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

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Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States

A Synthesis of the Literature

Companion Report to *Assessing Greenhouse Gas Mitigation Opportunities and Implementation Options for Agricultural Land Management in the United States*

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Contents

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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For more information see our website: http://www.nicholas.duke.edu/institute/t-agg.

Why a Second Edition?

This *Synthesis of the Literature* on agricultural land management GHG mitigation potential was first published in October 2010 in response to demand by policy and market design decision makers for summarized technical and scientific information on the topic. The second edition updates the side-by-side comparison of biophysical GHG mitigation potential of 37 agricultural land management activities with newly available and previously un-included data from field experiments, modeling, and expert review. Notable changes include:

1. Additional paragraph in *Introduction* to discuss the purpose of the side-by-side comparison, how this method is useful for contrasting many activities (including some that are data-poor or rely primarily on expert opinion), and how current and upcoming meta-analysis projects will provide further detail where data is sufficient.

2. Updates to the section on *Conservation Tillage*, with additional field experiments and refining of the separation between "no till" and "conservation till." This also included the removal of some field comparisons from the "conservation till" activity that were primarily "no till" reviews. The GHG mitigation estimate for conservation till is reduced somewhat.

3. Changes to the *Winter Cover Crops* section, with new additions of cover crop experiments in three different states, and a slightly altered discussion that mentions some of the regional variability in soil C sequestration potential.

4. Updates to the section on *Short Rotation Woody Crops*, mainly additional field experiment results, which provide more detailed information and greater confidence in the soil C sequestration rate that is not much changed from the previous edition.

5. Updates to the section on *Organic Soil Amendments* with additional field experiment results that lead to somewhat higher estimates of GHG mitigation potential. However, a more important addition is a discussion of the issues related to net GHG emission impacts of changing manure application practices. Unless the decomposition rates of organic compounds are decreased when manure is spread over additional land area, these improved manure management activities may not result in net C sequestration.

6. Updates on N management sections with revised applicable area, updated with most recent ARMS N application data (USDA ERS 2010a).

7. Some changes to the histosol management section, clarifying the delineation between histosol and wetland in this report and adding some estimates from the literature.

8. Additional references for the rotational grazing section, as well as more detailed discussion of net GHG impacts for the system, which are impacted by efficiency gains as well as the GHG flux changes in the local pasture.

9. Further information on the GHG impacts of grazing land set-aside, which highlights the high variability in available data, and the need for additional research.

10. Updated summary of GHG impacts for above activities and some adjustments to other values in the final table that illustrates the side-by-side comparison.

Introduction

This document is an appendix to the T-AGG report *Assessing Greenhouse Gas Mitigation Opportunities and Implementation Options for Agricultural Land Management in the United States* (hereafter called the "U.S. Assessment Report"). It is an extensive scientific literature review providing a side-by-side comparison of the biophysical greenhouse gas (GHG) mitigation potential of more than 40 agricultural land management activities in the United States.

The purpose of this T-AGG literature synthesis was to assess the viability of as many different management practices as possible for their mitigation potential. Prior to this effort the debate over agriculture's role in climate policies and programs had been fairly limited to changes in tillage and afforestation, with a rough list of other activities that various stakeholders wanted to include but very little sense of the potential or viability for large-scale programs. The synthesis provides an initial review of a larger range of practices, using data from other literature reviews, expert estimates, and

field experiments to provide a side-by-side comparison of biophysical GHG mitigation potential, and to help answer the question of which activities deserve further attention and assessment. For many activities considered, the data available are too scattered and incomplete for formal meta-analysis, even though meta-analysis would provide a robust assessment of mitigation potential and the variability that results from differences in soil, climate, or cropping conditions. Thus, caution should be used in interpreting the mitigation potentials, particularly those with few research comparisons. In this synthesis we note where data are severely limited (and the estimates therefore uncertain). Scientific certainty is further discussed in other T-AGG reports, U.S. Assessment Report and T-AGG Survey of Experts. Researchers in the USDA-ARS and universities are currently conducting meta-analyses to assess issues such as the soil C response to tillage changes as affected by sampling depth, region, soil type, and other factors (C. Rice, personal communication, January 2011; S. Ogle, personal communication, March 2011).[1]

While this report assesses individual activities, it assumes that agricultural production takes place within a system with multiple interconnections among processes, organisms, and nutrient and carbon pools. Viewing the field, farm, and region as an agro-ecosystem may also generate a longer-term perspective on energy and elemental transformations over time. Most broadly, agricultural land management is one component of the larger biosphere, where organisms and materials interact in ways that impact atmospheric GHG concentrations. At the farm or field level, multiple activities on the land interact with one another to affect the biogeochemical cycling of carbon (C), nitrogen (N), and other elements, affecting soil C storage and other GHG emissions.

Every farm is also a unique combination of multiple management decisions, many of which can have an effect on GHG emissions. The cascading effects of management decisions on the agricultural ecosystem can result in synergies where one activity enhances or is additive to the GHG mitigation potential of another, or tradeoffs where one activity reduces or eliminates the benefits of another. As much as possible all management decisions must be incorporated into the quantification of a farm's GHG impact. While this report disaggregates these practices to understand their individual impacts, it also discusses interactions that have been well documented. The U.S. Assessment Report also describes quantification techniques that can integrate across the multiple practices implemented on the landscape.

While biophysical GHG mitigation potential is a useful metric for a side-by-side comparison, alone it is not sufficient for assessing the relative viability of various activities. Total national biophysical GHG mitigation potential is affected by the amount of land area available for implementation, which in turn depends on land-use competition among these activities and other land uses. Where activities are mutually exclusive, the choice involves tradeoffs and other economic factors; this is addressed in the U.S. Assessment Report where the Forest and Agriculture Sector Optimization Model (FASOMGHG) considers land-use competition and market-force impacts on land management and land-use change. The companion report also considers scientific certainty for the biophysical potential values, as well as other economic factors, social and technical barriers, and ecological co-effects, all of which will impact the viability of these activities.

Methods for Literature Review

To determine biophysical GHG mitigation potential for individual agricultural land management activities, we conducted a review of the literature, using existing syntheses where possible and updating them with newer research. We take a geographic focus on the United States, but also include research from Canada and other regions with relevant agricultural systems and management activities, where information is missing or limited. Most activities target only one of the three major GHGs (with possible impacts on others): carbon dioxide (CO_2), by sequestering carbon in the soil; nitrous oxide (N_2O), by reducing emissions; and methane (CH_4), by reducing emissions or increasing uptake in the system. We focused on results from field studies, noting mitigation potential related to the *target* GHG[2] as well as other GHG impacts (per hectare). Modeled values or estimates based on expert opinion were used in select instances when field studies were not available.

We compiled a list of GHG-mitigating agricultural activities through a review of the literature and from sources already exploring market opportunities. The resulting activities impact agricultural land management, and tend to be related to extensive or nonpoint sources and sinks of GHGs. They generally apply to large land areas and have been slower to develop into valued GHG mitigation or offset protocols for a number of reasons, including that they require

1 Some of this ongoing work is part of the T-AGG project.

2 The *target* GHG is the main one of mitigation interest: CO_2 in the case of activities undertaken to encourage soil C sequestration; or N_2O or CH_4 when an activity aims to reduce emissions of these GHGs.

the involvement of numerous landowners in order to achieve appreciable impacts; that they lack certainty in GHG responses; and that regional differences can be significant or not well understood. Three main categories exist: (1) activities taking place on cropland, where products are removed by human harvest activities; (2) activities on grazing land, where the plant growth is removed by animal grazing, mostly cattle; and (3) activities that relate to land-use change, e.g., where cropland is converted to grazing land or wetlands are restored from agricultural use.

Some possible GHG mitigation activities related to agriculture are intentionally not included in this analysis. These include afforestation and manure storage management, both of which are already addressed in established protocols or projects, with good understanding of the technical issues. And while land-use changes that have negative GHG impacts (e.g., deforestation, grassland conversion to cultivated land) carry clear implications for the GHG balance and are important to consider at a para-sectoral level, they are not related to management of existing crop or pasture land, and are thus excluded from this review.

Other ideas for GHG mitigation in agriculture have also included organic farming, urban agriculture, biotechnology applications, and programs to support local farming and buy local (USGS 2009). Organic agriculture is a system of farming that often includes multiple activities with GHG implications (e.g., crop rotation diversity, cover crops, manure and compost application). Research comparing organic and conventional systems has found significantly greater organic soil carbon accumulation in the organic systems, both in the United States (Clark et al. 1998a; Lockeretz et al. 1981; Pimentel et al. 2005) and abroad (Freibauer et al. 2004). However, there are many different activities that contribute to these impacts, and separate consideration within this review enables understanding of the processes and driving factors. This allows consideration of a variety of activities within a farm system (organic or not), avoiding prescriptive application and allowing adaptation to individual soil, climate, and other characteristics. Urban agriculture may contribute to some GHG mitigation, but most benefits would likely be difficult to quantify (small areas, highly variable production). While not a specific activity on its own, advancements in biotechnology could have a wide range of impacts on GHGs. A separate section documenting some of the GHG implications of biotechnology is included at the end of the report.

Nitrogen fertilizer application increases yield and soil organic carbon (SOC) (Varvel 2006), prompting its proposal as a potential GHG mitigation technique (Snyder et al. 2009). However, with the majority of crops in the U.S. already receiving fertilizer N, increasing application rates above the baseline is unlikely to have any major C sequestration impact, and recent studies have determined little to no impact of additional N fertilizer application on SOC or CO_2 fluxes (Alluvione et al. 2009; Mosier et al. 2006). There are even reports of negative impacts, where N fertilizer encouraged organic matter decomposition (Khan et al. 2007), and the corresponding risk of increasing N_2O emissions generally outweighs any potential GHG mitigation benefit. Therefore, we will not explore this activity as a potential GHG offset.

Abbreviations

C (carbon); CH_4 (methane); CO_2 (carbon dioxide); CO_2e (carbon dioxide equivalents, i.e., having the same 100-yr global warming potential of indicated quantity of CO_2); CT (conventional till); GHG (greenhouse gas); N (nitrogen); N_2O (nitrous oxide); NT (no-till); SOC (soil organic carbon); SOM (soil organic matter)

Units of Measurement

ha (hectare); Mha (megahectare, i.e., million hectares = 10^6 ha); t (metric ton); Mt (megaton, i.e., million metric tons = 10^6 t)

For each activity considered, estimates of soil C sequestration or changes in nitrous oxide (N_2O) and methane (CH_4) fluxes (per hectare, per year) were gathered from the literature, and weighted by the number of comparisons reported in each study. If regional differences have been clearly established in the literature (e.g., soil C and N_2O emission impacts of no-till), weighting was based on regional GHG impact (per ha) and the applicable regional crop area.[3] The resulting mean and a range of observed values are detailed in each section and at the end of this report (Table 31). In order to advise an assessment of the national GHG mitigation potential of each activity, the maximum applicable land area (over and above current adoption rates, i.e., baseline area) was determined from the literature and available survey data.[4] Since applicable land area is a maximum value, and competition between different activities can result in implementation on less area, we avoided calculating a simple total national potential. More detailed economic land-use competition analysis and an assessment of interactions between activities are needed for any national total predictions.

3 The 48 coterminous states are divided into 10 generalized agricultural regions as per the Forest and Agriculture Sector Optimization Model (FASOMGHG).

4 Total crop areas and some relevant survey data were taken from the U.S. Agricultural Census. Current implementation rates from various sources were used to determine applicable crop area for each activity (see text in each section for relevant details).

For many activities examined, sequestration (storage) of soil C is the main mode of GHG mitigation, removing CO_2 from the atmosphere. For other activities, emissions of nitrous oxide (N_2O) and methane (CH_4) are the main target, and net GHG direct fluxes are important considerations for all mitigation programs or projects. Until recently (past 5–10 years), studies tended to only assess changes in soil carbon (C) and thus were missing data on other important greenhouse gases (N_2O and CH_4). Therefore, there are fewer studies reporting non-CO_2 gases; in the final table we have noted where the reported values are based on very limited data (i.e., three or fewer reports).

While upstream, process, and on-site fuel and energy use would not be included in an offsets program with an economy-wide cap-and-trade system in place (like those recently debated in the U.S. Congress), they would likely be counted under other policies or programs such as Farm Bill policies or corporate demand-driven supply-chain programs. Therefore, we also included fuel impacts and other upstream or process GHG effects. Where empirical data or expert assessments were not available, fuel- and fertilizer N-use estimates were based on knowledge of field operations, published cost-and-return reports from cooperative extension in different states, and assumed N fertilizer rate changes. Thus, if an activity involved a reduction in area devoted to crop production (e.g., windbreaks or conservation set-asides), the upstream fertilizer-related GHG emissions and fuel GHG emissions for field operations for that proportion of the land area comprise the upstream and process impacts. For activities where fertilizer N reduction occurs (e.g., organic amendments and nitrification inhibitors), the fertilizer-related GHG emissions avoided are counted. With limited data in the peer-reviewed literature, many of the upstream and process emissions values reported in the final table are indicated as coming from limited references. All attempts were made to maintain conservative assumptions and estimates contributing to these values (as per ISO 14064-2 standards).

Some key background assumptions include total N fertilizer consumption of 13.6 Mt N yr^{-1} (Millar et al. 2010; USDA ERS 2010a), total annual N_2O emissions from U.S. fields of 204 Mt CO_2e yr^{-1} (Paustian et al. 2004), and total cropland area of 124 Mha (USDA NASS 2007a). The average fuel use for agricultural field operations[5] emits an estimated 0.36 t CO_2e ha^{-1} yr^{-1}. Significant variation exists between crop types, with California crop production data indicating a range in field operations fuel emissions from 0.13–0.713 t CO_2e ha^{-1} yr^{-1} (corn > hay > wheat).[6] In addition to field N_2O emissions, the C cost of N fertilizer (for manufacture, distribution, and transportation) is approximately 3.2–4.5 t CO_2 per ton[7] of N fertilizer manufactured (Izaurralde et al. 1998; West and Marland 2002). Therefore, the maximum possible GHG mitigation related to process and upstream emissions is 1.8 t CO_2e ha^{-1} yr^{-1} (the difference in process and upstream emissions between [1] grain corn with 250 kg N ha^{-1} and [2] set-aside).

Conservation Tillage (Including No-Till)

Of all agricultural land management activities suggested for GHG mitigation, conservation tillage has been the most widely applied[8] and studied, with the majority of research investigating no-till (NT). Given the significance of NT in the literature and in practice, we will treat it as a separate activity in this synthesis, and use the term "conservation tillage" more narrowly to denote any reduced-tillage practice other than NT. Researchers predict that, nationwide, conservation tillage and NT could sequester between 29 and 173 Mt CO_2e yr^{-1} (Table 1). In calculating national potential, we assume a baseline of 58% of cropland conventionally tilled, from recent 2004–2008 data

GHG Impact Summary

GHG Category	Conventional to NT	Conventional to Conservation Till
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**1.08** (-0.26 – 2.60)	**0.91** (0.00 – 1.82)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**-0.18** (-0.91 – 0.72)	**0.07** (0.00 – 0.38)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.12** (0.07 – 0.18)	**0.08** (0.03 – 0.10)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**1.01**	**1.06**
Maximum U.S. applicable area, Mha	**72**	**72**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

5 National agricultural fuel use comes from Schnepf (2004), allocating all gasoline and diesel to field operations, and using conversions to CO_2 from the U.S. Energy Information Administration, available at http://www.eia.doe.gov/oiaf/1605/coefficients.html (Accessed 23 September 2010).

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

7 The term *ton* (abbreviated *t*) in this report refers to the metric ton (1 ton [or *tonne*] = 1,000 kg = 2,204.62 lbs). Hence, the abbreviation *Mt* refers to the megaton (1 million metric tons).

8 While much research has investigated GHG impacts, conservation tillage has been implemented for reasons other than GHG mitigation—soil erosion control, soil quality enhancement, and reduced fertilizer needs (related to SOC retention).

(CTIC 2008)[9], and subsequent full extension of tillage-practice change to all 124 Mha of U.S. cropland (USDA NASS 2007a). With applicability on much of the nation's cropland, tillage changes, especially to NT, have significant potential for GHG mitigation.

Shifting tillage practices from the traditional moldboard plow (inversion of the soil profile) to some form of reduced tillage or NT has become important for erosion control, maintaining soil fertility, and improved crop health. Equipment and chemical development has also played a significant role, allowing seed placement without a prepared seedbed and weed control without soil disturbance. Conservation tillage can take various forms, ranging in levels of soil disturbance. In NT (also called zero-till) systems, crops are seeded directly into the previous season's stubble, with an implement cutting into the soil only enough to plant the seeds. Other conservation tillage practices include (1) ridge-till, where crop rows are planted on top of ridges that are scraped off for planting and rebuilt during the growing season; (2) strip-till, where only the seed row zone is disturbed (tilled); and (3) mulch-till, a form of reduced tillage with residue retained and spread out, but tillage activity just prior to planting. More discussion of the gradient of soil disturbance will be found in the upcoming report on soil C management.

Table 1. Estimates of soil C sequestration potential for no-till (NT) and other conservation tillage practices (compared to conventional till [CT]), U.S.

Citation	Specific Tillage Type	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
West and Post (2002)	NT	79 comparisons, mostly U.S.; reduced-till showed no impact	Low: 1.58 High: 2.60	
Alvarez (2005)	reduced-till and NT	11 reduced-till and 45 no-till comparisons, all 10+ years; no significant differences between reduced and no-till	0.95	
Six et al. (2004)	NT	254 SOC comparison data points; mostly U.S.; avg 20 yrs; low: dry climate; high: humid climate	Low: 0.36 High: 0.81	Low: 28.5 High: 65.3
Franzluebbers and Follett (2005)	NT	Calculated from estimates of 5 different regions	From -0.26 (Northeast) to +1.76 (Corn Belt)	Mean: 95.1
Lal et al. (1999)	mulch-till, ridge-till, and NT	Based on reviews and expert opinion	mulch-till and NT: 1.83 ridge-till: 2.20	
Follett (2001a)	includes mulch-till, ridge-till and NT	Reviews and expert opinion, assumes 60 Mha total area possible	Low: 1.09 High: 2.20	Low: 65.3 High: 131.0
Sperow et al. (2003)	NT and reduced-till	Modeled; low: 50% NT, 50% reduced-till; high: NT on all cropland; 129 Mha		Low: 135.8 High: 172.5
Franzluebbers (2010)	conservation tillage	147 comparisons, Southeast	1.65	
Martens et al. (2005)	NT and reduced till	15 studies in the Southwest; NT evaluated with other types of conservation till	1.03	
De Gryze et al. (2009)	conservation tillage	3 locations in California; 3-yr study	0.33	
Veenstra et al. (2007)	conservation tillage	Longest-running study in California, but only 5 yrs	0.00	

Source: Calculated from source to common units; cropland area, if needed for national calculation and not in given reference, is from U.S. agricultural census.

Since European immigrants settled in North America, much land has been under continuous cultivation, leading to significant reductions in soil organic matter (SOM) levels with respect to those under native conditions.[10] Current soil organic carbon (SOC) levels for agricultural land are 22%–36% lower than uncultivated land (Franzluebbers and Follett 2005; VandenBygaart et al. 2003). With soil exposed to the elements, erosion by wind and water removed organic material, and with it, crop nutrients. Lower SOM can result in lower soil fertility, with declines in crop production and greater reliance on fertilizer. In many regions, tillage reductions (NT and other conservation tillage) have been adopted to prevent or control soil erosion and to improve soil quality. Research and experience show that this reduced soil disturbance also decreases SOM decomposition rates, as demonstrated by a comparison of ^{13}C signatures in SOC from

9 While the CTIC data contain the best baseline information with respect to tillage practices, there are suggestions that this overestimates the actual adoption of NT. Fields that are in "rotational NT"—where one crop is direct-seeded but the next may not be—are counted as NT in the CTIC survey, which is based on "drive-by" field observations (C. Rice, personal communication, 28 September 2010).

10 At early stages of soil formation, organic matter accumulates at much higher rates than decomposition. Over time, as was generally the case for North American soil plowed by European settlers, the organic matter level appears to stabilize, with accumulation and decomposition rates equilibrated. Although there is never a "true" steady state or equilibrium, the changes are very small compared to what happens when the soil environment experiences a significant shift, such as tillage or different vegetation cover.

NT and conventional sites (Six and Jastrow 2006). This explains the observation of soil C sequestration that have been noted in many studies (see Table 1), reversing the trend initiated by the early agricultural settlers.[11]

While the average soil C sequestration for conservation tillage tends to be positive, some researchers have questioned this conclusion. For example, in a review of 67 studies, West and Post (2002) concluded that conservation tillage other than NT yields very little consistent soil C sequestration. However, this could depend on soil and climatic conditions, since cooler and wetter soils—for example, those in the upper Midwest U.S.—may achieve maximum C storage with occasional (e.g., biennial) tillage (Venterea et al. 2006). Clear definitions of practice and residue retention[12] may explain why NT tends to exhibit more consistent potential for soil C sequestration (Six et al. 2004; West and Post 2002).

Regional differences in climatic conditions within North America suggest that the greatest potential for NT to sequester soil C occurs in subhumid regions (precipitation-to-potential evapotranspiration ratios of 1.1–1.4 mm mm^{-1}; such as midwestern and southeastern U.S.). Average soil C sequestration rates for the Southeast are the highest, at 1.65 t CO_2e ha^{-1} yr^{-1} (Franzluebbers 2010), and other regions demonstrate average rates of up to 1.10 t CO_2e ha^{-1} yr^{-1} (Johnson et al. 2005; Liebig et al. 2005b; Martens et al. 2005; Six et al. 2004).

Lowest (or negative) sequestration rates with NT occur in the cold northern states (Dolan et al. 2006; Venterea et al. 2006) and arid western states (Franzluebbers and Steiner 2002; Martens et al. 2005). Rates of decrease in soil C following NT adoption in the colder, humid northeast are based primarily on field data from Ontario and Quebec (Gregorich et al. 2005), but have been extended to include the U.S. Northeast, bounded on the southwest by West Virginia, Maryland, Pennsylvania, and Delaware. Even though the average soil C change was negative, there was high variability in the results, some of which depends on soil and crop type (VandenBygaart et al. 2003). Negative soil C response to NT may be due to depressed corn yield (and thus residue amount) that result from reduced aeration at annual precipitation rates of >800 mm, or because of higher nightcrawler earthworm populations under NT that may enhance decomposition in eastern soils (VandenBygaart et al. 2003).[13] However, although limited in data, there is some evidence that NT practiced on farms in Pennsylvania can sequester SOC, at least in surface soil (Dell et al. 2008).

The negative soil C response to NT in the northeast may also be associated with increased N_2O emissions (Rochette et al. 2008a). With low-to-negative soil C sequestration and the potential for increased N_2O emissions, NT would likely have little GHG benefit in the northeastern U.S. Given that the Northeast accounts for only 4% of the country's total crop area, the region could be excluded from a NT incentive or offsets program, while still leaving the vast majority of U.S. cropland eligible.

Differences in SOC dynamics and C sequestration potential can also vary within a region, based on topography, water regime, and agricultural history. Some of this variability is due to disparate conventional tillage practices between regions (moldboard plow is common in some areas but not others) and the wide range of soil disturbance levels that can be classed as conservation tillage (more details can be found in the upcoming soil C management report). Greater levels of soil disturbance tend to result in lower SOC levels over time. For example, Lal et al. (1994) measured mean total soil C (0–15 cm depth) of 31.5, 25.5, and 21.6 t ha^{-1} for moldboard plow, chisel plow, and no-till, respectively, after 19 years. Reducing tillage from full-inversion moldboard plow is therefore likely to net a greater SOC sequestration response than where business as usual consists of chisel plowing or disc cultivating. Thus, landscape characteristics and management must be understood when extending estimates of mitigation potential.

Elevated N_2O emissions can also be a concern in regions other than the northeastern United States. Weather, soil characteristics, and time are all important factors, and results are variable, with some systems (high clay content, damp climate, wet soils, poor aeration) showing large increases in N_2O emissions after implementation of NT (Rochette 2008; Six et al. 2002b) and others finding little or no significant difference (Grandy et al. 2006; Li et al. 2005a; Parkin and Kaspar 2006; Robertson et al. 2000). Increased aggregate stability and improved drainage in some systems leads

11 Total soil C stocks are reported at different depths by different researchers. This makes direct comparisons between some studies less than ideal, unless correcting for soil sample depth, and some researchers propose that the shallow sample depths overstate soil C changes due to conservation tillage (Baker et al. 2007). In this literature compilation, sample depth was recorded where available, but given little consensus as to the acceptable level for depth and also the inability to extend results from a one depth to others (e.g., cannot say that concentration or bulk density is the same in the top 15 and the next 15 cm), we simplify to use the values available in the literature at the prevailing sampling depths. Further discussion regarding sampling depth and soil C comparisons between tillage regimes can be found in the upcoming T-AGG report on soil carbon management.

12 Planting in a NT system takes place in a narrow seedbed or a slot created by disc openers. Soil is undisturbed from harvest until planting, except for strips of less than 20%–30% of the row width. Residue cannot be burned and must be uniformly distributed over the field (USDA NRCS 2010).

13 Nightcrawler earthworms are not found in western Canadian soils.

to reduced N_2O emissions, but higher levels of soil C and N, higher bulk density, and greater soil water content can increase emission rates (D'Haene et al. 2008; Rochette 2008). A review of 44 data points noted higher N_2O emissions in initial years following transition to NT, but reduced emissions when compared to CT after the system has been in place for 10 years or more (Six et al. 2004). This concurs with observations of soil structure improvements following 4–6 years of NT. While there is little consensus about the impact of reduced tillage on N_2O emissions (Venterea et al. 2005), research seems to indicate that these negative GHG impacts are generally limited to poorly aerated soils (Rochette 2008), and time seems to play an important role. The impact of NT on N_2O emissions may also be affected by the type of N fertilizer. In one study NT (versus CT) reduced N_2O emissions by almost 50% following anhydrous ammonia, had no impact with urea ammonium nitrate, but increased N_2O emissions with broadcast urea fertilizer (Venterea et al. 2005); fertilizer type effects on NO_2^- accumulation appear to play an important role in the differences (Venterea and Stanenas 2008). Overall, soil aeration and drainage are important, interactions with fertilizer type may be possible, and the initial years after practice change could be problematic with regard to N_2O emissions and NT; further study may be needed to solidify and confirm these interactions. In contrast to NT, conservation tillage (some soil disturbance) tends to have no impact on N_2O emissions (Drury et al. 2006; Johnson et al. 2010; Kong et al. 2009; Venterea et al. 2005) or reduces emissions (Drury et al. 2006; Jacinthe and Dick 1997; Li 1995).[14]

When compared with conventional tillage, increased CH_4 uptake has sometimes (Six et al. 2004; Venterea et al. 2005), but not always (Robertson et al. 2000), been noted for both reduced till and NT. The total GHG impact is in any case marginal in contrast to soil C changes and N_2O fluxes. Any enhanced uptake is likely related to more stable and porous soil structure with a better environment for methanotrophic bacteria.

Upstream and process emission impacts resulting from no-till and other reduced-tillage systems are dominated by reduced field operations. Fuel reductions equivalent to 0.03–0.10 $CO_2e\ ha^{-1}\ yr^{-1}$ have been achieved for conversion from conventional to conservation tillage (Archer et al. 2002; West and Marland 2002) and 0.07–0.18 t $CO_2e\ ha^{-1}\ yr^{-1}$ to NT (Frye 1984; West and Marland 2002). While the yearly sequestration potential of conservation tillage and NT tends to diminish until soil C comes to a new equilibrium point over time (Six et al. 2002a), these process emission reductions are a perpetual benefit, even though the value may not be large in comparison to other options. Negative upstream impacts result from additional chemical herbicides for weed control after the traditional mechanical weed control (tillage) has been eliminated. While the GHG impacts of this increase in herbicides are not significant, other ecological and social factors are important to consider.

Fallow Management

Fallow periods, during which there is no crop on the land, can be reduced or managed to increase soil C stocks, especially if fallow coincides with conditions that could permit some vegetative growth (primary productivity). Depending on the region and cropping system, both the elimination of summer fallow and the use of winter cover crops have significant GHG mitigation potential.

Eliminate or reduce summer fallow

Summer fallow is the practice of leaving cropland unplanted for a summer, and is often practiced every second or third year for water conservation purposes on lands susceptible to crop failure from drought (Janzen 2001). It can also allow for better weed control and seedbed conditions in the eastern Pacific Northwest (Machado et al. 2006). The practice of summer fallow is most predominant in winter wheat grown in the dry lands of the central Great Plains. Under conventional tillage, summer fallow with wheat accumulates nutrients and is cost-effective where annual rainfall is less than 325 mm (Machado et al. 2006).

Elimination of summer fallow can sequester soil C at rates of over 2.0 t $CO_2\ ha^{-1}\ yr^{-1}$ (Table 2), depending on tillage regime, climate, and other

GHG Impact Summary

GHG Category	Eliminate Summer Fallow
Soil Carbon, t $CO_2e\ ha^{-1}\ yr^{-1}$	**0.48** (-0.88 – 2.35)
Land Emissions, N_2O and CH_4, t $CO_2e\ ha^{-1}\ yr^{-1}$	**-0.03** (-0.30 – 0.16)
Process and Upstream, t $CO_2e\ ha^{-1}\ yr^{-1}$	**-0.12** (-0.23 – -0.07)
Average Net Impact, t $CO_2e\ ha^{-1}\ yr^{-1}$	**0.32**
Maximum U.S. applicable area, Mha	**20**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

14 An exception to this rule was in a corn-tomato system in California, where Kong et al. (2009) detected an elevated N_2O flux response from minimum-till in one of three cropping systems—the system receiving the most commercial N fertilizer. The systems with cover crops and fertilized with manure showed no such impact.

Soil Carbon Impacts of Crop Residue Management

Crop residues—the stalks, straw, and leaves left over after crops are harvested—represent most of the available carbon inputs on agricultural lands. In 2001, the total residue generated in the U.S. from 21 major grain and food crops was estimated to be 488 Mt/yr (Lal 2005). Using generalized estimates to calculate C sequestration (where 40%–42% of residue is C and 5%–20% of that C can be sequestered)[a] the soil C sequestration through residue retention is 36–150 Mt CO_2e yr^{-1}.

Several studies have measured a linear relationship between residue retention and soil carbon sequestration with a variety of cropping treatments, tillage scenarios, and geographic locations (Campbell et al. 2002; Carter 2002; Follett 2001a; Leifeld et al. 2009; Robinson et al. 1996), suggesting that residue input is a strong predictor of soil carbon content when management practices are equal. For the purpose of carbon accounting, the baseline practice is assumed to be complete residue retention, so any removals must be accounted for in a GHG mitigation project. In some NT or other conservation-tillage systems, full residue retention may be challenging, since, for a crop like grain corn, the large amounts of residue may not be easy to manage, and often require tillage for incorporation or removal. However, equipment and practice development continue to work out logistics.

The question of how much residue can be harvested without decreasing the existing carbon stock is important, especially in light of future cellulosic biofuel production. Unless other factors are simultaneously changed to decrease decomposition rates and offset the change in organic matter inputs into soil, residue removal will directly reduce soil C. In fact, higher soil C decomposition rates have been measured with corn stover removal (Clapp et al. 2000; 2005). In the past, residue harvest thresholds have been based on erosion prevention, but it is generally accepted that soil C maintenance will require more retention. The DOE estimated threshold for using corn stover for biofuels is 30% (i.e., 70% stays on the field) (Follett 2001a; 2008). Residue harvest thresholds, based on erosion control management rather than soil carbon balance, vary from 20%–30% (Nelson 2002). In some areas, the residue generated by certain crops is not sufficient to sustain SOC levels (Wilhelm et al. 2007; Wilhelm et al. 2004). Climatic factors have a significant impact in these determinations, and the amount of residue required to maintain consistent levels of SOC ranges from a low of < 1 t ha^{-1} yr^{-1} in Montana to a high of > 9.25 t ha^{-1} yr^{-1} in Minnesota (Wilhelm et al. 2004). By lowering nutrient inputs, residue removal can also negatively impact yields, further reducing carbon sequestration potential (Wilhelm et al. 1986).

a. A 5% rate of sequestration based on C isotope measurements in a dryland temperate soil has been observed by Paul et al. (1997) and Lal et al. (2003). A sequestration rate of crop C input of up to 20% has been observed in a 10-year study in Texas (Franzluebbers et al. 1998).

factors. With slight negative impacts on soil N_2O and process emissions, this leads to an average net GHG mitigation potential of 0.32 t CO_2e ha^{-1} yr^{-1} (or an average of 0.64 t CO_2e ha^{-1} yr^{-1}, if implemented only in NT systems).[15] As of 2007, there were 6.3 Mha of cropland under summer fallow in the U.S., a drop from 6.7 Mha in 2002 (USDA NASS 2007a). This is in agreement with Sperow et al.'s (2003) estimate that 20 Mha of cropland is summer fallowed at some point during a crop rotation (generally every two or three years).

However, summer fallow also reduces SOC. The elimination of plant C inputs during the fallow period can enhance soil C mineralization through increased moisture and temperature (Haas et al. 1974), and/or increase decomposition if tillage is used during the fallow period (Janzen et al. 1998). Summer fallow can also accelerate soil C loss through erosion, although this may actually redistribute C locally rather than release it to the atmosphere (Gregorich et al. 1998).

Simple elimination of summer fallow in wheat-fallow systems has not always had positive yield or soil C results—especially under conventional tillage or where water availability remains limited (West and Post 2002). In some cases, reducing, rather than eliminating summer fallow in the Great Plains may be a viable option for soil C benefits, as there is a direct relationship between relative rate of change in SOC and cropping frequency (Campbell et al. 2005). Sherrod et al. (2003), however, found that median SOC values were similar for fallow-crop-crop and fallow-crop-crop-crop rotations. Another option is to increase diversification, so that crop mixes include something other than wheat, such as corn, millet, or sunflower (Halvorson et al. 2002a; Sherrod et al. 2003). With winter wheat, the need to plant in the fall may make short season forage crops like triticale or foxtail millet attractive summer fallow replacements (Lyon et al. 2007). Diversified rotations have resulted in soil C increases (2.71 ± 1.91 CO_2e ha^{-1} yr^{-1}) that are more than eight times that of continuous wheat (West and Post 2002).

Summer fallow reduction or elimination has the most effective and consistent soil C benefits when combined with NT. By retaining more crop residue and reducing water loss from the soil profile, NT can provide sufficient moisture for annual crop production. In a review of 67 studies, West and Post (2002) found that moving from CT to NT in wheat-fallow rotations showed no significant increase in SOC, but conversion to NT in continuous wheat systems was generally positive and increased soil C by 0.92 ± 0.95 t CO_2e ha^{-1} yr^{-1} (10 paired treatments). The only reported evidence of

15 As seen in following paragraphs, research indicates that summer fallow elimination sequesters more carbon in NT systems (compared to conventional). One key reason is that NT prevents water from evaporating, which would have otherwise been conserved by not growing a crop in the fallow season (the impetus for summer fallow in dry-land farming). The estimate of 0.64 t CO_2e ha^{-1} yr^{-1} assumes that summer fallow is eliminated, and that the land is in NT; this estimate is just that due to the change in summer fallow. If the land was previously conventionally tilled, additional soil C sequestration would occur with conversion to NT, and the interaction effect of the two activities would need to be assessed. To improve estimates of potential, it would be useful to have better baseline data that detail current NT and summer-fallow, separate and together.

SOC reductions after eliminating summer fallow have been in combination with conventional tillage (Halvorson et al. 2002a; Sainju et al. 2006).

Table 2. Estimates of soil C sequestration potential from eliminating or reducing summer fallow, U.S.

Citation	Tillage Regime	Crop Type/Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Sperow et al. (2003)	Not specified	All of U.S., mainly wheat	Model and IPCC method, 20 Mha	0.59	11.7
Lal et al. (2003)		All of U.S.	Assumes 9.4 Mha	0.73	6.9
West and Post (2002)	Various	Global, mostly U.S.	11 studies	Low: - 0.07 High: 0.51	
Halvorson et al. (2002a)	CT, NT, minimum till	Northern Plains winter wheat	0–30 cm depth	CT: - 0.46 Minimum-till: 1.04 NT: 2.35	
Horner (1960)	Not specified	Pacific NW	Long-term experiment	0.83	
Machado et al. (2006)	CT	Eastern Pacific Northwest	0–40 cm depth Fertilized	1.21	
Potter et al. (1997)	NT and stubble-mulch	Southern Plains wheat	0–20 cm depth fertilized	NT: 1.54 SM: 0.37	
Sainju et al. (2006)	CT, NT NT	Rocky Mountains spring wheat Rocky Mountains spring wheat	0–20 cm depth, eliminate SF Reduce SF; W-F > W-W-F	CT: -0.88 NT: 1.03 2.56	
Sherrod et al. (2003)		Wheat, corn or sorghum, millet	Fallow every 2 > every 3 or 4 yrs	Positive trend, but insignificant	

The increase in SOC from transition to NT in continuous wheat cropping may not be immediate. Due to the limited water supply in the Great Plains, the amount of crop residue returned to the soil is lower than in other regions, and requires more time to provide a significant increase. In some cases, significant increases in SOC were not observed even after four to eight years (Halvorson et al. 2002b; Ortega et al. 2002). Also, further examination of regional differences may be warranted. Many studies have been conducted in Canada and the central Great Plains (Halvorson et al. 2002b; Ortega et al. 2002; Sherrod et al. 2003), but these studies may not be applicable to the northern Great Plains due to differences in temperature, rainfall, and growing-degree days (Sainju et al. 2006). Cold weather in the northern plains may also delay decomposition, thus having a positive soil C impact.

Eliminating summer fallow has been observed to increase (Boehm et al. 2004) or decrease (Grant et al. 2004) N_2O emissions, but the general conclusion seems to be that there is little consistent impact (Del Grosso et al. 2002; Desjardins et al. 2005). We assume that field operations are not impacted (both fallow and cropping require equipment passes), but an additional crop of wheat in a two- or three-year rotation will lead to more N fertilizer use,[16] increasing process emissions by an average of 0.13 $t\ CO_2e\ ha^{-1}\ yr^{-1}$.

In summary, since NT (with associated chemical weed control) makes it possible in many areas to conserve water without summer fallow, while also maintaining and enhancing soil C and soil fertility, it seems the most viable approach to achieving the best possible GHG benefits associated with summer fallow reduction or elimination.

16 Assuming that N fertilizer use for wheat is between 30% and 60% of the rate for corn.

Use winter cover crops

Adding winter cover crops to a crop rotation can increase levels of soil C (Table 3), and also reduce N_2O and fertilizer-related emissions. Cover crops are typically grown in combination with main summer annuals such as corn, soybean, and spring cereals to control nitrate leaching, provide nutrients (especially N) as "green manure," conserve water resources, reduce insect and pathogen damage, and improve soil quality (Hargrove 1991; Laub and Luna 1992; Sperow et al. 2003; Stivers and Shennan 1991).

GHG Impact Summary

GHG Category	Winter Cover Crops
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**0.84** (0.37 – 3.24)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.20** (0.00 – 1.05)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.56** (0.41 – 0.81)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**1.50**
Maximum U.S. applicable area, Mha	**74**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Some experts deem the use of winter cover crops feasible in most areas of the U.S. (up to 79% of active crop area, i.e., 100 Mha), with the exception of semi-arid regions where soil moisture can limit crop growth (Sperow et al. 2003).[17] Others suggest lower potential, with area estimates by Lal et al. (1999), who anticipated the use of cover crops on 51 Mha of land in the U.S., and Donigian et al. (1995), who applied projections to 87 Mha. For this analysis, we use an average of these three estimates minus the baseline (79 – 5 = 74 Mha). In drier regions, it is also necessary to consider the net benefit of cover crops if irrigation is necessary for their inclusion. Adding winter cover crops can result in soil C increases of more than 3 t CO_2e ha^{-1} yr^{-1} (De Gryze et al. 2009; Sainju et al. 2002; Veenstra et al. 2007), with the highest rates in warmer-winter locations such as California and Georgia. Despite these favorable results, as of 1995, cover crops were used on only 4% of the major field crop area (Paustian et al. 2004).

Cover crops can also increase N and water-use efficiencies, resulting from higher return of vegetative residues to the soil (Teasdale et al. 2000). Studies show that cover crops can significantly reduce the need for chemically derived N fertilizer, since both legumes and grass species will scavenge and recycle from 150 to 300 lbs of mineral N ha^{-1} yr^{-1} that would otherwise be lost by leaching (Delgado et al. 2007), making those nutrients available for subsequent crops upon decomposition of the cover crop,[18] as well as avoiding off-site N_2O emissions. Leguminous cover crops also fix atmospheric N into plant-useable forms (Gregorich et al. 2005), allowing further N fertilizer savings. Alluvione et al. (2010) and Utomo et al. (1990) were able to eliminate fertilizer N and completely meet crop N needs with vetch winter cover crop in northwestern Italy and Kentucky, respectively. With this reduction in soil mineral N concentrations during otherwise fallow seasons, field N_2O emissions might also be reduced (Alluvione et al. 2010; Delgado et al. 2007; Paustian et al. 2004). However, the additional C from winter cover crop residues may in some cases be used by microbial populations to immobilize available nutrients (such as N) in the microbial biomass (Wyland et al. 1995), so it has been suggested that agricultural management of cover crops should be carefully monitored for the synchronization of N release with subsequent crop need.

17 There are also areas where growing season length makes it technically challenging to seed winter cover crops in the short window of time that follows harvest of the main crop while a cover crop can still germinate and grow sufficiently prior to cooler (or cold) winter weather. If planting a cover crop necessitates earlier harvest of grain (and thus requires grain drying), the GHG gains from the cover crop could be quickly negated.

18 Cover crop biomass is not removed, but retained through soil incorporation or other methods of residue management.

Table 3. Estimates of soil C sequestration potential from use of winter cover crops, U.S.

Citation	Crop Type/Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Sperow et al. (2003)	All of U.S., except dry regions (98.5 Mha)	Used model and IPCC method		83.5
Lal et al. (1999)	26 states with suitable climate, minus winter wheat area (51 Mha)	Expert opinion	Min: 0.37 Max: 1.10	37.4
Franzluebbers (2010)	SE, various crop types, fertilization regimes, etc.	Synthesis of studies, primarily NT	Min: 0.51 Max: 1.32	
Donigian et al. (1995)	Midwest, extend to national (87 Mha)	CENTURY model	Midwest: 0.84 U.S. total: 1.25	
Lee et al. (1993)	Corn Belt	EPIC simulation model (100 yrs), NT	0.15	
Teasdale et al. (2007)	Maryland; corn, soybean, and wheat rotation; hairy vetch and rye cover crops	8 yrs, 30 cm depth, NT	Increased SOC concentrations[a]	
Senthilkumar et al. (2009)	Corn-soybean-wheat with legume cover crops/Michigan	Organic system, 18 yrs, conventional till	0.55	
Dell et al. (2008)	Rye cover crop/Pennsylvania	6-13 yrs	0.00	
Kaspar et al. (2006)	Small grain cover crop/Iowa	NT corn/soybean rotation, 6 yrs	0.00	
De Gryze et al. (2009)	Legume winter cover crop/ California	9-11 yrs	2.17	
Veenstra et al. (2007)	Cereal-legume mix/ California	5 yrs, conservation and conventional till	3.24	

a. While this study demonstrated increased SOC concentrations due to cover crops, the soil bulk densities were not determined, so the values cannot be calculated on a per-hectare basis.

Adding cover crops to rotations may increase fossil C use through the increase in field operations (Paustian et al. 2004), although the resulting GHG impacts are likely quite small in comparison to fertilizer N savings.[19] If the inclusion of cover crops necessitates earlier grain harvest and increased grain drying, the fuel-related increased emissions may be significant (D. Miller, personal communication, April 2010), although this has not been quantified. Other changes to the main crop must also be considered, especially as they may impact the net GHG flux. In summary, with significant soil C sequestration potential, reductions in N_2O emissions on and off the field, and reduced energy use for fertilizer production, cover crops have significant promise as a GHG mitigation activity.

Changes to Crop Rotations

There are two ways to shift annual crop rotations: (1) replacing some planting of the main crop with a different crop—i.e., diversification of the crop rotation, and (2) adding another crop within an annual cycle—i.e., intensification of the cropping system. Winter cover crops (see previous section) are a special case of intensification, since cover crops are generally not harvested for product, but are tilled under or otherwise returned to the soil. Summer fallow reduction or elimination (see previous section) is another special case because intensification can use the same crop. In nearly all other cases, intensification involves diversification, since it is usually only possible to grow a second crop (or third, in some cases) that has shorter growing-season needs or otherwise different growth requirements than the main summer crop. Adding perennial crops to an annual rotation typically comprises both intensification (more days of the year with an actively growing crop) and diversification, but perennial crops will be treated separately in this paper because of unique characteristics. While perhaps not directly changing crop rotations, improved crop varieties may make such adaptations more feasible.

19 Fertilizer N savings of 150 kg N $ha^{-1}\ yr^{-1}$ would result in decreased process emissions of 0.56 t $CO_2e\ ha^{-1}\ yr^{-1}$, while field operations (assuming planting and cultivation similar to that of wheat for grain) would increase emissions by 0.13 t $CO_2e\ ha^{-1}\ yr^{-1}$.

Diversify annual crop rotations

Crop species can vary significantly in growth patterns, biomass production, water requirements, and decomposition rates, all of which impact net GHGs. Therefore, alternate species or varieties of annual crops can be added into rotations to promote soil C sequestration (Table 4)—increasing root and/or residue biomass, increasing root exudates, or slowing decomposition—or otherwise reduce emissions.[20]

GHG Impact Summary

GHG Category	Diversify Annual Crop Rotations
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**0.58** (-2.50 – 3.01)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.07** (-0.04 – 0.33)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**no impact**
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**0.65**
Maximum U.S. applicable area, Mha	**99**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Crop rotation diversification most often involves changing from a continuously cropped cereal or simple rotation to multiple crops over multiple years of a crop rotation. Total GHG impacts of crop rotations are dominated by soil C. For example, the SOC impact of vegetables = cotton = tobacco ≤ flax < wheat = lentil < fall rye ≤ hay (Hutchinson et al. 2007; Ogle et al. 2005).

While in general, crops with greater biomass production (see Table 5 for select crop residue yields) have more soil C sequestration potential, the relationship is also impacted by other factors. For example, even though oats produce significantly less biomass than corn, a long-term corn-oats rotation had greater soil C than continuous corn after 79 years of comparison in the Corn Belt (Khan et al. 2007). Inclusion of legumes—other than soybeans—in a rotation often has a significant positive SOC impact. In a 20-year study of crop rotations in Nebraska (Western Corn Belt) there was no SOC benefit to two-year rotations (corn-soybean and sorghum-soybean) over continuous monocropping, but four-year rotations with oats and clover significantly increased SOC content by 12.4, 16.8, and 17.7 t CO_2 ha^{-1} after 10, 16, and 20 years (average of 1.24, 1.05, and 0.89 t CO_2^{-1} ha^{-1} yr^{-1}) (Varvel 2006). Therefore, residue amount, residue composition (e.g., N content), crop root exudates, differential decomposition rates, and crop impacts on soil water all play important roles.

Table 4. Estimates of soil C sequestration potential for diversifying crop rotations.

Citation	Region/Crop Type	Comments or Caveats	Potential (t CO_2e ha^{-1} yr^{-1})
West and Post (2002)	Global review, most from U.S., grain systems, enhance crop rotation	CT, 48 comparisons NT, 14 comparisons	Low: 0.07 High: 1.10 Low: -1.10 High: 3.01
Franzluebbers and Follett (2005)	North American review, more complex rotations	4 regions	Low: 0.44 High: 1.06
Johnson et al. (2005)	Midwestern U.S., grain systems	9 comparisons	Low: -1.80 High: 3.92
Franzluebbers et al. (1998)	Texas, wheat-soybean and sorghum-wheat-soybean vs. continuous	4 comparisons	Low: 0.51 High: 1.88
Alluvione et al. (2009)	Colorado, semi-arid irrigated, corn-based, add barley or dry bean	Measured CO_2 flux; compare with continuous corn	Barley: -0.11 Dry bean: 0.25
Omonode et al. (2007)	Indiana, corn-soybean vs. continuous corn	measured CO_2 flux	0.90
Varvel (2006)	Nebraska, 4-yr corn-based rotation with legume vs. continuous corn 2-yr rotation vs. cont. corn	Sequestration rate highest at 10 yrs, slowed after that	10-yr mean: 1.25 16-yr mean: 1.06 20-yr mean: 0.88 10-yr mean: 0.00
Khan et al. (2007)	Illinois, corn-oats or corn-soybean vs. continuous corn	Morrow Plots (est. 1876); 79-yr study	Corn-oats: 0.83 Corn-soybean: -0.67

Source: Values are as in citation (units adjusted, if necessary) or calculated from reported per unit area values; areas, if not in original citation, are from U.S. agricultural census.

As in other agricultural activities, net GHGs are also impacted by interactions with other land management practices. Within NT cropping systems, diversified crop rotations experience increased SOC by up to 0.75 t CO_2 ha^{-1} yr^{-1}, while very little impact has been observed when under conventional tillage (Franzluebbers 2010; West and Post 2002). Since crop rotation impacts on SOC may be small relative to the impacts of other management changes, it may take

20 Since the main GHG-mitigation goal of shifting crop patterns is to sequester soil C, impacts on N_2O and CH_4 emissions will be considered secondarily.

time (eight years plus) before the changes become apparent when using standard sampling and analytical approaches (Alluvione et al. 2009; Sainju et al. 2006).

Table 5. Residue production of selected crops, U.S.

Crop	Residue yield ($t\ ha^{-1}$)	2001 U.S. residue[a] production (Mt/yr)
Corn	10.1[a]	241.5
Barley	4.3[a]	8.1
Oat	5.6[a]	1.7
Soybean	4.3[b]	78.7
Sorghum	8.4[a]	19.7
Wheat	5.0[a]	80
Rice	6.7[a]	14.6
Cotton	6.7[a]	16.8
Sugar beet	5.6[a]	5.9

a. Source: Lal (2005).
b. Source: Allmaras et al. (1998) as cited by Follett (2001a).

Changes in crop rotations tend to have insignificant or very minimal impact on N_2O and CH_4 in most experiments (Alluvione et al. 2009; Johnson et al. 2010; Rochette et al. 2004; Venterea et al. 2010), although in one case where SOC gained 0.25 t ha^{-1} yr^{-1} in a corn–dry bean rotation (when compared with continuous corn), higher N_2O emissions erased 16% of that gain (Halvorson et al. 2008a). Process and upstream emissions would also be little impacted by most crop rotation adjustments, assuming similar fertilizer application and field operations.

Increase cropping intensity

Increasing the number of crops per year (or the amount of time that land is not fallow) adds to biomass inputs and can also reduce decomposition rates (Ogle et al. 2005).[21] Most research on changing cropping intensity relates to fallow reductions and winter cover crops (Liebig et al. 2010; Ogle et al. 2005; Peterson et al. 1998; Sherrod et al. 2003), but in some more temperate regions of the country, double- and triple-cropping are being explored for productivity gains, additional nutrient utilization (in the case of manure N especially), and soil C sequestration. In a 10-year cropping study in Texas, each additional month of cropping during a year resulted in increased SOC at a rate of 0.27 t CO_2e ha^{-1} yr^{-1} (Franzluebbers et al. 1998). Crop breeding programs and biotechnology could play a significant role in intensification, as crops that require a shorter growing season may make multiple crops or cover crops more feasible. Increased plant cover over a longer period of time through the year will utilize soil N and reduce N losses, although in some cases, additional N fertilizer may be needed for the second (or third) crop, which could increase N_2O losses. However, a paucity of research, plus the large diversity of cropping systems that would need to be examined, makes estimation of GHG impacts difficult.

Include perennial crops in rotations

Incorporating one to three years of a perennial crop (often alfalfa or grass hay) into annual crop rotations can result in significant SOC increase (Table 6), although it may be difficult to separate the impact of crop changes from tillage-reduction effects.[22] 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. They also increase evapotranspiration, drying soils and lowering soil C decomposition rates (Paustian et al. 2000). While good for maintaining SOC, in the long run this can be problematic in dry climates with rain-fed agriculture, as high water demand could lead to low-yielding

GHG Impact Summary

GHG Category	Include Perennial Crops in Rotation	Change from Annual to Perennial Crop
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**0.57** (-1.75 – 2.20)	**2.26** (0.00 – 4.67)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.03** (-0.55 – 0.55)	**0.12** (-0.55 – 0.84)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.17** (0.14 – 0.26)	**0.52** (0.37 – 0.67)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**0.76**	**2.90**
Maximum U.S. applicable area, Mha	**56**	**13**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

21 Slower decomposition rates with intensification may be a result of reduced soil water content due to greater evapotranspiration.

22 Inclusion of perennial crops is most often associated with fewer tillage operations, since seedbed preparation is dramatically reduced, and management generally does not involve growing-season tillage for weed control.

annual crops in following seasons (Paustian et al. 1997; Paustian et al. 2000). In more humid regions, these considerations are unimportant, and perennial crops can sequester substantial soil C. For irrigated cropland, the impact on water requirements (and associated energy and GHGs) will need to be considered.

Most crop rotation research studies in the peer-reviewed literature that include perennial hay have focused on Europe (Freibauer et al. 2004) or Canada (Hutchinson et al. 2007; VandenBygaart et al. 2003), so estimates of technical potential rely somewhat on these international values in addition to the limited U.S. data. Perennial plantings that involve land-use change (set-aside, crop-grazing land, short-rotation woody perennials) are discussed in separate sections below.

Table 6. Soil C sequestration potential of including perennials in crop rotations.

Citation	Region/Crop Type	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)
Johnson et al. (2005)	Midwestern U.S., corn-oat-meadow vs. monoculture corn	3 comparisons	Low: -1.58 High: 4.29
Khan et al. (2007)	Illinois, include hay in corn-oats rotation	Morrow Plots (est. 1876); 79-yr study	0.40
Lal et al. (1994)	Ohio, include hay in rotation	19-yr study; 3 tillage systems, only one showed increase in soil C	NT: -4.52 Chisel plow: -2.04 Moldboard plow: 1.32
Gregorich et al. (2001)	Ontario, monoculture corn vs corn-oats-alfalfa-alfalfa	35-yr study	2.03
Freibauer et al. (2004)	Europe, perennial grasses and permanent crop		2.20
Hutchinson et al. (2007)	Canada, hay in wheat rotation	Soil with low SOC at beginning gained at higher rate	Low: 0.29 High: 0.60
VandenBygaart et al. (2003)	Canada, review of 9 studies, hay in fallow-wheat rotation		0.81

Research into non-CO_2 GHGs suggests that changing crop rotations has a limited impact on N_2O and CH_4 fluxes (Johnson et al. 2010; Omonode et al. 2007). As in annual crop intensification, increases in plant cover (and growing plant roots) over a longer period of time through the year will scavenge mineral N and reduce N losses, potentially also reducing N_2O losses (Delgado et al. 2007; Robertson et al. 2000). Most often, perennial crops will have similar or lower fertilizer N requirements, and perennial legumes may decrease N_2O emissions while reducing fertilizer N needs. Rochette et al. (2004) found that N_2O emissions with legume crops are much lower than would be estimated from calculations of N additions through fixation. For alfalfa and soybean, an average of 0.48% ± 0.33% and 0.39% ± 0.27%, respectively, of fixed N was emitted as N_2O, versus the assumed 1.25% from the IPCC Tier I factor. Even with much higher soil mineral N concentrations under legume crops (compared with timothy grass), the N_2O emissions with legume crops were similar to that with the grass. Field operations depend largely on the previous and subsequent crop types, and because of the significant variability expected, we assume no consistent process emission impact in this analysis. In conclusion, this activity warrants consideration for GHG mitigation projects or programs.

Change from annual to perennial crops

Full conversion from annual crops to perennials doubles the soil C sequestration potential of including perennials within an annual rotation, with variable results depending on the crop type (Table 7). Typical examples include forages such as alfalfa and grasses or perennials grown for biofuels production. The mechanism for soil C change and impacts on N_2O and CH_4 emissions as well as field operations has already been discussed above. This high potential for GHG mitigation, without even including the fossil fuel offset of biofuels, has led some to conclude that farmers have much to gain with such systems. Lemus and Lal (2005) estimated that 13 Mha of potential area is available for transition to biofuels cropland in the U.S. over the next 50 years; this is the maximum area assumed available for transition from annual to perennial crops.

Table 7. Soil C sequestration potential of converting from annual to perennial crops (not including grazing land).

Citation	Crop Type	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)
Grandy and Robertson (2007)	Alfalfa compared with corn-wheat-soybean conventional till	Michigan	1.04
Liebig et al. (2005a)	Switchgrass vs. cultivated crops	Great plains and N Corn Belt	4.67
Freibauer et al. (2004)	Perennial grasses and permanent crops	Europe – a review	2.20
Potter and Derner (2006)	Restored grassland vs. continued cropping	Texas	0.00
Lemus and Lal (2005)	Switchgrass	U.S. – a review	2.93
Post and Kwan (2000)	Cropland to grassland	U.S. – a review	1.22

Short-Rotation Woody Crops

While most plantings of trees on agricultural or otherwise nonforested land are termed "afforestation," rotation lengths of less than 30 years are generally excluded from forestry. Therefore, even though short-rotation woody crops (SRWCs) tend to be very different from other agricultural crops—being perennials, but not providing food—they are included in this assessment as an agricultural land management practice. The short rotation period means that producing SRWCs may also be more attractive to farmers as a land-use option, since their management "feels" more agricultural.

GHG Impact Summary

GHG Category	Short-Rotation Woody Crops
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**2.60** (0.00 – 10.63)
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.76** (0.00 – 1.52)
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.65** (0.41 – 0.90)
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**4.02**
Maximum U.S. applicable area, Mha	**40**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

SRWCs include poplar, willow, mesquite, alder, Chinese tallow, and other fast-growth woody perennials, with a wide range of adaptability and disease resistance (Lemus and Lal 2005). Researchers estimate that between 40 and 60 Mha of land in the U.S. are available—from highly eroded land or abandoned mine land—for planting in fast-growth vegetation, including SWRC and herbaceous crops (Lemus and Lal 2005; Tuskan and Walsh 2001). The end purpose at harvest could be pulp/paper or bioenergy production.

The primary carbon sequestration in woody biomass plantations is within above ground material (Ranney et al. 1991), although end use essentially determines whether and how the aboveground biomass is counted in the GHG balance. As a conservative estimate, we assume no GHG benefit for aboveground biomass, limiting the focus to soil C. On average, the soil C sequestration potential is greater than that possible with annual crop species, although the estimates are highly variable and impacted by factors such as species choice and climate (Table 8). Some research has found that soil C decreases during the initial years of SRWC establishment, with a subsequent increase over time (Grigal and Berguson 1998; Hansen 1993).

Nitrous oxide and methane fluxes are more difficult to determine. In a review across Europe, Machefert et al. (2002) noted much lower N_2O emissions in forested versus agricultural land, while others have found little differences between annual crops and poplar plantations (Scheer et al. 2008). Therefore, while we assume some N_2O emission reduction, certainty in this estimate is tentative, and more research is needed to strengthen conclusions. Field operations and N fertilizer application rates are significantly reduced, leading to GHG benefits of 1.8 $t\ CO_2e\ ha^{-1}\ yr^{-1}$.

In addition to soil C sequestration, SRWCs also have the opportunity to displace fossil fuel if used for bioenergy production, but this is only of GHG benefit if the C absorbed by the plants is "additional" to that which would otherwise be absorbed (Searchinger 2010). The estimated potential of bioenergy displacement of fossil fuels from SRWCs is 18–20 $t\ CO_2e\ ha^{-1}\ yr^{-1}$ (Graham et al. 1992; Tuskan and Walsh 2001). Lemus and Lal (2005) estimate that nearly 10% of U.S. fossil fuel emissions could be offset by the use of willow and poplar for biofuels, assuming 28.6 Mha of severely to highly eroded land were put into SRWC production and an additional 29.8 Mha were allocated to switchgrass. In all cases of converting current cropland to SRWCs, indirect land-use change impacts may limit the real GHG mitigation potential due to displacement of crops onto other land currently in perennial crops, grassland, or forest production.

Table 8. Soil C sequestration physical potential from planting short-rotation woody crops.

Citation	Crop type/region	Comments	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Sartori et al. (2006)	Various species	Review of 7 field studies, 12+ comparisons, 3–18 yrs	2.93	
Lemus and Lal (2005)	Willow and poplar/ Quebec	Review of 3 field studies	Willow: 3.30 Poplar: 4.03	
Nabuurs and Mohren (1993)	Productive, fast-growth forests	Modeled with CO_2FIX, 45 yr poplar rotation, 30-yr pine rotation	Poplar: 5.46 Loblolly pine: 10.63	
Tuskan and Walsh (2001)	Various species/U.S.	Modeled, suggests applicability to 40 Mha	6.60	263.8
Schlamadinger and Marland (1996)		Modeled, 7-yr rotation	0.66	
Wright and Hughes (1993)	SRWC/North Central U.S.	Modeled, estimated that 14–28 Mha of cropland available for energy crops	1.10	23.1
Coleman et al. (2004)	Poplar/Midwest	27 adjacent sites of poplar and cropland, oldest poplar stand was 12 yrs	No significant difference	
Hansen (1993)	Hybrid poplar/Midwest	9 adjacent sites, 12- to 18-yr-old stands	5.97	
Heller et al. (2003)	Willow/New York	2- to 12-yr-old willow chronosequence	No significant difference	
Grandy and Robertson (2007)	Poplar/Michigan	12 yrs old	0.70	

Agroforestry (Windbreaks, Buffers, etc.)

While agroforestry is most commonly implemented in the tropics—with high C sequestration potential when compared to other agricultural land uses—it is also gaining some interest in North America. The Association for Temperate Agroforestry (AFTA) defines agroforestry as an intensive land management system that "optimizes the benefits from the biological interactions created when trees and/or shrubs are deliberately combined with crops and/or livestock" (AFTA 2010). The soil C sequestration potential can be significant, and although GHG impact calculations in this analysis do not include aboveground biomass, this can comprise a large C pool (see Table 9 for examples), but the net impact depends on the end use. Non-CO_2 gas fluxes, process emissions, and N fertilizer effects are assumed to be the same as those for SRWCs, above.

GHG Impact Summary

GHG Category	Agroforestry
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**2.71** (0.84 – 4.23)
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.76** (0.00 – 1.52)
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.39** (0.31 – 0.47)
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**3.85**
Maximum U.S. applicable area, Mha	**10**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

As estimated by Heath et al. (2003), a total area of 80 Mha of land in the U.S. may have the potential to accommodate alley cropping (20% of land in trees), plus 85 Mha of land with windbreaks (5% of land in trees), plus 70 Mha in silvopasture systems. In contrast with other agroforestry in temperate systems, silvopasture (trees planted on grazing land) tends to generate lower soil C sequestration rates (0.1 to 0.47 t $CO_2\ ha^{-1}\ yr^{-1}$, Heath et al. [2003]; Nair and Nair [2003]). While we document some results from silvopasture studies in comparison with other agroforestry (Table 9), the calculated biophysical GHG mitigation potential in this analysis focus only on cropland agroforestry. Therefore, we estimate between 5% and 20% of 80 Mha as the maximum applicable area, that is, the land area under trees alone, assuming that the adjoining crop area is unaffected.[23] As with SRWCs, indirect land-use change impacts (leakage) may significantly decrease the potential.

There are five basic types of agroforestry practices today in the United States: windbreaks, alley cropping, silvopasture, riparian buffers, and forest farming. Multiple benefits are associated with agroforestry, such as diversified income

23 There is very little data on the adjacent cropland area effects, and research seems to indicate that the impacts on soil C and other emissions in the area directly affected by trees would nevertheless overshadow any adjacent impacts. The exception to this rule is in silvopasture, where potential impacts refer to the entire area, since the integration of trees and pasture is difficult to disentangle.

sources, increased biological production, better water quality, improved habitat for both humans and wildlife, and C sequestration. If the land management goal is C sequestration, agroforestry can be optimal because land-use systems that include trees can result in higher C sequestration rates than those that are limited to annual crops, pastures, or grasslands. The C sequestration potential of agroforestry varies widely depending on the specific practice, individual site characteristics, and the time frame. In general, alley cropping, silvopasture, and forest farming have higher rates of sequestration than windbreaks and riparian buffers, since they are more intensively planted with trees.

Competition for light, nutrients, and water can make tree systems undesirable near cropland. Direct competition between trees and crops can be addressed by retaining tree strips only in the middle of larger field margins, where grass strips provide a buffer between tall-canopy trees and the annual crop, even though there may be some water competition between the grass strip and the crop (see, for example Falloon et al. 2004).

Table 9. Estimates of soil C sequestration potential for agroforestry practices in the U.S.

Citation	Activity	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Nair and Nair (2003)	Silvopasture Alley cropping Riparian buffers Windbreaks	U.S. estimates, SOC and biomass		33.0 270.8 7.3 14.7
Lal et al. (2003)	Alley cropping Riparian buffers Windbreaks	Only soil C		67.7 1.8 3.4
Dixon et al. (1994)	Agroforestry	Above- and belowground, U.S.	10.57	
Sharrow and Ismail (2004)	Silvopasture	Oregon; 11-yr study	1.91	
Bailey et al. (2009)	Corn-soybean with tree-grass buffer	Missouri; 13-yr study comparing arable to tree-grass buffer	1.56	
Albrecht and Kandji (2003)	Silvopastoral system	Tropical North America, includes aboveground	6.6–14.5	

As with SRWCs, N_2O emissions are reduced in the land area planted to trees (Machefert et al. 2002) and fuel- and fertilizer-related upstream and process emissions are also reduced. The net GHG mitigation values apply specifically to the land area planted to trees, since the area ratio of trees:crop can vary widely.[24] In summary, while the leakage issue of shifting some crop area out of production must be considered carefully, the GHG mitigation potential of agroforestry makes some versions of these varied practices worth considering.

Application of Organic Material (Especially Manure)

The U.S. produces a large amount of organic material, including livestock manure, municipal solid waste, and biosolids, that can be used as soil amendments to fertilize croplands and pasture (Table 10). The most commonly used organic soil amendment applied to agricultural lands is animal waste, such as poultry litter and cattle manure. In 2007, there were approximately 9 Mha of cropland treated with manure fertilizers in the U.S. (USDA NASS 2007a). This is less than 8% of total cropland, with manure applied most often to land in corn production (USDA ERS 2009).

Factors such as the decreasing cost of inorganic fertilizer, the increased average farm size and specialization, adoption of confined animal feeding operations, and policy and government incentives aimed at crop yield increases per land unit have led to decreased use of organic fertilizers (Chesworth 2008). High nutrient variability in manure makes efficient nutrient management more complex than with commercial fertilizer. Nevertheless, the nutrient benefits and fertilizer savings, combined with potential for mitigating GHGs, is leading to growing interest in the use of organic soil amendments.

GHG Impact Summary

GHG Category	Organic Material
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**2.19** (0.18 – 5.10)
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.19** (-1.35 – 1.81)
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**-0.18** (-0.57 – 0.21)
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**2.20**
Maximum U.S. applicable area, Mha	**8.7**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

24 See, for example, Falloon et al. (2004), where between 2.3% and 21.3% of the total cropland is shifted out of production and into field margins.

Table 10. Annual production of organic waste, U.S.

Organic Material	Organic materials production in the U.S. (dry Mt yr^{-1})[a]
Animal manure	156
Municipal refuse	130
Logging and milling waste	32
Sewage sludge	4
Food processing waste	3
Industrial organics	7

a. Adapted from Chesworth (2008).

Numerous studies have measured increases in soil C after application of manure (Table 11), leading to an average soil C sequestration potential that is significantly greater than that from tillage changes or winter cover crops. These experiments do not address the related soil C impact on land that may no longer be receiving the manure application. Therefore, whether this soil C increase can be counted as additional C storage for GHG mitigation, however, depends on the baseline situation – i.e., what would have been done with that organic material otherwise. If the manure is simply moved from one location to another – so that a different site experiences the soil C increase, the net change in soil C over the whole system is unchanged. Therefore, full life-cycle analysis is especially important with this activity, and improved nutrient distribution (with air and water quality benefits) might carry more incentive for manure application adjustments than GHG implications.

Compost: Net GHG Impacts

Compost application to agricultural soils can reduce net GHG emissions in two ways. First, by the displacement of more typical anaerobic storage options with aerobic decomposition of organic material, CH_4 emissions can be reduced. Second, soil application can sequester soil C and displace N fertilizer use, also potentially reducing field N_2O emissions. Some of these benefits have already been recognized in efforts to divert organic waste from landfills. For example, the Climate Action Reserve (2010) has published a GHG reduction protocol dealing specifically with organic waste composting. However, the focus of this report remains on agricultural land management, rather than point-source GHG emission reductions.

When livestock farm systems are producing organic nutrients in excess of crop needs on the receiving land, there is an opportunity for net GHG mitigation by applying this excess to other cropland as organic soil amendments, thereby increasing soil C sequestration and displacing fertilizer N, which hopefully leads to lower N_2O emissions (Brown et al. 2008; LaSalle and Hepperly 2008; Smith et al. 2001). Manure is a common feedstock for compost, and also a significant source of organic material in the U.S. Therefore, to examine the GHG mitigation potential for compost application, we compare the net GHG impacts of direct application of manure (from typical storage conditions) versus composting of the manure prior to land application.

What are the potential GHG benefits associated with composting of manure prior to land application? By stabilizing the organic matter through a largely aerobic process, composting of manure can generate much lower net GHG emissions during the storage period and after land application, compared to standard anaerobic manure storage in stockpiles or manure storage lagoons. Pattey et al. (2005) found that, compared to untreated manure storage, composting reduced total GHG emissions (CH_4 plus N_2O) prior to land application by 31% to 78%, depending on C:N ratio, moisture content, and aeration status. The impact of composting on emissions post-land-application is of further interest. Fronning et al. (2008) examined GHG fluxes following land application of solid beef manure and composted dairy manure over a three-year period. Net CH_4 flux was minimal (< 0.01 t CO_2e ha^{-1} yr^{-1}), and untreated manure generated higher N_2O emissions than did compost (0.9 versus 0.7 t CO_2e ha^{-1} yr^{-1}). However, these land emission impacts were small when compared to soil C sequestration rates, which were 1.8 times greater for compost than for manure, suggesting that the organic matter stabilization during the compost process reduces post-application respiration losses. The net C sequestration difference of untreated versus composted manure may also be affected by respiration losses during the composting process, so further research may be needed to address these life-cycle issues.

While soil C sequestration rates tend to be positively related to the rate of manure application, climate also plays a role, with lower carbon retention rates in warm climates (7 ± 5% of applied manure C retained in soil) compared with cooler ones (23 ± 15%)[25] (Risse et al. 2006), although soil moisture seems to have little effect (Johnson et al. 2007). Within a particular climatic region, a key question is whether decomposition rates of manure-source C are impacted by differential application rates. If the rate is affected, GHG mitigation would be maximized at the application rate where the greatest proportion of manure C is retained in the soil. Angers and N'Dayegamiye (1991) found a greater proportion of the manure C retained in the soil for 40 Mg ha^{-1} versus 80 Mg ha^{-1} applications every two years.[26] Chang et al. (1991) compared three different levels of cattle feedlot manure application on two types of cropland (30, 60, and 90 Mg manure ha^{-1} yr^{-1} on dryland and 60, 120, and 180 Mg manure ha^{-1} on irrigated land) and found that the soil C increased by

25 For these values, Risse et al. (2006) do not indicate the amount of time elapsed, but it is reasonable to assume that they are comparable.

26 Ten yrs of application of 40 Mg manure ha^{-1} increased soil C—in the 0–15cm layer—by 8.1 g kg^{-1} and 80 Mg manure ha^{-1} increased soil C by 12.2 g kg^{-1}.

similar proportions of the total organic C added in the manure, regardless of the application rate. However, on the same site, 16 years after those manure applications ceased, Indraratne et al. (2009) present decay model evidence of higher organic matter decay rates (soil organic N) for the sites that received the highest manure application. These results suggest greater organic matter (including C) stabilization with lower application rates, and thus GHG mitigation potential. On the other hand, C storage is not guaranteed with manure application, and Angers et al. (2010) noted increased native soil C decomposition with 20 years of nutrient-rich swine manure application to grassland soil (at low rates), and higher application rates were needed to maintain soil C levels.

Since the majority of manure is already land-applied, it is necessary to estimate the total amount in excess—currently applied at rates higher than crop nutrient needs—and thus available for application on additional land area. For effective nutrient management, manure application rates should be determined based on either N or phosphorus (P), depending on crop needs and manure nutrient content. A 2001 USDA report indicated that, on average in the U.S., 60% of manure N and 70% of manure P was in excess of the optimal application rate for the originating farm (Gollehon et al. 2001), and thus would be available for other land. Most of the excess manure is produced on the 2% of farms in the largest farm size class. The report indicated that 1.2 Mt of manure N and 0.7 Mt of manure P are generated each year. If N were the main limiting factor, assuming a national average rate of N application equivalent to 105 kg N ha^{-1}, would indicate that an additional 6.39 Mha could receive manure fertilizer, replacing commercial N sources. If P were the main limiting factor, assuming a national average rate of P application equivalent to 39 kg P ha^{-1}, would indicate that 10.8 Mha could receive manure fertilizer, replacing other P fertilizer sources.

Table 11. Estimates of soil C sequestration potential with land application of organic materials, primarily manure.

Citation	Treatment/Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Follett (2001a)	Livestock/manure	Application rate of 250 kg N ha^{-1} yr^{-1} equiv. where economically feasible	Low: 0.73 High: 1.84	Low: 6.2 High: 15.5[a]
Lal et al. (1999)	Organic materials/U.S.	10% avg sequestration rate; animal manure, municipal solid waste, biosolids		Low: 11 High: 33[b]
Gilley and Risse (2000)	Manure/Midwest and Northeast U.S.	Long-term study		21.3
Franzluebbers (2005)	Poultry litter/Southeastern U.S.	19 studies; 5–21 yrs; range in C sequestration of 17%±15%	Low: 0.18 High: 5.1	
Kingery et al. (1994)	Poultry litter/Alabama	21± 4 yrs study sites	1.10	
Li (1995)	Livestock manure	DNDC model on 6 U.S. sites in IA, IL, KS, NE, CA, and FL; 1,000 kg C ha^{-1} yr^{-1} applied. Sequestration rates approximately doubled when 2,000 kg C ha^{-1} yr^{-1} applied.	Low: 1.9 High: 3.5	
Buyanovsky and Wagner (1998)	Livestock manure/Missouri	100-yr study, Sanborn Field	Wheat: 1.21 Maize: 1.95	
Collins et al. (1992)	Livestock manure/Oregon	56-yr study	0.7	

a. In this paper, Follett gives higher rates of national sequestration potential (avg of 23.1 Mt $CO_2e\ yr^{-1}$) than presented here, assuming that the land currently receiving manure also sequesters additional C (thus a total area of 18 Mha). In our calculations, we assume that only the land area that is "new" to manure application would qualify for GHG mitigation credit above the baseline in a program or project.
b. For manure application, it is difficult to estimate a potential sequestration rate per unit of area, since the main limiting factor is not the area available, but the amount of manure and other organic materials available. Lal et al. (1999) rightly address this issue by estimating a total national potential for soil C sequestration, but not calculating that into a per ha estimate.

Gollehon et al. (2001) also indicated that while 80% of excess manure N (77% of P) could be utilized within the county of origin, the remainder would need to find other uses or transportation out of county. Transporting manure short distances has been shown to be economically feasible, from 15 km distance in single-axle trucks or pull-type manure spreaders for beef feedlot manure (Freeze and Sommerfeldt 1985) to much larger scales for poultry litter (Bosch and Napit 1992). The GHG impacts of transport need further evaluation,[27] but if commercial fertilizer were being displaced, the prevented process and transport emissions of the displaced fertilizer should also be considered.

27 We estimated an average of 0.57 t $CO_2e\ ha^{-1}\ yr^{-1}$ in transport-related emissions, assuming 100 km transport distance, 24.5 t load, and 380 g C km^{-1} emissions (emission values from Smith and Smith [2000]).

In practice, manure application can, but does not necessarily, lead to full displacement of commercial fertilizer. For corn production, 61% of farmers reported cutting their commercial nitrogen applications by an average of 58% when applying manure.[28] For other crops, such as oats and soybeans, the substitution percentage is less. Only 35% of oat farmers and 29% of soybean farmers reported reducing their chemical nitrogen applications by a substantial amount (reduced by 76% for oats and 85% for soybeans). However, the survey that measured these reductions did not account for the fact that some producers may not have been using commercial fertilizer to begin with, so they had nothing to reduce (USDA ERS 2009).

In summary, there are three possible pathways for GHG mitigation due to organic amendments. As stated, much livestock manure is applied in excess, resulting in lost opportunity for broader land application on surrounding farmland that could result in greater C sequestration and lower land emissions. Second, using organic amendments in place of commercial fertilizer, rather than in addition to it, will result in fewer upstream and process emissions associated with fertilizer production. Finally, composting organic amendments prior to land application could have the potential to increase C sequestration potential and reduce upstream emissions (see "Compost" box on p. 19).

Methane and N_2O impacts for organic additions to soil (including manure) are highly variable. Nitrous oxide emissions are positively correlated with native soil C content because C supports microbial activity and the processes that produce N_2O (Rochette et al. 2000), but negatively related to the C content of the manure or other organic source because the added C causes the microbial community to immobilize available N (Gregorich et al. 2005). Where manure can replace fertilizer N as the main N source, N_2O emissions tend to be lower (Alluvione et al. 2010), although this is not always the case, and depends on whether N_2O emissions are limited by available mineral N or by a carbon source for the microbes. Chantigny et al. (2010) found that in clay soil, manure had lower N_2O emissions, but in loam soil it had higher N_2O emissions compared with fertilizer N application. In the loam soil, the C in the manure provided the substrate for denitrifying bacteria. Another important consideration when accounting for the GHG impacts of manure use is that, compared to alternative manure handling, such as long-term storage in anaerobic lagoons or stockpiles, more frequent land application of manure can significantly decrease CH_4 emissions (Johnson et al. 2007).

Biochar Application

Biochar is produced by pyrolysis, the incomplete combustion of biomass into charred organic matter. While the pyrolysis process can be designed to capture heat and co-generate electricity as biofuel, the end product can also be used for soil application, with potential to increase soil C via three mechanisms: (1) by storing recalcitrant C in biochar soil amendments, (2) by stabilizing existing C in the soil, and (3) by increasing biomass production aboveground, thereby increasing C inputs into soil (Gaunt and Driver 2010). It can also have impacts on other GHGs as described below. Research suggests that this black carbon, or *terra preta*, thought to be charcoal from burning of organic matter hundreds of years ago, is a key factor for organic matter persistence in the tropics (Glaser et al. 2001; Lehmann et al. 2004). While few field studies exist in the U.S., similar effects are anticipated for temperate regions. As a result, soil C sequestration potential of biochar has been estimated based on calculations of available feedstock and expected C stability in the biochar (Table 12). Due to high variability and uncertainty, GHG impacts of possible productivity gains are not included in these calculations.

GHG Impact Summary

GHG Category	Biochar
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**3.37** (0.13 – 8.92)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**1.14** (0.82 – 2.93)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.70** (0.12 – 1.05)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**5.22**
Maximum U.S. applicable area, Mha	**124**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Possible biomass sources include milling residues (e.g., rice husks, nut shells, sugar cane bagasse), crop residues, biofuel crops, urban municipal wastes, animal manure, and logging residues, although their suitability is dependent on lignin content (Lehmann et al. 2006; Verheijen et al. 2009). Most research into biochar has focused on wood feedstocks in (sub) tropical regions, and scientific understanding of the properties of biochar from other feedstocks and in other regions remains limited (Verheijen et al. 2009). Not all forms of biochar

28 This somewhat low fertilizer adjustment response to applying manure nutrients may in some cases be influenced by the need to determine manure application rates based on phosphorus (P) content rather than N. In general, when manure is applied according to crop P needs, the N in manure is insufficient for that crop (because a greater proportion of N is lost in manure storage, hauling, and after field application).

have equivalent rates of C storage or stabilization, which are dependent on factors such as feedstock source and temperature, and rate or residence time of the pyrolysis process (Gaunt and Driver 2010). Therefore, further research into biochar application in U.S. cropping systems is needed to examine whether the anticipated impacts can be realized in large-scale implementation.

Table 12. Estimates of soil C sequestration potential for biochar application in the U.S.

Citation	Biomass Source	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
McCarl et al. (2009)	Crop residue	Sequestration rate per t of feedstock given; estimated residue of 5.5 t $ha^{-1}\ yr^{-1}$ on 120 Mha from (Lehmann 2007); assumes 20% residue removal[a]; fast pyrolysis yields lower sequestration than slow pyrolysis	Low: 0.13 High: 1.06	Low: 16.1 High: 127.1
Laird (2008) and Perlack et al. (2005)	U.S. harvestable forest and croplands	Assumes the U.S. can sustainably produce 1.1 x 10^9 Mg of biomass at 10% moisture annually from forest and cropland		510.1
Lehmann (2007)	Crop residue Fast-growth vegetation	Estimated residue of 5.5 t $ha^{-1}\ yr^{-1}$ on 120 Mha Estimated biomass of 20 t $ha^{-1}\ yr^{-1}$ on 30 Mha of idle farmland in U.S.	~4.9 ~19.6	~587 ~587
Roberts et al. (2010)b	Unused crop residue (U.S.)	141.1 Mt of unused crop residue, 0.53–0.57 t $CO_2e\ t^{-1}$ feedstock as sequestered soil C		77.7
Gaunt and Lehmann (2008)	Switchgrass, miscanthus, corn stover	Estimates for the UK for slow pyrolysis; low is corn stover, high is bioenergy crop with greater feedstock/area	Low: 4.3 High: 10.9	

a. We impose a limit of 20% of residue available for removal and pyrolysis into biochar. This is a conservative estimate of the threshold before erosion and soil carbon are affected. Also assume that biochar is returned to the same area from which residue was removed.

The response of soil to biochar amendments has to be biochar- and ecosystem-specific (Shneour 1966; Spokas and Reicosky 2009). When most plant biomass is decomposed, less than 10%–20% of the original C remains after 5–10 years (Lehmann et al. 2006). By comparison, research has found biochar to be highly stable, with mean residence time of hundreds to thousands of years (Lehmann et al. 2008; Roberts et al. 2010; Verheijen et al. 2009). Assuming that biochar application retains up to 50% of biomass C as a stable residue, Lehmann et al. (2006) estimated that up to 514 t CO_2e ha^{-1} could be stored under typical soil and plant species conditions. However, the characteristics of the applied biochar and the method of storage will greatly influence biochar recalcitrance. As with other organic material, biochar decay is facilitated by decomposition, microbial co-metabolism, abiotic processes, and physical breakdown and influenced by temperature, depth of burial, and soil cultivation (De Gryze et al. 2010). The complex interactions among these factors have not been studied extensively, so biochar recalcitrance remains widely variable in the literature. With biochar application, we also need to consider the impact of removing residue from the field in order to convert to biochar. If the biomass source is the same location as the receiving field, the true comparison is between leaving residue on the field (conventional) and removing the residue, converting it to biochar and then returning it to the field. If biomass is sourced from elsewhere, the GHG impacts of that movement need to be considered—similar to the issues discussed with manure application.

Beyond sequestration, biochar may have potential to mitigate GHGs by decreasing the need for fertilizer and other inputs, reducing upstream emissions (Gaunt and Driver 2010; Lehmann et al. 2006), and by reducing emissions of N_2O and CH_4 (Suddick et al. 2010), possibly due to production of ethylene, which inhibits microbial processes (Spokas et al. 2010). However, there are currently no peer-reviewed studies documenting suppression of N_2O or CH_4 emissions in the field (Sohi et al. 2010), even though a number of short-term studies and laboratory experiments have noted N_2O emission reductions of 50%–80% and near complete suppression of CH_4 with biochar additions (Fowles 2007; Lehmann et al. 2006; Renner 2007; Rogovska et al. 2008; Yanai et al. 2007). The potential seems particularly large in tropical soils, as shown by a study by Rondon et al. (2005) where 50% reduction of N_2O emissions was reported in soybean plots and 80% in grass stands. Yet, in another laboratory experiment, Yanai (2007) found that the impact of biochar on N_2O emissions was highly dependent on soil hydrology, where N_2O emissions varied from an 89% reduction in very wet soil to a 51% increase in N_2O emissions in drier soil. Another recent study found no reduction in N_2O production after urine application to pasture soils (Clough et al. 2010). Residence time of biochar in the soil may also be important, as increases in N_2O emissions have been noted when biochar is first applied to soil, with a shift to N_2O emission reduction over time as sorption capacity of biochar was enhanced with aging (Singh et al. 2010).

With many potentially variable factors,[29] full lifecycle assessments of biochar energy and GHG impacts are useful for comparing scenarios. For example, Roberts et al. (2010) calculate net GHG emission reductions of 0.86 t CO_2e t^{-1} of feedstock for corn stover. McCarl et al. (2009) estimate a net mitigation potential of 0.823 t CO_2e t^{-1} of feedstock for fast pyrolysis and 1.113 t CO_2e t^{-1} of feedstock for slow pyrolysis, accounting for emissions from collection, hauling, pyrolysis, and replacing nutrients. Laird (2008) estimates a net potential of 0.33 t CO_2e t^{-1} of feedstock through displacement of fossil fuel by bio-oil in a bioenergy pyrolysis platform, for a total national potential GHG mitigation of 822 Mt CO_2e yr^{-1} combined with sequestration potential (510 Mt CO_2e yr^{-1}) from harvestable forest and crop residues. Additional process emissions reductions may result from decreased need for fertilizer or lime (e.g., 20% reduction estimate in Laird (2008)), and other inputs due to biochar's soil-amending characteristics (Fowles 2007; Laird 2008; Lehmann et al. 2006; Roberts et al. 2010). A detailed comparison of feedstock alternatives, pyrolysis methods, and tradeoffs, and other costs of biochar production can be found in a report produced for the Climate Action Reserve (De Gryze et al. 2010).

Change Irrigation Practices

In many parts of the U.S., irrigated cropland is much more productive than dry land in terms of total biomass produced, thus increasing the amount of irrigated land provides potential for additional soil C sequestration (Table 13). However, this increase in C storage—in product and soil—needs to be weighed against increased land emissions of N_2O and CH_4 and the GHG impacts of using more energy for irrigation and other related management. In general, irrigation reduces soil aeration and stimulates microbial activity, thus increasing the potential for N_2O emissions (Bremer 2006; Rochette et al. 2008b). The GHG emissions from electricity used for irrigation pumping range from 0.31–1.21 t CO_2e ha^{-1} yr^{-1} (Follett 2001a; West and Marland 2002), which nearly always outweighs any sequestration. Another concern is that irrigation of semi-arid land with high-pH soils can release C to the atmosphere when calcium carbonate ($CaCO_3$) is dissolved, approximately 0.29 t CO_2e ha^{-1} yr^{-1} (Martens et al. 2005; Schlesinger 2000). Increasing irrigation, and thus water use, in drier regions where water is likely already in limited supply also creates tradeoffs with other uses of this water, including human consumption and ecological flows to support aquatic species. Therefore, the benefits of increasing irrigation area are unlikely to outweigh the costs.

GHG Impact Summary

GHG Category	Convert Dryland to Irrigated	Irrigation Improvements
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**1.46** (1.14 – 4.77)	**0.34** (0.26 – 0.42)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**-0.42** (-1.05 – -0.05)	**0.66** (0.14 – 0.94)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**-1.38** (-3.34 – -0.41)	**0.23** (0.19 – 0.27)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**-0.34**	**1.19**
Maximum U.S. applicable area, Mha	**n/a**	**20**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

By reducing the total amount of water applied and optimizing water distribution to root zones, irrigation efficiency gains can provide water savings as well as GHG benefits. While the soil C impacts are difficult to quantify, there are suggestions that irrigation improvements may sequester soil C by increasing crop production or reducing soil erosion (Follett 2001a; Lal 2004). On the other hand, N_2O emission reduction through irrigation improvements looks promising and has been somewhat better documented (Amos et al. 2005; Kallenbach et al. 2010; Scheer et al. 2008). Burger et al. (2005) noted higher N_2O emissions following each irrigation event, but when water filled pore space (WFPS) went below 60%, the N_2O emissions dropped significantly. Studies on subsurface drip irrigation have found that WFPS is higher than 60% only within a few centimeters of the drip tape, with overall low WFPS (20%–30%) in these systems (Kallenbach et al. 2010). Scheer et al. (2008) found that reducing irrigation intensity (irrigating cotton when soil moisture was at 65% instead of 75% of field capacity) reduced N_2O emissions by almost 50% (0.94 t CO_2e ha^{-1} yr^{-1}). Similar effects were observed on winter wheat, although with less total impact because of lower baseline emissions. Conversion to subsurface drip irrigation from furrow irrigation can also reduce N_2O emissions (Amos et al. 2005; Kallenbach et al. 2010). This research is supported by previous studies of drip irrigation, which demonstrated sustained or increased yields and reduced N fertilizer requirements (Camp 1998), along with improved N use efficiency (Halvorson et al. 2008b).

29 Factors that could be varied include: feedstock source, amount of crop residue harvested for biochar pyrolysis, type of pyrolysis, area of land for application versus area of feedstock source, and indirect land use change impacts.

Many systems have changed from the less efficient furrow irrigation to central-pivot sprinklers. Further efficiency gains can be obtained with drip irrigation, which requires 25%–72% less water than furrow irrigation in agronomic and horticultural crops, with no negative yield impact (Camp 1998; Halvorson et al. 2008b; Lamm et al. 1995), thus providing significant energy and emission savings. Using a conservative estimate of 25% water savings for widely implemented drip irrigation or other efficiency improvements on the current 15.5 Mha of cropland irrigated through pumping (the remaining 4.7 Mha is gravity-fed), the emission reductions from energy savings alone[30] would be approximately 2.8 Mt CO_2e yr^{-1}.

Table 13. Soil C sequestration and N_2O emission impacts of irrigation changes on agricultural land.

Citation	Region/ Crops	Comments	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)
Irrigation vs. Dry land			
Follett (2001a)	U.S.	Sequestration potential from increased biomass in irrigated systems; value also used by IPCC (2000) and Lal et al. (2007); only soil C[a]	0.18 –0.55
Liebig et al. (2005b)	continuous corn/ Colorado	Irrigation compared to no irrigation; only soil C	1.95
Bordovsky et al. (1999)	wheat/Central Great Plains	Irrigated vs. rainfed; only soil C	1.22
Entry et al. (2002)	Idaho	Irrigated vs. rainfed; only soil C	3.00
Smith et al. (2008)	Global estimate	Increase irrigation; only soil C	Low: -0.55 High: 2.82
Rochette et al. (2008b)	Canada	At typical fertilizer N application rates, irrigation increases N_2O emissions	-0.79
Bremer (2006)	Turfgrass/Kansas	Irrigation significantly increased N_2O fluxes	-0.05
Improved Irrigation Systems			
Lal et al. (2003)	U.S.	Sub-irrigation on poorly drained soils (recycle drainage water), soil C	Low: 0.26 High: 0.42
Scheer et al. (2008)	Uzbekistan	Reduce irrigation intensity; N_2O emissions decrease	Wheat: 0.14 Cotton: 0.94

a. For those sources investigating soil C sequestration only, the increased emissions from energy inputs or N_2O and CH_4 flux are not included.

Reduce Chemical Inputs (Other than N Fertilizer)

Practices such as integrated pest management, intercropping for weed control, and the use of some genetically modified crops could reduce agricultural inputs of non-fertilizer chemicals. The majority of GHG emissions from the use of chemical inputs stems from the production of these chemicals from fossil fuels (mostly ethylene, propylene, or methane) (Helsel 1992; West and Marland 2002). Therefore, the GHG benefit of reducing chemical inputs is primarily in the upstream part of the lifecycle (Table 18).

In regulatory scenarios that control or cap emissions at the source, the price impact on agricultural chemicals would provide incentive to reduce application rates, and reduced use of chemicals would not be available for credit as a GHG offset. However, the upstream impacts may still be important considerations for other federal programs and voluntary markets. The production and application of pesticides uses less than 15% of the total energy in agriculture (Helsel 2007). And, although the production of pesticides uses 2–5 times more energy (on a per-weight basis) than N-fertilizer production, the GHG impacts (on a per-hectare basis) are small in comparison.

Table 18. Upstream GHG emissions impacts of reducing use of non-fertilizer agricultural chemicals.

Citation	Activity	Comments	Emissions impact ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Emissions Impact ($Mt\ CO_2e\ yr^{-1}$)
Audsley et al. (2009)	Pesticide reduction	Pesticide lifecycle, including fungicides, herbicides, insecticides, molluscides, growth regulator and seed treatment	0.09[a]	12.1
Lal et al. (2003)	Pesticide reduction	Herbicides, fungicides, and insecticides	0.03	3.8
West and Marland (2002)	Pesticide reduction	Production and application emissions, herbicides and insecticides	0.07	9.6

a. These numbers represent full implementation, so 100% reduction/elimination of pesticide use on a per ha basis.

30 This assumes that current adoption of these improved irrigation systems is minimal. The calculation uses estimates from Follett (2001a), assuming irrigation pumping emissions of 0.31–1.23 t CO_2e ha^{-1} yr^{-1} (average of 0.72).

Based on the relative amounts of 64 pesticides used on corn, wheat, and soybean crops in the U.S. (1996), production emissions (including fossil fuel associated CO_2 emissions) for herbicides, insecticides, and fungicides total approximately 17.3, 18.0, and 19.1 t CO_2e yr^{-1} of pesticide, respectively (West and Marland 2002). The authors suggest that these values may be within 10% for "some of the best known and most widely used pesticides," based on estimations by Green (1987). Audsley (2009) estimated an average pesticide energy input to arable crops of 0.09 t CO_2e ha^{-1}. These results suggest that pesticide manufacturing represents about 3% of the total GHG emissions from crop production (using values from Green 1987). However, when compared with other GHG mitigation opportunities, the potential impact of reducing chemical use is very small, and would likely be better considered for non-GHG reasons.

Nitrous Oxide Emission Reduction with Nitrogen Management

Total annual direct and indirect nitrous oxide (N_2O) emissions from U.S. fields are estimated at 215.9 Mt CO_2e, approximately 3.1% of all U.S. GHG emissions (U.S. EPA 2010). Nitrous oxide is predominantly the product/by-product of two N transformation processes—denitrification and nitrification—that are performed by soil microorganisms. Emission rates are positively correlated with low pH, higher temperatures, high water-filled pore space, soil compaction, available C substrate, and available mineral N (Chantigny et al. 2010; Farahbakhshazad et al. 2008; Venterea and Rolston 2000). Given these dynamics, mineral N often considered the main limiting factor, and N_2O emissions from agricultural land are significantly related to the application of inorganic and organic nitrogen (N) fertilizer, legume-derived N, and other factors that impact the availability of soluble mineral N in the soil. Improvements in N use efficiency (i.e., less N applied for the same crop productivity) have potential to significantly reduce N_2O emissions. Residual soil mineral N concentrations are also positively correlated to NO_3^- leaching and emissions of nitric oxide and ammonia (Mosier et al. 1998a). These degrade water and air quality, and leached NO_3^- also increases the potential for off-site N_2O emissions.

GHG Impact Summary

GHG Category	Reduce Nitrogen Fertilizer Rate
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**no impact**
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.38** (0.14 – 1.32)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.07** (0.05 – 0.09)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**0.45**
Maximum U.S. applicable area, Mha	**106**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

N_2O fluxes are highly variable over time. In one study, almost one-third of the annual N_2O emissions occurred in the one-month period following N fertilization (Liu et al. 2010), and Parkin and Kaspar (2006) observed 45%–49% of the cumulative N_2O flux from corn during two peak periods that followed rainfall. Mosier et al. (2006) found significantly different N_2O flux rates between years, with the same cropping system and fertilizer N rates. Elevated emissions are also common during freeze/thaw cycles in winter/spring (Gregorich et al. 2005; Wagner-Riddle et al. 2007). However, even with such high variability at the smaller scale, it is possible to determine impacts of management changes with larger-scale sampling and existing models (Desjardins et al. 2010). For the purposes of this report, we have divided N_2O emission management strategies into eight sections, with the first five addressing synthetic fertilizer: managing rate, source, placement, and timing; and using nitrification inhibitors. This fertilizer N management fits into the 4 R framework described by Roberts (2006)—right rate, right product, right time, and right place. The subsequent sections address the potential to mitigate N_2O emissions through improvements in manure management, irrigation practices, and drainage practices in humid areas.

Reduce fertilizer N application rate

Field studies in cropland agriculture have found that emissions of N_2O correlate well with fertilizer N rate (e.g., Halvorson et al. 2008a; MacKenzie et al. 1998; McSwiney and Robertson 2005; Mosier et al. 2006). In all of these studies, increasing the amount of N added to soil resulted in increasing emissions of N_2O. In recognition of this relationship, the IPCC Tier I method uses a direct linear multiplier of 1.0% of total applied fertilizer N lost as N_2O-N (IPCC 2006), although in field studies researchers have noted proportions ranging from <0.2% of fertilizer N to >1.6% (Lemke et al. 2003; Mosier et al. 2006; Stehfest and Bouwman 2006), depending on the soil, climate, season, and other factors. Using Tier I default factors, indirect emissions of N_2O are also calculated as a proportion of total N application, bringing the

total N_2O-N emission rates to 1.1%–1.3% of fertilizer N applied.[31] Such linear relationships may be appropriate at large scales and lower fertilizer N application rates; the estimated direct annual N_2O emissions from synthetic fertilizer (40.8 Mt CO_2e on cropland plus 4.0 Mt CO_2e on grassland) are equal to 0.7% of national synthetic fertilizer use (Millar et al. 2010; USDA ERS 2010b). However, N_2O emission rates in the field—as a function of amount of N applied—have been shown to rise in a nonlinear fashion after crop N needs have been met (Grace et al. 2011; Grant et al. 2006; Hoben et al. 2011; Malhi et al. 2006; McSwiney and Robertson 2005; van Groenigen et al. 2010). However, while the theoretical potential for N_2O emissions depends on availability of excess N, soil moisture or C substrate availability may instead be limiting factors so that this nonlinear response is not observed (R. Lemke and P. Rochette, personal communication, 13 September 2010). A wide range in the ratio of N_2O emissions to fertilizer N application (0.3%–7%) is reflected in GHG mitigation potential estimates resulting from N fertilizer rate reductions (Table 14). While some of the emissions are related to other N sources (manure, legume-derived, atmospheric deposition, and mineralized soil N), this also suggests that, at least in certain cases, the N_2O emissions rate rises significantly above that predicted by the IPCC Tier I factor (see Grant et al. 2006; McSwiney and Robertson 2005).

With increasing demand for food (due to increasing population and consumption), any shifting management of N must also sustain crop yield (Snyder et al. 2009). Thus the primary objective is N use efficiency gains (i.e., productivity per unit of N applied). If reductions in N fertilizer decrease crop yields, GHG emissions could actually increase, since production that compensates for yield losses could shift to less efficient regions or production systems. Incentives should therefore avoid reducing yield too much in highly efficient systems. Output-based accounting approaches, discussed in a companion T-AGG paper,[32] can capture the yield impacts, reduce negative leakage, and reward positive leakage.

In some studies, fertilizer N application has been associated with reductions in soil C (Khan et al. 2007; Mulvaney et al. 2009), although Powlson et al. (2010) countered these claims by questioning both the experimental methods and the conclusions. In fact, there are many different factors at play that produce varying results. For example, Poirier et al. (2009) found that high fertilizer N application rates reduced soil C with moldboard plow, but not under no-till. The fertilizer N application speeded decomposition in the plow treatment, but the additional productivity from fertilizer N in the no-till treatment generated more plant more residue that enhanced soil C. Upstream and process GHG emissions, on the other hand, will certainly be reduced with N fertilizer rate decreases, although the total impact is generally far less than the N_2O emission effect.

Table 14. GHG mitigation potential estimates for fertilizer N rate reduction.

Citation	Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential (Mt $CO_2e\ yr^{-1}$)
Paustian et al. (2004)	National estimate	Estimated reduction of 30%–40% with efficient use of N inputs	0.62	65.8
Millar et al. (2010)	National estimate	Assumes 15% reduction of N application		43.6
Stehfest and Bouwman (2006)	Global	Model from field estimates (n=840); fertilizer rates of 50 to >250 kg N ha^{-1}	0.11	
Smith et al. (2008)	Global	Reduce N application by 20%, dry vs. moist climate, very wide range in potential	Dry: 0.33 Moist: 0.62	
Bremer (2006)[a]	Kansas	Reduced application of urea fertilizer	0.02	
Halvorson et al. (2008a)[a]	Colorado	Low is continuous corn and corn-barley; high is corn-dry bean	Low: 0.04 High: 0.11	
Millar et al. (2010)	Michigan	Linear vs. nonlinear N_2O emissions response; corn-corn and corn-soybean rotations	Linear: 0.13 Nonlinear: 0.79	
McSwiney and Robertson (2005)[a]	Michigan	2%–7% of each additional kg N lost as N_2O	0.33	

a. The potential mitigation is calculated from the relationship between N_2O and fertilizer N rate given by these studies, then assuming at 15% rate reduction

There is no consensus among agricultural scientists as to whether current N application rates are above or at the agronomic (or economic, for that matter) optimum. While some assert that farmers are already using the lowest possible N rates, others suggest that extra fertilizer N is often applied as "insurance." Those that do think lower rates are pos-

31 Leaching losses (NO_3^--N) = 30% of N applied where irrigation or rainfall exceed soil water holding capacity; otherwise zero. Of the leached NO_3^--N, 0.75% is assumed to be emitted as N_2O-N. Volatilization as ammonia (NH_3-N) = 10% of total applied fertilizer N and 20% of applied organic N (manure etc.), for both of which 1.0% is emitted as N_2O-N.

32 This paper (Murray and Baker 2011) has been published in the February 2011 issue of *Greenhouse Gas Measurement and Management* and can be accessed at http://www.ingentaconnect.com/content/earthscan/ghgmm/2011/00000001/00000001/art00006.

sible have estimated that fertilizer N could be reduced by 12%–20% without severely negative yield impacts.[33] Some reductions in fertilizer N application rates are possible in conjunction with the N use efficiency gains that result from changes in the other three Rs (placement, timing, and source) or with nitrification inhibitors. Thus, the N fertilizer rate can function as an integrator of multiple practices, and GHG mitigation potential for the different N management practices cannot be additive; interactions must be considered carefully. Assuming continued implementation, the reduced emissions of N_2O generate benefits in perpetuity, and there is no risk of reversal, as in soil C sequestration.

Change fertilizer N source

In some regions and cropping systems, fertilizer source has been shown to significantly impact N_2O emissions, but the results are quite variable, depending on region and climate. In this analysis, fertilizer source management is separated into (1) changing between common sources, such as anhydrous ammonia and urea, and (2) using slow-release fertilizers (Table 15).

GHG Impact Summary

GHG Category	Change N Source from NH_4-based to Urea	Change N Source to Slow-release
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**no impact**	**no impact**
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.42** (-0.48 – 2.80)	**0.46** (0.00 – 1.43)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**no impact**	**0.07** (0.04 – 0.08)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**0.42**	**0.51**
Maximum U.S. applicable area, Mha	**37**	**79**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

About 85% of total N fertilizer used in the U.S. is ammonium-based; from 2006 to 2010, anhydrous ammonia prices were 62%–80% of those for urea fertilizer per unit of N (USDA ERS 2010a). Using 1,125 agricultural field measurements for N_2O, Stehfest and Bouwman (2006) expanded on earlier work (Bouwman et al. 2002), to conclude that anhydrous ammonia use resulted in no consistent difference in N_2O emissions when compared with urea or urea ammonium nitrate (the three most common N fertilizer sources in the U.S.).[34] Earlier work by Bremner et al. (1981) did conclude that emissions following anhydrous ammonia were substantially higher than following urea. However, neither the Stehfest and Bouwman (2006) nor Bremner et al. (1981) studies were based on contemporaneous side-by-side treatment comparisons. In side-by-side comparisons, researchers in Scotland found higher N_2O emissions from urea than from ammonium sulfate on grassland and barley (Clayton et al. 1997; McTaggart et al. 1997). A similar effect was noted by Tenuta and Beauchamp (2003) in an incubation experiment, but only under aerobic conditions; urea fertilizer had lower emissions when soil was saturated. Studies in Tennessee and southern Minnesota corn systems (Thornton et al. 1996; Venterea et al. 2005; Venterea et al. 2010) measured significantly lower emissions from broadcast urea than from anhydrous ammonia. Some of this effect may also be related to placement (urea is broadcast, but AA injected) or differential ammonia volatilization losses between fertilizer types (which impacts the amount of remaining N that could be lost as N_2O). Venterea et al. (2005) also noted interactions with tillage. While anhydrous ammonia always generated higher emissions, NT had lower emissions than CT under anhydrous ammonia but greater emissions with urea. To date, the direct studies showing no difference between anhydrous ammonia and urea (e.g., Burton et al. 2008a) have been limited to crops (e.g., wheat) that received substantially lower N application rates than the studies done with corn. Therefore, although there may be a trend toward lower emissions with urea as opposed to anhydrous ammonia, further research is necessary before extending to additional crops and climatic conditions.

There is great interest in using enhanced-efficiency N fertilizers (EEF), such as slow- and controlled-release, and stabilized N fertilizers to enhance crop recovery of N and minimize N losses to the environment (Snyder et al. 2009). To date, however, there are very few long-term studies investigating their effect on emissions of N_2O, although some recent work suggests potential for coated and urea-based slow-release fertilizer to improve N use efficiency and reduce N_2O emissions (Halvorson et al. 2010; Hyatt et al. 2010). The somewhat increased cost of production and transportation (due to greater mass and bulk) may be worth the price due to GHG benefits, efficiency gains, as well as the reduced impact on downstream water quality. More research is needed to evaluate impacts on N_2O emissions for a range of systems.

33 Smith et al. (2008) estimated that 20% reductions in fertilizer N application rates were feasible, and Millar et al. (2010) estimated that 12% to 15% reductions are possible by shifting from the high to the low end of the profitable N rate range for grain corn.

34 These two reviews compared observations from many different experiments, and so were not side-by-side comparisons that kept other factors constant. Also, high variability contributed to statistical insignificance.

Table 15. N_2O emission reduction potential with changes in N fertilizer source.

Citation	Crop Type / Region	Results and Comments	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)[a]
Changing between Common N Sources			
Venterea et al. (2005; 2010)	Corn-soybean (CS) and cont. corn (CC)/S. Minnesota	Lower emissions with broadcast urea vs. anhydrous ammonia	2005: 0.92 2010 CS: 0.25 2010 CC: 0.50
Thornton et al. (1996)	No-till corn/Tennessee	Lower emissions with urea vs. anhydrous ammonium	2.80
Burton et al. (2008a)	Wheat/Manitoba	Anhydrous ammonia and urea	no difference
Enhanced-efficiency N Fertilizers			
Delgado and Mosier (1996)	Colorado	Polyolefin-coated urea decreased N_2O emissions by 16% vs. urea (3 months)	
Akiyama et al. (2000)	Japan	Polyolefin-coated urea vs. urea (3 months)	
Halvorson et al. (2010)	Colorado	Average percent reduction in CT and NT systems for enhanced-efficiency urea sources	ESN[b]: 0.05 Super U[c]: 0.09
Hyatt et al. (2010)	Minnesota	Polymer-coated urea	1.43

a. National GHG mitigation potential is not included in this table because none of the references made national conclusions. National estimates were later calculated, making assumptions about applicable crop area.
b. ESN is a controlled-release polymer-coated urea fertilizer.
c. Super U is a stabilized urea source.

Change fertilizer N placement

The placement of synthetic fertilizer near the zone of active root uptake may both reduce surface N loss and increase plant N use, resulting in less N_2O emitted (Table 16). Improved placement of N fertilizer can constitute banding of fertilizer along crop rows or rate modification for different areas of a field based on yield expectations (e.g., precision agriculture using global positioning systems or other field area delineation). Since factors other than N availability (i.e., soil pH, water, etc.) lead to yield variability across a crop field, yield—and thus N uptake—can vary. Even N application often means over-application in areas of fields that tend to be lower-yielding. By reducing the fertilizer N rate by 25 kg N ha^{-1} for a low-yielding portion of a field, Sehy et al. (2003) measured N_2O emissions reduction of 2.3 t $CO_2e\ ha^{-1}$ in the lower-yielding field area (due to lower NO_3^- concentrations), with an average emission reduction of 0.4 t $CO_2e\ ha^{-1}$ for the entire field. Banded placement may reduce immobilization of N, causing delayed leaching or denitrification (Snyder et al. 2009). When compared with broadcast placement mid-row, banding reduced emissions in one study in Saskatchewan (Hultgreen and Leduc 2003), but had no significant impact in other research (Sehy et al. 2003).

GHG Impact Summary

GHG Category	Change N Fertilizer Placement	Change N Fertilizer Timing	Use Nitrification Inhibitors
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**no impact**	**no impact**	**no impact**
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.33** (0.12 – 0.47)	**0.35** (0.01 – 0.52)	**1.01** (0.00 – 2.23)
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**no impact**	**no impact**	**no data**
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**0.33**	**0.35**	**1.01**
Maximum U.S. applicable area, Mha	**85**	**53**	**92**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

In contrast, shallow versus deep injection has shown contradictory GHG flux impacts, with reduced N_2O emissions from shallow placement of ammonium nitrate in Ontario (Drury et al. 2006), but increased emissions from shallow placement of liquid UAN[35] in Colorado (Liu et al. 2006). Shallow placement of anhydrous ammonia also decreased emissions in Iowa corn, but only at lower fertilizer rate applications (Breitenbeck and Bremner 1986). Drury et al. (2006) concluded that shallow N placement appears to reduce N_2O emissions from corn crops on fine-textured soils in cool, humid climates. Further research is needed to fully understand the different interactions of soil type and other conditions that affect N_2O emissions with varied fertilizer N placement.

35 UAN = Urea ammonium nitrate.

Change fertilizer N timing

Crop N uptake capacity is generally low at the beginning of the growing season, increasing rapidly during vegetative growth, and dropping sharply as the crop nears maturity. Synchronous timing of fertilizer N application with plant N demand may help reduce N losses, including N_2O emissions. Several studies have found lower N_2O emissions associated with spring application compared to fall (Hao et al. 2001; Hultgreen and Leduc 2003), and up to 30% of U.S. corn is fertilized in the fall (Paustian et al. 2004). Results from studies of split application have varied, but it appears that lower emissions may occur (Table 16), especially in areas with greater rainfall or irrigation (Burton et al. 2008b). Timing fertilizer application to crop uptake is one of the most effective ways to reduce N loss as N_2O.

Table 16. Physical GHG mitigation potential from changes in fertilizer N placement and timing.

Citation	Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)[a]
Placement of Fertilizer N			
Drury et al. (2006)	Ontario	Shallow N placement	0.48
Liu et al. (2006)	NE Colorado	Deep injection in CT (low) and NT (high)	Low: 0.12 High: 0.32
Sehy et al. (2003)		No significant impact for banded application, but reduction for precision field application, with lower rates on low-yield areas	0.40
Timing of Fertilizer N Application			
Hao et al. (2001)	Southern Alberta	Lower emissions from spring application vs. fall	0.48
Hultgreen and Leduc (2003)	Saskatchewan	Reduced emissions from spring application vs. fall	0.52
Burton et al. (2008b)	New Brunswick	Split application	0.40
Burton et al. (2008b)	Manitoba	Urea, lower emissions from spring application vs. fall	0.01
Burton et al. (2008b)	Manitoba	Anhydrous ammonia, lower emissions from spring application	0.34

a. National GHG mitigation potential is not included in this table because none of the references made national conclusions. National estimates were later calculated, making assumptions about applicable crop area.

Integrating the 4 R's

As mentioned earlier, N rate can be an integrator of the 4 R's, since all N use efficiency improvements lead to lower N fertilizer needs per unit of production. Nitrogen rate reductions can be obtained with the application of precision agriculture techniques that accommodate within-field spatial and season-to-season temporal variability in N availability, thereby improving N management decisions for crop production. Two of the main goals of precision agriculture are to optimize the use of available resources to increase the profitability and sustainability of agricultural operations, and to reduce negative environmental impact (Gebbers and Adamchuk 2010). Schmidt et al. (2009) showed that crop canopy reflectance measured with an "on-the-go" sensor was a good indicator of crop N needs, making it possible to adjust N rates during growing-season N fertilizer application. When compared with uniform N rates for winter wheat in Oklahoma, Raun et al. (2002) found an on-board sensor resulted in a more than 15% increase in nitrogen use efficiency (NUE), and Li et al. (2009) noted 20% NUE improvement using a sensor instead of basing application rates on soil N testing. These decreases in N fertilizer application promise to be some of the most effective in reducing N_2O emissions, since the "excess" fertilizer above crop needs is highly susceptible to losses, being in the exponential portion of the curve that relates N_2O flux to N application rate.

Use nitrification inhibitors

Nitrification inhibitors increase the cost of fertilizer by 9% (Snyder et al. 2009), but can significantly improve fertilizer N recovery and reduce nitrate leaching. By slowing nitrification, the release of soluble mineral N is slowed, and thus lower N_2O emissions are also anticipated and have been observed in some studies. In combination with urea fertilizers, various nitrification inhibitors have reduced N_2O emissions by 9%–95% (Bhatia et al. 2010; Bronson et al. 1992; Snyder et al. 2009), although the short length of some of these studies may have overestimated the impact (Snyder et al. 2009). Research in Scotland also noted significant emission reductions when nitrification inhibitors were added to urea fertilizer, and some reductions with ammonium sulfate (McTaggart et al. 1997). In this same study, the nitrification inhibitor also retained effectiveness in August following an April application, indicating that long-term (even post–growing season) fluxes should be monitored.

However, in other cases, nitrification inhibitors used with ammonium sulfate and anhydrous ammonium fertilizer delayed nitrification, reducing N_2O emissions in the near term, but the annual flux was unaffected (Parkin and Hatfield 2010). Therefore, while nitrification inhibitors do change N dynamics with anhydrous ammonium fertilizer (Cochran

[1973] noted increased N uptake by plants), translation into N_2O flux impact is not always certain. Effects appear to be related to fertilizer source, timing, placement, depth (Parkin and Hatfield 2010), soil temperature, and pH (Kyveryga et al. 2004). From the limited long-term field studies available, it appears there is potential for the use of nitrification inhibitors to reduce emissions of N_2O, but further research is needed to understand the interactions with fertilization and soil conditions. Nitrification inhibitors are effective for non-nitrate N fertilizers, which comprise approximately 90% of the total commercial fertilizer N utilized in the U.S. (USDA ERS 2010b).

Improve manure management to minimize N_2O emissions

A significant amount of the N in manure can be lost as ammonia (NH_3), nitrate (NO_3^-), or N_2O after land application; estimates of up to 50% of the mineral N are not uncommon (Mosier et al. 1998a). Most gaseous losses are in the form of NH_3, causing air quality problems and N deposition in natural ecosystems. Leaching losses of NO_3^- reduce water quality. Because the N loss pathways are connected, efforts to control direct N_2O emissions from manure application provides the environmental co-benefit of lower NH_3 and NO_3^- losses (the latter of which also contributes significantly to indirect N_2O emissions). Estimates of national N_2O emissions from managed manure range from 2.6–30.6 Mt CO_2e yr^{-1} (U.S. EPA 2009; USDA 2008a). A number of improved management activities have been proposed to reduce these emissions (Table 17). Although other crops should not be ignored, improved management of manure on corn cropland alone will generate significant results, since corn comprises 58% of all manured land,[36] 79% of manured field crop area, and 87% of total manure N in 2009 (USDA ERS 2009).

GHG Impact Summary

GHG Category	Improve Manure Management to Minimize N_2O emissions
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**no data**
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.89** (0.37 – 1.22)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**no impact**
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**0.89**
Maximum U.S. applicable area, Mha	**12**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Table 17. GHG mitigation potential of altered manure management to reduce N_2O emissions.

Citation	Activity	Comments or Caveats	Potential t CO_2e ha^{-1} yr^{-1})	National Potential (Mt CO_2e yr^{-1})
Paustian et al. (2004)	Improved "waste" disposition	10% reduction of current N_2O emissions from manure	1.17	14.0
Pork Technical Working Group (2005)	Apply to dry areas rather than wet	50% reduction in N_2O emissions; Canada	Low: 0.37 High: 0.82	
Rochette et al. (2000)	Apply lower rate of pig slurry	Reduce % N denitrified from 1.65% to 1.23%; Canada	1.22	
Gregorich et al. (2005)	Apply solid manure rather than liquid	Canada	0.86	

Nitrous oxide emission rates are highly variable, depending on time elapsed since manure applications, type of manure applied, climatic conditions, and the amount of water available (in the soil or with the manure) (Saggar et al. 2004). The proportion of denitrified N lost as N_2O (rather than N_2) is greatest directly after liquid manure application (Saggar et al. 2004), so timing application to coincide with drier soil and lower temperatures could reduce losses. Nitrification inhibitors may reduce N_2O emissions (Saggar et al. 2004), and using anaerobic storage instead of aerobic also significantly reduces losses of N_2O, both in storage and upon field application (Mosier et al. 1998a). However, the most promising place to start may be adjusting commercial N application rates to account for N addition in the manure. Almost 40% of farmers do not make such adjustments (USDA ERS 2009). This would also lead to lower fertilizer costs and related emissions. As with fertilizer N, the tradeoff between N_2O emission reductions and crop yield needs to be considered if reducing manure N application rates (Rochette et al. 2000).

36 Hay and grass are second, with 26% of total manured land area.

Histosol Management

Between 10 and 15 Mha of land in the U.S. are classified as histosol or organic soils (peat), mostly occurring in Michigan, Wisconsin, Minnesota, and Florida (Lal et al. 2003; Morgan et al. 2010). About 7.5% of these soils have been drained for agriculture, half in California and Florida and the remainder mostly in the Lake States and the east coast (Morgan et al. 2010). Histosols and wetlands are somewhat confounded and a bit confusing in the literature. While most soils are primarily made up of mineral particles (sand, silt, clay), histosols are a unique soil type, contain at least 20%–30% organic matter—by mass—in at least the first 40 cm of depth from the surface, which is why they are also called organic soils. The organic material is most often *Sphagnum* moss. Many—but not all—histosols are also wetlands or were wetlands until drainage for human uses, and some former wetlands available for restoration are not histosols, but are composed primarily of mineral material. In the context of this synthesis, histosols are separated from wetlands as a special case, since there is somewhat more information available, and wetlands are highly variable. "Wetlands restoration," treated in a separate section, refers to all non-histosol water-influenced areas. In their natural state, histosols emit CH_4 and sequester C in buried biomass, although net GHG flux varies. However, organic soils that are drained for agriculture emit significant amounts of CO_2 and N_2O, but become CH_4 sinks (Elder and Lal 2008; Rochette et al. 2010), turning farmed histosols into a significant GHG source, between 3.5 and 53 t CO_2e ha^{-1} yr^{-1} (Freibauer et al. 2004; Morgan et al. 2010), with high variability depending on practice, soil characteristics, and climate.

GHG Impact Summary

GHG Category	Convert Histosol to Natural	Alter Histosol Management
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**28.48** (2.20 – 73.33)	**5.29** (0.00 – 15.03)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**6.80** (1.50 – 12.10)	**11.17** (2.23 – 28.18)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.74** (0.35 – 1.14)	**no impact**
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**36.02**	**16.45**
Maximum U.S. applicable area, Mha	**0.8**	**0.8**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

Alternative practices to reduce net GHG emissions include restoring organic soils to a natural state or altering tillage and cropping practices that retain the land in agriculture (Table 19). In formerly forested organic soils in Finland, the methane flux differences between cropped soils and those abandoned for conservation is very small in comparison to the CO_2 and N_2O impacts, with a total GHG benefit for the set-aside land of 10.3 t CO_2e ha^{-1} yr^{-1} (Alm et al. 2007). Most of the research on histosol management has been conducted in Europe, but Rochette et al. (2010) observed that organic soils in Canada exhibited similar GHG fluxes as those seen in Europe. Therefore, the lessons learned from Europe should be applicable in the North American context.

IIf histosols are removed from agricultural production, avoided emissions of CO_2 amount to more than 21 t CO_2e ha^{-1} yr^{-1}, the highest potential on an area basis of any activity considered in this report. By eliminating field operations and fertilizer N, upstream and process emissions lead to more GHG benefits, although the production would likely shift elsewhere, so these benefits may not be realized. The net land emissions of CH_4 and N_2O are expected to be highly variable; some unfarmed organic soils are significant sources of CH_4 (Morgan et al. 2010), but abandoned farmland has been found in some cases to be a CH_4 sink (Alm et al. 2007). Emissions of N_2O are most likely to decrease with conversion to either grassland or natural ecosystems (Alm et al. 2007), but maintaining higher water tables to reduce CO_2 emissions will likely stimulate greater CH_4 and perhaps greater N_2O emissions (Morgan et al. 2010).

Where it is difficult to remove these sensitive soils from agricultural production, management that reduces soil disturbance and avoids drainage can have large GHG benefits in contrast to row crop cultivation (Freibauer et al. 2004). No-till of histosols was also found to dramatically reduce N_2O emissions in one study (Elder and Lal 2008). However, further research in this area is needed to confirm CH_4 and N_2O impacts.

Table 19. GHG mitigation potential from alternative management of histosols (organic soils).

Citation	Crop Type/Region	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Removal from Agricultural Production				
Lal et al. (2003)	Restore organic soils/ U.S.	Assumes 19 Mha available for restoration	2.2	41.8
Alm et al. (2007)	Abandon for conservation/ Finland		8.8	
Smith et al. (2008)	Restore organic soils/ global	Cool-dry and moist Warm-dry and moist	36.7 73.3	
Freibauer et al. (2004)	Farmed organic soils/ Europe	Convert to woodland Abandon for conservation Protect and restore	3.5 8.1 16.9	
Alter Management of Farmed Organic Soils				
Alm et al. (2007)	Finland	Cereal crop to grassland	2.8	
Freibauer et al. (2004)	Farmed organic soils/ Europe	Change from heavy-tillage crop (e.g., potatoes) to lesser-tilled crop Maintain shallow water table Cultivated to grassland Avoid deep plowing Sheep grazing, undrained land	5.9 10.1 5.1 5.1 8.1	

Drain Agricultural Lands in Humid Areas

There is very little information in the scientific literature about the potential of draining agricultural land for N_2O emission reduction. In a 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 difference of 0.19 t $CO_2e\ ha^{-1}\ yr^{-1}$). However, as these were not side-by-side comparisons, other factors may have also played a role; and it is unclear as to whether poorly drained soils in the U.S. could be remediated with drainage, and if so to how much land area this practice could be extended. The expense of installing tile drains or other systems also means that GHG mitigation would have to be very high or combined with other crop production benefits to be economically feasible.

Reduce Methane Emissions from Rice

Rice soils emit methane because microbial respiration in flooded conditions reduces oxygen potential, creating anaerobic conditions that lead to methane production. In 2009, the worldwide planted rice area totaled 155.7 Mha. Of this, the U.S. contributed 1.29 Mha, 0.8% of the worldwide rice area (USDA NASS 2009b). Annual rice-related methane emissions total 6.2 Mt CO_2e (2007), almost 1% of the total methane emitted in the U.S. (U.S. EPA 2009). This can be compared to worldwide methane emissions from rice, which are estimated at 708 Mt CO_2e for 2010 (U.S. EPA 2006), 11% of global GHG emissions from the agricultural sector. While methane from rice production is an important source of GHGs worldwide, it is not a very big source for the U.S. However, potential mitigation per unit area for water management, variety development, or reducing total rice production area is only exceeded by restoration of organic soils, and the anticipated cost is low per t CO_2e (Smith et al. 2008). U.S. research on management practices could guide developments in other parts of the world, contributing to GHG mitigation in other countries where rice systems are more prevalent.

GHG Impact Summary

GHG Category	Rice Water Management for CH_4	Rice Variety Development for CH_4
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**no impact**	**no impact**
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**1.56** (-0.88 – 5.22)	**1.17** (0.00 – 2.71)
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**no data**	**no impact**
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**1.56**	**1.17**
Maximum U.S. applicable area, Mha	**1.3**	**1.3**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

GHG mitigation potential in rice systems varies dramatically by management practice, as well as from site to site, and the reductions in CH_4 emissions can range between 7%–80% (Wassmann et al. 2000). Mid-season drainage is one of the more promising emission-reducing activities (Table 20). Li et al. (2004) propose that the widespread shift from

continuous flooding to mid-season drainage, in the 1990s in China, accounted for much of the slowed growth in atmospheric methane concentrations during that time. This water management was adopted in order to save water and increase yields. In Asian rice systems, Wassmann et al. (2001b) found that a single mid-season drainage could reduce CH_4 emissions by 7%–43%, statistically significant in seven out of eight experiments. Dual drainage at mid-tillering and pre-harvest could reduce CH_4 emissions by up to 80% (Wassmann et al. 2000). Sass et al. (1997) found that a single mid-harvest drainage, for rice cultivated in Texas, could reduce total emissions by about 50%, and a two-day drainage period every 3 weeks could reduce emissions to an insignificant amount (<0.25 t CO_2e ha^{-1}). Other studies from around the world have had similar findings, with most research in China.

However, in some regions, particularly those with high soil C content, increased N_2O emissions can follow mid-season drainage, eliminating any net GHG benefit (Li et al. 2005b). The increased N_2O emissions in some areas reached levels of >7.5 t CO_2e ha^{-1} yr^{-1}. Therefore, the implementation of rice water management for GHG mitigation needs to avoid or at least check N_2O emissions on likely (high C) soil types. This could be done through model validation or perhaps by setting a level of soil C above which N_2O emissions will have to be considered when implementing water management for GHG mitigation. In California rice growing regions, preliminary data suggest that N_2O emissions are not elevated with multiple drainages or other alternative water management (W.R. Horwath, personal communication, 18 June 2010).

Water management during the non–growing season can also impact gaseous flux, necessitating full-year CH_4 emission accounting. Fitzgerald et al. (2000) found half of all CH_4 emissions occurred in the winter, but that winter flooding reduced winter emissions, with a large flux upon drainage. In contrast, flooding of Chinese rice fields in winter increases CH_4 emissions (Kang et al. 2002; Xu et al. 2000). Therefore, the impacts of temperature and soil conditions may need to be better understood before winter water management can be recommended for GHG mitigation.

Process emission impacts of water management changes depend on the energy requirements for transportation of water in and out of fields, and would be minimal in gravity-fed irrigation systems. Where irrigation water is pumped, rather than gravity-fed, increased fuel use associated with mid-season drainage (and subsequent re-flooding) may offset some of the benefits from CH_4 emission reduction, but there are no data readily available with which to make reasonable estimates.

Table 20. Potential GHG mitigation with rice system management changes.

Citation	Comments or Caveats	Activity	Physical Potential (t CO_2e ha^{-1} yr^{-1})	National Potential (Mt CO_2e yr^{-1})
Sass and Fisher (1997)	Texas	Mid-season drainage Drainage every 3 wks, 100% reduction Low-emission cultivar	1.1 2.3 2.7[b]	3.1[a] 6.2[a]
Li et al. (2004)	DNDC model, China	Mid-season drainage	4.7–5.2	
Li et al. (2005b)	DNDC model, China	Mid-season drainage	4.2	
Towprayoon et al. (2005)	Thailand	Mid-season drainage Multiple drainages	1.5 2.1	
Wassmann et al. (2000)	Asia	Mid-season drainage, 7%–43% reduction		0.4–2.7

a. National potential is calculated by multiplying the national emissions estimate (U.S. EPA 2009) by the proportional CH_4 flux reduction in the given reference.
b. This value represents the difference between the two most widely different cultivars, using average national emissions. At this location, the total emissions were higher than national average, ranging from 4.5–10.3 t CO_2e ha^{-1} yr^{-1}, but emission reductions of this level (~5.8 t CO_2e ha^{-1} yr^{-1}) are not expected at a national scale.

Another important management issue is the incorporation of rice straw. Emissions of CH_4 increase by 2–5 times when rice straw is incorporated rather than burned (Bossio et al. 1999; Redeker et al. 2000), as the additional organic material encourages microbial activity, including methanogenesis. However, by incorporating straw rather than burning it, air quality and nutrient cycling are improved (Eagle et al. 2000). Thus, the tradeoffs between GHG mitigation, plant nutrients provided by straw, and other factors require further examination.

High-yield cultivars can reduce emissions when compared to lower-yielding varieties, by directing more C to grain production rather than root processes, where respiration results in CH_4 production (Denier van der Gon et al. 2002; Sass and Cicerone 2002). Cultivar emission differences may be a factor of varying abilities of rice aerenchyma to transport CH_4 from the roots or oxygen to the roots, impacting soil redox potential (Sass and Fisher 1997). Others propose that emission rate differences relate mainly to availability of substrate for methanogens, especially root exudates (Huang et

al. 1998) (Aulakh et al. 2001a). However, over multiple seasons, Wassman et al. (2002) found inconsistent emission rate differences between cultivars, especially on variable soils. In summary, before specific rice cultivars can be promoted for GHG mitigation purposes, additional region-specific research may be needed.[37]

In some situations where CH_4 emissions are very high and alternative crops or land set-aside are feasible, removal of rice cropping area from rice production could also provide GHG mitigation benefits. On average, the eliminated CH_4 emissions would be worth 4.8 t CO_2e ha^{-1} yr^{-1}. However, the full system GHG impacts are important to consider, since net emissions will depend on the subsequent crop or landcover, and the need to grow additional rice elsewhere (at perhaps lower efficiency) may more than offset any local mitigation gains.

Grazing Land Management

Of the 766 million hectares (Mha) of land in the 48 contiguous United States, approximately 336 Mha (44%) are used for grazing (Lal et al. 2003) with 214 Mha under nonfederal ownership (USDA 2007). Because of the substantial coverage of grazing lands in both the U.S. and the world, they are considered an important C sink—storing up to 10%–30% of the world's SOC (Schuman et al. 2002). Grazing lands can be divided into two distinct classes: (1) rangelands or uncultivated land with minimal inputs, consisting of natural or naturalized plant species and which are extensively grazed; and (2) pastures which feature periodic agronomic inputs like cultivation, intentional species planting, irrigation, and fertilizers and which are more intensively managed (Follett and Reed 2010). These systems have very different levels of productivity and thus C storage, with pasture having much higher biomass production per unit area. Of nonfederal grazing land, 22.5% (48 Mha) is classed as pasture, and the remaining majority rangeland (USDA 2007). Improved pasture in the U.S. is mostly located east of the Missouri River (Schnabel et al. 2001).

GHG Impact Summary

GHG Category	Improved Grazing Management, Rangeland	Improved Grazing Management, Pasture
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**0.93** (-0.10 – 4.99)	**2.71** (0.55 – 5.86)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**0.28** (0.27 – 0.31)	**0.28** (0.27 – 0.31)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**1.22**	**2.99**
Maximum U.S. applicable area, Mha	**166**	**48**

Positive numbers depict removal of GHG from atmosphere or prevented emissions. Tables comparing all practices can be found on pages 45–46.

Much of the GHG mitigation potential discussed for rangeland and pasture is related to soil C sequestration, with greater increases coming from lands that were in degraded or marginal conditions. Highly productive, well-managed land with high SOC levels will show minimal soil C sequestration (Follett and Reed 2010). Understanding the state of the range or pasture land before implementing practices will help determine the C sequestration potential (Bremer 2009). It's only been over the last decade that scientists have begun to document soil C and other GHG dynamics on grazing lands as a result of different management practices. Earlier reports of grazing land productivity can provide useful information and some initial soil C sequestration estimates used this information to advise expert opinion estimates.

Improve grazing management, rangeland

Compared with more highly productive pasture, C sequestration rates on rangelands are relatively low on a per unit basis, but because of their vast area, it's been predicted they could capture up to 2%–4% of annual anthropogenic GHG emissions on a global basis, and 20% of the CO_2 released annually from global deforestation and land-use change (Derner and Schuman 2007; Follett and Reed 2010). The majority of this C (greater than 90%) is stored as SOC. These systems are characterized by an inherently high degree of variability in soils, topography, plant communities and/or dominant species, precipitation, and climate.

Grazing practices are central to healthy rangeland management. Grazing facilitates litter decomposition to SOC by the removal of aboveground biomass, and churning of surface soil by animal hooves. Further, removing excess aboveground material regenerates root growth and hastens the onset of spring regrowth and photosynthesis (LeCain et al. 2000).

37 Other rice management practices and site characteristics also impact methane emissions, with decreases noted from application of silicate fertilizer (16%–28% reduction in CH_4 flux, Ali et al. 2008a; Ali et al. 2008b).

Unlike removal for hay, grazing returns the majority of nutrients back to the soil via excreta (Schnabel et al. 2001); the total sequestration due to grazing activity can be up to 4.0 t CO_2e ha^{-1} yr^{-1} (Frank 2004; Schuman et al. 1999), although excess grazing can result in negative soil C change (Fuhlendorf et al. 2002; Lal 2001b). This is especially a problem when a moisture deficit limits production (Schnabel et al. 2001). The most promising soil C sequestration practices include improved grazing management (appropriate stocking rate/forage utilization, timing of grazing to avoid the months of high C uptake and adjusted frequency of grazing), destocking during drought conditions, introduction of natural or naturalized legumes, control of undesirable species, and in some cases, addition of fertilizer N (Follett and Reed 2010; Morgan et al. 2010).

In this report, the main GHG mitigation activity considered for rangeland is improved grazing management on rangeland (Table 21), which tends to involve reductions in stocking rates, since many of the poorly managed rangelands have been overgrazed. Aside from stocking rates, attention to drought conditions and grazing timing and frequency are also important. While grazing management has an impact, SOC dynamics are strongly related to precipitation. From Derner and Schuman (2007), the following relationships were seen for the U.S., indicating that rangeland from the Southwest to the Northeast see increases in C sequestration potential with improved management: (1) arid rangeland (<250 mm) = 0.07–0.29 t CO_2 ha^{-1} yr^{-1}; (2) semi-arid rangeland (250–500 mm) = 0.11–0.44 t CO_2 ha^{-1} yr^{-1}; and (3) semi-humid and subhumid (500–1000 mm) = 0.29–0.73 t CO_2 ha^{-1} yr^{-1}.

Schuman et al. (2001; 2002) compiled information on the state of U.S. rangelands from USDA-NRCS (1998), USDI-BLM (1998), and the USDA-FS (1999) and found that 67% of private rangelands have serious ecological or management problems and would achieve higher C sequestration rates with improved management. Of federally managed rangelands (Bureau of Land Management), over 63% are in fair to poor condition. In contrast, Conant and Paustian (2002) estimated that only 4.0% of all North American grassland was overgrazed. Researchers have identified large potential for C sequestration on poorly managed lands (Table 22), and even the maintenance of well-managed grasslands represents a potential 62 Mt CO_2e yr^{-1} of avoided losses, compared to shifting to cropland (Schuman et al. 2001; Schuman et al. 2002). Following this line of logic, 84 Mt CO_2e yr^{-1} of avoided loss would occur even if current grazing strategies were maintained on the poorly managed grazing lands as opposed to conversion to annual cropland.

Long-term trend analysis on rangelands is showing that in wetter years management may have little impact on soil C sequestration. But in drought years, management effects can have a significant impact on C sequestration rates. Zhang et al. (2010) found that rangelands can become a C source if >65% of the area is in drought conditions. Net ecosystem C exchange patterns show that from the West to the East, rangelands change from largely being a source of C to a sink under moister conditions (Table 23). When less than 50% of the lands are experiencing drought, the range can still manage to be a sink, sequestering C (Svejcar et al. 2008). Since rangelands are characterized by C sequestration that occurs in short periods (2–4 months) of high C uptake and long periods of steady-state C balance or small losses, the intensity and frequency of grazing is critical. Significant C loss can occur with heavy grazing over time in drier years. Therefore, proper grazing management during the C uptake periods and during drought years is critical.

Longer-term grazing studies in the Northern Great Plains of the U.S. have found that where increases in SOC have occurred, the species composition changes from cool season, mid-grasses to more of the warmer-season C4 grasses (predominantly some shrubs and *Bouteloua gracilis*, Reeder et al. 2004). *B. gracilis*, with its high root:shoot ratio, stores more of its C belowground than other species, and therefore may be the reason why higher soil sequestration rates are found. The authors found that the heavy grazing treatments were 23.8 t ha^{-1} higher in total soil C (0–90 cm) than the nongrazed treatments, with 68% attributable to higher soil inorganic C levels, and 32% to higher SOC. Although largely overlapping, the typical potential soil C sequestration reported for U.S. rangeland seems to be greater than in Canada or other global sites, with averages of 1.18 t CO_2 and 0.57 t CO_2, respectively (Conant et al. 2001; Smith et al. 2008).

Table 21. Soil C sequestration potential for grazing management on rangeland and pasture.

Citation	Activity	Comments or Caveats	Physical Potential (t CO_2 ha^{-1} yr^{-1})	National Potential (Mt CO_2yr^{-1})
Improved Rangeland Management				
Conant and Paustian (2002)	Decrease grazing intensity on overgrazed land	Statistical model for North America rangeland	Low[a]: 0.57 High: 1.30	8.1[b]
Follett et al. (2001)	Improved rangeland management	Estimate for all of U.S.	Low: 0.18 High: 0.55	78.5
Follett and Reed (2010)	Grazing management on rangeland	Rocky Mountains and Great Plains	Low: 0.26 High: 0.44	
Reeder et al. (2004)	Heavy grazing vs. no/light grazing	Northeastern Colorado	1.56	
Liebig et al. (2010)	Decrease grazing intensity	N. Dakota; although SOC impact was negative, CH_4 emissions were reduced by 0.31 t CO_2 e ha^{-1} yr^{-1}; net impact was 0.16	-0.10	
Franzluebbers and Stuedemann (2009)	Reduce grazing pressure on fescue/bermudagrass	Georgia Piedmont (0–30 cm depth)	2.42	
Lal (2001b)	Moderate grazing rather than heavy grazing	Texas rolling plains	Low: 0.66 High: 4.99	
Improved Pasture Management				
Follett et al. (2001)	Grazing management on pasture	National estimate assumes 10.2 Mha	Low: 1.10 High: 4.77	29.9
Franzluebbers (2001)	Increased grazing intensity	Georgia	0.00	
Stuedemann et al. (1998), cited in Schnabel et al. (2001)	Increased grazing intensity on pasture	Southeastern U.S.	5.87	
Improved Rangeland OR Pasture Management				
Conant et al. (2001)	Improved grassland management	North America	2.20	
Lal (2001a), Lal et al. (2003)	Restore eroded grazing lands	U.S. estimate, national assumes 123 Mha	Low: 0.18 High: 0.73	67.7

a. In this case, lowest potential occurs when changing from "moderately overgrazed" to "improved/moderate grazing" and highest potential occurs when changing from "strongly overgrazed."
b. The low national value from this study results from counting only the 14 Mha of overgrazed land for this treatment.

When evaluating different management practices, it's important to understand the net effect on GHG emissions. There are only a few studies that attempt to assess the net effect of management on all three GHGs, and otherwise IPCC equations have been utilized for CH_4 and N_2O to infer the net effect. Grazing management (i.e., adjusting number of animals per unit of land) does not tend to affect rates of N_2O emission on rangelands (J.D. Derner, personal communication, March 2010; B.H. Ellert, personal communication, March 2010) or increases emissions by less than 0.05 t CO_2e ha^{-1} yr^{-1} (Paustian et al. 2004; Liebig et al. 2010; Wolf et al. 2010), a minimal impact. Methane emissions from the soil are minimal in all systems, so they are not impacted. Enteric fermentation methane emissions are mainly affected by animal density on the land, and while improved management can reduce CH_4 emissions by lowering animal numbers, the transfer of those animals elsewhere may result in no real impact.

Table 22. Estimated potential soil C sequestration on U.S. rangeland and potential avoided losses. Rates based on Great Plains region.

Status of Grazing Lands	Area (Mha)	Rate (t CO_2 ha^{-1} yr^{-1})	Total Rate (Mt CO_2 yr^{-1})
Potential Mitigation Gains			
Well managed	57	0.0	0
Poorly managed	113	0.4	40[a]
CRP grasslands[b]	13	2.2	29
TOTAL			70
Potential Avoided Losses (by Keeping in Grazing vs. Cropland)			
Well managed	57	1.1	62
Poorly managed	113	0.7	84
CRP grasslands[c]	13	1.1	15
TOTAL			158

Adopted from Schuman et al. (2001; 2002).
a. Total rate may not equal area X rate and columns may not add up exactly, due to rounding in the "rate" column.
b. Data based on Bruce et al. (1999).
c. Data based on Doran et al. (1998) and compared to conversion to NT wheat-fallow system.

Table 23. Net ecosystem GHG exchange for different rangelands in the U.S.

Location	Vegetation	Mean (and Range) Annual Net Ecosystem Exchange ($t\ CO_2\ ha^{-1}\ yr^{-1}$)
Las Cruces, NM	Desert grassland[a]	-5.9 (-9.3 to 3.4)
Lucky Hills, AZ	Desert shrub[a]	-3.4 (-5.9 to 2.0)
Burns, OR	Sagebrush steppe	2.7 (-2.2 to 8.4)
Dubois, ID	Sagebrush steppe	3.0 (-1.7 to 9.5)
Mandan, ND	Northern mixed prairie	1.9 (-1.0 to 4.4)
Nunn, CO	Shortgrass steppe	3.9 (0.1 to 8.3)

Adapted from Svejcar (2008).
a. Influence of carbonates in the soils of the desert Southwest causes net C source (negative numbers).

By way of example, Liebig et al. (2010) conducted a Northern Great Plains case study that estimated net GHG effects for two long-term grazing management systems near Mandan, North Dakota, one with moderate grazing (2.6 ha/steer) and the other with heavy grazing (0.9 ha/steer). Using similar methodology, Derner (personal communication, March 2010) compared two grazing systems near Cheyenne, Wyoming—a lightly grazed system (5 ha/steer) and a heavily grazed system (2.25 ha/steer). The results (Table 24) show, that depending on the system and the location, there may be substantial differences in net GHGs—where one system can be a source, the other a net sink.

Table 24. Case studies showing net effects on GHG emissions/removals.[a]

Mandan, ND	Moderately Grazed	Heavily Grazed
(44 yrs of treatment) $t\ CO_2e\ ha^{-1}\ yr^{-1}$		
SOC change	-1.42 (0.19)[b]	-1.52 (0.19)
Enteric fermentation	0.18 (0.03)	0.48 (0.08)
Soil CH_4 flux	-0.06 (0.01)	-0.06 (0.01)
Soil N_2O flux	0.52 (0.09)	0.48 (0.04)
NET GWP[c]	***-0.78 (0.03)***	***-0.62 (0.08)***
Cheyenne, WY	**Lightly Grazed**	**Heavily Grazed**
$t\ CO_2e\ ha^{-1}\ yr^{-1}$		
SOC change	-0.66	0.00
Enteric fermentation	0.10	0.22
Soil CH_4 flux	-0.06	-0.06
Soil N_2O flux	0.52	0.52
NET GWP[b]	***-0.11***	***0.67***

a. Adapted from Liebig et al. (2010) and Derner (personal communication, March 2010).
b. Values in Parentheses indicate standard error of the mean; negative values imply net CO_2e uptake.
c. Net GWP for Mandan, ND is not significantly different at $p<=0.05$.

Improve grazing management, pasture

In pasturelands, applying fertilizer or other inputs can increase C sequestration rates, primarily through increases in annual net primary productivity, by anywhere between 0.40 and 11.16 $t\ CO_2\ ha^{-1}\ yr^{-1}$ with a mean of 1.98 $t\ CO_2\ ha^{-1}\ yr^{-1}$ (Conant et al. 2001). However, there is a paucity of studies that address grazing intensity for pasture in relation to soil C sequestration. The challenge in pasturelands is that the management factors introduce complexity across the soil-animal-plant interactions, increasing the spatial variability of the analysis immensely. The relatively higher C sequestration rates would need to be balanced by the net effect of this improved management on other trace gases (N_2O and CH_4), which could be significant, but there is a dearth of studies available in this regard as well. In temperate climates, most forage-based animal agriculture grazes animals on pasture for 5 to 12 months of the year. Thus, stored forages are an important part of the mix, and this complexity must be taken into account at the landscape level in future GHG studies (Follett and Reed 2010).

For pasture, the most promising C sequestration practices are improved grazing management (which, as for rangeland, often [Lynch et al. 2005]—but not always [Schnabel et al. 2001]—involves reducing stocking rates), altered species composition (legumes; altering forage quality), fertilization, and irrigation. Morgan et al. (2010) estimate that the C sequestration potential of improved pasturelands can be double that of cropland, particularly in the mesic systems of the eastern United States. The greater allocation of plant biomass C to belowground soil C sequestration under pasturelands, combined with an extended growing season, less soil disturbance, and better utilization of soil water is, in part, responsible for the higher soil sequestration rates, compared to harvested croplands. Under best management practices, C sequestration rates can vary from 1.1–5.1 $t\ CO_2\ ha^{-1}\ yr^{-1}$ (Morgan et al. 2010). This range in sequestration rates is a reflection of regional characteristics, such as soil composition, topography, climate, and existing grass species,

as well as whether CO_2, N_2O, or CH_4 are being mitigated (Conant et al. 2005). As on rangeland, grazing management on pasture is assumed to have very little N_2O effect; and CH_4 emissions are affected primarily by enteric fermentation, and thus the grazing intensity.

Implement rotational grazing

Rotational grazing (also known as management-intensive Rotational grazing (also known as management-intensive grazing, MIG) differs from continuous grazing in that land is separated into smaller paddocks, and the group of animals is moved regularly between paddocks. This intensifies grazing pressure for a smaller period of time (e.g., 1–3 days for ultra-high stocking density or 3–14 days for typical rotationally grazed), leaving a rest period for regrowth inbetween. Little research is available in North America, but there are reports of soil C sequestration with rotational grazing on pasture in mesic climates (Baron and Basarb 2010; Conant et al. 2003). The U.S. DOE technical guidelines for voluntary GHG reporting (1605(b) program) assume soil C sequestration rates of 2.9 t CO_2e ha^{-1} yr^{-1} under rotational grazing (U.S. DOE 2007), although this value is taken from the expert opinions expressed by Lal et al. (1999) and Follett (2001a), in which the particular rate is assigned to all improved pasture management, including rotational grazing. In contrast, rotationally grazed grass/legume pastures in Canada's prairie grazing land area resulted in C sequestration rates of 0.23 t CO_2e ha^{-1} yr^{-1}, versus the continuously grazed rate of 0.28 t CO_2e ha^{-1} yr^{-1} (Lynch et al. 2005), a very small (but negative) impact.

GHG Impact Summary

GHG Category	Rotational Grazing (MIG) on pasture	Fertilize Grazing Lands
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**1.45** (-0.05 – 2.90)	**1.05** (0.37 – 5.86)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**-0.82** (-1.70 – 0.05)	**-0.75** (-0.89 – -0.60)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**3.57** no range	**-0.94** (-1.36 – -0.59)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**4.20**	**-0.63**
Maximum U.S. applicable area, Mha	**42**	**80**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

On highly productive pasture, rotational grazing maintains the utilized forage at a relatively young and even growth stage, allowing cattle to utilize better-quality, lower-fiber-content forages. This lowers methane emissions from grazing animals by up to 22% when compared with continuous grazing (DeRamus et al. 2003). Rotational grazing pasture also tends to be more productive in terms of total available forage (e.g., Bosch et al. (2008) noted that grass consumption nearly doubled), so that less land is required for equivalent cattle weight gain, (Baron and Basarb 2010; Bosch et al. 2008). With better quality forage there are also fewer open (non-pregnant) cows, further improving efficiency (Bosch et al. 2008). Efficiency gains may allow shifts of pasture land to afforestation or other high C sequestration activities (Baron and Basarb 2010), and if GHG mitigation is calculated on an output-basis (see Murray and Baker 2011), the emission reduction resulting from efficiency gains can be as much as or greater than that from soil C sequestration (Bosch et al. 2008), so that the primary GHG benefits to rotational grazing may be non-soil C related. Therefore, although increased stocking density resulted in 1.7 t CO_2e ha^{-1} yr^{-1} of greater CH_4 and N_2O emissions, these are not really problematic if offset by efficiency gains, as they are in this case. Note that the upstream and process emission reductions reported for this synthesis (3.57 t CO_2e ha^{-1} yr^{-1}) are based on this one study, and the real value is expected to be highly variable, even though no range is available at this point.

Current adoption of rotational grazing is generally small. For example, surveys in dairy grazing systems in the northeastern U.S. found that between 13% and 19% of grazing animals were in MIG systems (Foltz and Lang 2005; Winsten et al. 2010). Therefore, we estimate the total applicable land area to be, at most, 87% of the 48 Mha of pasture in the U.S.

In contrast to pasture, continuous grazing on rangeland is equal to or out-performs rotational grazing in animal production per head and per area and in plant production (Briske et al. 2008; Derner et al. 2008; Manley et al. 1997). In Wyoming, cattle weight gain was 6% lower in the rotational grazing system than under continuous grazing (Derner et al. 2008). In fact, stocking rates and precipitation overshadow most differences between grazing systems on rangeland (Derner et al. 2008; Jacobo et al. 2006).

Table 25. Estimates of soil C sequestration potential for rotational grazing on pasture.

Citation	Activity/Region	Comments or Caveats	Physical Potential ($t\ CO_2\ ha^{-1}\ yr^{-1}$)
Conant et al. (2003)	MIG/Virginia	Four farm locations	1.50
Lynch et al. (2005)	Rotational grazing/Alberta, Canada	Praries, grass-legume pasture	-0.05
U.S. DOE (2007)	rotational grazing/1605 (b) technical guidelines for voluntary reporting	assumes steady increase in soil C over 20-yr period	2.90

Other grazing land management practices

In general, grasslands are not fertilized heavily—unlike grain and row crops—and fertilization (not just nitrogen, but also micronutrients) can increase productivity of the grazing land, contributing to root and aboveground growth that can become soil C (Table 26). Lynch et al. (2005) found that fertilizing pasturelands on the Canadian prairies (100 kg N ha^{-1}) resulted in a SOC gain of 0.81 t $CO_2e\ ha^{-1}\ yr^{-1}$. Conant et al. (2005) summarized the results of several papers and found that an average of 6.1 kg of C was sequestered for every kg of nitrogen applied. Franzluebbers and Stuedemann (2009) found that C sequestration rates for Georgia pasture in the surface 30 cm of soil were relatively unaffected by whether fertilizer was applied all as inorganic (2.44 ± 1.40 t $CO_2\ ha^{-1}\ yr^{-1}$), part inorganic and part organic (3.37 ± 2.12 t $CO_2\ ha^{-1}\ yr^{-1}$), or all organic as poultry litter (3.29 ± 2.48 t $CO_2\ ha^{-1}\ yr^{-1}$). And, while fertilization may reduce the overall uptake of CH_4 (Mosier et al. 1998b), it can also stimulate N_2O emissions—effectively offsetting a substantial portion of the gains from any soil C sequestration (Paustian et al. 2004; Lynch et al. 2005). Using IPCC Tier I estimates, 250 kg N fertilizer ha^{-1} would increase N_2O emissions by 0.94 t $CO_2e\ ha^{-1}$.

GHG Impact Summary

GHG Category	Irrigation on Grazing Lands	Species Composition on Grazing Lands
Soil Carbon, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**1.04** (0.00 – 1.83)	**2.44** (0.18 – 4.84)
Land Emissions, N_2O and CH_4, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**-0.42** (-1.05 – -0.05)	**-0.94** no range
Process and Upstream, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**-1.08** (-3.34 – -0.41)	**no impact**
Average Net Impact, $t\ CO_2e\ ha^{-1}\ yr^{-1}$	**-0.47**	**1.50**
Maximum U.S. applicable area, Mha	**n/a**	**80**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Grasslands are also most often not irrigated, yet this practice also increases productivity in dryland conditions, thereby increasing soil C inputs. In a review of 17 grassland management studies (87% of which were from Australia, the UK, New Zealand, Canada, Brazil and the U.S.), Conant et al. (2001) found only one study with measurement of irrigation impacts. Nevertheless, of all management changes, irrigation resulted in the greatest levels of C sequestration, 0.4 t CO_2e ha- yr-. In this five-year Australian study, soil C accumulation with irrigation is likely highly correlated with mat production (Rixon 1966). However, there was also a high proportion of low-fertility, allophonic soils with highly decomposable existing surface organic matter, so this finding may not be generalizable for longer periods of time or to other regions. Martens et al. (2005) noted that after many years of agricultural activity in Idaho, irrigated grasslands had more SOC than native dry land (difference of 37–147 t $CO_2e\ ha^{-1}$). However, in another study in New Zealand, no long-term soil C effects were found from irrigation, possibly due to variability of land management and spatial conditions (Houlbrooke et al. 2008). Irrigation water can also contain dissolved CO_2, thereby changing the soil inorganic C dynamics and potentially precipitating CaCO3 and releasing it back into the atmosphere or leaching deeper into the soil profile (Martens et al. 2005; Sahrawat 2003). And, as with irrigation of cropland (above), when considering the energy-related emissions from pumping of irrigation water and the increased N_2O emissions upon irrigation (Rochette et al. 2008b), the net GHG impacts of grazing land irrigation are most likely negative.

Species composition can serve an important role in C sequestration on both rangeland and pasture, and with pasture and rangeland covering 48 and 166 Mha of land, respectively, in the U.S.,, the potential for total sequestration gains is worth considering. In a review, Conant et al. (2001) found an average soil C increase of 4.85 t $CO_2\ ha^{-1}\ yr^{-1}$ with improved grass species on pasture,[38] and soil C increase of 2.75 t $CO_2\ ha^{-1}\ yr^{-1}$ after legume introduction to grass-only pasture. Similarly,

38 In this comparison of four studies, the reported average increase in soil C was 11.2 t CO_2 ha–1 yr–1. However, this finding was skewed by a study of deep-rooted African grasses in the Colombian savanna (Fisher et al. 1994) and, excluding this outlier, the SOC increased by 4.85 t CO_2 ha–1 yr–1.

long-term interseeding of alfalfa on a northern mixed-grass rangela nd in South Dakota, yielded an average increase in SOC of 3.11 t CO_2 ha^{-1} yr^{-1} over a 36-year period, diminishing over time (Mortenson et al. 2004). It is important to note, however, that it can be difficult to separate the soil C impact of species composition changes from other activities, since grazing behavior and grazing intensity are very inter-linked with species composition (V. Baron, personal communication, 6 April 2010). Additional considerations of interseeding include potential emissions associated with seeding due to soil disturbance, evidence of enteric emissions reductions from cattle on grass/legume pastures compared to pure grass stands (McCaughey et al. 1997), and lower N_2O emissions from legumes compared with grasses (Rochette et al. 2004).

Table 26. Estimates of soil C sequestration potential for fertilization, irrigation, and interseeding on grazing land.

Citation	Activity	Comments or Caveats	Physical Potential (t CO_2 ha^{-1} yr^{-1})
Fertilization			
Conant et al. (2001)	Grassland fertilization	Review of 42 global studies	1.10
Follett et al. (2001)	Lime and fertilizer N	U.S., estimates based on review	0.55
Reeder et al. (1998)	N fertilization	Wyoming	1.75
Rice (2000), in Follett and Reed (2010)	N fertilization	Kansas grasslands	5.87
Schnabel et al. (2001)	High vs. low fertilization	Tall fescue, Georgia Piedmont	0.64
Irrigation			
Houlbrooke et al. (2008)	Irrigation of grassland/New Zealand	No significant impact	0.00
Martens et al. (2005)	Irrigation of pasture	Idaho, long-term comparison between irrigated and native	Low: 0.73 High: 2.94
Rixon (1966)	Irrigation of grassland	Australia	Low: 0.51 High: 0.94
Species Management			
Lynch et al. (2005)	Seeded grasslands and legumes/ Southern Canadian Prairie	Low is continuously grazed, high is rotationally grazed	Low: 0.23 High: 0.28
Liebig et al. (2010)	Seeded with wheatgrass and heavily grazed	North Dakota, 44-yr study	0.18
Conant et al. (2001)	Plant improved species	Global review	Legume: 2.75 Improved grass species: 4.84
Follett et al. (2001)	Plant improved species	U.S., estimates based on review	Low: 0.37 High: 1.10
Mortensen et al. (2004)	Interseed native rangeland with legume	South Dakota, sequestration rate decreased over time	4 yrs: 4.31 14 yrs: 2.48 36 yrs: 1.21

The use of fire as a management tool on grazing lands is expected to have a minimal to detrimental effect on GHG mitigation. Periodic burns can promote overall health and growth of rangelands; for example, in tall grass prairie the increased plant productivity after the burn more than compensates for the loss of plant C by ignition. However, most studies found that SOC stays about the same or even decreases following repeated burns (Rice and Owensby 2001). Furthermore, there are several negative co-effects associated with burning (methane, smoke, aerosols) that are linked to climate change, making it even less attractive as a GHG mitigation option (Smith et al. 2008). Therefore, fire management will not be considered further.

Non–carbon dioxide GHGs: nitrous oxide and methane

With respect to N_2O and CH_4 emissions from grazing land, mitigation-oriented research is concentrated in New Zealand, although the mechanisms are likely applicable in the U.S. as well. Soil compaction by grazing action can significantly increase N_2O emissions (Bhandral et al. 2007), while grazing on NT (versus recently tilled pasture or cropland) or during lower field-water capacity conditions can reduce these emissions (Thomas et al. 2008). In addition, more efficient management of mineral N (e.g., sufficient, but reduced N content of animal feed) would likely reduce emissions, but few data exist to support or quantify this hypothesis (Mosier et al. 1998a).

Issues of increased methane emissions from enteric fermentation due to higher stocking rates are stimulating research that explores the link between maintaining forage of a certain quality and rumen methane. Seeding legumes to pasture or otherwise improving grazed forage quality can reduce methane emission by over 20% (DeRamus et al. 2003). A further strategy involves seeding higher tannin-containing legumes that show potential for suppressing methanogenesis in the rumen. Further study is needed to assess the effectiveness of these strategies.

Convert Cropland to Pasture

Conversion of annually cropped land to perennial grass/legumes through set-asides or conversion to pasture can result in higher rates of sequestration than those mentioned above for grazing land management. The conversion of cropland to grazing land has a relatively high per-hectare potential for soil C sequestration (Table 27), regardless of whether the cropland is managed for annual row crops or for hayed perennial grasses. Additional GHG mitigation can come from reductions in fuel use and upstream GHG costs (fertilizer and other inputs) as well as fewer N_2O emissions. The net GHG balance for land emissions is also impacted by increased CH^4 flux from enteric fermentation after addition of grazing animals to the land. The tradeoff of lower agricultural productivity may only make this feasible on more marginal cropland or if incorporating perennial forages into diverse, long-term crop rotations.

GHG Impact Summary

GHG Category	Convert Cropland to Pasture
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**2.82** (0.00 – 4.70)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**1.00** (-0.70 – 4.96)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.45** (0.23 – 0.68)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**4.37**
Maximum U.S. applicable area, Mha	**unknown**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

By converting from cropland to pasture, the associated fuel use can be brought close to zero, and reducing GHG emissions by 0.13–0.71 t CO_2e ha^{-1} yr^{-1}. Fertilizer use on pasture tends to be somewhat lower than on cropland, although with fertilizer N rates on pasture ranging from occasional (Machado et al. 2006) to 600 kg N ha^{-1} yr^{-1} (intensively grazed pasture in New Zealand, Bhandral et al. 2007), a clear idea of the differences is difficult to assess. We assume a conservative 25% reduction in total N fertilizer in calculating the net upstream GHG impact.[39]

Table 27. Soil C sequestration potential of conversion from cropland to pasture, U.S.

Citation	Activity	Comments or Caveats	Potential (t CO_2e ha^{-1} yr^{-1})	National Potential (Mt CO_2e yr^{-1})
Murray et al. (2005)	Cropland to grassland	U.S., rates from CRP	Low: 2.23 High: 4.69	
Lal (2003)	Cropland to pasture	U.S., review, assumed 4.8 Mha	Low: 1.47 High: 4.39	Low: 7.0 High: 21.1
McPherson et al. (2006)	Cultivated soils to perennial grass cover	Colorado and Kansas; used Comet VR to generate potential at MLRA scale	Low: 3.30 High: 5.87	
Reeder et al. (1998)	Cropland to pasture	Wyoming; not grazed after grassland established	Low: -0.46 High: 2.98	
Martens et al. (2005)	Tilled row crops to pasture	Texas	Low: -0.66 High: 3.00	
White et al. (1976)[a]	Cultivation to improved pasture	South Dakota; 4 types of improved pasture, with much variation in species	0.77	
Robles and Burke (1998)[a]	Cultivation to seeded grass	Wyoming, semi-arid grassland	No significant change	
Franzluebbers (2010)	Conventionally tilled cropland convert to perennial pasture	Southeastern U.S.	Low: 2.67 High: 3.48	
Franzluebbers and Stuedmann (2000)	Grazed vs. hayed bermudagrass	Georgia Piedmont	1.58	
Franzluebbers and Stuedemann (2009)	High-pressure (HP) and low-pressure (LP) grazing vs. hayed	Georgia Piedmont, 12 yrs of management	HP: 2.68 LP: 5.10	

a. Studies compiled in the review paper by Post and Kwon (2000).
b. In this study, three grazed treatments and one hayed treatment were planted in "Tifton 44" variety hybrid bermudagrass, whereas the other two hayed treatments were "Coastal" hybrid bermudagrass.

39 Therefore, the assumption is that pastureland receives only 75% of that applied to cropland.

Cropland Set-aside and Herbaceous Buffers

In addition to histosols (discussed above), other cropland areas may have high potential for GHG mitigation when converted back to natural landscape or unharvested vegetation (Table 28). These tend to be sensitive or marginal agricultural land, prone to erosion or flooding, or with the potential to provide significant environmental co-benefits upon conversion. Such land can take the form of herbaceous buffers (grass strips) within a field or along a riparian area, or could consist of larger tracts of land. A significant amount of former cropland has already been taken out of production through the Conservation Reserve Program (CRP), but experts estimate an additional 9–25 Mha could provide significant benefits if set aside from agriculture (Bruce et al. 1999; Sperow et al. 2003). While the co-benefits and non-GHG reasons for implementation are varied—wildlife habitat, erosion prevention, water quality protection, aesthetics—the GHG impacts of converting cropland to set-aside follow similar patterns, with significant potential for soil C sequestration and N_2O emission reduction.

GHG Impact Summary

GHG Category	Cropland to Set-aside or Herbaceous Buffers	Convert Pasture to Set-aside
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**2.49** (-0.15 – 4.74)	**0.32** (-3.37 – 5.39)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**1.41** (0.36 – 5.06)	**no data**
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.74** (0.35 – 1.14)	**no data**
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**4.65**	**0.32**
Maximum U.S. applicable area, Mha	**14**	**unknown**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

Herbaceous buffers are strips of land planted for the purpose of reducing wind and water erosion, reducing N and P runoff into waterways, and for wildlife habitat. These buffer strips (filter strips, conservation buffers, field borders, contour buffer strips) range in width from 6 to 30 m. For the purpose of this paper, herbaceous buffers and other set-aside areas are distinguished from agroforestry by vegetation type. Lands adjacent to cropland planted with woody shrubs and/or trees are considered agroforestry.

The unharvested vegetation in set-aside land sequesters C in two ways: through retention of sediment from agricultural runoff, and through capture and sequestration in biomass. Planting herbaceous vegetation can be more appealing to farmers than trees, due to the lower capital investment and labor. This vegetation is also easier to remove once a program ends, easing implementation, but also raising concerns about long-term C sequestration (permanence).

The physical potential of set-asides to sequester C depends on their size, vegetation, former land use, and structure, making it difficult to generalize. Currently, there are approximately 13 Mha enrolled in government erosion-reduction practices through the CRP. Over 200 million tons of sediment is captured through CRP practices (USDA 2008a). Additionally, more than 1 Mha are enrolled in buffers through the Natural Resources Conservation Service and other state incentive programs. The USDA estimates a soil C sequestration rate of 48 Mt of CO_2 yr^{-1} through the CRP program alone. An additional 9 Mt CO_2 yr^{-1} are offset through energy and fertilizer savings (USDA 2008a).

On the buffer area itself or other set-aside land, there is limited impact on CH_4, as determined by Kim et al. (2010), who measured CH_4 flux in three different buffer vegetation types and adjacent cropland. However, the N_2O emission reduction can be significant, and buffers can reduce N_2O emissions by capturing NO_3^- before it reaches surface- or groundwater and is denitrified off-site. The potential for this benefit will depend on the characteristics of the buffer and N transfers. Different buffers have varying abilities to capture N and also different tendencies to lose that N as N_2O. Hefting et al. (2003) found that forested buffers emitted 10 times more N_2O than grass buffers (as proportion of total N and total quantity), in conditions of high lateral nitrate loading (4,700 kg N ha^{-1} yr^{-1}, in the Netherlands). At equivalent fertilizer N rates, Stehfest and Bouwman's (2006) global model of N_2O emissions shows grassland emissions were 0.16 t CO_2e ha^{-1} yr^{-1} less than cereal crops, and other reviews have concluded that emissions from perennial grassland are much lower than from annual crops (Grant et al. 2004; Machefert et al. 2002; Smith et al. 2008). Eliminating fertilizer N will also reduce land-based N_2O emissions and contribute to upstream GHG savings. The net impact on N_2O will depend on baseline emissions in the land to be removed from production. With the high variability and multiple influential factors, it is difficult to generalize the N_2O emissions reductions for a typical buffer; hence the high range of values in the GHG summary. Each situation would likely need to be modeled, allowing for hydrologic and other input specification.

Table 28. GHG mitigation potential of converting sensitive cropland to set-aside.

Citation	Activity Specific	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)	National Potential ($Mt\ CO_2e\ yr^{-1}$)
Bruce et al. (1999)	Additional CRP	U.S.	2.95	25.7
Follett (2001b)	Additional CRP	U.S.	Low: 2.22 High: 3.33	5.5
Johnson et al. (2005)	Conversion of cropland to grass in CPR	U.S., 5 comparisons	Low: -0.15 High: 4.35	
Lal et al. (2003)	Conservation buffers CRP	U.S. U.S.	Low: 1.10 High: 2.57 Low: 2.20 High: 3.30	3.7–7.3 33.0–47.7
Sperow et al. (2003)	Conversion of all highly erodible land to perennial grass set-aside	U.S., modeled, would remove 25.8 Mha of cropland from production	1.49	38.5
Franzluebbers and Stuedemann (2009)	Unharvested land	Georgia Piedmont (0–30 cm), 12-yr study	3.15	
Burke et al. (1995)	From cultivated to abandoned field	Colorado, 10 yrs	0.11	
Gregorich et al. (2001)	Continuous bluegrass vs. corn	Grass not harvested	4.74	

Convert Pasture to Set-aside

In a review of 276 global cases of grazed versus ungrazed comparisons, Milchunas and Lauenroth (1993) determined that while more sites showed a positive (rather than negative) response of root mass to grazing, there was no consistent impact on soil organic matter. The impacts seem to depend on ecosystem type. Annual forage productivity can be significantly greater in grazed versus ungrazed grasslands (Haan et al. 2007); during a 5-year bermudagrass study in Georgia, forage productivity increased with grazed management compared with unharvested management by 1.3–1.5 $t\ ha^{-1}\ yr^{-1}$ (Franzluebbers et al. 2004). As a result, appropriately managed grazing tends to have a positive soil C impact when compared with ungrazed natural grassland (Derner and Schuman 2007). Grazing activity can stimulate shoot and root growth (Haan et al. 2007; Reeder et al. 2004) and organic acid root exudation, the latter of which can increase inorganic C in arid rangeland soils through carbonate precipitation (Reeder et al. 2004). As well, vegetation breakdown of ungrazed pastures may also lead to increased runoff and erosion (Webber et al. 2010).

In contrast to the native grasslands of the Great Plains, set-asides from grazing in the arid rangeland of the Southwest may have a positive SOC impact due to shrub encroachment. Of 11 studies comparing mesquite and similar vegetation to neighboring grassland, 9 found higher SOC in the shrub/mesquite area with average SOC sequestration rates of 0.79 (± 1.08) $t\ CO_2e\ ha^{-1}\ yr^{-1}$ (Martens et al. 2005). Martens et al. (2005) concluded that the comparison of these studies "suggests an east to west gradient of C accumulation under shrubs across the southwestern USA." Further research may be needed to delineate the most appropriate regions if either grazing or set-aside are to be implemented as GHG mitigation activities.

Table 29. Soil C sequestration potential of grazing land set-aside.

Citation	Activity Specific	Comments or Caveats	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)
Martens et al. (2005)	Allow shrub (mesquite) encroachment on range	Review of 11 studies on western arid land, high variability	0.79
Franzluebbers et al. (2005)	Exclosure from grazing	Review of 3 studies, Oklahoma and Southeast, very high variability	1.17 (-3.37 – 5.39)
Derner and Schuman (2007)	Exclosure from grazing on semi-arid grassland	Review of 5 studies, Great Plains	-0.57
Liebig et al. (2005b)	Exclosure from grazing	Review of 2 studies, Great Plains	-0.59
Manley et al. (1995)	Exclosure from grazing on grassland	Wyoming	-1.20

Information on N_2O and CH_4 flux in grazing land is limited, including the difference between grazing versus set-aside, although urine deposition from cattle can increase N_2O emissions (Liebig et al. 2005b). While the soil of grazed grassland may capture more CH_4 (Franzluebbers 2005), set-aside of grazing land reduces enteric fermentation emissions, at least locally. However, since these cattle will likely be grazed elsewhere with unknown impact on soil C, leakage issues make the narrow focus of the field-level an incomplete accounting. With the net GHG impacts so variable and regionally dependent, non-GHG reasons are most likely to dominate in decisions to convert pasture or rangeland to ungrazed natural grassland. These could be related to streamside protection from trampling (high traffic pressure near water

sources can cause overuse and soil breakdown), habitat protection (endangered species may need protection during critical time periods), or installation of vegetative buffers on hillsides to reduce runoff (Webber et al. 2010).

Wetland Restoration

Often comprised of organic soils (histosols)—but not always—wetlands in North America contain large amounts of stored C, and are estimated to sequester up to 180 Mt CO_2e yr^{-1} (Bridgham et al. 2006). Wetlands are highly variable, in the amount—and characteristics—of organic matter, water level, vegetation, and other factors. It is unclear whether U.S. wetlands on the whole, are net GHG sources or sinks, as there are large uncertainties in all GHG flux estimates (Bridgham et al. 2006). However, it is well understood that draining wetlands—often for agricultural purposes—changes the balance of emissions so that CH_4 emissions nearly cease while CO_2 emissions accelerate with very high SOC oxidation rates. Restoration of these wetlands can reverse this effect. As a unique type of wetland, cropped histosol set-aside has been discussed above. This section will focus on non-histosol wetlands, with the majority of relevant data coming from the prairie pothole region of the Great Plains. With this in mind, we assume total land area available as that of the prairie pothole region (3.8 Mha), although the applicable area may be greater than that, since Lal et al. (2003) suggest that a total (histosol plus wetland) area of 19 Mha is available within the U.S. for restoration.

GHG Impact Summary

GHG Category	Wetland Restoration
Soil Carbon, t CO_2e ha^{-1} yr^{-1}	**3.71** (0.45 – 7.70)
Land Emissions, N_2O and CH_4, t CO_2e ha^{-1} yr^{-1}	**-1.35** (-2.70 – 0.00)
Process and Upstream, t CO_2e ha^{-1} yr^{-1}	**0.74** (0.35 – 1.14)
Average Net Impact, t CO_2e ha^{-1} yr^{-1}	**3.10**
Maximum U.S. applicable area, Mha	**3.8**

Positive numbers depict removal of GHG from atmosphere or prevented emissions.
Tables comparing all practices can be found on pages 45–46.

The GHG impacts of wetland restoration can be estimated by comparing GHG balances of restored formerly cultivated land with that still in cultivation (Table 30). Prairie pothole restoration may have the potential to sequester significant amounts of soil C and reduce N_2O emissions, while increasing CH_4 emissions somewhat, with a net GHG benefit of 5.0 t CO_2e ha^{-1} yr^{-1} (Badiou et al. 2010). The opposite effect can occur in some regions – one study found native marshland in China generated 0.4–0.5 t CO_2e ha^{-1} yr^{-1} greater net GHG emissions than marshland converted to cropland (Huang et al. 2010). However, these negative impacts are an order of magnitude smaller than the potential benefits, so while they are a reminder that each situation must be tested carefully, we conclude that wetland restoration is likely to have significant GHG mitigation potential. In a related manner, the conservation of existing wetlands (although not an agricultural land management practice per se, and thus not directly addressed in this synthesis) will most often prevent significant GHG emissions.

Table 30. Soil C sequestration potential of wetland restoration from agricultural land.

Citation	Activity Specific	Comments or Caveats	Potential (t CO_2e ha^{-1} yr^{-1})	National Potential (Mt CO_2e yr^{-1})
IPCC (2000)	Wetland restoration	Presents a range of global values	2.02	
Euliss et al. (2006)	Wetland restoration	Estimate of all U.S. wetland restoration	2.99	11.5
Lal et al. (2003)	Wetland reserve program	U.S.	Low: 0.73 High: 1.10	1.5–2.2
Badiou et al. (2010)	Restore prairie pothole wetlands	Prairies of Canada	7.70	
Gleason et al. (2009)	Cropped wetland restored to grass wetland (CRP)	Prairie potholes in North Dakota	1.91	

Comparison of Biophysical Potential

In Table 31 we present a side-by-side comparison of biophysical potential for a wide range of agricultural GHG mitigation activities. The estimates given are U.S. averages. The range of values comes from the literature and indicates the degree of variability for each estimate. Depending on the activity, this variability is a result of regional, soil, climate, or crop differences; and/or related to uncertainty in existing measurement or other determination of soil carbon or GHG flux.

Three categories of GHG impacts are included in the comparison: (1) change in soil C, (2) change in land emissions (N_2O and CH_4), and (3) change in upstream and process emissions (fuel, fertilizer and other). A national subtotal is estimated from direct on-site impact and excludes upstream and process emissions, while the final average net impact

includes all three categories. Finally, the maximum applicable area in the U.S. is estimated. Given a limited land base and competing uses for this land, it is likely that less than the total "applicable" land would be converted to these specified management activities. We did not calculate maximum national potential for each activity in acknowledgement of the impacts of land competition, the significant degree of uncertainty inherent in determining the potential applicable area for implementation, and the need for economic analysis and assessment of co-effects and other modifying factors in the prioritization of these activities. These other factors are assessed in the U.S. Assessment Report.

The activities in Table 31 are separated into two groups. The first group (Table 31a) consists of those with a reasonable body of scientific data supporting the estimate of GHG mitigation potential for the GHG that is the primary focus of the mitigation activity, while the second (Table 31b) includes activities for which the estimates reported were determined from expert opinion alone or from three or fewer field or laboratory comparisons. Further exploration of the potential due to these activities would be prudent prior to project or program implementation. Activities with negative GHG mitigation potential are also included in the second group. In this table, positive values represent soil C sequestration and/or GHG emission reduction, while negative values indicate increases in GHG emissions, including loss of soil C.

Table 31a. Comparison of GHG mitigation potential for agricultural land management practices in the United States, summarized from scientific literature. Management activities with more extensive scientific data. All GHG units are in equivalents of carbon dioxide (CO_2e) with 100-year time horizon global warming potential.

Activity	Soil Carbon ($t\ ha^{-1}\ yr^{-1}$)			Land Emissions (N_2O and CH_4, $t\ ha^{-1}\ yr^{-1}$)			Direct Impact ($t\ ha^{-1}\ yr^{-1}$)	Process and Upstream Emissions ($t\ ha^{-1}\ yr^{-1}$)			Average Net Impact ($t\ ha^{-1}\ yr^{-1}$)	Maximum Area (Mha)
	Mean	**Max**	**Min**	**Mean**	**Max**	**Min**		**Mean**	**Max**	**Min**		
Conventional to no-till	1.08	2.60	-0.26	-0.18	0.72	-0.91	0.90	0.12	0.18	0.07	1.01	72
Conventional to conservation till	0.91	1.82	0.00	0.07	0.38	0.00	0.98	0.08	0.10	0.03	1.06	72
Eliminate summer fallow	0.48	2.35	-0.88	-0.03	0.16	-0.30	0.45	-0.12	-0.07	-0.23	0.32	20
Use winter cover crops	0.84	3.24	0.37	0.20	1.05	0.00	1.03	0.56	0.81	0.41	1.50	74
Diversify annual crop rotations	0.58	3.01	-2.50	0.07	0.33	-0.04	0.65	0.00	0.00	0.00	0.65	99
Include perennial crops in rotations	0.57	2.20	-1.75	0.03	0.55	-0.55	0.59	0.17[a]	0.26	0.14	0.76	56
Change from annual to perennial crops	2.26	4.67	0.00	0.12	0.84	-0.55	2.38	0.52	0.67	0.37	2.90	13
Application of organic materials (esp. manure)	2.19	5.10	0.18	0.19	1.81	-1.35	2.38	-0.18	0.21	-0.57	2.20	8.7
Reduce chemical use (other than N)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.09	0.03	0.07	122
Reduce fertilizer N application rates	0.00	0.00	0.00	0.38	1.32	0.14	0.38	0.07	0.09	0.05	0.45	106
Change fertilizer N source – slow-release	0.00	0.00	0.00	0.46	1.43	0.00	0.46	0.07	0.12	0.06	0.51	79
Use nitrification inhibitors	0.00	0.00	0.00	1.01	2.23	0.00	1.01	no data			1.01	92
Improve manure management (N_2O)	no data			0.89	1.22	0.37	0.89	0.00	0.00	0.00	0.89	12
Rice water management for CH_4	0.00	0.00	0.00	1.56	5.22	-0.88	1.56	no data			1.56	1.3
Rice variety development for CH_4	0.00	0.00	0.00	1.17	2.71	0.00	1.17	0.00	0.00	0.00	1.17	1.3
Reduced rice area[d]	no data			4.82	10.26	2.32	4.82	0.00	0.00	0.00	4.82	1.3
Improved grazing management, rangeland[b]	0.93	4.99	-0.10	0.28	0.31	0.27	1.22	no data			1.22	166
Improved grazing management, pasture	2.71	5.86	0.55	0.28	0.31	0.27	2.99	no data			2.99	48
Manage species composition on grazing lands	2.44	4.84	0.18	-0.94	no range[c]		1.50	0.00	0.00	0.00	1.50	80
Cropland to pasture	2.92	4.70	0.00	1.00	4.96	-0.70	3.92	0.45	0.68	0.23	4.37	unknown
Cropland to set-aside or herbaceous buffers	2.49	4.74	-0.15	1.41	5.06	0.36	3.91	0.74	1.14	0.35	4.65	14
Wetland restoration	3.71	7.70	0.45	-1.35	0.00	-2.70	2.36	0.74	1.14	0.35	3.10	3.8

a. Cells that are shaded indicate limited scientific data available (i.e., estimate based on expert opinion or on three or fewer field or laboratory comparisons).
b. Area for rangeland management does not include any land of federal ownership.
c. "No range" is indicated where the value comes from one scientific expert opinion or other estimate, with no indication of variability.
d. Impact of reduced rice acreage depends on subsequent land use. These estimates account for elimination of current CH_4 emissions.
e. For histosols, the total area farmed is highly variable in the literature.

Table 31b. Comparison of GHG mitigation potential for agricultural land management practices in the United States, summarized from scientific literature. Management activities with limited scientific data or negative potential. All GHG units are in equivalents of carbon dioxide (CO_2e) with 100-year time horizon global warming potential.

Activity	Soil Carbon ($t\ ha^{-1}\ yr^{-1}$)			Land Emissions (N_2O and CH_4, $t\ ha^{-1}\ yr^{-1}$)			Direct Impact ($t\ ha^{-1}\ yr^{-1}$)	Process and Upstream Emissions ($t\ ha^{-1}\ yr^{-1}$)			Average Net Impact ($t\ ha^{-1}\ yr^{-1}$)	Maximum Area (Mha)
	Mean	Max	Min	Mean	Max	Min		Mean	Max	Min		
Short-rotation woody crops[f]	2.60	10.63	0.00	0.76	1.52	0.00	3.36	0.65	0.90	0.41	4.02	40
Agroforestry (windbreaks, buffers, etc.)	2.71	4.23	0.84	0.76	1.52	0.00	3.47	0.39	0.47	0.31	3.85	10
Biochar application	3.37	8.92	0.13	1.14	2.93	0.82	4.51	0.70	1.05	0.12	5.22	124
Convert dryland to irrigated	1.46	4.77	1.14	-0.42	-0.05	-1.05	1.04	-1.38	-0.41	-3.34	-0.34	n/a[g]
Irrigation improvements (drip, supplemental, etc.)	0.34	0.42	0.26	0.66	0.94	0.14	1.00	0.23	0.27	0.19	1.19	20
Manage farmed histosols to reduce GHG emissions	5.29	15.03	2.75	11.17	28.18	2.23	16.45	0.00	0.00	0.00	16.45	0.8
Convert histosol cropland to natural[e]	28.48	73.33	2.20	6.80	12.10	1.50	35.28	0.74	1.14	0.35	36.02	0.8
Change fert. N source – from NH_4-based to urea	0.00	0.00	0.00	0.42	2.80	-0.48	0.42	0.00	0.00	0.00	0.42	37
Change fertilizer N placement	0.00	0.00	0.00	0.33	0.47	0.12	0.33	0.00	0.00	0.00	0.33	85
Change fertilizer N timing	0.00	0.00	0.00	0.35	0.52	0.01	0.35	0.00	0.00	0.00	0.35	53
Rotational grazing, rangeland	0.00	0.00	0.00	no data				no data			0.00	166
Rotational grazing, pasture	1.45	2.90	-0.05	-0.82	0.05	-1.70	0.63	3.57	--no range--		4.20	42
Fertilize grazing lands	1.05	5.86	0.37	-0.75	-0.60	-0.89	0.30	-0.94	-0.59	-1.36	-0.63	n/a
Irrigation on grazing lands[h]	1.04	1.83	0.00	-0.42	-0.05	-1.05	0.62	-1.08	-0.41	-3.34	-0.47	n/a
Convert pasture to set-aside	0.32	5.39	-3.37	no data			0.32	no data			0.32	unknown

f. For SRWCs and agroforestry, upstream and process emissions do not take into account the end use for the aboveground biomass,because the GHG impact is highly variable (even though there is a likely GHG benefit). In practice, such lifecycle GHG impacts would need to be assessed.
g. n/a = not applicable, national total is not calculated because the net GHG benefit is negative.
h. Land emissions assume same N_2O response as for irrigated cropland.

Specialty Crops[40]

Farmers in the U.S. grow more than 250 types of specialty crops, including fruits and vegetables, tree nuts, dried fruits, and nursery crops (including floriculture), as defined by Section 3 of the Specialty Crops Competitiveness Act of 2004 (Public Law 108-465, 2004). Another way to view specialty crops is simply any agricultural crop that is not—or has not been—included in federal farm programs (i.e., not wheat, feed grains, oilseeds, cotton, rice, peanuts, or tobacco) (Public Law 107-25, 2001). Grown in all 50 states, specialty crops span approximately 5.6 Mha, of which 3.4 Mha (62%) are irrigated (USDA NASS 2007a). According to the 2007 Agricultural Census, there were a total of 247,772 farms growing specialty crops with a total harvested area of 3.9 Mha (2.0 Mha for orchards and 1.9 Mha for vegetables). This is 3.2% of total U.S. harvested cropland.

The farmgate value (cash receipts) of specialty crops forecasted for 2010 is approximately $83 billion (52%) out of a total crop value of $160 billion (USDA ERS 2010b). California leads specialty crop production in both area and market value (approximately 30% and 35% respectively of total national values), followed (in market value) by Florida, Washington, Oregon, North Dakota, Michigan, Texas, Idaho, Minnesota, North Carolina, New York, Arizona, Wisconsin, and Georgia (Lucier et al. 2006; USDA NASS 2009a; WGA n.d.). The top five fruit, vegetable, or nut commodities produced in the U.S. are grapes, potatoes, lettuce, tomatoes, and almonds (WGA, n.d.).

Much impetus for GHG mitigation action in specialty crops has been coming recently from buyer-driven supply-chain initiatives, rather than C markets or broad-based GHG mitigation programs. For example, the Stewardship Index for Specialty Crops incorporates the monitoring of GHG emissions with other sustainability factors (e.g., air and water quality, biodiversity, energy use, and pesticides) (Stewardship Index for Specialty Crops 2010). Various agricultural land management activities with GHG mitigation potential can be applied to specialty crops, including many that sequester soil C and/or reduce emissions of N_2O and CH_4. However, mitigation potential values applicable to corn or wheat, for example, cannot be directly translated to specialty crops. Perhaps the most significant challenge with specialty crops is that the mitigation potential for different activities can vary by crop (and there are many), and in turn, this variety of crops makes it difficult to determine the optimal techniques for GHG mitigation. Achieving a measurable soil C increase in specialty crops may be challenging due to specialized field management practices requiring tillage, diverse

40 This section was made possible by the research contributions of Candice Chow (Environmental Defense Fund, Sacramento, California).

rotations, and the importance of optimized timing for bringing crops to market (Morgan et al. 2010). For instance, the nature of planting and harvesting of some vegetables, potatoes, and sugar beets results in frequent and intensive soil disturbance, which can enhance N mineralization and may limit carbon sequestration opportunities (Freibauer et al. 2004).

The limited research available suggests that – while perhaps less significant and more challenging to manage than more "common" crops – there is some soil C storage potential due to shifting practices for specialty crops. Cover cropping and increased grain rotations in potato-grain crop systems on sandy loam soil increased soil C content and reduced erosion (Al-Sheikh et al. 2005), and cover cropping and no-till showed increased soil C in California vineyards (Steenwerth and Belina 2008). In contrast, while cover cropping in a Mediterranean tomato-cotton rotation in California increased soil carbon, no-till did not (Veenstra et al. 2007); and in a tomato system in Georgia, no-till only increased soil C when combined with cover crop and N fertilization (Sainju et al. 2002). Compost substitution for synthetic fertilizer also has potential, as shown in a maize-vegetable-wheat rotation in Pennsylvania where compost increased soil carbon by 16%–27% over 9 years compared to a decline with synthetic fertilizer (Hepperly et al. 2009).

Some practices adopted in certified organic agriculture (crop diversity, crop rotation, and organic matter amendments) may also demonstrate GHG mitigation benefits, but depending on cover crops or timing of organic amendment applications, an increase in N_2O emissions is also possible. In a study of large-production Salinas Valley vegetable farms transitioning to organic production, Smukler et al. (2008) noted yield increases of 45%–95% after 3 years, increasing cropping efficiency and thus creating potential for reverse leakage as well as increased soil C and reduced soil nitrate levels (which translates to likely N_2O emission reduction). Reduced chemical use in such systems can also have a small GHG beneficial impact, with little to no yield-reduction effect (Clark et al. 1998b).

Table 32. N fertilizer applied for top specialty crops (by acres), California.

Crop	Rate per application (kg N ha^{-1})	Rate per year (kg N ha^{-1})	Location	Citation and Comments
Lettuce		560–580	Central Coast, CA	Smith et al. (2009a; 2009b)– assuming 2 crops per calendar year
Head lettuce	76	289	CA and AZ	USDA NASS (2007b) – 70,400 hectares
Lettuce, broccoli, celery		124–371	CA	Burger et al. (2009)
Broccoli	84	242	CA	USDA NASS (2007b) – 52,000 hectares
Tomatoes, processing		124–297	CA	Burger et al. (2009)
Tomatoes, fresh	29	242	CA, FL, GA, NJ, NC, OH, TN	USDA NASS (2007b) – 42,700 hectares
Almonds		22–313	CA	Freeman et al. (2008) – note that higher rates are for producing years, lower for establishment
Grapes		6–56	CA	Vasquez et al. (2007) – note that higher rates are for producing years, lower for establishment
Fall potatoes	58	242	U.S.	USDA NASS (2007b) – 342,000 acres

Management practices such as irrigation and precision agriculture that are used for all crops—but more common among specialty crops—can also impact on N_2O and other GHG emissions. Cover cropping may also decrease N_2O emissions in some systems, as in a study of lettuce in a Midwestern sandy loam, where N recovery was double in the cover-cropped system as in a winter bare soil (Wyland et al. 1995). It is uncertain how widespread some of these alternative management practices are used among specialty crops, but would be helpful in understanding the mitigation potential from specialty crops alone.

The biggest mitigation gains in specialty crops may lie in N fertilizer management which can also address water quality concerns related to high application rates in some vegetable crops (Table 32). For example, compare N fertilizer application for lettuce of almost 600 kg N ha^{-1} yr^{-1} to corn N fertilizer application of 280 kg N ha^{-1} yr^{-1} as estimated in UC Cooperative Extension Cost & Return Studies (Brittan et al. 2008; Frate et al. 2008; Smith et al. 2009a; Smith et al. 2009b), even though lettuce crop N removal is significantly below that for corn (Osmond and Kang 2008). As in other farming systems, 4R Nutrient Stewardship (right rate, source, place, and time) plays an important role in N_2O emissions for specialty crops. Decreases in N_2O emissions may be achieved by using the same alternative application practices used for other crops (e.g., split application [Burton et al. 2008b], or using slow-release fertilizers like polymer-coated urea [Hyatt et al. 2010]).

In the U.S. and globally, fruits and vegetables use 4.4% and 15.6%, respectively of total fertilizer N (Heffer 2009). Other non-grain/oilseed/cotton/sugar crops (which can include pasture, and pulses, but also some specialty crops) use 24.2% and 16.0% in the U.S. and globally, respectively. There is no evidence to suggest that emission factors for specialty crops vary significantly from those for corn, wheat, and others; and N_2O emissions are most likely impacted by C substrate and N availability as well as soil moisture conditions. Thus, based on fertilizer N use alone, U.S. specialty crops could be responsible for 5%–20% of fertilizer-related N_2O emissions from agriculture and perhaps a similar proportion of emissions from legume and manure or compost-derived N. Current high fuel use rates for some specialty crops may provide room for efficiency improvements to generate lower process and upstream emissions.

Other challenges in quantifying GHG emissions arise from the sheer diversity of crops and understanding how these alternative management practices differ in GHG emissions. While process-based modeling, such as the DNDC model, can be used for most crops and can track the interactions of many different management practices with GHG emissions, validating the model for each crop type in a variety of different environments may be prohibitively expensive. However, modelers indicate that a significant amount of relevant GHG impact data in specialty crop systems already exists, but has not yet been incorporated into models due to lack of funds for hiring staff to do this work (S.J. Del Grosso, personal communication, 22 April 2010). Even if this was possible, there is likely a need to categorize crops into larger groups so that specialty crop growers can access existing models, as an intermediate step toward having commodity- and region-specific models.

GHG Impacts of Plant Breeding and Biotechnology Advances[41]

Biotechnology, defined as "any technological application that uses biological systems, living organisms, or derivatives thereof, to make or modify products or processes for specific use" (UNCBD 2010), 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. Agricultural biotechnology includes traditional practices, such as selective breeding and hybridization, and advanced technologies, such as Marker Assisted Selection (MAS) and genetic modification or engineering (GM or GE) using recombinant DNA technology (Buttazzoni 2009).

Yield increases are a major driver for agricultural efficiency and have fostered as much as 591 Gt CO_2e emissions avoidances since 1961 (Burney et al. 2010). However, higher yields do not always correlate with reductions in agricultural land use or preclude agricultural expansion, due to ever-increasing demand (Balmford et al. 2005; Burney et al. 2010; Ewers et al. 2009; Green et al. 2005; Matson and Vitousek 2006; Rudel et al. 2009). In this context of mitigating future agricultural GHG emissions, yield increases are a key strategy to meet the growing global food demand, which is expected to increase 70% by 2050 (FAO 2006).

Grain yields in the United States and globally have risen significantly since the mid-1900s, with plant breeding contributing about 50% of the increase and improved management the other 50% (Duvick 2005). While much of the discussion about increased future yield potential centers around GE crops, some reports suggest that these crops in the U.S. have delivered lower yield increases than traditional breeding (Duvick 2005; Gurian-Sherman 2009; Ortiz-Monasterio et al. 1997). However, in the most comprehensive study to date of the impacts of the use of GE crops in the United States, the National Research Council of the National Academy of Sciences of the United States found that GE crops have helped improve water and soil quality, reduce GHG emissions, decrease the use of insecticides, and lower costs of production because of higher yield returns (National Research Council 2010).

One of the mechanisms for increased yield in wheat and other grains is breeding for stronger and shorter stems to reduce lodging (falling over) (Reitz 1970). Traditional breeding and hybridization, which involve controlled mating of elite germplasm selected for desirable genetic traits, have particularly increased yields in corn (Duvick 2005; Ortiz-Monasterio et al. 1997). Advanced technology in variety selection (without GE) has also improved lodging resistance in corn (Flint-Garcia et al. 2003). New GE crop varieties have also exhibited yield improvements through improved pest and disease resistance (Carpenter 2010; Edgerton 2009; National Research Council 2010). Advances in traditional breeding and marker-assisted selection for pest and disease resistance are also ongoing (Flint-Garcia et al. 2003).

Bacillus thuringiensis (Bt) crop varieties have provided more effective pest control than conventional pesticide usage and have reduced the environmental impact of agriculture by reducing the use of harmful pesticides (Pray et al. 2002;

41 This section was made possible by the research contributions of Pradip K. Das (Monsanto).

Qaim and De Janvry 2005). Brookes and Barfoot (2010) estimate that, since 1996, biotech (GM) crop areas have reduced insecticide and herbicide use by a total of 352 million kg (8.4%) globally as compared with conventional systems, with developed countries responsible for 50% of these benefits. The largest environmental gains were seen in cotton, but significant gains were also seen in the soybean, corn, and canola sectors. Bt plant varieties resistant to corn rootworm and other pests may also exhibit enhanced root strength, larger root balls, and reduced lodging leading to increased aboveground biomass, possibly leading to increased carbon sequestration potential (Coulter et al. 2010). Improved rooting structures in corn (from traditional breeding or GE) also enable better crop growth under NT (F. Yoder, personal communication, 30 April 2010), extending the GHG mitigation impact beyond yield improvements to increase feasibility of NT management, with the associated soil C sequestration benefit.

GE crop varieties with herbicide tolerance (HT), such as glyphosate-resistant canola, wheat, corn, and soybean have helped reduce tillage needs and soil compaction, albeit accompanied by the increased use of glyphosate. Within the U.S., the most rapid adoption of GM seeds has been in areas planted under NT management (GM cultivars comprised around 99% of total NT soybeans in 2008). Brookes and Barfoot (2010) estimate that the average level of carbon sequestered per hectare as a result of this soybean conversion to NT, facilitated by the use of GM HT cultivars, is 0.16 t CO_2e $ha^{-1}yr^{-1}$.

Variety development (both traditional and GE) for shorter growing seasons and other characteristics can also directly impact GHG mitigation by increasing the viability of using cover crops and other intensified rotations in applicable regions. Efforts are also under way to develop new plant varieties that carry characteristics that could help increase soil carbon storage, improve N use efficiency (NUE), or reduce irrigation requirements. In fact, there have been developments in transgenic canola, rice, maize, and wheat that demonstrate improved NUE (Beatty et al. 2009). Canola varieties development by Good et al. (2007) required 40% less fertilizer N to achieve yields that were equivalent to original varieties. Other relevant crop breeding and development activities include variety development of rice for lower CH_4 emissions (Aulakh et al. 2001a; Wassmann et al. 2002) and genetic improvement in short-rotation woody crops such as willow (Smart et al. 2005).

Crops that are optimized for nutrient use can reduce N_2O emissions and also reduce other N losses (leaching and runoff), with lower reliance on fertilizer N. The NUE of corn crops in the U.S. has improved 36% over the past several decades (Gurian-Sherman and Gurwick 2009) and traditional and enhanced breeding has prompted a 42% gain in NUE for wheat in Mexico (Ortiz-Monasterio et al. 1997). Other countries have seen similar increases for rice, cereals, and grains (Gurian-Sherman and Gurwick 2009). Genetic engineering of crops for improved NUE involves gene insertion to increase nitrogen metabolism, but research in this field is still limited and commercial potential of this technology is unknown.

One additional biotechnology under development is optimization of crops for water use. Improved water use productivity could have small GHG benefits resulting from improved yields in water-stressed areas and less irrigation with associated input emissions. Traditional plant breeding for yield improvement has succeeded in improving water use due to achievements in reduced crop growth duration. For instance, the modern "IRRI varieties" of rice have improved water use threefold since the green revolution (Farooq et al. 2009; Kijne et al. 2002). Further promising opportunities include genetic selection of plants to reduce transpiration without lost productivity, or to increase productivity while maintaining current transpiration rates (Kijne et al. 2002). Biotechnology developments for water-deficit tolerance have also been achieved (Castiglioni et al. 2008; Nelson et al. 2007). However, in some cases, breeding for drought resistance results in moderated growth, reduced leaf area, and short growth duration which could negate the benefits of reduced water use (Blum 2005).

Much of the discussion about the potential for biotechnology to increase yields and mitigate climate change centers on GE products, as opposed to traditional breeding or MAS. Research to date has shown that GE crops, on the whole, have beneficial effects on the environment by displacing toxic herbicides and insecticides, stimulating conservation tillage, and bolstering farm income and efficiency (Dale et al. 2002; National Research Council 2010). However, possible negative environmental co-effects or social barriers to such development may modify biophysical GHG mitigation potential of these practices. These issues—discussed in greater detail in the co-effects and barriers sections of the U.S. Assessment Report—include the emergence of "superweeds" resistant to herbicides, negative effects of monoculture cropping, and social/ethical resistance to advanced genetic manipulation.

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