issues can also apply for N_2O and CH_4 emission quantification.

Ideally, the tester will make educated-guess estimates of both these values.

Figure B1 provides calculations for three possible scenarios: (1) when the variability among the sites is less than variability of samples within the sites; (2) when the two sources of variability are approximately equal, and (3) when variability among the sites is substantially higher than the variability within the sites. The first scenario would be appropriate when the sites are relatively similar but sampling uncertainty is high due to local within-site GHG variations or high variability in lab procedures. The last scenario would be appropriate when the sites are markedly different. For the first scenario, site variance is, say, 50% of the sample (within-site) variance. For the second scenario, the two variances are assumed to be equal. For the third scenario, the site variance is 150% of the sample variance. Because no simple equation can be used to determine the necessary numbers of samples in this case, the tester will generate the numbers of samples, *r* and *n*, for the three scenarios for three different magnitudes of the expected difference between conventional and new management. The size of the expected difference will be presented as a standardized effect size (ES). Here, ES is calculated as the ratio of the hypothesized change, expressed in percent, and the coefficient of variation of the whole set of data collected at the initial time point, expressed in percent. For example, if the coefficient of variation for soil C concentration is equal to 10%, and the increase in C concentration after conservational management implementation is expected to be 20% of the original level, the ES value is equal to 2.0. Coefficients of variation for soil C concentrations in the top soil layer are commonly 10%–20% (e.g., Syswerda et al. 2011).

Figure B1 shows numbers of paired fields (or areas within fields), *r*, and numbers of samples per field (area), *n*, that must be collected to detect as statistically significant an effect of size ES with probability of Type I error of 5% and power of 90%. Sections a, b, and c within the figure correspond to the three variability (local versus field) scenarios described above. For example, if local variability is higher than field variability, and the ES value is equal to 2.0, and if three pairs of fields or areas within fields will be used, approximately eight samples should be taken from each of them at the beginning and at the end of the study. If the number of paired areas is four or five, the number of samples per area will be equal to three and two, respectively. When field variability is equal to or greater than local variability, three paired areas will not be sufficient to detect the ES of 2.0. In that case, more than three paired areas must be used, or the tester must accepted that only an ES of 2.2 or greater will be detectable.

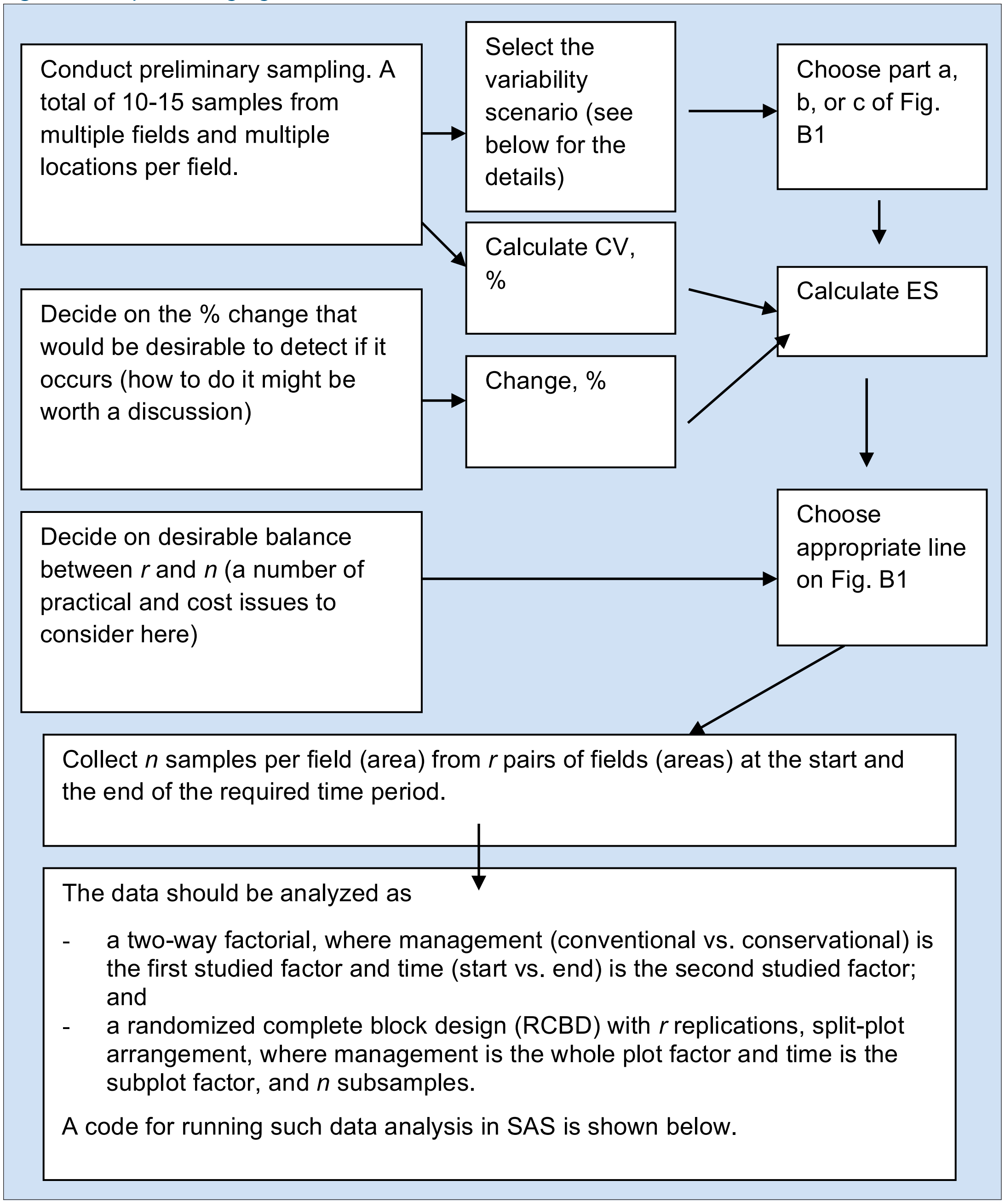
Figure B1. Number of samples, *n*, that must be taken per field at the beginning and at the end of the evaluation period when 3, 4, and 5 pairs of fields or areas within fields are considered



Note: One member of each pair is under conventional management and one under conservational management. The data will be analyzed as a randomized complete block design (RCBD) with *r* replications and split-plot arrangement with probability of Type I error of 0.05. The figures are constructed for power>0.90. ES is the standardized effect size equal to the ratio of the hypothesized change from the initial level expressed in % and the coefficient of variation of the entire set of initial data expressed in %.

Figure B1 shows the difference between two management practices at the end of the evaluation period. Figure B2 outlines suggested steps for using Figure B1 to decide on *r* and *n*.

Figure B2. Steps for using Figure B1 to decide on *r* and *n*



Selecting the variability scenario

The tester must collect preliminary data from r_{prelim} fields (>2) and multiple locations n_{prelim} per field (>2). (These data can then be included in the main data set). Then the tester should conduct random-effect model analysis to get estimates of the field variance and sample variance. Shown below are an example data set (three fields, three samples per field), the SAS code used to run the random effect model, and the estimates of the field and sample variances.

```
*********************************************;
* Example for variability scenario selection;
*********************************************;
data varsc;
input Field$    Sample$    Carbon;
cards;
1     1     8.6
1     2     10.1
1     3     9.7
2     1     12.3
2     2     11.2
2     3     10.8
3     1     15.6
3     2     14.5
3     3     16.0
;
proc mixed data=varsc;
class field;
model carbon = ;
random field;
run;
```

Covariance Parameter Estimates

Cov Parm Estimate

Field 8.8237

Residual 0.6033

In this example, the estimates of variances are 8.8 for field variance and 0.6 for sample variance. Field variance >> local variance, thus it can be assumed that differences among pairs of fields will be substantially higher than the differences among samples within a field. Thus part c of Figure B1 should be used.

A simplified procedure for assessing the size of among-field and within-field variability would be to calculate averages for each field and the variance for those fields and the variances for the samples within each field. For this example:

Field	Sample	Carbon, g/kg soil	Field averages	Among-field standard deviation	Within-field standard deviation
1	1	8.6	9.5	**3.0**	**0.8**
1	2	10.1			
1	3	9.7			
2	1	12.3	11.4		**0.8**
2	2	11.2			
2	3	10.8			
3	1	15.6	15.4		**0.8**
3	2	14.5			

Because the among-field variability is substantially higher (standard deviation is equal to 3.0) than the within-field variability (standard deviations are equal to 0.8), part c of Figure B1 would again be used.

Statistical model for the data analysis

The statistical model for the data analysis will consist of the following components (degrees of freedom for each term are written under it):

Y = **μ+Rep+Management+e1 +**

*r*2*2* *1 (r-1) 1 r-1*

+Time+Time*Management + e2 + e3

1 1 2r-2 4r(n-1)

Here:

- **μ** is the grand mean;.
- **Rep** is the random effect of the pair of the fields.
- **Management** is the fixed effect of the management practice.
- **e1** is the random effect of the individual fields.
- **Time** and **Time*Management** are fixed effects of time and time by management interaction, respectively.
- **e2** is the random effect of the individual fields at different sampling times.
- **e3** is the random effect of the individual soil samples.
- Error term **e2** is what will be used for detecting any difference between the two management outcomes at the end of the evaluation period.

The calculations in Figure B1 use the total variability and split it into field and local values as described above. It is assumed that the three random components related to field variability, that is Rep, e1, and e2, equally contribute to the overall field variability.

Suggested SAS code

```
proc mixed data=a ;
class management time r;
model c= management time management*time;
random r r*management r*management*time;
estimate 'managements 1 and 2 in time 2' management -1 1
management*time 0 -1 0 1;
lsmeans management*time/pdiff;
run;
```

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