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

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 Strategies for Agricultural Land Management in the United States*

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

The **Technical Working Group on Agricultural Greenhouse Gases (T-AGG)** began work in November 2009 to assemble the scientific and analytical foundation for implementation of high-quality agricultural GHG mitigation activities. 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 mitigation programs and markets. Much research to date has concentrated on manure management and on forests on agricultural lands. Conversely, the T-AGG focus is on production agriculture and grazing lands, for which a number of new mitigation protocols are now being devised.

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 it regularly engages the expertise of a science advisory committee and cross-organizational advisory board (details below). Its work is made possible by a grant from the David and Lucile Packard Foundation.

T-AGG has produced a series of reports that survey and prioritize agricultural mitigation opportunities in the United States and abroad to provide a roadmap for protocol development, providing in-depth assessments of the most promising approaches for protocol development. Experts and scientists provided guidance throughout the process, through advisory groups, other meetings, and individual outreach. In addition, T-AGG has sought the agricultural community's feedback and guidance on the approaches it assesses.

T-AGG's reports provide the fundamental information necessary for the development and review of protocols for agricultural GHG mitigation projects and for the design of broader programs that wish to address GHG mitigation (e.g., programs under the U.S. farm bill). Accordingly, the reports are intended to be of use to private or voluntary carbon markets and registries as well as to regulatory agencies that may oversee GHG mitigation programs or the development of regulatory carbon markets.

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For more information see our website: http://nicholasinstitute.duke.edu/ecosystem/t-agg.

The Third Edition

This synthesis of the literature on the GHG mitigation potential of agricultural land management 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.

This third edition updates the side-by-side comparison of the biophysical GHG mitigation potential of 42 agricultural land management activities with newly available and previously un-included data from field experiments, modeling, and expert review. Rather than directly using summarized values from previous literature reviews, T-AGG now derives estimates of mitigation potential from original field data sources, where available. To calculate average anticipated GHG mitigation potential, we use individual data points of side-by-side experiments (previous editions also included expert and modeling estimates and applied different weighting factors). When possible, the values are regionally weighted by the total cropland area of the different regions. Where few experimental observations are available, when the GHG mitigation potential is low or negative, when evidence suggests that life-cycle GHG concerns exist, or where the available data appear inconsistent, the average was not calculated. Instead, a range of observed, modeled, and expert estimates of mitigation potential is reported. Net GHG effects are calculated for activities with sufficient data for the target greenhouse gas, and as such information is available.

Another change from previous editions of this literature synthesis is a more detailed categorization of activities by the amount of available research. This change facilitates organization of activities for implementation and future research prioritization. In this report, specific research gaps are highlighted in the sections describing individual activities.

Introduction

This document is an appendix to the T-AGG report *Assessing Greenhouse Gas Mitigation Opportunities and Implementation Strategies for Agricultural Land Management in the United States* (hereafter called the "U.S. Assessment Report"). An extensive scientific literature review, this report provides a side-by-side comparison of the biophysical greenhouse gas (GHG) mitigation potential of 42 agricultural land management activities in the United States. 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; stakeholders had little sense of the potential or viability of other management activities on a large scale.

By providing an initial review of a larger range of activities, this synthesis identifies those activities that deserve further attention. For many activities, the available data—expert estimates and information from other literature reviews and field experiments—are too scattered and incomplete to support a formal meta-analysis, which would provide a robust assessment of mitigation potentials 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, severely limited data are noted. Scientific certainty is further discussed in other T-AGG reports, the U.S. Assessment Report and the T-AGG Survey of Experts. Researchers at North American universities and in the Agricultural Research Service (ARS) of the U.S. Department of Agriculture (USDA) are currently conducting meta-analyses to assess issues such as the soil carbon response to tillage changes as affected by sampling depth, region, soil type, and other factors.[1]

This report assesses individual activities, but it also 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 supports a long-term perspective on energy and elemental transformations. Most broadly, agricultural land management is one component of the larger biosphere, where organisms and materials interact in ways that affect 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 and thus influence soil C storage and other GHG emissions. Every farm reflects a unique combination of multiple management decisions. The cascading effects of these decisions on the agricultural ecosystem can result in synergies whereby one activity enhances or is additive to the GHG mitigation potential of another or in tradeoffs whereby one activity reduces or eliminates the benefits of another. To the extent possible, all management decisions must be incorporated in the quantification of a farm's GHG impact. Although this

1. C. Rice, personal communication, January 2011; S. Ogle, personal communication, March 2011. Some of this ongoing work is also part of the T-AGG project.

report disaggregates activities to understand their individual impact, it also discusses their interactions, some of which have been well documented. The U.S. Assessment Report describes techniques for quantifying the GHG mitigation impact of activities and their interactions.

Although a useful metric for a side-by-side comparison, biophysical GHG mitigation potential alone is insufficient for assessing the relative viability of various mitigation activities. Total national biophysical GHG mitigation potential is affected by the amount of land area available for implementation of mitigation activities, which in turn depends on competition for land among the activities and among other land uses. Where activities are mutually exclusive, the choice involves tradeoffs and other economic factors; this choice is addressed in the U.S. Assessment Report, which describes economic models that consider land-use competition and market-force impacts on land management and land-use change (as affected by GHG mitigation policies and markets). The assessment report also considers social and technical barriers to and ecological co-effects of GHG mitigation activities, all of which will affect the viability of these activities.

Methods for Literature Review

While other greenhouse gases may also be affected, most agricultural land management activities target only one of the three major GHGs: 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 their uptake in the system. To determine the biophysical GHG mitigation potential of individual activities, we conducted a review of the literature, using existing syntheses where possible and updating them with newer research. The geographic focus is on the United States, but the review also includes research from Canada and other regions with relevant agricultural systems and management activities, where information is missing or limited. The data focus is on results from field studies; the review notes 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 when field studies were unavailable.

We compiled a list of GHG-mitigating agricultural activities through a review of the literature and from sources that have explored associated market opportunities. These activities tend to be related to extensive or nonpoint sources and sinks of greenhouse gases. They generally apply to large land areas and have been relatively slow to develop into valued GHG mitigation or offset protocols, perhaps because they require the involvement of numerous landowners to achieve appreciable impacts, their mitigation potential is uncertain, or regional differences in that potential can be significant or are not well understood. The activities can be divided into three main categories: (1) those taking place on cropland, where products are removed by human harvest activities; (2) those on grazing land, where animals, mainly cattle, remove plant growth; and (3) those that relate to land-use change, e.g., conversion of cropland to grazing land or restoration of former agricultural lands to wetlands.

Some GHG mitigation activities related to agriculture are intentionally not included in this analysis. These activities include afforestation and manure storage management, both of which are already addressed in established protocols or projects. Land-use changes that have negative GHG impacts (e.g., deforestation, grassland conversion to cultivated land) are important to consider at a parasectoral level, but they are not related to management of existing crop or pasture land and thus are also excluded from this review.

Other possible GHG-mitigating agricultural activities not included in this review include organic farming, urban agriculture, biotechnology applications, and programs to support local farm-product sales (USGS 2009). Research comparing organic farming and conventional farming systems has found significantly greater soil organic carbon (SOC) accumulation in the organic systems, both in the United States (Clark et al. 1998b; Lockeretz et al. 1981; Pimentel et al. 2005) and abroad (Freibauer et al. 2004). However, organic agriculture as a farming system often encompasses multiple activities that have GHG implications (e.g., crop rotation diversity, cover crop use, manure and compost application), the interactions of which are not generally well understood. In this review, the effects of different activities are analyzed separately to facilitate understanding of the processes and driving factors and to allow consideration of the activities within any farming system (organic or not); thereby 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). Advancements in biotechnology could have a wide range of GHG effects, some of which are documented in a separate section at the end of this report.

2. The target is the main greenhouse gas of mitigation interest: CO_2 (either soil C changes or upstream and process emissions), N_2O, or CH_4.

The promotion of local agriculture deals more with marketing and supply-chain issues than specific land-use decisions; it thus falls outside of the scope of this review.

Nitrogen fertilizer application increases yield and SOC (Varvel 2006), prompting its proposal as a potential GHG mitigation technique (Snyder et al. 2009). But because the majority of crops in the United States already receive N fertilizer, increasing application rates above the baseline is unlikely to have any major C sequestration impact, and recent studies have found that additional N fertilizer application above typical levels has little to no impact on SOC or on CO_2 fluxes (Alluvione et al. 2009; Mosier et al. 2006). At least one study (Khan et al. 2007) reported that N fertilizer application encouraged organic matter decomposition; and the corresponding risk of increasing N_2O emissions likely outweighs any potential GHG mitigation benefit, at least in typical U.S agricultural systems. Therefore, this activity is not explored here 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., 1 million hectares = 10^6 ha); **t** (tonne, or metric ton); **Mt** (megatonne, i.e., 1 million metric tons = 10^6 t)

We used existing syntheses to identify original field comparison studies and supplemented these with newer research to calculate an estimate (and range) of mitigation potential, per hectare, for the target GHG. When fewer than 30 U.S. field observations were available, we utilized data from Canada (many within 200 km of the U.S. border), and if necessary, from other international research. In the case of activities for which (1) fewer than 9 observations exist,[3] (2) evidence suggests life-cycle GHG concerns, (3) mitigation potential is low or negative, or (4) available data appear inconsistent, a range of observed, modeled, and expert estimates for mitigation potential is reported. For these lower-certainty activities, no national average was calculated, and nontarget GHG effects were not estimated.

For the three activities with regionally distributed research—switch to no-till, reduce or eliminate summer fallow, and diversify annual crop rotations—the per hectare impact on the target greenhouse gas was estimated for all applicable U.S. regions,[4] and then scaled up to a regionally weighted national average on the basis of the cropland area in each region.[5] For the other activities with sufficient data points on a national basis and positive mitigation potential, experimental data were too sparse to calculate regionally specific estimates of mitigation potential, so the average was calculated as the mean of all field comparisons. Significant outliers were removed from the analyses, to avoid skewing of the results. In this process, experimental data points for each activity with a modified z-score of more than 3.5 were eliminated prior to calculation of the mean and range (Peat and Barton 2005). The reported range contains 80% of the observed experimental results, and thus gives a fair picture of GHG effects that could be observed under conditions in various U.S. regions. Also reported are other relevant GHG impacts, so that the net GHG effect is the sum of estimates for soil C, N_2O, CH_4, and upstream and process effects, to the extent that these data were available.

For many of the examined activities, soil C sequestration (storage) is the main mode of GHG mitigation, removing CO_2 from the atmosphere. For other activities, emissions of N_2O and CH_4 are the main target. Net GHG direct fluxes are important considerations for all mitigation programs or projects. Until the early 2000s, many studies tended to assess only soil C changes and thus were missing data on other important greenhouse gases (N_2O and CH_4). Therefore, relatively few studies report non-CO_2 gases; Table 33 notes where the reported values are based on three or fewer reports.

Upstream and process GHG emissions—in most cases resulting from changes in N fertilizer application rates or from adjustments in fuel for field operations and irrigation—would not be included in an offsets program under an economy-wide cap-and-trade system (like the systems recently debated in the U.S. Congress). However, they would likely be counted under other policies or programs such as the Farm Bill or corporate demand-driven supply-chain programs. Such flux effects have been directly estimated in the scientific literature for only a few of the activities assessed here (i.e., no-till, conservation till, conversion of dry land to irrigated land, reduction of nonfertilizer chemical application). For other activities, the GHG flux effect of changes in N fertilizer rates (e.g., reduced rates with switches to winter cover crops or perennial legume crops) was estimated here from anticipated proportional changes, and GHG equivalents of

3. The level of 9 observations was chosen as a natural break in the number of data points; above that number, observations tended to be consistent with one another for most activities. Any exceptions reflect concerns about biophysical GHG mitigation potential related to the quality, rather than the volume, of available research.
4. The 48 coterminous states are divided into 9 generalized agricultural regions.
5. These three activities have at least five observations in all applicable regions, allowing the calculation of regionally weighted averages.

relative fuel use (e.g., reduced tillage operations for perennial crops) were calculated using national statistical reports and published cost-and-return reports from cooperative extension services at the state level.

As an example, crop production is associated with both fuel- and fertilizer-related upstream and process emissions. If U.S. agricultural fuel use (total amount from Schnepf 2004) is equally allocated to all 124 Mha of cropland (USDA NASS 2007b), the average fuel use for agricultural field operations emits an estimated 0.36 t CO_2e ha^{-1} yr^{-1}.[6] Further, the carbon cost of N fertilizer (for manufacture, distribution, and transportation) is approximately 3.2–4.5 t CO_2 per tonne of N fertilizer manufactured (Izaurralde et al. 1998; West and Marland 2002). If the total N fertilizer consumption of 13.6 Mt N yr^{-1} (Millar et al. 2010; USDA ERS 2010a) is equally allocated to all U.S. cropland, the average fertilizer N application is 103 kg N ha^{-1}, and related process emissions are 0.39 t CO_2e ha^{-1} yr^{-1}. Setting aside agricultural land from production would thus reduce upstream and process emissions by an average of 0.74 t CO_2e ha^{-1} yr^{-1}; other activities with fuel or fertilizer use rates below the average could reduce such emissions by a portion of this total. Emission reductions for an individual project will depend on the baseline cropping system.[7] Throughout this process, all attempts were made to maintain conservative assumptions, as per ISO 14064-2 (2006) standards.

The maximum applicable land area for the mitigation activities assessed here (over and above current adoption rates, i.e., baseline area) was also determined from the literature and available survey data.[8] This land area is affected by crop types, current management practices, and regional or climate variations. Because multiple activities may compete for the same land area, the practical area available for implementation will likely be lower, at least for the activities that are more expensive or challenging to adopt. More detailed economic land-use competition analysis and an assessment of interactions among activities are needed for any national predictions of total mitigation potential.

Some management activities that mitigate greenhouse gases can significantly affect yield or production (e.g., fertilizer rate reductions, changes in crop mix and animal numbers), and the GHG impacts beyond the field or farm—or even beyond the country—will need to be considered in program or protocol development. These "leakage" implications were not incorporated in the estimates presented here. Leakage is positive (or "good") when activities increase productivity or otherwise indirectly reduce GHG emissions in other locations. Negative (or "bad") leakage occurs when activities cause shifts in production that increase emissions elsewhere.

Conservation Tillage (Including No-Till)

Of all agricultural land management activities suggested for GHG mitigation, conservation tillage has been the most widely applied[9] and studied; the majority of research has investigated no-till (NT) management. Given the significance of NT in the literature and in practice, this synthesis treats it as a separate activity and uses the term *conservation tillage* more narrowly to denote any reduced-tillage practice other than NT. With over 280 field comparisons of soil C response to no-till (Table 1), the average mitigation potential for that practice is estimated at 1.2 t CO_2e ha^{-1} yr^{-1} (a range of -0.2 to 3.2). With slight decreases in N_2O and process emissions and no effect on CH_4, the net GHG mitigation potential due to NT is 1.5 t CO_2e ha^{-1} yr^{-1}. Using data from 70 field comparisons, the soil C sequestration potential of other conservation tillage practices

GHG Impact Summary

GHG category	Switch to no-till	Switch to other conservation tillage
# of observations	**282**	**70**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**1.22** (-0.24–3.22)	**0.44** (-0.54–1.38)
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.12**	**0.18**
CH_4, t CO_2e ha^{-1} yr^{-1}	**0.01**	**0.00**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.12**	**0.08**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**1.47** (0.01–3.46)	**0.70** (-0.29–1.63)
Maximum U.S. applicable area, Mha	**94**	**72**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

6. Conversions from amounts of gasoline and diesel to CO_2 are drawn from the U.S. Energy Information Administration, available at http://www.eia.doe.gov/oiaf/1605/coefficients.html carbon content of fuel (accessed 23 September 2010).
7. Fuel-related emissions during field operations vary significantly when comparing crop types; California crop production data indicate a range in fuel emissions of 0.13–0.71 t CO_2e ha^{-1} yr^{-1} (corn > hay > wheat). This range is calculated from crop production cost reports published by University of California Cooperative Extension (http://coststudies.ucdavis.edu) and the carbon content of fuel.
8. Total crop areas and relevant survey data were taken from the U.S. Agricultural Census. Current implementation rates from various sources were used to determine the applicable crop area for each activity (see text in each section for relevant details).
9. Conservation tillage has been implemented for reasons other than GHG mitigation, including soil erosion control, soil quality enhancement, and reduced fertilizer requirements (related to SOC retention).

averages 0.4 t CO_2e ha^{-1} yr^{-1} (a range of -0.5 to 1.4). Slight decreases in N_2O and process emissions result in a net GHG mitigation potential of 0.7 t CO_2e ha^{-1} yr^{-1}.

Some form of conservation tillage is now applied on more than 40% of U.S. cropland; 24% to 35% of cropland is under NT management (CTIC 2008; Horowitz et al. 2010). Therefore, conservatively estimated, the maximum area applicable for additional conservation tillage adoption is 72 Mha[10] and that for NT adoption is 94 Mha. These land areas are not additive, because land intended for other conservation tillage would no longer be available for NT management. Although field surveys focus almost exclusively on continuous NT management, some of the area they count as NT area is actually under NT management for only one year or two consecutive years, not continuous NT as is reported in the research studies on soil C effects (Horowitz et al. 2010).[11] Thus, shifting from intermittent NT to permanent or semipermanent NT management may create additional opportunities for mitigation.

Since European immigrants settled in North America, much land has been under continuous cultivation, leading to significant reductions in soil organic matter (SOM) levels relative to SOM levels in land under native conditions.[12] Current soil organic carbon (SOC) levels for agricultural land are 22% to 36% lower than SOC levels in 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 (and thus SOC) can reduce soil fertility and therefore cause decreases in crop production and greater reliance on fertilizer.

Reducing tillage from the traditional moldboard plow (inversion of the soil profile) has become important for controlling erosion, maintaining soil fertility, and improving crop health. Equipment and chemical development have 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 tillage, whereby crop rows are planted on top of ridges that are scraped off for planting and rebuilt during the growing season; (2) strip tillage, whereby only the seed row zone is disturbed (tilled); and (3) mulch tillage, a form of reduced tillage in which residue is retained and spread out but the soil is tilled just prior to planting.

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

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})
Switch to no-till				
Lal et al. (1999b)	U.S. general	Based on reviews and expert opinion	Expert estimate	*1.83*
Six et al. (2002b)	Temperate and tropical soils assessed together	Modeled to 30 cm depth	Modeled, based on >55 field comparisons	*1.19*
Sperow et al. (2003)	U.S. general	Modeled using IPCC method	Modeled estimate	*2.39*
Six et al. (2004)	Considers humid and dry regions separately	Modeled; 20 yrs; effects over time also examined	Modeled, based on 254 comparisons	*Humid: 0.81* *Dry: 0.36*
Dell et al. (2008)	Pennsylvania	Compared farmers' fields; not side-by-side comparisons	3 counties	*1.57*
Peterson et al. (1998)	Colorado and Texas	Review	6	0.65
West and Post (2002)	Regionally dispersed	Review; reported mean of 2.09 includes some non-U.S. studies	44*	1.76
Alvarez (2005)	Regionally dispersed	Review; reported mean of 0.95c	35*	0.93
Liebig et al. (2005b)	Great Plains	Review; reported mean of 0.99; only 1 unique comparison (others included above)	1	0.81

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

11. In the Mississippi River basin, the NRI-CEAP multi-year cropland study found that only 50% of the corn and soybean crop area reported to be under NT management was continuously under such management during the three-year study period (Horowitz et al. 2010). The remaining area was under NT management for only one or two crop cycles.

12. At early stages of soil formation, organic matter accumulates at much higher rates than decomposition. Over time, the level of organic matter appears to stabilize; that is, accumulation and decomposition rates equilibrate. At this point, any further changes are very small compared with those occurring when the soil environment experiences a significant shift, such as tillage or a change in vegetation cover.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})
Johnson et al. (2005)	Corn Belt	Review; reported mean of 1.47	23*	1.85
Martens et al. (2005)	Southwestern United States	Review; reported mean of 1.10	28	0.81
Franzluebbers (2005)	Southeastern United States	Review; reported mean of 1.54	59*	1.37
Franzluebbers (2010)	Southeastern United States	Review; reported mean of 1.65, which also included observations in Franzluebbers (2005)	28*	1.67
Luo et al. (2010)	Regionally dispersed	Review; reported slight decline in SOC when counting soil profile up to 40 cm and numerous international observations	11*	0.85
Potter et al. (1998)	Texas, corn and cotton	15 yrs, 20 cm	9	0.74
Six et al. (1999)	Michigan, Kentucky, Nebraska, Ohio; corn, soybean and winter wheat	38 yrs, 20 cm	4	0.54
Denef et al. (2004)	Nebraska, winter wheat-fallow	One location, 33 yrs, 18 cm	1	0.21
Puget and Lal (2005)	Ohio, corn-soybean rotation	8 yrs, 80 cm	2	2.98
Dolan et al. (2006)	Minnesota, corn and soybean	One location, 23 yrs, 45 cm	4	-0.44
Venterea et al. (2006)	Minnesota, corn-soybean rotation	One location, 15 yrs, 60 cm, not statistically significant	1	-3.15
Vyn et al. (2006)	Indiana, corn and soybean rotations	Converted from moldboard and chisel plow, all data reported as average, 28 yrs, 100 cm	1	1.05
Huggins et al. (2007)	Minnesota, corn-soybean monocrops and rotations	From moldboard and chisel plow, 14 yrs, 45 cm	3*	1.44
Senthilkumar et al. (2009b)	Michigan, corn-soybean-wheat rotations	One location, 18 yrs, 40 cm	2	1.42
Archer and Halvorson (2010)	Colorado, corn	One location, 4 yrs, 30 cm	3	1.20
Jagadamma and Lal (2010)	Ohio, corn and soybean	From chisel plow, 42 yrs, 45 cm	1	0.36
Stone and Schlegel (2010)	Kansas, wheat-sorghum-fallow	12 yrs, 10 cm	1	0.44
Varvel and Wilhelm (2010)	Nebraska, corn-soybean monocrops and rotations	Converted from subtill and chisel, moldboard, and disc plow; 24 yrs, 30 cm	12	1.34
Wortmann et al. (2010)	Nebraska, corn-soybean and grain sorghum-soybean rotations	Converted from moldboard and mini-moldboard plow, 5 yrs, 30 cm	3	0.57
Switch to other conservation tillage				
Lal et al. (1999b)		Mulch tillage and ridge tillage; based on reviews and expert opinion	Expert estimate	*Mulch tillage: 1.83* *Ridge tillage: 2.20*
McConkey et al. 1999 (as cited by Follett 2001)	Prairies in Canada	Conversion to minimum tillage	Expert estimate	*Low: 0.37* *High: 1.10*
Follett and McConkey 2000 (as cited by Follett 2001)	U.S. Great Plains	Estimated that soil C change from no-till, mulch tillage and ridge tillage would all be in same range	Expert estimate	*Low: 1.10* *High: 2.20*
Sperow et al. (2003)	U.S. general	Assumes 50% no-till and 50% reduced tillage on 129 Mha	Modeled	*1.09*
West and Post (2002)	Regionally dispersed, mostly wheat or corn rotations	Review; reported no significant impact when including some non-U.S. studies	19*	0.53
Alvarez (2005)	Rocky Mountains and Northern Plains, wheat rotations	Review; reported mean of 0.95 (same as for NT management)	4*	0.34
Martens et al. (2005)	Texas, grain sorghum or wheat rotations	Review; reported mean of 1.03	10	0.59
Franzluebbers (2005)	Georgia and South Carolina	Review; some reported studies have been since updated and are now included separately	2	-1.07
Sainju et al. (2006b)	Georgia, corn or grain sorghum rotations	One location, reduced tillage, 3 yrs, 120 cm depth	12	0.30

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})
Novak et al. (2007)	South Carolina	Conventional tillage versus subsoil only, 24 yrs, 30 cm depth	1	1.66
Veenstra et al. (2007)	California, corn-tomato rotation	Longest-running conservation tillage study in California, 5 yrs, no significant difference	2	-0.17
Sainju et al. (2008b)	Alabama, corn-cotton rotations	One location, mulch tillage, 10 yrs, 20 cm	2	0.38
Novak et al. (2009)	South Carolina	Conventional tillage versus subsoil only, 6 yrs, 15 cm depth, more soil C loss in poorly drained areas of field	2*	-0.94
De Gryze et al. (2009)	California	Conservation tillage; 3-yr study	3	0.24
Stone and Schlegel (2010)	Kansas, wheat-sorghum rotation	Reduced tillage	1	0.17
Varvel and Wilhelm (2010)	Nebraska, corn-soybean monocrops and rotations	Ridge tillage and reduced tillage versus moldboard and chisel plow, 24 yrs, 30 cm depth	12	0.91

a. For activities in this table and many of the following tables, there are sufficient field observations available to calculate mitigation potential. In these cases, expert estimates and modeled values are not included in calculations of the average and range of GHG flux effect; the values are presented for comparison only, and appear above the dotted line in the table, with mitigation potential estimates in *italics*. While some reviews included non-U.S. studies, this synthesis extracts the data from U.S. studies only (unless U.S. data are insufficient). Where original data were reported by more than one review, they were counted only in the first instance (hence some later reviews appear to have fewer observations than would be expected). Outliers removed from the analysis (see "Methods for Literature Review") are not included in these totals; any citations from which outliers were removed are indicated with an asterix (*). A list of all original comparisons is available upon request from the authors.

Reducing soil disturbance not only controls soil erosion and improves soil quality, but also decreases SOM decomposition rates. These outcomes have been demonstrated by a comparison of ^{13}C signatures in SOC from NT and conventional sites (Six and Jastrow 2006) and by the observation of soil C sequestration in many studies.

SOC dynamics and C sequestration potential can also vary in accordance with land's agricultural history. For example, conventional tillage practices differ from region to region (moldboard plowing is common in some areas but not others), and conservation tillage encompasses a wide range of soil disturbance levels. Greater levels of soil disturbance tend to result in lower SOC levels over time, and reducing tillage from conventional practice of full-inversion moldboard plowing is likely to net a greater SOC sequestration response than a conventional practice of chisel plowing or disc cultivating.

Although the data seem to be convincing, some researchers have questioned whether the average soil C change following a switch to NT management and other conservation tillage is actually positive. For example, West and Post (2002) concluded that conservation tillage other than NT management yields very little consistent soil C sequestration. Clear definitions of practice and residue retention[13] may explain why NT management tends to exhibit more consistent potential for soil C sequestration than other conservation tillage (Six et al. 2004; West and Post 2002). Other researchers have proposed that the prevalence of shallow sample depths in much of the reported data tends to overstate soil C changes due to NT management (Baker et al. 2007; Luo et al. 2010).[14] However, these latter assertions have generated significant discussion among the scientific community. The lack of statistical significance with some deeper soil samples could be related to high variability and the need for greater differences before detection is possible (Franzluebbers 2010; Kravchenko and Robertson 2011). More regionally specific assessments are needed.[15]

NT management appears to have the greatest potential to sequester soil C in subhumid regions (with precipitation-to-potential evapotranspiration ratios of 1.1–1.4 mm mm^{-1}) such as the midwestern and southeastern United States. Average soil C sequestration rates for the Southeast are the highest at 1.65 t CO_2e ha^{-1} yr^{-1} (Franzluebbers 2010); other regions have 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). In cooler and wetter soils—for example, those in Minnesota or Wisconsin—maximum C storage may be achieved with occasional (e.g., biennial) tillage rather than NT (Venterea et al. 2006).

NT management yields some of the lowest (or negative) sequestration rates 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). According to

13. In NT systems, crops are planted in a narrow seedbed or a slot created by disc openers. With the exception of these strips, which comprise less than 20%–30% of the row width, soil is undisturbed from harvest until planting. Residue cannot be burned and must be uniformly distributed over the field (USDA NRCS 2010).
14. In this literature compilation, no consensus on the acceptable level of sample depth was found. Thus, the values available in the literature at the prevailing sampling depths are used.
15. For example, the five available experiments cited by Baker et al. (2007) in which NT management resulted in lower SOC at depth were all from Eastern Canada, where cold and humid conditions tend to make such management a less-than-optimum practice for crop production and soil C sequestration.

field data, the average soil C effect of NT management is negative in Ontario and Quebec (Gregorich et al. 2005)—a region near and very similar to the U.S. Northeast, which is bounded on the southwest by West Virginia, Maryland, Pennsylvania, and Delaware. In this region, even though the average soil C change was negative, results were highly variable, depending somewhat on soil and crop type (VandenBygaart et al. 2003). Negative soil C response to NT management may be related to depressed corn yield (and thus residue amount) that results from reduced aeration at annual precipitation rates of >800 mm or to higher nightcrawler earthworm populations in NT systems, the latter of which may enhance decomposition in these eastern soils (VandenBygaart et al. 2003).[16] However, there is some evidence that NT practices on farms in Pennsylvania can sequester SOC, at least in surface soil (Dell et al. 2008).

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

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; in some systems (high clay content, damp climate, wet soils, poor aeration), N_2O emissions increase greatly after implementation of NT management as a result of higher bulk density, more soil C and N, and greater soil water content (D'Haene et al. 2008; Rochette 2008; Six et al. 2002b). Others have found little or no significant difference in N_2O emissions (Grandy et al. 2006; Li et al. 2005a; Parkin and Kaspar 2006; Robertson et al. 2000), and in some drier and warmer regions, the increased aggregate stability and improved drainage leads to reduced N_2O emissions under NT management (Halvorson et al. 2010; Omonode et al. 2011). Therefore, negative GHG impacts are generally limited to poorly aerated soils (Rochette 2008), and time also appears to play an important role. A review of 44 data points revealed higher N_2O emissions in the initial years following transition to NT management, but reduced emissions after 10 or more years when compared with CT management (Six et al. 2004). This finding accords with observations of improved soil structure following 4–6 years of NT management.

The impact of NT management on N_2O emissions may also be affected by the type of N fertilizer used. In one study, NT (versus CT) management reduced N_2O emissions by almost 50% following application of anhydrous ammonia, had no impact with application of urea ammonium nitrate, but increased N_2O emissions with application of 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). In contrast to NT management, other conservation tillage (with 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 to reduce those emissions (Drury et al. 2006; Jacinthe and Dick 1997; Li 1995).[17]

When compared with conventional tillage, both NT and other conservation tillage have been observed to increase CH_4 uptake (Six et al. 2004; Venterea et al. 2005), although this is not always the case (Robertson et al. 2000). The total GHG impact of any CH_4 flux change is, in any case, marginal in contrast to soil C and N_2O flux effects. Any enhanced CH_4 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 NT and conservation 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 by conversion from conventional to conservation tillage (Archer et al. 2002; West and Marland 2002) and to 0.07–0.18 t CO_2e ha^{-1} yr^{-1} by conversion to NT management (Frye 1984; West and Marland 2002). The yearly sequestration potential of conservation tillage and NT management tends to diminish until soil C comes to a new equilibrium point over time (Six et al. 2002a), but the process emissions reductions are a perpetual benefit, even though their value may not be large by comparison. Somewhat small negative upstream GHG impacts may result from application of additional chemical herbicides for weed control, because traditional mechanical weed control (tillage) has been eliminated. However, although the GHG impacts of this increase in herbicides are not significant, other ecological and social factors may be important to consider.

16. Nightcrawler earthworms are not found in western Canadian soils.
17. An exception to this rule was observed in a corn-tomato system in California, where Kong et al. (2009) detected an elevated N_2O flux response from minimum tillage 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.

Fallow Management

Fallow periods, during which no crop is growing, can be reduced or managed to increase soil C stocks, especially if those periods coincide 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.

GHG Impact Summary

GHG category	Eliminate summer fallow	Use winter cover crops
Number of observations	**33**	**31**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**0.60** (-0.22–1.20)	**1.34** (-0.07–3.22)
N_2O, t CO_2e ha^{-1} yr^{-1}	**-0.03**	**0.12**
CH_4, t CO_2e ha^{-1} yr^{-1}	**0.00**	**no data**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**-0.12**	**0.46**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.44** (-0.38–1.05)	**1.92** (0.51–3.81)
Maximum U.S. applicable area, Mha	**20**	**66**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Eliminate or reduce summer fallow

Summer fallow, leaving cropland unplanted for a summer, is often practiced every second or third year for water conservation purposes on 20 Mha of U.S. cropland otherwise susceptible to crop failure from drought (Janzen 2001; Sperow et al. 2003). It can also improve weed control and seedbed conditions (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, the Pacific Northwest, and the Rocky Mountain region. 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). As of 2007, 6.3 Mha of U.S. cropland were under summer fallow, a drop from 6.7 Mha in 2002 (USDA NASS 2007b). This figure accords 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).

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 United States from 21 major grain and food crops was estimated to be 488 Mt/yr (Lal 2005). Using generalized estimates (in which 40% to 42% of residue is C and 5% to 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 2001; Leifeld et al. 2009; Robinson et al. 1996), suggesting that residue input is a strong predictor of soil carbon content when other management practices are held constant. 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; for a crop like grain corn, the large amounts of residue may be difficult to manage and often require tillage for incorporation or removal. However, equipment and practice development will 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, although this threshold varies significantly by year (due to yield and climate variability), by region, and by crop type (Nelson 2002). Soil C maintenance will most likely further restrict harvest amounts (Wilhelm et al. 2007). The DOE estimated that the threshold for using corn stover for biofuels is 30% (i.e., 70% stays on the field) (Follett 2001). Johnson et al. (2006) estimated that between 16% and 50% of corn residue could be harvested from a grain corn crop with a 10.0 Mg ha^{-1} grain yield, while still maintaining soil C levels. More crop residue can be harvested without erosion or negative soil C implications if no-till or other conservation tillage systems are utilized for soil stabilization (Johnson et al. 2006; Nelson 2002). In some areas, the residue generated by certain crops is insufficient to sustain SOC levels (Wilhelm et al. 2004; Wilhelm et al. 2007). Climatic factors have a significant effect on 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 affect yields, further reducing C sequestration potential (Wilhelm et al. 1986).

a. Based on C isotope measurements in a dryland temperate soil, Paul et al. (1997) and Lal et al. (2003) observed a 5% rate of sequestration. A 10-year study in Texas (Franzluebbers et al. 1998) detected up to 20% of crop C input was stored as soil C.

Summer fallow tends to reduce SOC. The elimination of plant C inputs during the fallow period can enhance soil C mineralization by increasing moisture and temperature (Haas et al. 1974), and tillage during the fallow period can raise decomposition rates (Janzen et al. 1998). Summer fallow can also accelerate soil C loss through erosion, although this process may actually redistribute C locally rather than release it to the atmosphere (Gregorich et al. 1998).

With 33 data points (Table 2)—from all applicable regions—the regionally weighted average soil C sequestration rate for eliminating summer fallow is 0.6 t CO_2 ha^{-1} yr^{-1} (a range of -0.2 to 1.2 t CO_2 ha^{-1} yr^{-1}). Small increases in upstream and process emissions are related to additional fertilizer N requirements for the crop that replaces the fallow, resulting in an average net GHG potential of 0.4 t CO_2e ha^{-1} yr^{-1}. In most cases, total crop production will increase,[18] with positive leakage implications.

Table 2. Estimates of soil C sequestration potential for eliminating or reducing summer fallow

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Sperow et al. (2003)	Semi-arid regions of U.S.	Assumes 20 Mha applicable area	Modeled	*0.59*
Follett (2001)	U.S. general		Expert estimate	*Low: 1.10* *High: 2.20*
Lal et al. (2003)	U.S. general	Assumes 9.4 Mha	Expert estimate	*Low: 0.37* *High: 1.10*
West and Post (2002)	Rocky Mountains, Southern Plains, Prairies (Canada)	Review; all studies are wheat	19	0.24
Horner (1960)	Pacific Northwest (Washington)	Long-term experiment	4	0.73
Potter et al. (1997)	Southern Plains wheat	0–20 cm depth; fertilized; NT & stubble-mulch	4	NT: 1.54 SM: 0.37
Rasmussen and Albrecht (1998)	Pacific Northwest (Oregon)	60-yr comparison	1	0.38
Bowman et al. (1999)	Colorado	Full elimination of summer fallow stored more C than reducing it to once every 3 or 4 yrs	1	1.05
Machado et al. (2006)	Eastern Pacific Northwest	0–40 cm depth; fertilized; CT	1	1.21
Sainju et al. (2006a)	Rocky Mountains; spring wheat	0–20 cm depth, reducing summer fallow to every 3 yrs stored more soil C than eliminating summer fallow (in NT system)	3*	CT: -0.24 NT: 1.10

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Summer fallow reduction or elimination has the most effective and consistent soil C benefits when combined with NT management rather than conventional tillage (Potter et al. 1997; Sainju et al. 2006a).[19] By retaining more crop residue and reducing water loss from the soil profile, such management 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 management in wheat-fallow rotations yielded no significant increase in SOC, but conversion to NT management 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).

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). Where summer fallow is still useful for water conservation, its reduction may have greater soil sequestration potential than its elimination (Sainju et al. 2006a). Sherrod et al. (2003) 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. 2002b; 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). When eliminating summer fallow, diversified rotations have resulted in soil C increases (2.7 ± 1.9 CO_2e ha^{-1} yr^{-1}) more than eight times those of continuous wheat cropping (West and Post 2002).

The increase in SOC from transition to continuous wheat cropping or to NT in these systems 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 it requires more time to provide a significant increase. In some cases, significant increases in SOC were not

18. Any summer fallow elimination that is not accompanied by an increase in total productivity is relatively less likely to achieve soil C gains and would be economically inefficient and impractical. Therefore, the whole system must maintain or increase sufficient yield.
19. The soil C change in these cases is just that resulting from the change in summer fallow activity. If the land was previously conventionally tilled, soil C could also accrue as a result of the tillage reduction; the interaction effect of the two activities would then need to be assessed.

observed even after four to eight years (Halvorson et al. 2002a; Ortega et al. 2002). In addition, further examination of regional differences may be warranted. The many studies conducted in Canada and the central Great Plains (Halvorson et al. 2002a; Ortega et al. 2002; Sherrod et al. 2003) may not be applicable to the northern Great Plains due to differences in temperature, rainfall, and growing-degree days (Sainju et al. 2006a). Cold weather in the northern plains may also delay decomposition of any increased plant biomass, 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 its effect is inconsistent (Del Grosso et al. 2002; Desjardins et al. 2005). This review assumes that field operations are not affected (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, increasing process emissions by an average of 0.1 t CO_2e ha^{-1} yr^{-1}.

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

Use winter cover crops

Planting of winter cover crops during the normally fallow winter season increases total primary productivity. As a result, this activity can generate soil C gains of more than 3 t CO_2e ha^{-1} yr^{-1} (De Gryze et al. 2009; Sainju et al. 2002; Veenstra et al. 2007)—increases that are highest in warmer winter locations such as California and Georgia. A total of 31 field observations (Table 3) yields an average soil C sequestration rate of 1.3 t CO_2 ha^{-1} yr^{-1} (a range of -0.1 to 3.2 t CO_2 ha^{-1} yr^{-1}). Adding winter cover crops to a crop rotation can also reduce N_2O and fertilizer-related emissions, and when these emission reductions are also considered, the net GHG mitigation potential for winter cover crops is 1.9 t CO_2e ha^{-1} yr^{-1}. 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).

Experts estimate that winter cover crops can be implemented in most or all moist regions of the United States for a total area of 51–99 Mha of U.S. cropland (Donigian et al. 1995; Lal et al. 1999b; Sperow et al. 2003). Four percent of U.S. cropland was planted to winter cover crops as of 1995, the latest available estimate (Paustian et al. 2004). Excluding dry regions (Rocky Mountains, Great Plains, and Pacific Southwest), the area in winter wheat, and that already adopted, this review estimates that at most 66 Mha of additional cropland could be planted to winter cover crops. Winter cover crops may be less feasible in regions with a relatively short growing season, but crop development and experimentation have shown benefits even in large areas of North Dakota.[20] Cover crops under irrigation also hold some potential for GHG benefits, although care must be taken to ensure that these benefits are not negated by the net GHG impacts of irrigation.

Cover crops can also increase N and water-use efficiencies, which is related to the increased 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, because both legumes and grass species will scavenge and recycle 170–340 kg of mineral N ha^{-1} yr^{-1} that would otherwise be lost through leaching (Delgado et al. 2007), making those nutrients available for subsequent crops upon decomposition of the cover crop[21] as well as avoiding off-site N_2O emissions. Leguminous cover crops also fix atmospheric nitrogen into plant-useable forms (Gregorich et al. 2005), allowing further N fertilizer savings. Alluvione et al. (2010) and Utumo et al. (1990) were able to eliminate N fertilizer and completely meet the N needs of the subsequent crop when using a vetch winter cover crop in northwestern Italy and Kentucky, respectively. With lowered concentrations of soil mineral N 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 carbon from winter cover crop residues could also be used by microbial populations to immobilize available nutrients (such as nitrogen) 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 N need.

20. S. Samson-Liebig, personal communication, March 2011.
21. 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 for use of winter cover crops

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Lee et al. (1993)	Corn Belt	EPIC simulation model (100 yrs), NT	Modeled	*0.15*
Donigian et al. (1995)	Midwest modeled, assumed to extend to United States (87 Mha)	CENTURY model	Modeled	*Midwest: 0.84* *U.S. total: 1.25*
Lal et al. (1999b)	26 states with suitable climate, minus winter wheat area (51 Mha)		Expert estimate	*Low: 0.37* *High: 1.10*
Sperow et al. (2003)	Nationwide, except dry regions (98.5 Mha)	Used model and IPCC method	Modeled	*0.85*
Dell et al. (2008)	Pennsylvania, rye CC	6–13 yrs, not a side-by-side comparison	n/a	*0.00*
Franzluebbers (2010)	Southeast United States, various crop types	87 studies with CC and 60 studies without CC (not side-by-side comparison), primarily NT	n/a	*Min: 0.51* *Max: 1.32*
Siri Prieto et al. (2002)	Alabama	98 yrs, cotton with crimson clover CC, no added N fertilizer	2	0.34
Sainju et al. (2002)	Georgia	5 or 6 yr studies, CC stored more soil C under NT	12	1.38
Kaspar et al. (2006)	Iowa; small grain CC	NT corn-soybean rotation, 6 yrs	3	-0.19
Sainju et al. (2006b)	Georgia	7 yrs	6	1.27
Teasdale et al. (2007)	Maryland; corn, soybean, and wheat rotation; hairy vetch and rye CCs	8 yrs, 30 cm depth, NT	0	Increased SOC concentrations[b]
Veenstra et al. (2007)	California, cereal-legume mix	5 yrs, conservation tillage and conventional tillage	2	3.24
Senthilkumar et al. (2009b)	Michigan, corn-soybean-wheat with legume winter CC	Organic CT system with cover crops versus conventional CT system without cover, 18 yrs, different field positions, 40 cm depth	2	1.83
Senthilkumar et al. (2009a)	Michigan, corn-soybean-wheat with legume winter CC	Organic CT system with cover crops versus conventional CT system without cover, 18 yrs, 15 cm depth	1	0.56
De Gryze et al. (2009)	California, legume winter CC	9–11 yrs	3	2.17

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.
b. 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.

Adoption of winter cover crops may necessitate additional field operations, and consequent increases in fuel-source GHG emissions (Paustian et al. 2004), although the resulting GHG impacts are likely quite small in comparison to fertilizer N savings.[22] If the inclusion of cover crops necessitates earlier grain harvest and increased grain drying, the fuel-related increase in emissions may be significant,[23] although this increase has not been quantified. Other changes to the main crop must also be considered, especially if they would affect 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.

22. For example, 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 the additional field operations (assuming planting and cultivation similar to that of wheat for grain) would increase process emissions by 0.13 t CO_2e ha^{-1} yr^{-1}.
23. D. Miller, personal communication, April 2010.

Crop Rotation Changes

Either diversification—replacing the main crop with a different crop for one or more seasons—or intensification—adding another crop to the annual (or biennial, triennial, and so on) cycle to increase the number of days during which crops are growing—can be used to increase total productivity or otherwise reduce GHG emissions from annual crop rotations. Improved crop varieties from crop breeding programs and biotechnology may make such adaptations more feasible. For example, crops with shorter growing-season requirements may make intensification or cover crops more feasible or may provide more flexibility with regard to planting and harvest time.

GHG Impact Summary

GHG category	Increase cropping intensity	Diversify annual crop rotations
Number of observations	**none**	**88**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.00** (-1.69–1.66)
N_2O, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.17**
CH_4, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.00**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.00**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.17** (-1.52–1.83)
Maximum U.S. applicable area, Mha	**unknown**	**46**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Increase cropping intensity

Most research on increased cropping intensity relates to fallow reductions and winter cover crops (Liebig et al. 2010a; Ogle et al. 2005; Peterson et al. 1998; Sherrod et al. 2003), but in some more temperate regions of the United States, double- and triple-cropping are being explored for productivity gains, additional nutrient utilization (especially in the case of N in manure), and soil C sequestration. In most cases, intensification of annual crop rotations is combined with diversification, because growing only a second (or third) crop with shorter growing-season needs or growth requirements otherwise different than those of the main summer crop is most feasible. The shorter total fallow (nongrowing) period can lead to increased biomass inputs and reduced decomposition rates (Ogle et al. 2005),[24] with positive soil C implications. 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). 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, side-by-side comparison data of soil C response to simple intensification are not available, so estimation of GHG impacts is difficult.

Diversify annual crop rotations

Crop species can vary significantly in growth patterns, biomass production, water requirements, and decomposition rates, all of which affect net GHG emissions. Therefore, many rotations could be adapted with alternative species or varieties of annual crops to promote soil C sequestration—increasing root and residue biomass, increasing root exudates, or slowing decomposition—or otherwise reduce emissions (Table 4). Total GHG impacts of crop rotations are dominated by soil C, which is affected by both total amount and quality of the crop residue and root biomass. For example, the SOC impact of vegetables = cotton = tobacco ≤ flax < wheat = lentil < fall rye ≤ hay (Hutchinson et al. 2007; Ogle et al. 2005).

Field studies demonstrate that although certain rotations can sequester carbon, the soil C response to diversification is highly variable. Nearly 90 comparisons yielded an average soil C change near zero, although for rotations other than corn-soybean, diversification from a monocrop results in an average gain of about 0.1 t CO_2e ha^{-1} yr^{-1}. Some reductions in N_2O emissions are anticipated, and the average net GHG mitigation potential for all diversification is calculated as 0.2 t CO_2e ha^{-1} yr^{-1}. In three states (Iowa, Illinois, and Nebraska) where data are available, 2% to 5% of the total crop area is in continuous corn, and 8% to 12% is planted to corn every 4 of 5 years (Boryan et al. 2009).[25] We estimate that between 25% and 50% of U.S. annual crop production area could be further diversified (~46 Mha), with possible soil C sequestration and N_2O emission reductions. However, other environmental and productivity issues (e.g., weeds, diseases) may provide the greatest incentive for diversification—GHG mitigation may be considered a side benefit rather than the primary driver for change.

24. Slowed decomposition rates with intensification may be a result of reduced soil water content caused by increased evapotranspiration.
25. Certain counties have up to 38% of total cropland area in continuous corn cropping systems.

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

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Franzluebbers and Follett (2005)	North American review, more complex rotations	4 regions, no individual data points given	Expert estimate	*Low: 0.44* *High: 1.06*
West and Post (2002)	Mostly U.S. grain systems	Review, continuous corn to corn-soybean Review, other than corn to corn-soybean	14 48	-0.58 -0.20
Johnson et al. (2005)	Midwestern United States, grain systems	Review	4*	0.73
Franzluebbers et al. (1998)	Texas, wheat-soybean and sorghum-wheat-soybean vs. continuous	9-yr study	8	1.62
Sainju et al. (2006a)	Montana, wheat system	6-yr study	CT: 2 NT: 2	CT: 1.37 NT: -1.53
Varvel (2006)	Nebraska, corn-based rotations	2-yr rotation versus continuous corn 4-yr versus 2-yr rotations – the sequestration rate was highest at 10 yrs, and slowed afterward	1 3	0.00 1.06
Omonode et al. (2007)	Indiana, corn-soybean vs. continuous corn	Measured CO_2 flux	1	0.90
Khan et al. (2007)	Illinois, corn-oats (corn-soybean since 1957) vs. continuous corn	Morrow Plots (est. 1876), 79-yr study, 3 fertilizer-level treatments	3	0.40
Alluvione et al. (2009)	Colorado, semi-arid irrigated, add barley or dry bean to corn	Measured CO_2 flux; compared with continuous corn cropping	1 1	Barley: -0.11 Dry bean: 0.25

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Crop rotation diversification most often involves moving from a continuously cropped cereal to multiple crops within a rotation. In general, crops with greater biomass production (see Table 5 for select crop residue yields) have more soil C sequestration potential, but other factors also affect this relationship. For example, a 79-year comparison of corn-oats rotation with continuous corn cropping revealed that the former yielded greater soil C than than the latter even though oats produce significantly less biomass than corn (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), two-year rotations (corn-soybean and sorghum-soybean) were shown to offer no SOC benefit over continuous monocropping, but four-year rotations with oats and clover increased SOC content by 12.4, 16.8, and 17.7 t CO_2 ha^{-1} after 10, 16, and 20 years (an 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.

As with other agricultural activities, net greenhouse gases are also affected by interactions with other land management practices. Within NT cropping systems, diversified crop rotations yield SOC increases up to 0.75 t CO_2 ha^{-1} yr^{-1}; very little SOC impact has been observed when these cropping systems are under conventional tillage (Franzluebbers 2010; West and Post 2002). Because the effects of annual crop-rotation changes on SOC may be small relative to the SOC effects of other management changes, it may take up to eight years or more for standard sampling and analytical approaches to reveal them (Alluvione et al. 2009; Sainju et al. 2006a).

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

Crop	Residue yield (t/ha)	2001 U.S. residue production (Mt/yr)[a]
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]	80.0
Rice	6.7[a]	14.6
Cotton	6.7[a]	16.8
Sugarbeet	5.6[a]	5.9

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

Changes in annual crop rotations tend to have insignificant or minimal impact on nitrous oxide and methane in most experiments (Alluvione et al. 2009; Johnson et al. 2010; Rochette et al. 2004; Venterea et al. 2010). For example, in one case in which SOC increased 0.25 t CO_2e ha^{-1} yr^{-1} in a corn-dry bean rotation (when compared with continuous corn cropping), higher N_2O emissions erased only 16% (0.04 t CO_2e ha^{-1} yr^{-1}) of that gain (Halvorson et al. 2008b). In corn-soybean rotations, which may reduce soil C when compared with continuous corn cropping,[26] such losses are offset by lower N_2O emissions of between 0.03 and 0.56 t CO_2e ha^{-1} yr^{-1} (MacKenzie et al. 1998; Omonode et al. 2011; Venterea et al. 2010). Process and upstream emissions would be little affected by most crop-rotation adjustments, assuming similar fertilizer application and field operations.

Include perennials in crop rotations

Incorporating one to three years of a perennial crop (often alfalfa or grass hay) into an annual crop rotation both intensifies and diversifies the rotation and can also sequester soil C; however, separating the impact of crop changes from tillage-reduction effects may be difficult.[27] On the basis of 28 observations (Table 6), this review estimates that incorporating one to three years of a perennial crop such as alfalfa or grass hay into annual crop rotations captures soil C at an average rate of 0.5 t CO_2 ha^{-1} yr^{-1} (a range of 0 to 1.2 t CO_2 ha^{-1} yr^{-1}). Including perennials in crop rotations reduces N fertilizer needs, field operations, and N_2O emissions, resulting in an estimated net GHG mitigation of 0.7 t CO_2e ha^{-1} yr^{-1}. Because U.S. data are somewhat limited, these estimates are supplemented by research from Canada (e.g., Gregorich et al. 2001; Hutchinson et al. 2007; VandenBygaart et al. 2003). Up to 56 Mha of U.S. land (the moist regions) is estimated to be available for incorporating perennials into existing crop rotations.

GHG Impact Summary

GHG category	Include perennials in crop rotations	Replace annuals with perennial crops
Number of observations	**28**	**17**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**0.52** (-0.01–1.20)	**0.67** (-0.86–2.00)
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.03**	**0.24**
CH_4, t CO_2e ha^{-1} yr^{-1}	**0.00**	**0.00**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.17**	**0.52**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.71** (0.19–1.39)	**1.43** (-0.10–2.76)
Maximum U.S. applicable area, Mha	**56**	**13**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Table 6. Estimates of soil C sequestration potential of including perennials in crop rotations, U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
West and Post (2002)	U.S. data drawn from larger review, corn rotations	Review	6	0.21
VandenBygaart et al. (2003)	Canadian prairies, hay in fallow-wheat rotation	Review	10	0.44
Robinson et al. (1996)	Iowa, corn rotations	Study reported soil C concentration only; Johnson et al. (2005) calculated mass	5	0.40
Campbell et al. (2000)	Canada, hay in wheat rotation	Soil with lower SOC at time zero gained C at a higher rate	1	0.60
Lal et al. (1994)	Ohio, includes hay in rotation	19-yr study; moldboard plow[b]	1*	1.32
Gregorich et al. (2001)	Ontario, monoculture corn vs. corn-oats-alfalfa-alfalfa rotation	35-yr study	1*	2.03
Khan et al. (2007)	Illinois, includes hay in corn-oats rotation	Morrow Plots (est. 1876); 79-yr study	4	0.89

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.
b. In this study, soil C was also monitored when perennials were included in NT and chisel plow systems, with negative soil C response. However, these were outliers in the current analysis, so not included in this table.

Compared with annual crops, perennials (especially grasses) tend to allocate a relatively high proportion of carbon underground and to have a greater number of days per year of active plant primary productivity, resulting in increased

26. This review's compilation of 20 comparisons of corn-soybean versus continuous corn cropping systems reveals high variability in soil C response; diversification of this type yields an average decrease of 0.21 t CO_2e ha^{-1} yr^{-1} (a range of -3.5 to 2.9 t CO_2e ha^{-1} yr^{-1}).
27. Perennial crops most often have lower tillage requirements, because the need for seedbed preparation is dramatically lowered, and management generally involves no growing-season tillage for weed control.

potential biomass production and SOC storage. Perennials can also generate more total evapotranspiration, drying soils and lowering soil C decomposition rates (Paustian et al. 2000). In the long run, this greater evapotranspiration may become problematic in dry climates with rain-fed agriculture, because high water demand could lead to low-yielding annual crops in successive seasons (Paustian et al. 1997; Paustian et al. 2000). With irrigated cropland, the impact of perennials on water requirements (and associated energy and greenhouse gases) must also be considered.

In general, altered crop rotations have a limited effect on N_2O and CH_4 fluxes (Johnson et al. 2010; Omonode et al. 2007). However, with perennial crops, the increases in plant cover (and deeper root development) over a longer period of time throughout the year will lead to mineral N scavenging and reduce N losses, including losses via N_2O emissions (Delgado et al. 2007; Robertson et al. 2000).

In contrast to annual crops, perennial crops have similar or lower fertilizer N requirements; legumes, in particular, not only require less N fertilizer but also tend to reduce N_2O emissions. 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. Alfalfa and soybeans emitted an average of 0.48% ± 0.33% and 0.39% ± 0.27%, respectively, of fixed N as N_2O, significantly lower than the IPCC Tier I factor used for fertilizer and other N additions, which is 1.25% (Rochette et al. 2004). In this experiment, even though legume crops were associated with higher soil mineral N concentrations than timothy grass, the N_2O emissions of these crops and the grass were similar.

In a perennial-annual crop rotation, the seed-bed preparation and fuel requirements for harvest of the perennial crop are lower than those of the annual crop. For example, California cost studies find that fuel costs for grain corn are three times that for alfalfa hay (Frate et al. 2008; Mueller et al. 2008). Hence, the process-related GHG emissions of the perennial crop portion of the rotation are lower than those of the annual crop portion.

Replace annuals with perennial crops

On the basis of 17 observations, the average soil C sequestration potential of fully replacing annuals with perennials (Table 7) is estimated to be 0.7 t CO_2 ha^{-1} yr^{-1} (a range of -0.8 to 2.0 t CO_2 ha^{-1} yr^{-1}), nearly double the soil C sequestration rate obtained when perennials are included as only part of a rotation. Results vary depending on the crop type and other factors. By lessening N fertilizer and field operation requirements and reducing N_2O emissions, conversion to perennial crops is estimated to yield a net GHG mitigation potential of 1.4 t CO_2e ha^{-1} yr^{-1}. These estimates are based on conversion to cropped perennials, such as alfalfa or grass forages, or to biofuel grasses; perennial plantings that involve land-use change (setting land aside from agriculture, switching from cropland to grazing land, and introducing short-rotation woody perennials) are discussed in separate sections below.

Table 7. Estimates of soil C sequestration potential of replacing annuals with perennial crops (not including grazing land), U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Freibauer et al. (2004)	Europe, perennial grasses and permanent crops	Review, no separate data points	n/a	*2.20*
Liebig et al. (2005a)	Great Plains and northern Corn Belt, switchgrass versus cultivated crops	42 obs, individual data not available	n/a	*4.67*
Lemus and Lal (2005)	U.S.-wide, switchgrass	Review	Expert estimate	*2.93*
VandenBygaart et al. (2003)	Canadian prairies, crested wheat grass	Review	5	0.43
Franzluebbers (2010)	Cropland to grassland	Review	8	1.47
Grandy and Robertson (2007)	Michigan, alfalfa versus corn-wheat-soybean	Conventional tillage	1	1.04
Potter and Derner (2006)	Restored grassland (for hay) versus continued cropping	Texas	3	0.00

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Previous reviews have determined that N_2O emissions from perennial grassland are much lower than those from annual crops (Grant et al. 2004; Machefert et al. 2002; Smith et al. 2008). However, at similar levels of N fertilizer input, emissions from grass and cereals do not appear to be significantly different (Stehfest and Bouwman 2006). Fossil fuel offset of biofuels made from perennial crops may offer further GHG mitigation potential, but due to high variability and policy uncertainty, that potential is not included in the estimates in this review. Significant conversion to perennial crops could decrease total commodity volume and farm income. Therefore, it may be more costly than activities that maintain approximately similar crop mixes.

Lemus and Lal (2005) estimated that 13 Mha are potentially available for transition to biofuels cropland in the United States over the next 50 years; this area is the maximum area assumed to be available for transition from annual to perennial crops.

Switch to Short-Rotation Woody Crops

Although most tree plantings 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—their planting is included in this assessment as an agricultural land management practice. As a GHG mitigation activity, production of SRWCs may be more attractive to farmers than longer-term forestry options, because the short rotation of the crops makes this production “feel” more agricultural.

GHG Impact Summary

GHG category	Switch to short-rotation woody crops	Establish agroforestry
Number of observations	**35**	**3/3/0**[a]
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**2.51** (-7.34–13.26)	**0.84–6.87**
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.76**	—
CH_4, t CO_2e ha^{-1} yr^{-1}	**no data**	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.65**	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**3.92** (-5.93–14.67)	—
Maximum U.S. applicable area, Mha	**40**	**21**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.
a. For activities with < 9 comparisons, number of observations indicates “field comparisons/expert estimates/model estimates.”

On the basis of data from 35 field-based observations (Table 8), SRWCs are estimated to sequester soil C at an average rate of 2.5 t CO_2 ha^{-1} yr^{-1} (a range of -7.3 to 13.3 t CO_2 ha^{-1} yr^{-1}). On cropland, SRWCs could also generate substantial reductions in fertilizer and fuel use and could reduce N_2O emissions, for a net GHG mitigation potential of 3.9 t CO_2e ha^{-1} yr^{-1}. As much as 40 Mha of highly eroded, degraded, or mining lands could be planted to SRWCs with limited negative impact on the production of key food and fiber crops (Tuskan and Walsh 2001).

Table 8. Soil C sequestration physical potential of planting short-rotation woody crops, U.S.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Schlamadinger and Marland (1996)		Modeled, 7-yr rotation	Modeled	*0.66*
Heller et al. (2003)	New York, willow	2- to 12-yr old willow chronosequence; assumes no increase	Expert estimate	*0.00*
Tuskan and Walsh (2001)	United States, various species	Modeled, suggests applicability to 40 Mha	Modeled	*6.60*
Nabuurs and Mohren (1993)	Southeastern United States, productive, fast-growth forests	Modeled with CO_2FIX, 45-yr poplar rotation, 30-yr loblolly pine rotation	Modeled	*Poplar: 5.46* *Pine: 10.63*
Wright and Hughes (1993)	North Central United States, various SRWCs	Modeled, estimated 14–28 Mha of cropland available for energy crops	Modeled	*1.10*
Sartori et al. (2006)	Various species	Review, 3–18 yrs	3	2.38
Hansen (1993)	Midwest, hybrid poplar	12- to 18-yr old stands	8*	3.11
Coleman et al. (2004)	Midwest, poplar	Oldest poplar stand was 12 yrs, soil C decrease in younger stands	23*	2.40
Grandy and Robertson (2007)	Michigan, poplar	12 yrs old	1	0.70

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

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). The primary carbon storage in woody biomass plantations is in aboveground material (Ranney et al. 1991), although end use, which could be pulp/paper or bioenergy production, essentially determines whether and how the aboveground biomass is counted in the GHG balance. As a conservative estimate, this review assumes no GHG benefit for this biomass, limiting its focus to soil C. On average, the soil C sequestration potential of SRWCs is greater than that of management options that maintain annual crop species, although the estimates are highly variable and affected by species, climate, and other factors. In some cases, soil C decreases during the initial years of SRWC establishment and increases only later (Grigal and Berguson 1998; Hansen 1993).

Nitrous oxide and methane flux effects of SRWCs are unclear. In a review of these effects across Europe, Machefert et al. (2002) noted much lower N_2O emissions in forested versus agricultural land, but other researchers have found little difference in the N_2O emissions of annual crops versus those of poplar plantations (Scheer et al. 2008). Therefore, although the estimate in this review assumes some N_2O emissions reduction because of the lower fertilizer N application requirements and the nutrient scavenging capability of SRWCs, this estimate is somewhat tentative.

In addition to sequestering soil C, SRWCs could displace fossil fuel if used for bioenergy production, but, depending on the accounting measure, this activity may only be counted as mitigation if the carbon absorbed by the plants is "additional" to that which would otherwise be absorbed (Searchinger 2010). The estimated bioenergy displacement of fossil fuels from SRWCs could be as much as 18–20 t CO_2e ha^{-1} yr^{-1} (Graham et al. 1992; Tuskan and Walsh 2001). On the other hand, when current cropland is converted to SRWCs, indirect land-use change impacts (i.e., leakage) may limit real GHG mitigation potential as crop production moves to other land currently in perennial crops, grassland, or forest production.

Establish Agroforestry

Although agroforestry is most commonly implemented in the tropics—with high C sequestration potential when compared with other agricultural land uses—it is 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). On current U.S. cropland, agroforestry could entail alley cropping, windbreaks, or riparian buffers. Experts estimate agroforestry's soil C sequestration potential on this cropland at 0.8 to 6.9 t CO_2 ha^{-1} yr^{-1} (Table 9).[28] This potential varies widely, depending on the specific practice, individual site characteristics, and time frame. As with SRWCs, aboveground biomasss can also comprise a large C pool, but because the net effect depends on the end use, it is not included in this assessment. Non-CO_2 gas fluxes, process emissions, and N fertilizer effects are likely similar to those for SRWCs.

Table 9. Estimates of soil C sequestration potential of establishing agroforestry, U.S.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Nair and Nair (2003)	Alley cropping Riparian buffers Windbreaks	Nationwide; SOC estimated as 25% of total C stored	Expert estimate	4.23 6.87 3.45
Lal et al. (2003)	Alley cropping Windbreaks	Nationwide; only soil C	Expert estimate	4.22 0.84
Dixon et al. (1994)	Agroforestry	Nationwide; SOC estimated as 25% of total C stored	Expert estimate	2.64
Bailey et al. (2009)	Missouri, corn-soybean with tree-grass buffer	13-yr field study	Unknown	1.56

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Nair and Nair (2003) estimated that as much as 80 Mha of U.S. land could accommodate alley cropping (20% of land in trees), 85 Mha of land could utilize windbreaks (5% of land in trees), and 0.8 Mha of land could be planted in forested riparian buffers. The total land area under trees alone is estimated to be 21 Mha, and soil C sequestration potential estimates assume that the adjoining crop area is unaffected.[29] As with SRWCs, indirect land-use change impacts (leakage) may significantly decrease the net GHG mitigation potential.

While agroforestry can provide water quality and habitat benefits, competition for light, nutrients, and water can make tree systems undesirable near cropland. Some direct competition between trees and crops can be addressed by retaining

28. Mitigation potential is reported as a range of all available field observation data, model estimates, and expert estimates for agroforestry and other activities for which data are lacking or GHG mitigation potential is low.

29. Data on adjacent cropland area effects are few, and research appears to indicate that these effects would be overshadowed by impacts on soil C and other emissions in the area directly affected by trees. The exception is in silvopasture, where data reporting soil C effects refer to the entire area, because trees and pasture are highly integrated.

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 with this strategy, some water competition between the grass strip and the crop may exist (see, for example, Falloon et al. 2004).

Apply Organic Material (e.g., Manure)

The United States 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). Livestock manure is the material most commonly applied to agricultural lands. In 2007, approximately 9 Mha of U.S. cropland—less than 8% of total U.S. cropland—were treated with manure fertilizers (USDA NASS 2007b). Most of this land was in corn production (USDA ERS 2009).

Several factors have led to shifts from organic fertilizers to chemical alternatives; these factors include decreases in the cost of inorganic fertilizer, increases in average farm size and specialization, adoption of confined animal feeding operations, and policy and government incentives aimed at crop yield increases per land unit (Chesworth 2008). High nutrient variability in manure makes efficient nutrient management more complex than nutrient management with commercial fertilizer. Nevertheless, nutrient benefits and fertilizer savings, combined with GHG mitigation potential, are renewing interest in the use of organic soil amendments—manure and compost in particular.

GHG Impact Summary

GHG category	Apply organic material (e.g., manure)	Apply biochar to cropland
Number of observations	**28/1/2[a]**	**0/5/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**0.18–5.10**	**0.63–19.57**
N_2O, t CO_2e ha^{-1} yr^{-1}	—	—
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**8.5**	**124**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.
a. Although the number of observations exceeds nine, life-cycle concerns about this activity make calculation of average GHG mitigation potential unrealistic.

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

Organic material	Organic materials production (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. These studies indicate an average soil C sequestration potential of 0.2 to 5.1 t CO_2e ha^{-1} yr^{-1} (Table 11), a potential often greater than that of tillage changes or use of winter cover crops (e.g., Buyanovsky and Wagner 1998; Franzluebbers 2005; Sainju et al. 2008a). However, estimating a potential sequestration rate for manure application per unit of area can be difficult, because the main limiting factor is not the area available, but the amount of manure and other organic materials available. Lal et al. (1999a; 1999b) rightly address this issue by estimating a total national potential for soil C sequestration, but not translating that potential into a per ha estimate.

Experiments to test the soil C changes with manure application also do not address the related soil C impact on land that may no longer be receiving the manure application. Hence, the life-cycle GHG mitigation potential 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 the soil C increase occurs in an alternate location, the net change in soil C over the whole system is unchanged. Therefore, a full life-cycle analysis of this activity is especially important, and improved nutrient distribution (with air and water quality benefits) might provide a greater incentive for manure application adjustments than would GHG mitigation.

Soil C sequestration rates following manure application are lower in warm climates (7% ± 5% of applied manure C retained in soil) than those in cool climates (23% ± 15% of applied C retained)[30] (Risse et al. 2006). However, soil moisture appears 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 affected by differential application rates. If so, GHG mitigation would be maximized at the application rate at which the greatest proportion of manure C is retained in the soil. When Chang et al. (1991) compared three levels of cattle feedlot manure application on two types of cropland (30, 60, and 90 Mg manure ha^{-1} yr^{-1} on dry land and 60, 120, and 180 Mg manure ha^{-1} yr^{-1} on irrigated land), they found that soil C increased by 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) presented model evidence of higher organic matter decay rates (soil organic N) on the sites that received the highest manure application. Angers and N'Dayegamiye (1991) found that application of 40 Mg ha^{-1} every two years resulted in soil retention of a greater proportion of manure C than application of 80 Mg ha^{-1} every two years.[31] 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); higher application rates were needed to maintain soil C levels.

Because the majority of manure is already land applied, the total amount applied in excess—i.e., applied at rates higher than crop nutrient needs—and thus available for wider distribution must be estimated. For effective nutrient management, manure application rates should be based on either N or phosphorus (P) crop needs and manure nutrient content. A 2001 USDA report indicated that, on average in the United States, 60% of manure N and 70% of manure P was applied in excess of the optimal application rate for the originating farm (Gollehon et al. 2001) and thus would be available for other land. The total amount of manure N and P generated is approximately 1.1 Mt and 0.6 Mt, respectively, and most excess is produced on the 2% of farms in the largest farm-size class. If N were the main limiting factor, and assuming a national average N application rate of 105 kg N/ha, an additional 6.5 Mha could receive manure fertilizer, replacing commercial N sources. If P were the main limiting factor, and the average P application rate is 40 kg P/ha, an additional 10.5 Mha could receive manure fertilizer, replacing other P fertilizer sources. If the potential area falls somewhere in between, approximately 8.5 Mha of additional cropland could receive the excess manure.

Table 11. Estimates of soil C sequestration potential following land application of organic material, U.S.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Follett (2001)	United States	Apply 250 kg N ha^{-1} yr^{-1} equivalent of livestock manure where economically feasible	Expert estimate	Low: 0.73 High: 1.84
Li (1995)	6 U.S. sites: Iowa, Illinois, Kansas, Nebraska, California, and Florida	Livestock manure; DNDC model; 1000 kg C ha^{-1} yr^{-1} applied; sequestration rates ~ double with 2000 kg C ha^{-1} yr^{-1} applied	Modeled	Low: 1.90 High: 3.50
Collins et al. (1992)	Oregon	Livestock manure; 56-yr study	4	0.70
Kingery et al. (1994)	Alabama	Poultry litter, 21 ± 4 yrs on different study sites	3	1.10
Buyanovsky and Wagner (1998)	Missouri	Livestock manure, 100 yrs, Sanborn field	4 4	Wheat: 1.21 Maize: 1.95
Drinkwater et al. (1998)	Pennsylvania	Manure system received less crop residue	1	2.56
Franzluebbers (2005)	Southeastern U.S.	Poultry litter; 5–21 yrs, range in C sequestration of 17% ± 15%	19	2.64
Sainju et al. (2008a)	Alabama	Poultry litter, 10 yrs	1	1.87

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Approximately 80% of excess manure N (77% of P) in the United States could be utilized within the county of origin (Gollehon et al. 2001). Transporting manure—from short distances of 15 km 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)—has been shown to be economically feasible. The GHG impacts of transport require further evaluation.[32]

30. 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.
31. Ten years 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}.
32. The average estimate of 0.57 t CO_2e ha^{-1} yr^{-1} in transport-related emissions assumes a transport distance of 100 km, a load of 24.5 t, and 380 g C km^{-1} emissions (emission values from Smith and Smith 2000).

If manure application displaces commercial fertilizer, the upstream, process, and transport emissions not incurred by use of the commercial fertilizer should also be considered.

In practice, manure application can, but does not necessarily, lead to full displacement of commercial fertilizer. In a USDA ERS survey (2009), 61% of corn farmers reported cutting their commercial N applications when applying manure (an average reduction of 58%).[33] Only 35% of oats farmers and 29% of soybean farmers reduced their chemical N applications (by 76% and 85%, respectively), but these data reflect no correction for the possibility that some of these producers had not been using commercial fertilizer and so had nothing to reduce (USDA ERS 2009).

The impacts on CH_4 and N_2O flux of organic matter additions to soil are highly variable. Nitrous oxide emissions are positively correlated with native soil C content, because carbon supports microbial activity, including the processes that produce N_2O (Rochette et al. 2000). But these emissions tend to be negatively related to the C content of the manure or other organic source, because the added carbon causes the microbial community to immobilize available nitrogen (Gregorich et al. 2005). Where manure can replace N fertilizer as the main N source, N_2O emissions tend to be lower (Alluvione et al. 2010). However, whether these emissions actually are lower depends on whether they are limited by available mineral N or by a carbon source for the microbes. Chantigny et al. (2010) found that manure application led to lower N_2O emissions in clay soil, but higher emissions in loam soil, when compared with N fertilizer application. In the loam soil, the carbon in the manure provided the substrate for denitrifying bacteria. Another important GHG-related consideration is that more frequent land application of manure can significantly reduce CH_4 emissions because of shorter storage times in anaerobic lagoons or stockpiles (Johnson et al. 2007).

Compost: Net GHG impacts

Compost application on agricultural soils can reduce net GHG emissions in two ways. First, it can displace more typical anaerobic storage options with aerobic decomposition of organic material, reducing CH_4 emissions. Second, it 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.

When livestock farm systems are producing organic nutrients in excess of crop needs on the receiving land, the overage can be applied to other cropland as organic soil amendments, thereby increasing soil C sequestration and requiring less N fertilizer, which could lead 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 a significant source of organic material in the United States. Therefore, to examine the GHG mitigation potential of compost application, the following paragraphs compare the net GHG impacts of direct application of manure (from typical storage conditions) with composting of the manure prior to land application.

Several potential GHG benefits are associated with composting of manure prior to land application. By stabilizing manure's organic matter through a largely aerobic process, composting can generate much lower net GHG emissions during the storage period and after land application than standard anaerobic manure storage in stockpiles or manure storage lagoons. Pattey et al. (2005) found that, compared with untreated manure storage, composting reduced total GHG (CH_4 plus N_2O) emissions prior to land application by 31% to 78%, depending on the C:N ratio, moisture content, and aeration status. The impact of composting on emissions after land application is of further interest. Fronning et al. (2008) examined GHG fluxes following land application of solid beef manure and composting of 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 t CO_2e ha^{-1} yr^{-1} 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 stabilization of organic matter during the compost process reduces post-application respiration losses. However, the net C sequestration difference between untreated manure and composted manure may also be affected by respiration losses during the composting process, which were not recorded in the above-mentioned study. Further research may be needed to address these life-cycle issues.

Apply Biochar to Cropland

Biochar is produced by pyrolysis, the incomplete combustion of biomass into charred organic matter. The pyrolysis process can capture heat and co-generate electricity as a biofuel (with some GHG mitigation benefits), but the end product can also be applied to soil, potentially increasing soil C by (1) storing recalcitrant C in biochar soil amendments, (2) stabilizing existing C in the soil, and (3) increasing biomass production above ground, thereby increasing C inputs into soil (Gaunt and Driver 2010). Research suggests that this black carbon, or *terra preta*, likely 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). Estimates of the soil C sequestration potential of biochar application in the United

33. This somewhat low fertilizer-adjustment response to application of manure nutrients may in some cases be influenced by the need to determine manure application rates on the basis of phosphorus (P) rather than nitrogen content. In many cases, when manure is applied according to crop P needs, the N in that manure is insufficient for the crop (because manure loses a greater proportion of N compared with P during storage and hauling and after field application).

States range from 0.6 to 19.6 t CO_2e ha^{-1} yr^{-1} (Table 12) and have been based on calculations of available feedstock and expected C stability in the biochar (e.g., Gaunt and Lehmann 2008; Lehmann 2007; Roberts et al. 2010). If sufficient material were available, all U.S. cropland (124 Mha) could be available for biochar application. However, with high variability in quality and decomposition rates, affected by feedstock and other factors, and uncertain technical details for large-scale implementation, further research is needed to provide proof of biochar application's environmental benefit. Other GHG benefits (N_2O emissions reduction or improved fertilizer N use efficiency) and productivity gains may provide additional mitigation potential, but given variability in soil C implications (the main target), these effects are not quantified here.

Possible biomass sources for biochar 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 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 have equivalent rates of C storage or stabilization, which are dependent on feedstock source and on temperature, rate, and residence time of the pyrolysis process (Gaunt and Driver 2010).

The response of soil to biochar amendments is expected to be biochar- and ecosystem-specific (Shneour 1966; Spokas and Reicosky 2009). When most plant biomass is decomposed, less than 10% to 20% of the original C remains after 5 to 10 years (Lehmann et al. 2006). By comparison, biochar tends to be highly stable; the mean residence time is hundreds to thousands of years (Lehmann et al. 2008; Roberts et al. 2010; Verheijen et al. 2009). Assuming that biochar production retains up to 50% of biomass C as a stable residue, Lehmann et al. (2006) estimated that as much as 512 t CO_2e ha^{-1} could be stored under typical soil and plant species conditions over a long period. However, as with other organic material, biochar decay is facilitated by decomposition, microbial co-metabolism, abiotic processes, and physical breakdown. This process is influenced by biochar characteristics, temperature, depth of burial, and soil cultivation (De Gryze et al. 2010). The complex interactions among these factors have not been studied extensively; therefore, biochar recalcitrance remains widely variable in the literature.

Beyond sequestration, biochar may have potential to mitigate greenhouse gases by decreasing the need for fertilizer, lime, or other inputs, thereby reducing upstream and field emissions (Gaunt and Driver 2010; Lehmann et al. 2006). Lower N_2O and CH_4 field emissions following biochar application may be related to production of ethylene, which inhibits microbial processes (Spokas et al. 2010). Little research has documented suppression of nitrous oxide in the field (Taghizadeh-Toosi et al. 2011), although many short-term studies and laboratory experiments have noted N_2O emissions reductions of 50% to 80% and nearly complete suppression of methane with biochar additions (Fowles 2007; Lehmann et al. 2006; Renner 2007; Rogovska et al. 2008; Yanai et al. 2007). Yet, in another laboratory experiment, Yanai et al. (2007) found that the impact of biochar on N_2O emissions was highly dependent on soil hydrology; N_2O emission effects varied from an 89% reduction in very wet soil to a 51% increase in drier soil. Clough et al. (2010) and Singh et al. (2010) reported in separate studies that N_2O emissions were initially higher in biochar-amended soils but that after a period of time they were lower. Therefore, biochar absorption capacity may be enhanced with aging (Singh et al. 2010).

Table 12. Estimates of soil C sequestration potential of biochar application on U.S. cropland

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Lehmann (2007)	United States, crop residue	Estimated 5.5 t residue ha^{-1} yr^{-1} on 120 Mha	Expert estimates	4.89
	United States, fast-growth vegetation	Estimated 20 t biomass ha^{-1} yr^{-1} on 30 Mha of idle farmland		19.57
Laird (2008)	United States, harvestable forest and croplands	Assumes the United Statescan sustainably produce 1,100 Mt biomass yr^{-1} from forest and cropland (10% moisture), 509 Mt CO_2e yr^{-1}, no area estimate	Expert estimate	n/a
Gaunt and Lehmann (2008)	United Kingdom, switchgrass, miscanthus, corn stover	Estimates for slow pyrolysis; corn stover (A) and bioenergy crop (B)	Expert estimates	A: 4.61 B: 8.92
Roberts et al. (2010)	United States, unused crop residue	141.1 Mt of unused crop residue, 0.53–0.57 t CO_2e t^{-1} feedstock as sequestered soil C	Expert estimate	0.63

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Of perhaps greater importance than soil C or N_2O emission effects are the life-cycle GHG implications of biochar. For example, what are the soil C and other GHG effects of removing residue from the field to convert it to biochar? If the biomass source is also the receiving field, the effects of leaving residue on the field must be compared with those of removing the residue, converting it to biochar, and returning it to the field. If biomass is sourced from elsewhere, the GHG effects of that movement must be considered.

Biochar production both requires and produces energy, and full life-cycle assessments will consider these upstream and process emissions as well as fossil fuel displacement (if applicable). Roberts et al. (2010) calculate a net GHG emission reduction of 0.86 t CO_2e t^{-1} of corn stover feedstock. Similarly, McCarl et al. (2009) estimate a net mitigation potential of 0.82 t CO_2e t^{-1} of feedstock for fast pyrolysis and 1.11 t CO_2e t^{-1} of feedstock for slow pyrolysis, accounting for emissions from collection, hauling, pyrolysis, and nutrient replacement. 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. De Gryze et al. (2010) provide a detailed comparison of feedstock alternatives, pyrolysis methods, tradeoffs, and other costs of biochar production.

Histosol Management

Between 10 Mha and 15 Mha of U.S. land—mostly in Michigan, Wisconsin, Minnesota, California and Florida—are classified as histosols or organic soils (peat) (Lal et al. 2003; Morgan et al. 2010). About 7.5% of these soils (0.8 Mha)—half in California and Florida and the remainder mostly in the Lake States and the East Coast—have been drained for agriculture (Morgan et al. 2010). Histosols are a unique soil type, containing at least 20% to 30% organic matter—by mass—in at least the first 40 cm of depth from the surface. The organic material is most often *Sphagnum* moss. Many histosols are also wetlands or were wetlands until drainage for human uses; but some wetland soils are composed primarily of mineral material and thus are not histosols. In the context of this assessment, *wetland restoration* (treated in a separate section) refers to all nonhistosol water-influenced areas. In their natural state, histosols emit methane and sequester carbon in buried biomass, although net GHG flux varies. Organic soils that are drained for agriculture emit significant amounts of carbon dioxide and nitrous oxide but become CH_4 sinks (Elder and Lal 2008; Rochette et al. 2010), turning farmed histosols into a significant GHG source (Freibauer et al. 2004; Morgan et al. 2010).

GHG Impact Summary

GHG category	Set aside histosol cropland	Manage farmed histosols
Number of observations	**3/10/0**	**2/5/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**2.20–73.33**	**0.00–15.03**
N_2O, t CO_2e ha^{-1} yr^{-1}	—	—
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**0.8**	**0.8**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Setting aside histosol cropland (with associated restoration of the natural hydrologic cycle) has the potential to reduce CO_2 emissions by 2.2–73.3 t ha^{-1} yr^{-1} (Table 13), depending on practice, soil characteristics, and climate. U.S. estimates of current CO_2 flux from farmed histosols are available (Morgan et al. 2010), but much of the existing research on setting aside histosols was conducted in Europe. For example, in formerly forested organic soils in Finland, the difference between the CH_4 flux of cropped soils and that of soils abandoned for conservation is very small in comparison to the CO_2 and N_2O impacts; the total GHG benefit garnered by the set-aside land is 10.3 t CO_2e ha^{-1} yr^{-1} (Alm et al. 2007). Rochette et al. (2010) observed that organic soils in Canada exhibited GHG fluxes similar to those observed in Europe, making extension of the flux findings to the North American context reasonable.

The net CH_4 and N_2O land emissions following histosol set-aside are expected to be highly variable; some unfarmed organic soils are significant CH_4 sources (Morgan et al. 2010), but abandoned farmland has been found in some cases to be a CH_4 sink (Alm et al. 2007). Nitrous oxide emissions are most likely to decrease with conversion to grassland or natural ecosystems (Alm et al. 2007), but maintaining higher water tables to reduce CO_2 emissions will likely stimulate greater CH_4 emissions and perhaps greater N_2O emissions (Morgan et al. 2010). By eliminating field operations and fertilizer N application, the setting aside of histosols can reduce upstream and process emissions, with additional GHG benefits. But because agricultural production would likely shift elsewhere, the net benefit depends on emissions in other agriculturally productive regions.

Table 13. Estimates of GHG emissions effects of alternative histosol (organic soils) management

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Set aside histosol cropland				
Lal et al. (2003)	United States, restore organic soils	Assumes 19 Mha available for restoration	Expert estimate	2.20
Freibauer et al. (2004)	Europe, farmed organic soils	Convert to woodland Abandon for conservation Protect and restore	Review	3.48 8.06 16.85
Alm et al. (2007)	Finland, abandon for conservation	Reduce CO_2 and N_2O emissions, minimal increase in CH_4 emissions		CO_2: 8.84 N_2O: 1.34 CH_4: -0.16
Smith et al. (2008)	Global, restore organic soils	Cool-dry and cool-moist climates Warm-dry and warm-moist climates	Expert estimate	36.67 73.33
Morgan et al. (2010)	California and Florida	Summarized current CO_2 emissions rates, setting land aside could reduce or stop emissions	4 studies	41.49
Rochette et al. (2010)	Eastern Canada, farm organic soils	Eliminate current CO_2 emissions and reduce N_2O emissions	Field study	CO_2: 26.32 N_2O: 11.94
Manage farmed histosols				
Freibauer et al. (2004)	Europe, farmed organic soils	Switch from higher-tillage (e.g., potatoes) to lower-tillage crop Maintain shallow water table Convert cropland to grassland Avoid deep plowing Sheep grazing, undrained land	Review with no individual data	5.86 10.08 5.13 5.13 8.07
Alm et al. (2007)	Finland, convert cereal crop to grassland	10 to 35 yrs of treatment	5 study sites, no individual data	CO_2: 2.75 N_2O: 2.65 CH_4: -0.43
Elder and Lal (2008)	Ohio, intensively farmed histosol	Switch from conventional tillage to no tillage	Field study	Soil C: 0.00 N_2O: 28.61

Note: Unless otherwise indicated, estimates refer to reductions in CO_2 emissions (i.e., reduced soil C decomposition).
a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Where removing these sensitive soils from agricultural production is difficult, management options that reduce soil disturbance and avoid drainage can lead to GHG benefits. These options include reducing tillage (including deep plowing), switching to less intensively managed crops (vegetable crops are common in histosols), allowing a shallower water table, and converting from cropland to grassland. The high existing CO_2 emissions could be reduced by up to 15.0 t CO_2e ha^{-1} yr^{-1} (Alm et al. 2007; Freibauer et al. 2004). No-till management of histosols can also significantly reduce N_2O emissions when compared with conventional tillage on these soils (Alm et al. 2007; Elder and Lal 2008). However, further research is needed to confirm these effects.

Changes in Irrigation Practices

In 2007, 17% of U.S. cropland was irrigated (USDA NASS 2007b). The increased aboveground and belowground biomass production with irrigation can lead to soil C sequestration estimated at -0.6–2.8 t CO_2e ha^{-1} yr^{-1} (Table 14). Therefore, converting existing dryland agriculture to irrigated area has been proposed as a GHG mitigating activity. However, any increase in C storage—in products and soil—must be weighed against increased N_2O (and possibly CH_4) land emissions as well as the GHG impacts of increased energy use for irrigation.

By reducing soil aeration and stimulating microbial activity, irrigation increases the potential for N_2O emissions. Bremer (2006) measured increased N_2O flux of 0.05 t CO_2e ha^{-1} yr^{-1} on irrigated versus non-irrigated turf grass in Kansas. Liebig et al. (2005b) summarized N_2O flux measurements from

GHG Impact Summary

GHG category	Convert dry land to irrigated land	Improve irrigation management
Number of observations	**11/2/0**	**4/1/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**-0.55–2.82**	—
N_2O, t CO_2e ha^{-1} yr^{-1}	—	**0.14–0.94**
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**n/a**	**20**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

28 dryland and 13 irrigated land experiments and showed that irrigated fields emitted more nitrous oxide than dry land; the average difference was approximately 0.7 t CO_2e ha^{-1} yr^{-1}.

The GHG emissions from electricity or fossil fuel used for irrigation pumping are 0.03–3.1 t CO_2e ha^{-1} yr^{-1} (Follett 2001; Lal 2004; Schlesinger 2000; West and Marland 2002), which in most cases outweighs any C sequestration. The irrigation of semi-arid land with high-pH soils can also release carbon dioxide to the atmosphere when calcium carbonate ($CaCO_3$) is dissolved; such emissions approximately equal 0.3 t CO_2e ha^{-1} yr^{-1} (Martens et al. 2005; Schlesinger 2000). In addition, irrigation in regions where water is already in limited supply creates tradeoffs with other water uses, including human consumption and ecological flows to support aquatic species. Therefore, the GHG mitigation benefits of increasing irrigation area are unlikely to outweigh the costs.

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. Total N_2O emissions after a reduction in irrigation intensity have been measured to decrease by between 0.1 t CO_2e ha^{-1} yr^{-1} and 0.9 t CO_2e ha^{-1} yr^{-1} (Table 14). Studies by Kallenbach et al. (2010) and Amos et al. (2005) also documented significantly lower N_2O fluxes with drip and buried tape (versus surface) irrigation, although annual mitigation effects could not be determined because of relatively short-term flux measurements. Some irrigation improvements are likely possible on most of the 20 Mha of irrigated cropland in the United States.

Table 14. Estimates of soil C sequestration achieved by converting dry land to irrigated land and estimates of N_2O emissions reductions from irrigation system changes in the U.S.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Convert dry land to irrigated land				
Lal et al. (1999b)	U.S. general	Soil C sequestration; value also used by IPCC (2000) and Lal et al. (2007)	Expert estimate	*Low: 0.18* *High: 0.55*
Smith et al. (2008)	Global estimate	Water management (mainly increases in irrigation)	Expert estimate	*Low: -0.55* *High: 2.82*
Liebig et al. (2005b)	Colorado, continuous corn	Review	3	1.95
Bordovsky et al. (1999)	Texas, sorghum and wheat	Controlled experiment with conventional and reduced tillage treatments, residue retained or removed	8	0.87
Entry et al. (2002)	Idaho	Soil C on multiple sites with moldboard plow (A) and conservation tillage (B) compared with native land; irrigation- and fuel-related emissions of 1.2 t CO_2e ha^{-1} yr^{-1}	2	A: 1.69 B: 2.56
Improve irrigation management				
Rochette et al. (2008b)	Canada	N_2O emissions lower without irrigation, estimate assumes 75–150 kg N fert ha^{-1}	Empirical model	0.79
Scheer et al. (2008)	Uzbekistan	Reduce irrigation intensity, N_2O emissions decrease	Field study	Wheat: 0.14 Cotton: 0.94

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Burger et al. (2005) noted higher N_2O emissions immediately following irrigation events but a significant decrease after water-filled pore space (WFPS) went below 60%. Studies on subsurface drip irrigation have found that WFPS is higher than 60% only within a few centimeters of the drip tape; the overall low WFPS is 20% to 30% in these systems (Kallenbach et al. 2010). The resulting decrease in N_2O emissions complements previous drip irrigation studies that demonstrated sustained or increased yields and reduced N fertilizer requirements (Camp 1998) and improved N use efficiency (Halvorson et al. 2008a). Scheer et al. (2008) determined 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}), and similar effects were observed on winter wheat fields, although the total impact was lower because of lower baseline emissions.

Many systems have switched from inefficient furrow irrigation to central-pivot sprinklers. Further efficiency gains can be obtained with drip irrigation, which requires 25% to 72% less water than furrow irrigation in agronomic and horticultural crops, with no negative yield impact (Camp 1998; Halvorson et al. 2008a; Lamm et al. 1995), thus providing significant energy and emissions savings. Using a conservative estimate of 25% water savings for widely implemented drip irrigation or other similar improvements on the current 15.5 Mha of cropland irrigated through pumping (the remaining 4.7 Mha are gravity-fed), the emissions reductions from energy savings alone[34] would be approximately 2.8 Mt CO_2e yr^{-1} (0.2 t CO_2e ha^{-1} yr^{-1}).

Reduce Chemical Inputs

Various conservation practices, e.g., integrated pest management or intercropping for weed control, could reduce agricultural inputs of nonfertilizer chemicals, leading to upstream and process GHG emissions reductions. 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). Such upstream emissions reductions may be important for voluntary markets or for non–cap-and-trade regulatory programs, which would target emissions at the production site.

Although pesticide production 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, and reductions are likely to be of more importance for non-GHG reasons. The production and application of pesticides uses less than 15% of the total energy in agriculture (Helsel 2007). Total upstream and application emissions associated with herbicides, insecticides, and fungicides in the United States are 0.03–0.06 t CO_2e ha^{-1} yr^{-1} (Lal et al. 2003; West and Marland 2002).[35] By comparison, Audsley et al. (2009) estimated an average pesticide energy input to arable crops of 0.09 t CO_2e ha^{-1} for the United Kingdom.

Nitrous Oxide Emissions Reduction with Nitrogen Management

Total annual direct and indirect 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 or by-product of two N transformation processes performed by soil microorganisms—denitrification and nitrification. Emission rates are positively correlated with low pH, high 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). Mineral N is often considered the main limiting factor; N_2O emissions are significantly related to the application of inorganic and organic N fertilizer, legume-derived nitrogen, and other factors that affect the availability of soluble mineral N in the soil. Therefore, N use efficiency improvements (i.e., increased productivity for the same N application rate, or equivalent productivity with lower N application rate) can significantly lower N_2O emissions. Residual soil mineral N concentrations are also positively correlated with nitrate leaching and emissions of nitric oxide and ammonia to the air (Mosier et al. 1998b), which degrade water and air quality. Nitrate leaching also increases the potential for off-site (i.e., indirect) N_2O emissions.

Nitrous oxide 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). Parkin and Kaspar (2006) observed 45%–49% of the cumulative annual 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 and spring (Gregorich et al. 2005; Wagner-Riddle et al. 2007). However, even with such high variability at the small scale, determining the impacts of management changes with large-scale sampling and existing models is possible (Desjardins et al. 2010). Because cropland N_2O emissions tend to be higher in humid regions than in dry regions, the majority of emissions reduction potential is in the humid regions.

In this report, N_2O emissions management strategies are divided into seven categories; the first five address the rate, source, placement, and timing of synthetic fertilizer and the use of nitrification inhibitors. This N fertilizer management fits into the 4-R framework described by Roberts (2006)—right rate, right product, right time, and right place. The other categories address the potential to mitigate N_2O emissions through improvements in manure application and cropland drainage in humid areas. Irrigation water management, one other N_2O emission reduction activity considered in this assessment, is already discussed above.

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

35. The lower estimate (Lal et al. 2003) is calculated from all chemical-related emissions and divided among all U.S. cropland, whereas the upper estimate (West and Marland 2002) corrects for the proportion of land area with such chemical application. Using data from West and Marland (2002), the average emissions for corn (0.09 t CO_2e ha^{-1} yr^{-1}) are somewhat higher than those for wheat (0.03 t CO_2e ha^{-1} yr^{-1}).

Reduce N fertilizer rate

Using data from 32 field comparisons (Table 15), the average N_2O emission reduction for reducing fertilizer N application rates by 15% is 0.3 t CO_2e ha^{-1} yr^{-1} (a range of 0.03 t CO_2e ha^{-1} yr^{-1} to 0.8 t CO_2e ha^{-1} yr^{-1}). With no data (and no expected impact) on soil C and methane, and only small reductions in upstream emissions, the net GHG mitigation potential remains 0.3 t CO_2e ha^{-1} yr^{-1}. Field studies in cropland agriculture have found that N_2O emissions are positively correlated with N fertilizer rate (e.g., Halvorson et al. 2008b; MacKenzie et al. 1998; McSwiney and Robertson 2005; Mosier et al. 2006), even though varying processes affect emissions at different mineral N concentrations in the soil (McSwiney and Robertson 2005). Because a greater proportion of N fertilizer tends to be lost as N_2O in moist climates than in dry climates, the average N_2O emissions reduction potential is significantly greater in moist climates than in dry ones: 0.6 t CO_2e ha^{-1} yr^{-1} versus 0.05 t CO_2e ha^{-1} yr^{-1} (Bremer 2006; Halvorson et al. 2008b; McSwiney and Robertson 2005; Millar et al. 2010; Mosier et al. 2006). Nitrogen fertilizer is applied on nearly all U.S. cropland, and rate reductions may be possible on much of this area. Soil sampling helps farmers understand fertilizer needs, in most cases leading to lower fertilizer application rates,[36] but only 50% of corn cropland is tested for soil N availability (Paustian et al. 2004). If 50% to 60% of cropland is over-fertilized, rate reductions could be implemented on approximately 68 Mha.

GHG Impact Summary

GHG category	Reduce fertilizer N rate by 15%
Number of observations	**32**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**no data**
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.28** (0.03–0.82)
CH_4, t CO_2e ha^{-1} yr^{-1}	**no data**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.06**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.33** (0.08–0.88)
Maximum U.S. applicable area, Mha	**68**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

The Intergovernmental Panel on Climate Change (IPCC) Tier I method for calculating N_2O emissions uses a direct linear multiplier of 1% of total N fertilizer application lost as N_2O-N (IPCC 2006). However, in field studies, researchers have noted proportions ranging from <0.2% to >1.6% of N fertilizer, depending on the soil, climate, season, and other factors (Lemke et al. 2003; Mosier et al. 2006; Stehfest and Bouwman 2006). Some of these N_2O emissions are related to other N sources (manure, legumes, atmospheric deposition, and mineralized soil N), but in certain cases, the N_2O emissions rate from N fertilizer itself appears to rise significantly above that predicted by the IPCC Tier I factor (see Grant et al. 2006; McSwiney and Robertson 2005). Using Tier I default factors, indirect N_2O emissions are also calculated as a proportion of total N fertilizer application, bringing the total N_2O-N emissions rate to 1.1%–1.3% of N fertilizer application.[37]

Such linear relationships may be appropriate at large scales and low N fertilizer 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 2010a). However, N_2O emission rates at the field scale—as a function of the amount of nitrogen applied—have been shown in many cases 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). On the other hand, although the theoretical potential for N_2O emissions depends on excess nitrogen availability, soil moisture or C substrate availability may also be limiting factors so that, in certain situations, this nonlinear response is not observed.[38]

With increasing demand for food (due to increasing population and consumption), any shift in N management must sustain crop yield (Snyder et al. 2009). Thus, the primary objective is to improve N use efficiency (i.e., productivity per unit of N application). If reductions in N fertilizer decrease crop yields, GHG emissions could actually increase, because production that compensates for yield losses could shift to less efficient regions or production systems (negative leakage). Incentives for GHG mitigation should therefore avoid reducing yield by much in highly efficient systems.

36. Of the corn farmers who test for soil N availability, 80% apply the recommended rate (Paustian et al. 2004).
37. 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 N fertilizer applied and 20% of organic N applied (e.g., manure), for both of which 1.0% is emitted as N_2O-N.
38. R. Lemke and P. Rochette, personal communication, September 2010.

Output-based accounting approaches (see Murray and Baker 2011), can capture yield impacts, reduce negative leakage, and reward positive leakage.

Nitrogen fertilizer tends to increase productivity and biomass input and thus SOC (Varvel 2006), prompting suggestions that fertilizer application could be a GHG-mitigating technique (Snyder et al. 2009). Although this may be the case in N-limited regions internationally, most U.S. crops already receive N fertilizer, and higher application rates are unlikely to sequester much soil C. Recent studies have determined that additional N fertilizer application has little to no impact on SOC or CO_2 fluxes (Alluvione et al. 2009; Mosier et al. 2006). In some studies, N fertilizer has been associated with reductions in soil C (Khan et al. 2007; Mulvaney et al. 2009), although Powlson et al. (2010) questioned both the experimental methods and the conclusions of these studies. In fact, many factors could cause the different results. For example, Poirier et al. (2009) found that high N fertilizer application rates reduced soil C under moldboard plow, but not under no-till treatment. The N fertilizer application accelerated SOM decomposition in the plow treatment, but the additional productivity from N fertilizer in the no-till treatment generated more plant residue. Although soil C effects may be uncertain, some upstream and process GHG emissions savings will accompany any decrease in N fertilizer rate.

Table 15. Estimates of N_2O emissions reductions of 15% reductions in N fertilizer rate

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Paustian et al. (2004)	National estimate	Estimated reduction of 30%–40% with efficient use of N inputs	n/a	
Stehfest and Bouwman (2006)	Global	Model from field estimates (n=840); reduce N_2O by 8.2% at rates of 75–225 kg N ha^{-1}	n/a	
Smith et al. (2008)	Global	Reduce N application by 20%, dry vs. moist climate, wide range in potential	Expert estimate	*Dry: 0.33* *Moist: 0.62*
Millar et al. (2010)	Michigan, continuous corn; Corn Belt and Lake States, corn-soybean rotation	Calculated from field trials, assumes 15% reduction of N application	4 7	CC: 0.70 CS: 0.60
Bremer (2006)[b]	Kansas	Reduce application of urea fertilizer	1	0.02
Halvorson et al. (2008b)[b]	Colorado	Lower potential for continuous corn and for corn-barley rotation than for corn-dry bean rotation	2	0.08
Mosier et al. (2006)	Colorado	Conventional tillage (CT) and no-till (NT) systems, continuous corn (CC) or corn-dry bean (CB) rotation	6 6 3	CT/CC: 0.06 NT/CC: 0.05 NT/CB: 0.04
McSwiney and Robertson (2005)[b]	Michigan	2%–7% of each additional kg N lost as N_2O, no yield decrease as long as rate remained above 101 kg N ha^{-1}	3	0.31

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.
b. In these studies, the potential mitigation is based on the relationship between nitrous oxide and a given N fertilizer rate and then calculated under the assumption of a 15% rate reduction.

Researchers do not agree on whether, where, and how much excess fertilizer is being used at the farm or field level. Some assert that farmers are already applying N fertilizer at the lowest possible rates; others suggest that extra N fertilizer is often applied as "insurance," and thus may not be needed. The latter have estimated that N fertilizer could be reduced by 12%–20% without severely negative yield impacts.[39] Any rate reductions are likely possible only in conjunction with nitrification inhibitors or with the N use efficiency gains that result from changes in placement, timing, and source. Thus, N fertilizer rate could function as an integrator of multiple practices, and the GHG mitigation potential for different N management practices cannot be additive; interactions must be considered carefully. Assuming continued implementation of any improved N management practices, reduced N_2O emissions generate benefits in perpetuity without risk of reversal, as in soil C sequestration.

39. Smith et al. (2008) estimated that 20% reductions in N fertilizer 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.

Change N fertilizer source

In some regions and cropping systems, fertilizer source significantly affects N_2O emissions. In this analysis, fertilizer source management is separated into two activities: switching fertilizer N source from ammonium-based to urea and switching to slow-release fertilizers.

GHG Impact Summary

GHG category	Switch fertilizer N source from anhydrous ammonia to urea	Switch to slow-release fertilizer N source
Number of observations	**18**	**18**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.59** (0.03–1.47)	**0.12** (0.04–0.21)
CH_4, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**no data**	**0.06**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.59** (0.03–1.47)	**0.18** (0.10–0.27)
Maximum U.S. applicable area, Mha	**37**	**93**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

In the United States, approximately 36% of the total N fertilizer used is in ammonia form (mostly anhydrous ammonia with some aqueous ammonia); 22% is in urea form, 29% is in a nitrogen solution form (primarily urea-ammonium-nitrate), and the remainder comes from other various sources. Since the 1980s, the proportion of anhydrous ammonia-N use to urea-N use has decreased from 3.3 to 1.6. Farmers are switching to urea (likely for safety and availability reasons) even though anhydrous ammonia prices have been between 55% and 80% lower per unit of nitrogen (USDA ERS). Continued shifting from anhydrous ammonia to urea may have GHG benefits: 18 field observations in North America (Table 16) indicate an average N_2O emissions decrease of 0.6 t CO_2e ha^{-1} yr^{-1} (a range of 0.03 t CO_2e ha^{-1} yr^{-1} to 1.5 t CO_2e ha^{-1} yr^{-1}). Other GHG categories are not affected. With 20% of U.S. fertilizer sourced as anhydrous ammonia, the switch to urea form fertilizer could be implemented on as much as 37 Mha of cropland.

Globally, the relationship between the N_2O emissions of anhydrous ammonia application and those of urea application has not been consistent. Using 1,125 agricultural field measurements for nitrous oxide, 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 use of urea or urea ammonium nitrate (the three most common N fertilizer sources in the United States).[40] However, in earlier work, Bremner et al. (1981a) and Breitenbeck and Bremner (1986a) concluded that emissions following anhydrous ammonia use were substantially higher than those following urea use. While neither the Stehfest and Bouwman (2006) nor Bremner et al. (1981a) studies were based on contemporaneous side-by-side treatment comparisons, Breitenbeck and Bremner (1986a) reported a controlled side-by-side comparison, albeit without a crop present. Later studies in Tennessee, Iowa, and southern Minnesota corn systems (Fujinuma et al. 2011; 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, whereas anhydrous ammonia is injected) or to differential ammonia volatilization losses among fertilizer types (which affects the amount of remaining nitrogen that could be lost as nitrous oxide).

Other ammonium-based fertilizers may not have the same relationship. Researchers in Scotland found higher N_2O emissions from urea use than from ammonium sulfate use on grasslands and barley (Clayton et al. 1997; McTaggart et al. 1997). A similar difference between ammonium sulfate and urea emissions was noted by Tenuta and Beauchamp (2003) in an incubation experiment, but only under aerobic conditions; urea fertilizer application had lower emissions when soil was water saturated. Therefore, emission effects from these other fertilizer sources warrant further study.

Venterea et al. (2005) also noted that tillage affects emissions; although anhydrous ammonia always generated higher emissions than urea, there was a greater difference between the two fertilizer types in CT than in NT systems. To date, the direct studies showing no emissions 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 corn (which is the main crop in the studies that do experience a significant difference). Therefore, although urea use tends to produce

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

lower emissions than anhydrous ammonia use, further research is necessary to determine interactions with crop type and climatic conditions.

Table 16. Estimates of N_2O emissions reductions with changes in N fertilizer source

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})
Switch from anhydrous ammonia to urea N source				
Breitenbeck and Bremner (1986a)	Iowa, no crop present	Lower emissions for urea in all three soils tested; site previously planted to corn	3	1.02
Thornton et al. (1996)	Tennessee, NT corn	Lower emissions with urea use than with anhydrous ammonia use	1	2.80
Venterea et al. (2005)	Minnesota, corn-soybean rotation	Lower emissions with broadcast urea than with injected anhydrous ammonia	1 1	CT: 0.78 NT: 0.29
Burton et al. (2008a)	Manitoba, wheat	Emissions difference between anhydrous ammonia and urea not significant	4	0.02
Venterea et al. (2010)	Minnesota, corn-soybean (CS) and continuous corn (CC) rotations	Lower emissions with broadcast urea than with injected anhydrous ammonia	3 3	CC: 0.50 CS: 0.25
Fujinuma et al. (2011)	Minnesota, corn	Lower emissions with broadcast urea than with shallow (0.1m) or deep (0.2m) injected anhydrous ammonia (2 year average); however, NO emissions were higher with urea	1 1	Shallow: 0.47 Deep: 0.10
Switch to slow-release fertilizer N source				
Delgado and Mosier (1996)	Colorado, irrigated barley	Polyolefin-coated urea decreased N_2O emissions by 16% compared with urea (3 mo)	1	0.05
Burton et al. (2008a)	Manitoba, wheat	Polymer-coated urea	3	0.20
Halvorson et al. (2010)	Colorado, corn-dry bean-barley rotations	CT and NT systems; enhanced efficiency urea sources	6	0.13
Hyatt et al. (2010)	Minnesota, potato	Polymer-coated urea (PCU)	6	0.13
Venterea et al. (2011)	Minnesota, corn-soybean rotation	PCU and impregnated urea; no significant difference by area; PCU had lower yields, so more N_2O emissions per unit of crop yield	2	0.00

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Enhanced-efficiency N fertilizers (EEFs), such as slow- and controlled-release and stabilized N fertilizers, could increase crop recovery of nitrogen and minimize N losses to the environment (Snyder et al. 2009). Data from 18 field observations (Table 16) suggests that N_2O emissions can be reduced by 0.1 t CO_2e ha^{-1} yr^{-1} (a range of 0.04 t CO_2e ha^{-1} yr^{-1} to 0.2 t CO_2e ha^{-1} yr^{-1}). Small reductions in total N fertilizer net some upstream emission savings as well, so that the net GHG mitigation potential of EEFs is 0.2 t CO_2e ha^{-1} yr^{-1}. These fertilizers are applicable to most cropland; this assessment assumes a conservative estimate of 75% (93 Mha in the United States). The somewhat increased cost of production and transportation (due to greater mass and bulk) of EEFs may be worth the price, given the GHG benefits and efficiency gains as well as the reduced damage to downstream water quality. More research is needed to evaluate N_2O emissions response to EEFs for a range of regions and cropping systems.

Change fertilizer N placement

The placement of synthetic fertilizer near the zone of active root uptake may reduce surface N loss and increase plant N uptake for an estimated N_2O emissions reduction of 0.3 t CO_2e ha^{-1} yr^{-1} (a range of 0.0 t CO_2e ha^{-1} yr^{-1} to 0.7 t CO_2e ha^{-1} yr^{-1}) drawn from 21 field observations (Table 17). No data are available for other GHG categories, nor are any significant effects expected. In the 1990s, improved N fertilizer placement was achieved through banding on 40% of U.S. corn acreage (Paustian et al. 2004). Using corn as the best available approximation for all U.S. crops, this assessment estimates that 60% of U.S. cropland (63 Mha) could experience improved fertilizer N placement.

Banded, as opposed to broadcast, placement may reduce immobilization of nitrogen and delay leaching or denitrification (Snyder et al. 2009) as well as reduce N_2O emissions (Hultgreen and Leduc 2003). Improved placement can also entail rate modification for different areas of a field based on yield expectations (e.g., precision agriculture using global positioning systems). Because factors other than N availability (i.e., soil pH, water, and so on) affect crop growth, yield—and thus crop N demand and uptake—can vary across a crop field. Evenly applied N fertilizer often means over-application in areas of fields that tend to be lower yielding. After reducing the N fertilizer rate by 25 kg N ha^{-1} for a low-yielding portion of a field, Sehy et al. (2003) measured a N_2O emissions reduction of 2.3 t CO_2e ha^{-1} in that area (assumed to be related to lower soil NO_3^- concentrations); the average emission reductions for the entire field was 0.4 t CO_2e ha^{-1}.

Shallow versus deep N fertilizer injection has yielded contradictory GHG flux effects; reduced N_2O emissions resulted from shallow placement of ammonium nitrate in Ontario (Drury et al. 2006), but increased emissions resulted from shallow placement of liquid urea ammonium nitrate (UAN) 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 1986b). 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 elucidate the different interactions of soil type and other conditions that affect N_2O emissions when combined with different N fertilizer placement options.

GHG Impact Summary

GHG category	Change fertilizer N placement	Change fertilizer N timing	Use nitrification inhibitors
Number of observations	**21**	**19**	**35**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**	**no data**
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.25** (0.00–0.69)	**0.18** (0.00–0.53)	**0.41** (0.02–1.04)
CH_4, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**	**no data**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**no data**	**no data**	**no data**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.25** (0.00–0.69)	**0.18** (0.00–0.53)	**0.41** (0.02–1.04)
Maximum U.S. applicable area, Mha	**63**	**53**	**92**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Table 17. Estimates of N_2O emissions reductions from changes in N fertilizer placement and timing

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Change fertilizer N placement				
Hultgreen and Leduc (2003)	Saskatchewan, canola, flax, and wheat	Change from broadcast fertilizer to banded, 3 years of data per site Change from banded mid-row to side-row, 3 years of data per site	4 4	0.04 0.03
Drury et al. (2006)	Ontario, wheat-corn-soybean rotation	Shallow N placement (2 cm) yielded fewer emissions than deep placement (10 cm); sampled during corn phase; tillage affected emissions with deep-placed fertilizer (zone tillage < NT < CT)	9	0.47
Liu et al. (2006)	NE Colorado, corn	Urea-ammonium-nitrate, deep injection (10 or 15 cm) had lower emissions than shallow (0 or 5 cm), two tillage treatments	4	CT: 0.11 NT: 0.25
Change fertilizer N timing				
Hao et al. (2001)	Southern Alberta, wheat and canola	Irrigated, change from fall to spring application	2	0.73
Hultgreen and Leduc (2003)	Saskatchewan, canola, flax, and wheat	Dry land, change from fall to spring application with urea and anhydrous ammonia, 3 years of data per site	8	0.02
Burton et al. (2008b)	Manitoba	Dry land, change from fall to spring application of urea	2	0.00
Burton et al. (2008b)	Manitoba	Dry land, change from fall to spring application of anhydrous ammonia	2	0.16
Burton et al. (2008b)	New Brunswick, potatoes	Split application of ammonium nitrate	3	0.27
Zebarth et al. (2008)	New Brunswick, corn	Side-dress instead of preplant application of ammonium nitrate	2	0.38

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Change fertilizer N timing

Crop N uptake capacity is generally low at the beginning of the growing season, increases rapidly during vegetative growth, and drops sharply as the crop nears maturity. Synchronous timing of N fertilizer 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 with fall application (Hao et al. 2001; Hultgreen and Leduc 2003). Although study results vary, it appears that split application lowers emissions, especially in areas with high rainfall or a lot of irrigation (Burton et al. 2008b). On the basis of 19 field comparisons (Table 17), the average N_2O reduction potential due to improved fertilizer N timing is estimated to be 0.2 t CO_2e ha^{-1} yr^{-1} (a range of -0.03 t CO_2e ha^{-1} yr^{-1} to 0.5 t CO_2e ha^{-1} yr^{-1}). No data are available for other GHG categories, nor are any other significant effects expected. Thirty percent of U.S. corn is fertilized in the fall (Paustian et al. 2004), and additional cropland area could be improved with split fertilizer application. Therefore, this assessment estimates that 50% of U.S. cropland (53 Mha) could experience some form of improvement in fertilizer N timing.

Use nitrification inhibitors

Nitrification inhibitors increase the cost of fertilizer by 9% (Snyder et al. 2009), but they can significantly improve N recovery (Cochran et al. 1973) and reduce nitrate leaching when applied with urea or ammonium-based N fertilizer. Slowing nitrification reduces the release rate of soluble mineral N, leading to average N_2O emissions reductions of 0.4 t CO_2e ha^{-1} yr^{-1} (a range of 0.02 t CO_2e ha^{-1} yr^{-1} to 1.0 t CO_2e ha^{-1} yr^{-1}) from 35 field comparisons in the United States and Europe (Table 18). No data for other GHG categories are available. Nitrification inhibitors are currently utilized on only 3.4 Mha of U.S. cropland (USDA ERS 2010b), and because 90% of commercial fertilizer is urea or ammonium based (USDA ERS 2010a), a total area of 92 Mha is available for nitrification inhibitor application.

Table 18. Estimates of N_2O emissions reductions from using nitrification inhibitors on cropland and grassland

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Snyder et al. (2009)	Colorado and Germany	Review, only included field comparison results from the U.S. (7 obs.) and Europe (1 obs.)	8	0.69
Akiyama et al. (2010)	Iowa, Germany, United Kingdom, and Spain	Review, only included field comparison results from the U.S. (2 obs.) and Europe (22 obs.)	24*	0.39
Parkin and Hatfield (2010)	Iowa, corn Iowa, grassland	Fall-applied anhydrous ammonia, delayed N_2O emissions and increased corn yield, total difference not statistically significant No significant difference	2 1	-0.24 0.00

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Field studies worldwide have measured N_2O emissions reductions of 9%–95% when nitrification inhibitors were combined with urea, ammonium nitrate, and ammonium sulfate fertilizers (Akiyama et al. 2010; Snyder et al. 2009), although the short length of some of these studies may have overestimated the impact (Snyder et al. 2009). In a review of 85 observations of nitrification inhibitors, Akiyama et al. (2010) noted average emissions reduction of 38%; nitrapyrin, 3,4-dimethyl pyrazole phosphate (DMPP) and calcium-carbide were the most effective (average reductions of 50%). The greatest effectiveness seems to be achieved on grassland fields, as opposed to other upland or rice paddies (Akiyama et al. 2010). McTaggart et al. (1997) found that the nitrification inhibitor retained effectiveness in August following an April application, indicating that long-term (even post–growing season) fluxes should be monitored.

Early data from the Corn Belt indicated significant reductions in N_2O emissions when using nitrapyrin with anhydrous ammonia (Bremner et al. 1981b). However, annual N_2O flux was unaffected by nitrification inhibitors used with ammonium sulfate and anhydrous ammonium in more recent studies, even though nitrification was delayed and N_2O emissions reduced in the near term (Parkin and Hatfield 2010). Therefore, 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). Further research is needed to elucidate interactions with fertilization and soil conditions.

Integrating the four Rs

As mentioned earlier, N rate can be an integrator of the 4 Rs (right rate, right source, right placement, and right timing), because all N use efficiency improvements can reduce the N fertilizer needs per unit of production. Precision agriculture techniques can achieve N rate reductions by accommodating 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 for farmers to adjust N rates during growing-season N fertilizer application. When compared with uniform N rates based on soil testing, on-board sensors can improve N use efficiency by 15%–20% (Li et al. 2009; Raun et al. 2002). These decreases in N fertilizer rates may be some of the most effective in reducing N_2O emissions, because the "excess" fertilizer above crop needs is highly susceptible to losses.

Improve land manure application

A significant amount of the nitrogen in manure can be lost as ammonia (NH_3), nitrate (NO_3^-), or nitrous oxide after land application; loss estimates of up to 50% of the total nitrogen are not uncommon (Mosier et al. 1998b). Most gaseous losses are in the form of ammonia, causing air quality problems and N deposition in natural ecosystems. Leaching losses of nitrate reduce water quality. Because the N loss pathways are connected, most efforts to control direct N_2O emissions from manure application provide the environmental co-benefit of reduced 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 Mt CO_2e yr^{-1} to 30.6 Mt CO_2e yr^{-1} (U.S. EPA 2009; USDA (U.S. Department of Agriculture) 2008). With potential emissions reductions ranging from 0.4 t CO_2e ha^{-1} yr^{-1} to 1.2 t CO_2e ha^{-1} yr^{-1} (Table 19), researchers have studied various improvements in manure application, including reducing total application rates, applying solid rather than liquid manure, using nitrification inhibitors, and applying manure to dry rather than wet areas when air temperatures are low. Such improvements could be implemented on at least a portion of the 12 Mha of U.S. cropland currently receiving manure applications. Better management of manure on corn cropland alone will generate significant results, because corn comprises 58% of all manured land,[41] 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 reduce N_2O
Number of observations	**1/3/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	—
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.37–1.22**
CH_4, t CO_2e ha^{-1} yr^{-1}	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—
Maximum U.S. applicable area, Mha	**12**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Nitrous oxide emissions rates are highly variable and depend on elapsed time since manure application, type of manure, climatic conditions, and the amount of water available in the soil or with the manure (Saggar et al. 2004). The proportion of denitrified nitrogen lost as N_2O (rather than N_2) is greatest directly after liquid manure application (Saggar et al. 2004). Therefore, 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 instead of aerobic storage also significantly reduces N_2O losses, both during storage and following field application (Mosier et al. 1998b). However, the most promising starting place may be adjustments of commercial N application rates after accounting for N addition in the manure. Nearly 40% of farmers do not make such adjustments (USDA ERS 2009), which would also lower fertilizer costs and related emissions. As with N fertilizer, the tradeoff between N_2O emissions reductions and crop yield may need to be considered if manure N application rates are reduced (Rochette et al. 2000).

Table 19. Estimates of N_2O emissions reductions from improved manure application management

Citation	Region	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Paustian et al. (2004)	U.S. general	General estimate for improved "waste" disposition, 10% reduction in emissions	Expert estimate	1.17
Pork Technical Working Group (2005)	Canada	Apply to dry rather than wet areas, 50% reduction in N_2O emissions	Expert estimate	0.59
Gregorich et al. (2005)	Canada	Apply solid rather than liquid manure, review of 5 studies	Review, no individual data	0.86
Rochette et al. (2000)	Canada	Apply lower rate of pig slurry, reduces % N denitrified from 1.65% to 1.23%	Field study	1.22

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

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

Drain Agricultural Lands in Humid Areas

The scientific literature contains little information about the potential of draining agricultural land to obtain N_2O emissions reductions. 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 a difference of 0.19 t CO_2e ha^{-1} yr^{-1}). However, as these comparisons were not side-by-side comparisons, other factors may have also played a role; whether N_2O emissions from poorly drained soils in the United States could be remediated with drainage, and if so, how much land could be treated in this way remains unclear. 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

Microbial and plant respiration in flooded conditions reduces oxygen potential, creating anaerobic conditions in rice fields that lead to CH_4 production. In 2009, the worldwide planted rice area totaled 155.7 Mha, of which 1.29 Mha (0.8%) was in the United States (USDA NASS 2009a). Annual rice-related CH_4 emissions in the United States total 6.2 Mt CO_2e (2007), almost 1% of the national total from all sources—3% of the CH_4 emissions from agriculture (U.S. EPA 2009). In contrast, worldwide CH_4 emissions from rice are estimated at 708 Mt CO_2e for 2010 (U.S. EPA 2006)—comprising 11% of global agricultural GHG emissions. Although CH_4 emissions from rice production make up a small portion of U.S. emissions, their potential mitigation per unit area can be significant, and the anticipated cost per t CO_2e of that mitigation is low (Smith et al. 2008).

GHG Impact Summary

GHG category	Adjust rice water management	Plant rice cultivars that produce less CH_4
Number of observations	**26**	**19**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**-0.04**	**no data**
N_2O, t CO_2e ha^{-1} yr^{-1}	**-0.79**	**0.00**
CH_4, t CO_2e ha^{-1} yr^{-1}	**1.97** (0.08–5.31)	**0.97** (0.06–1.87)
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	no data	0.00
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**1.14** (-0.75–4.48)	**0.97** (0.06–1.87)
Maximum U.S. applicable area, Mha	**1.3**	**1.3**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Midseason drainage is one of the more promising emission-reducing activities (Li et al. 2005b; Sass and Fisher 1997), and data from 26 field comparisons (Table 20) indicate an average reduction in CH_4 emissions of 2.0 t CO_2e ha^{-1} yr^{-1} (a range of 0.1 t CO_2e ha^{-1} yr^{-1} to 5.3 t CO_2e ha^{-1} yr^{-1}). Small gains in CO_2 emissions and significant increases in N_2O emissions in some locations yield a net GHG mitigation potential of 1.1 t CO_2e ha^{-1} yr^{-1}. Li et al. (2004) propose that the widespread shift from continuous flooding to midseason drainage, during the 1990s in China accounted for much of the slowed growth in atmospheric CH_4 concentrations during that time. Such water management changes were adopted to save water and increase yields. In Asian rice systems, Wassmann et al. (2001) found that a single midseason drainage could reduce CH_4 emissions by 7%–43%, a statistically significant finding in seven of eight experiments. Dual drainage at midtillering and preharvest could reduce CH_4 emissions by as much as 80% (Wassmann et al. 2000). Sass and Fisher (1997) found that a single midharvest drainage, for rice cultivated in Texas, could reduce total emissions by about 50%, and a two-day drainage period every three weeks could reduce emissions to an insignificant amount (<0.25 t CO_2e ha^{-1}). Other studies from around the world but mainly in China have made similar findings.

However, in regions with high soil C content, increased N_2O emissions can follow midseason drainage, eliminating any net GHG benefit (Li et al. 2005b). The increased N_2O emissions in some areas have 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 monitor N_2O emissions on likely (high C) soil types. This task could be accomplished through model validation or perhaps by determining the level of soil C above which N_2O emissions will have to be considered. In California rice-growing regions, preliminary data suggest that N_2O emissions are not elevated with multiple drainages or other alternative water management.[42]

42. W.R. Horwath, personal communication, June 2010.

Water management during the nongrowing season can also affect gaseous flux, necessitating full-year CH_4 emissions accounting. With two years of monitoring, Fitzgerald et al. (2000) found that winter flooding increased annual CH_4 emissions from California rice fields by 2.0 t CO_2e ha^{-1} yr^{-1} (emissions in non-winter-flooded plots were 2.7 t CO_2e ha^{-1} yr^{-1}). In addition, about half of the emissions occurred during the flooded conditions in the winter, requiring full-year measurements to monitor the effects. Flooding of Chinese rice fields in winter also increased CH_4 emissions (Kang et al. 2002; Xu et al. 2000). Therefore, other agronomic advantages to winter flooding may be offset by the GHG implications.

Another important management issue is the incorporation of rice straw. Methane emissions increase by 2–5 times when rice straw is incorporated in soil rather than burned (Bossio et al. 1999; Redeker et al. 2000), because the additional organic material encourages microbial activity, including methanogenesis. However, this practice improves air quality and nutrient cycling (Eagle et al. 2000). Thus, the tradeoffs among GHG mitigation, addition of plant nutrients, and other factors may require further examination.

Process emissions impacts of water management change depending on the energy requirements for transport of water in and out of fields; emissions would be minimal in gravity-fed irrigation systems. Where irrigation water is pumped, rather than gravity-fed, increased fuel use associated with midseason drainage (and subsequent reflooding) may offset some of the benefits from CH_4 emissions reduction, but no data are readily available with which to make reasonable estimates.

Table 20. Estimates of CH_4 emission reductions with management changes in rice systems

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Adjust rice water management				
Li et al. (2004)	DNDC model, China	Midseason drainage	Modeled	*4.7–5.2*
Li et al. (2005b)	DNDC model, China	Midseason drainage	Modeled	*4.2*
Sass and Fisher (1997)	Texas	Midseason drainage drainage every 3 wks, 100% reduction	2 1	1.10 2.32
Towprayoon et al. (2005)	Thailand	Midseason drainage multiple drainages	1 1	1.65 2.07
Wassmann et al. (2000)	Asia	Midseason drainage, 7%–43% reduction	21	2.04
Plant rice cultivars that produce less CH_4				
Sass and Fisher (1997)	Texas	Tested 10 cultivars, estimate is difference between lowest and highest emissions	1	5.79[b]
Setyanto et al. (2000)	Central Java	Tested four cultivars	4	0.78
Wassman et al. (2002)	Philippines	Tested three cultivars, estimate is average of highest versus lowest	14*	0.68

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.
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 the national average, ranging from 4.5 t CO_2e ha^{-1} yr^{-1} to 10.3 t CO_2e ha^{-1} yr^{-1}, so emission reductions of this level (~5.8 t CO_2e ha^{-1} yr^{-1}) may not be possible at a national scale.

On the basis of 19 field comparisons (Table 20), this assessment estimates that development of low-emission cultivars can reduce CH_4 emissions by an average of 1.0 t CO_2e ha^{-1} yr^{-1} (a range of 0.1 t CO_2e ha^{-1} yr^{-1} to 1.9 t CO_2e ha^{-1} yr^{-1}). No data are available for other GHG categories, and significant effects on soil C and on N_2O and upstream and process emissions are not expected. High-yield cultivars can produce lower emissions than lower-yielding varieties by directing more carbon to grain production rather than to root processes, where respiration results in CH_4 production (Denier van der Gon et al. 2002; Sass and Cicerone 2002). Cultivar emissions differences may also be affected by the varying capacities of rice aerenchyma to transport methane from the roots or oxygen to the roots, thus affecting soil redox potential (Sass and Fisher 1997). Other researchers propose that emissions rate differences among cultivars relate mainly to the availability of substrate for methanogens, especially root exudates (Aulakh et al. 2001b; Huang et al. 1998). Identification of specific species choice may be complex, however, as Wassman et al. (2002) noted inconsistent emissions rate differences over multiple seasons, especially on different soil types. In summary, before specific rice cultivars can be promoted for GHG mitigation purposes, additional region-specific research is needed.

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

Other management activities for rice CH_4 reductions have also been proposed but are not examined here due to lack of research or little anticipated benefit in the U.S. context. However, in combination with water management or cultivar development, they may deserve additional attention. These activities include reducing water consumption by adjusting levees or land grading and upgrading irrigation pumping systems. Another potentially worthwhile activity is application of silicate fertilizer, which has reduced CH_4 emissions by 16%–28% in Asia (Ali et al. 2008a; Ali et al. 2008b).

Grazing Land Management

As much as 44% of land in the 48 contiguous United States is used for grazing (Lal et al. 2003). Worldwide, grazing lands are considered an important C sink—storing 10%–30% of the world's SOC (Schuman et al. 2002). Grazing lands can be divided into two distinct classes: (1) extensively grazed rangelands or uncultivated land with minimal inputs, consisting of natural or naturalized plant species, and (2) intensively managed pastures with inputs such as cultivation, intentional species planting, irrigation, and fertilizers (Follett and Reed 2010). Compared with rangeland, pasture most often has much higher biomass production per unit area and higher levels of soil C. In the United States, improved pasture is mostly located east of the Missouri River (Schnabel et al. 2001).

Higher soil C sequestration rates are anticipated on land that is in degraded or marginal conditions, whereas lower soil C sequestration rates are anticipated on highly productive, well-managed land with high SOC levels (Follett and Reed 2010). Therefore, the state of the range or pasture land will help determine the C sequestration potential of mitigation activities (Bremer 2009).

Improve grazing management on rangeland

Improved grazing management on rangeland (grazing land without tillage, seeding, or irrigation inputs) often involves reducing stocking rates on overgrazed land, avoiding grazing during drought conditions, and improving the timing of grazing and its frequency. We address species management, irrigation, rotational grazing, and fertilization as separate activities. Improved grazing management on rangeland is expected to capture a significant amount of carbon in the United States: broad reviews indicate potential soil C sequestration rates between 0.6 t CO_2e ha^{-1} yr^{-1} and 1.3 t CO_2e ha^{-1} yr^{-1} (Conant et al. 2001; Conant and Paustian 2002; Follett et al. 2001a). These reviews suggest that reduced stocking rate is the primary driver for this change, especially because many of the poorly managed rangelands have been overgrazed. Indeed, Fuhlendorf et al. (2002) measured a significant decrease in soil C concentration with stocking rates that were nearly double the USDA-recommended rate, compared with less intense grazing. However, the data from 10 field observations of reduced stocking rates on North American rangelands, for which soil C was quantified,[43] were extremely variable and suggest an average soil C *decrease* of approximately 1 t CO_2e ha^{-1} yr^{-1} (Frank et al. 1995; Liebig et al. 2010a; Manley et al. 1995; Naeth et al. 1991; Reeder et al. 2004; Schuman et al. 1999; Smoliak et al. 1972). See Table 21 for details.

GHG Impact Summary

GHG category	Improve grazing management on rangeland	Improve grazing management on pasture
Number of observations	**10/3/0**	**5/1/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**uncertain** (see text)	**-2.97–4.76**
N_2O, t CO_2e ha^{-1} yr^{-1}	—	—
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**227**	**48**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

In the United States, nonfederal grazing land area (i.e., owned privately or by state and other governments) is between 176 Mha (Lubowski et al. 2006) and 214 Mha (USDA NRCS 2007); federal grazing land area is 62 Mha (Lubowski et al. 2006). Of this land, up to 48 Mha of nonfederal land is pasture, i.e., grazing land with tillage, seeding, or irrigation

43. That is, these studies measured the mass, not just the concentration, of soil carbon.

inputs (USDA NRCS 2007). The government-owned land is primarily unimproved rangeland, mostly in the western states. Therefore, management could be improved on as much as 227 Mha of total rangeland.[44]

Table 21. Estimates of soil C sequestration potential of improved grazing management on rangeland

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Follett et al. (2001a)	United States	Improved rangeland management, national estimate	Expert estimate	*Low: 0.18* *High: 0.55*
Lal (2001)	Texas rolling plains	Reduce grazing pressure, recalculated from Pluhar et al. (1986)		*Low: 0.66* *High: 4.98*
Conant et al. (2001)	Global	Improved grazing management, based on review	Expert estimate	*1.28*
Manley et al. (1995)	Wyoming	Reduce grazing pressure	1	-7.26
Schuman et al. (1999)	Wyoming, rangeland	Reduce grazing pressure, increased plant C so that C change in whole system was not significant	1	-2.85
Conant and Paustian (2002)	North America	Decrease grazing intensity on overgrazed land	6	0.02
Reeder et al. (2004)	Northeastern Colorado	Reduce from heavy grazing to light grazing	1	-0.53
Liebig et al. (2010a)	North Dakota, native range	Decrease grazing intensity; CH_4 emissions reduced by 0.31 t CO_2e ha^{-1} yr^{-1}	1	-0.10

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Compared with more highly productive pasture, rangelands have low C sequestration rates on a per unit basis, but because of their vast area, they could capture 2%–4% of annual anthropogenic GHG emissions on a global basis, i.e., 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 carbon capture (greater than 90%) is in the form of SOC. Rangeland systems are characterized by an inherently high degree of variability in soils, topography, plant communities and dominant species, precipitation, and climate.

SOC dynamics are strongly related to precipitation. Lal (2000) observed the following differences in C sequestration potential for restorative measures on degraded rangeland: (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) semihumid and subhumid rangeland (500–1000 mm) = 0.29–0.73 t CO_2 ha^{-1} yr^{-1}.

Long-term trend analysis on rangelands shows that in wetter years, management may have little impact on soil C sequestration, but the opposite is true in drought years. Zhang et al. (2010) found that rangelands can become a C source if more than 65% of the area is in drought conditions. Net ecosystem C exchange patterns show that in U.S. rangelands, soils generally change from a C source to a C sink when moving from west (drier) to east (more moist) (Table 22). When less than 50% of the lands are experiencing drought, the range can still be a C sink (Svejcar et al. 2008). Because 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.

Table 22. Net ecosystem GHG exchange for different rangelands, 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)

Note: Positive numbers indicate net CO_2e removal from the atmosphere.
Source: Adapted from Svejcar (2008).
a. The influence of carbonates in the soils of the desert southwest causes a net C source (negative numbers).

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

Schuman et al. (2002) compiled information on the state of U.S. rangeland (grassland) from USDA-NRCS and USDI-BLM rangeland inventory and status reports and determined that 62% of this rangeland area has been poorly managed and has some constraints that limit productivity. Improving management in these areas could result in soil C sequestration; in well-managed grasslands, the soil C is relatively stable and has little potential for increase. In contrast, Conant and Paustian (2002) estimated that only 4% of all North American grassland was overgrazed. Although land in poor condition may have large C sequestration potential, even the maintenance of well-managed grasslands represents a potential 62 Mt CO_2e yr^{-1} of avoided losses, compared with shifting of grasslands to cropland (see Table 23).

Table 23. Estimated potential soil C sequestration on U.S. rangeland and potentially avoided losses

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
Potentially avoided losses (by not converting grazing land to cropland)			
Well managed	57	1.1	62
Poorly managed	113	0.7	84
CRP grasslands[c]	13	1.1	15
TOTAL			158

Note: Rates are based on the Great Plains region.
Source: 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 with conversion to a NT wheat-fallow system.

Longer-term grazing studies in the Northern Great Plains have found that where increases in SOC have occurred, species composition changes from cool season, mid-grasses to warmer-season C4 grasses (predominantly some shrubs and *Bouteloua gracilis*, Reeder et al. 2004). *B. gracilis*, with its high root to shoot ratio, stores more of its carbon below ground than other species and therefore may prompt higher soil C sequestration rates.

Few studies attempt to assess the net effect of grazing management on all three GHGs, and IPCC equations have otherwise been utilized for methane and nitrous oxide to infer the net effect. Stocking rate adjustment tends to have no effect on rangeland N_2O emissions[45] or to increase emissions by less than 0.05 t CO_2e ha^{-1} yr^{-1} (Liebig et al. 2010a; Paustian et al. 2004; Wolf et al. 2010), a minimal impact. Methane emissions from the soil are minimal in all systems, so they are not affected. Enteric fermentation CH_4 emissions are mainly affected by animal density on the land, and although improved management can reduce CH_4 emissions by lowering animal numbers, the transfer of those animals elsewhere may result in no real impact.

By way of example, Liebig et al. (2010a) conducted a Northern Great Plains case study that estimated net GHG effects for two long-term (44-year) 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 a similar methodology, Derner[46] 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 show that, depending on the system and the location, differences in net GHGs may be substantial (Table 24). In North Dakota, enteric fermentation emissions affected the net GHG flux more than the SOC change between systems; but in Wyoming, enteric emission rates were lower overall, and soil C sequestration was evident when grazing intensity was reduced.

45. J.D. Derner, personal communication, March 2010; B.H. Ellert, personal communication, March 2010.
46. J.D. Derner, personal communication, March 2010.

Table 24. Case studies showing net effects on GHG emissions or removals

Mandan, North Dakota	Moderately grazed	Heavily grazed
(44 yrs of treatment[a]) 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, Wyoming	**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

Note: Positive values indicate net CO_2e removal from the atmosphere.
a. Adapted from Liebig et al. (2010a) and Derner (personal communication, March 2010).
b. Values in parentheses indicate standard error of the mean; positive values imply net CO_2e uptake.
c. The net GWP for Mandan, North Dakota, is not significantly different at $p<=0.05$.

Improve grazing management on pasture

As in the case of rangeland, improved grazing management on pasture often (Lynch et al. 2005)—but not always (Schnabel et al. 2001)—involves reducing stocking rates. In some contexts, "improved grazing management" is used to describe agronomic inputs of fertilizer or irrigation, altered species composition, and rotational grazing—activities treated separately in this assessment. Few data document soil C change with different levels of grazing intensity for pasture, and the soil C response to reduced grazing pressure ranges from a loss of 3.0 t CO_2e ha^{-1} yr^{-1} to an increase of 4.8 t CO_2e ha^{-1} yr^{-1} (Table 25). In these data, the one instance of soil C decrease came from an unpublished study (Stuedemann et al. 1998) cited by Schnabel et al. (2001). This decrease may be an exception to the general trend of soil C gain with reduced grazing pressure. Improved pasture management may be possible on all 48 Mha of U.S. pasture.

Table 25. Estimates of soil C sequestration potential of improved grazing management on pasture, U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Stuedemann et al. (1998), as cited in Schnabel et al. (2001)	Southeastern United States, coastal bermudagrass	Reduced grazing pressure decreases C, based on soil C change when moving from 600 to 1200 grazing days/yr (mid-point)	Expert estimate, based on field study	-2.97
Follett et al. (2001a)	United States	Grazing management on pasture, assumes 10.2 Mha	Expert estimate	Low: 1.10 High: 4.77
Franzluebbers et al. (2001)	Georgia	Increased grazing intensity	Field study	0.00
Franzluebbers and Stuedemann (2009)	Georgia Piedmont, bermudagrass	Reduced grazing pressure on fescue, 30 cm depth, 12 yrs	Field study	2.42
Lynch et al. (2005)	Canadian prairies, tame pasture	Reduced stocking density	Field study	0.32

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Compared with conservation activities on harvested croplands, such activities on pasture yield higher soil C sequestration rates. The difference owes to pastures' greater allocation of plant biomass C to belowground soil C and the extended growing season, reduced soil disturbance, and better utilization of soil water. The range in sequestration rates is a reflection of regional characteristics, such as soil composition, topography, climate, and existing grass species, and net fluxes are also affected by N_2O, or CH_4 (Conant et al. 2005). As on rangelands, grazing management on pasture is assumed to have little N_2O effect. Methane emissions are affected primarily by enteric fermentation and thus grazing intensity. The challenge with pasturelands is that management factors also introduce complexity to soil-animal-plant interactions, immensely increasing the spatial variability of the analysis.

In temperate climates, most forage-based animal agriculture places grazing animals on pasture for 5 to 12 months of the year. Thus, stored forages can be an important part of the mix, in some cases the main mode of feeding. This complexity must be taken into account at the landscape level in future GHG studies (Follett and Reed 2010).

Implement rotational grazing

GHG Impact Summary

GHG category	Introduce rotational grazing on pasture	Introduce rotational grazing on rangeland
Number of observations	**4/1/1**	**3/0/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**-0.05–2.90**	**-5.27–1.90**
N_2O, t CO_2e ha^{-1} yr^{-1}	—	—
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**42**	**n/a**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Rotational grazing (also known as management-intensive grazing, MIG) differs from continuous grazing in that land is divided into paddocks, among which animals are regularly moved. This practice intensifies grazing pressure for a relatively short period of time (e.g., 1–3 days for ultra-high stocking density or 3–14 days for typical rotational grazing), leaving a rest period for regrowth in between rotations. Little research on the practice is available in North America (see Table 26), but it appears likely to lead to soil C sequestration on pasture (Conant et al. 2003). The U.S. DOE technical guidelines for voluntary GHG reporting (1605(b) program) assume a soil C sequestration rate of 2.9 t CO_2e ha^{-1} yr^{-1} under rotational grazing (U.S. DOE 2007). However, this value originates from expert estimates for *all* improved pasture management activities, which include—but are not exclusive of—rotational grazing (Follett 2001; Lal et al. 1999b). Rotational grazing on grass/legume pastures in Canada's prairie grazing land area resulted in a C sequestration rate of 0.23 t CO_2e ha^{-1} yr^{-1}, compared with 0.28 t CO_2e ha^{-1} yr^{-1}, the rate for continuous grazing (Lynch et al. 2005). Given this small (but negative) impact, additional research is necessary.

Compared with continuous grazing, rotational grazing maintains forage at a relatively younger and more even growth stage, resulting in higher-quality, lower-fiber-content forages. This lowers grazing animals' CH_4 emissions per unit of beef gain by up to 22% on highly productive pasture (DeRamus et al. 2003). Rotational grazing pasture also tends to be more productive in terms of total available forage—grass consumption nearly doubled in one study (Bosch et al. 2008)—thereby reducing the land area required for equivalent cattle weight gain (Baron and Basarb 2010; Bosch et al. 2008). With better-quality forage, open (nonpregnant) cows are less common, 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). Therefore, any elevated CH_4 and N_2O emissions resulting from increased stocking density may not be problematic if offset by efficiency gains.

Current adoption of rotational grazing is generally limited, given necessary investments in fencing, management, and labor. Surveys in dairy grazing systems in the northeastern United States found that between 13% and 19% of grazing animals were in MIG systems (Foltz and Lang 2005; Winsten et al. 2010). Using 13% as a baseline, an estimated 42 Mha of additional U.S. pasture area could be converted to rotational grazing.

Table 26. Estimates of soil C sequestration potential of rotational grazing on pasture and rangeland, U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Rotational grazing on pasture				
Conant et al. (2003)	Virginia, MIG	Four farm locations, not a side-by-side comparison	Farm-scale study	1.50
Lynch et al. (2005)	Alberta, rotational grazing	Prairies, grass-legume pasture	Modeled	-0.05
U.S. DOE (2007)	United States, rotational grazing	1605(b) technical guidelines for voluntary reporting, assumes steady soil C increase over 20 yrs	Expert estimate	2.90
Rotational grazing on rangeland				
Manley et al. (1995)	Wyoming		Field study, 2 observations	-4.67
Teague et al. (2010)	Texas	Woody savanna with herbaceous undercover, also tested burning	1	1.90

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

The response to rotational grazing may interact with water availability; the more moist pasture seems to respond more favorably than rangeland in terms of overall forage production and soil C. In contrast to continuous grazing on pasture, continuous grazing on rangeland is equal to or outperforms rotational grazing in plant production and in animal production per head and per area (Briske et al. 2008; Derner et al. 2008). In Wyoming, cattle weight gain was 6% lower under rotational grazing than under continuous grazing (Derner et al. 2008). Measured soil C response to rangeland rotational grazing varies from losses of 5.3 t CO_2e ha^{-1} yr^{-1} (Manley et al. 1995) to gains of 1.9 t CO_2e ha^{-1} yr^{-1} (Teague et al. 2010). Further research is needed to elucidate this relationship. In addition, practical implementation of rotational grazing on rangeland may be relatively difficult, with little means available for fencing and other resources due to low forage productivity per unit area.

Other grazing land management practices

On pasturelands, applying fertilizer or other inputs can increase annual net primary productivity, and soil C sequestration has been measured at rates between 0.4 t CO_2 ha^{-1} yr^{-1} and 5.9 t CO_2 ha^{-1} yr^{-1} (Table 27). Grazing land is often fertilized at lower rates than grain and row crops, but rates between 200 and 300 kg N ha^{-1} are not uncommon (Follett et al. 2001a). Lynch et al. (2005) measured a SOC gain of 0.81 t CO_2e ha^{-1} yr^{-1} on the Canadian prairies following pasture fertilization of 100 kg N ha^{-1}. Conant et al. (2005) summarized several studies to determine that an average of 6.1 kg of carbon 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 the applied fertilizer was inorganic (2.44 ± 1.40 Mt CO_2 ha^{-1} yr^{-1}), part inorganic and part organic (3.37 ± 2.12 Mt CO_2 ha^{-1} yr^{-1}), or all organic as poultry litter (3.29 ± 2.48 Mt CO_2 ha^{-1} yr^{-1}). Although fertilization may sequester carbon and reduce the overall uptake of methane (Mosier et al. 1998a), it can also stimulate N_2O emissions—effectively offsetting a substantial portion of the gains from any soil C sequestration (Lynch et al. 2005; Paustian et al. 2004). No direct data are available, but calculations using IPCC Tier I estimates suggest that 250 kg N fertilizer ha^{-1} would increase N_2O emissions by 0.7 t CO_2e ha^{-1}. Upstream emissions of 0.9 t CO_2e ha^{-1} yr^{-1} for this amount of N fertilizer would further decrease net GHG benefits.

GHG Impact Summary

GHG category	Fertilize grazing land	Irrigate grazing land
Number of observations	**7/2/1**	**8/1/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**0.37–5.86**	**0.00–1.83**
N_2O, t CO_2e ha^{-1} yr^{-1}	—	—
CH_4, t CO_2e ha^{-1} yr^{-1}	—	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	—	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	—	—
Maximum U.S. applicable area, Mha	**n/a**	**n/a**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Like fertilizer application, irrigation increases grassland productivity, particularly in dryland conditions, and thereby increases soil C inputs. With limited data available, estimates of soil C sequestration range from zero to 2.94 t CO_2e ha^{-1} yr^{-1} (Table 27). Rixon (1966) found soil C change to be highly correlated with mat production, and the lack of long-term soil C effects in a New Zealand study was possibly due to variability in land management and spatial conditions (Houlbrooke et al. 2008). Martens et al. (2005) noted that after many years of agricultural activity in Idaho, irrigated grasslands contained more SOC than native dry land (a difference of 37–147 t CO_2e ha^{-1}). If this SOC buildup takes place at a constant rate over 50 years, the soil C

GHG Impact Summary

GHG category	Manage species composition on grazing land	Establish agroforestry on grazing land
Number of observations	**9**	**1/3/0**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**1.46** (0.18–3.12)	**0.47–3.63**
N_2O, t CO_2e ha^{-1} yr^{-1}	**-0.86**	—
CH_4, t CO_2e ha^{-1} yr^{-1}	**-0.03**	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**no data**	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**0.57** (-0.71–2.23)	—
Maximum U.S. applicable area, Mha	**80**	**70**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

sequestration rate is between 0.7 t CO_2e ha^{-1} yr^{-1} and 2.9 t CO_2e ha^{-1} yr^{-1}. Irrigation water can contain dissolved carbon dioxide. If so, it changes the soil inorganic C dynamics, potentially precipitating calcium carbonate, which can be released back into the atmosphere or leached deeper into the soil profile (Martens et al. 2005; Sahrawat 2003). When considering the energy-related emissions from pumping of irrigation water and the increased N_2O emissions on irrigation (Rochette et al. 2008b), the net GHG effects of grazing land irrigation are most likely negative.

Table 27. Estimates of soil C sequestration potential of fertilizing and irrigating grazing land

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Fertilize grazing land				
Conant et al. (2001)	Global	Grassland fertilization	42 (individual data points not available)	1.10
Follett et al. (2001a)	United States	Lime and N fertilizer	Estimates based on review	0.55
Lynch et al. (2005)	Southern Canadian prairie	Concluded that net GHG effect was negative because of inputs	Modeled	0.81
Nyborg et al. (1994), as cited in Follett and Reed (2010)	Saskatchewan	N and S fertilizer	2	2.14
Reeder et al. (1998)	Wyoming	N fertilization, ungrazed grassland	2	1.75
Rice (2000), as cited in Follett and Reed (2010)	Kansas, grasslands	N fertilization	2	5.86
Schnabel et al. (2001)	Georgia Piedmont, tall fescue	High vs. low fertilization	1	0.64
Irrigate grazing land				
Rixon (1966)	Australia	Irrigation of grassland, 6 types of pasture	Field study	Low: 0.51 High: 0.94
Martens et al. (2005)	Idaho	Long-term comparison of irrigated and native lands		Low: 0.73 High: 2.94
Houlbrooke et al. (2008)	New Zealand	Irrigation of grassland; no significant impact		0.00

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Species composition can serve an important role in C sequestration on both rangeland and pasture. Data from nine field comparisons (Table 28) revealed that seeding of improved grass or legume species resulted in an average soil C gain of 1.5 t CO_2e ha^{-1} yr^{-1} (a range of 0.2 t CO_2e ha^{-1} yr^{-1} to 3.1 t CO_2e ha^{-1} yr^{-1}). Measured increases in N_2O emissions with overseeding on rangeland (Liebig et al. 2010a) lead to a net GHG mitigation potential of 0.6 t CO_2e ha^{-1} yr^{-1}.[47] With both pasture and rangeland possible subjects of overseeding or interseeding, this activity could be used to store carbon on as much as 80 Mha of land.

Soil C storage rates tend to decrease over time. For example, Mortenson et al. (2004) measured gains of 1.2 t CO_2 ha^{-1} yr^{-1}, 2.4 t CO_2 ha^{-1} yr^{-1}, and 5.7 t CO_2 ha^{-1} yr^{-1} in soil carbon 36, 14, and 3 years, respectively, after alfalfa interseeding on a northern mixed-grass rangeland in South Dakota. Separating the soil C impact of species composition changes from other activities may be difficult, because grazing behavior and grazing intensity are very interlinked with species composition.[48] 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 with pure grass stands (McCaughey et al. 1997), and lower N_2O emissions from legumes compared with grasses (Rochette et al. 2004).

Because increases in stocking rates lead to increases in enteric fermentation and thus to increases in CH_4 emissions, researchers are exploring the link between rumen methane and maintaining forage of a certain quality. Seeding legumes to pasture or otherwise improving the quality of grazed forage can reduce CH_4 emissions by more than 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.

47. This N_2O emission response to overseeding was the only available example, and other systems may not react in the same way. Further research is needed to confirm that overseeding has such a significant effect on N_2O flux.
48. V. Baron, personal communication, April 2010.

Table 28. Estimates of soil C sequestration potential of managing species composition on grazing land and agroforestry on grazing land (silvopasture), U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Manage species composition on grazing land				
Follett et al. (2001a)	United States, plant improved species	Estimates based on review	Expert estimate	*Low: 0.37* *High: 1.10*
Conant et al. (2001)	Plant improved species	Global review, eliminated tropical observations for this research	2 1	Legumes: 1.31 Grasses: 0.48
Mortenson et al. (2004)	Interseed native rangeland with legume	South Dakota, sequestration rate decreased over time, 3 to 36 yrs	3	3.11
Lynch et al. (2005)	Canadian prairie, seeded grasslands and legumes	Low is continuously grazed, high is rotationally grazed	2	0.25
Liebig et al. (2010a)	North Dakota, seeded with wheatgrass and heavily grazed	44 yrs	1	0.18
Establish agroforestry on grazing land				
Dixon (1995)	United States	Humid temporal low (A) and dry lowlands (B), 25% of C storage is below ground.	Expert estimate	A: 2.77 B: 2.43
Nair and Nair (2003)	U.S. estimates	Assumed 70 Mha of land	Expert estimate	0.47
Sharrow and Ismail (2004)	Oregon	Compared with grassland pasture, 12% of C storage is above ground, 11 yrs	Field study	1.68

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Through soil C and aboveground C storage, silvopasture (trees planted on grazing land) may also have GHG mitigation potential on up to 70 Mha of grazing land (Nair and Nair 2003). With few field research data, the estimated soil C sequestration rates of 0.5 t CO_2 ha^{-1} yr^{-1} to 3.6 t CO_2 ha^{-1} yr^{-1} (Table 28) are largely based on expert opinion. Therefore, further assessments of the effects on life-cycle GHG balance are warranted.

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 the overall health and growth of rangelands; for example, in tall grass prairie, increased plant productivity after the burn more than compensates for the loss of plant carbon by ignition. However, most studies found that SOC stays about the same or even decreases following repeated burns (Rice and Owensby 2001). Furthermore, other negative co-effects (methane, smoke, aerosols) are also linked to climate change, making burning even less attractive as a GHG mitigation option (Smith et al. 2008). Therefore, anecdotal evidence and the lack of side-by-side comparison data make rangeland fire management a poor candidate for GHG mitigation.

Specific activities may also have the potential to reduce N_2O emissions from grazing land, but few, if any, data are available for quantification. Soil compaction by grazing action can significantly increase N_2O emissions (Bhandral et al. 2007), but grazing on NT (versus recently tilled) pasture or cropland or during low field-water capacity conditions can reduce these emissions (Thomas et al. 2008). Improved manure and mineral N management, including reducing the N content in animal feed (Mosier et al. 1998b), may also lower N_2O emissions, but a lack of data precludes any estimate of impact.

Convert Cropland to Pasture

Converting cropland to perennial grass or legume pasture can increase soil C by 2.4 t CO_2e ha^{-1} yr^{-1} (a range of -0.4 t CO_2e ha^{-1} yr^{-1} to 4.2 t CO_2e ha^{-1} yr^{-1}). The greater total production achieved with perennials, as opposed to annuals, plus the trampling and fertilizing related to grazing activity provide mechanisms for this soil C sequestration. Just over half of the 26 observations used in this estimate are from the Southeast (Franzluebbers et al. 2000; Franzluebbers 2010); higher rates observed in the Southeast compared with other regions (an average of 2.9 t CO_2e ha^{-1} yr^{-1} versus an average of 2.1 t CO_2e ha^{-1} yr^{-1}) are likely due to greater total yearly biomass productivity (see Table 29). Including other GHG categories, net GHG mitigation potential is estimated at 3.1 t CO_2e ha^{-1} yr^{-1}.

GHG Impact Summary

GHG category	Convert cropland to pasture	Set aside grazing land
Number of observations	**26**	**28**
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**2.39** (0.40–4.18)	**-0.53** (-2.84–0.80)
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.46**	—
CH_4, t CO_2e ha^{-1} yr^{-1}	**-0.25**	—
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.45**	—
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**3.06** (1.07–4.85)	—
Maximum U.S. applicable area, Mha	**unknown**	**n/a**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.

Grassland most often (Desjardins et al. 2005; Grant et al. 2004; Kessavalou et al. 1998), but not always (Stehfest and Bouwman 2006) experiences lower N_2O emissions than cropland in the same location, with average savings of 1.0 t CO_2e ha^{-1} yr^{-1}. Methane emissions from the land are not affected (Falloon et al. 2004; Kim et al. 2010), but enteric fermentation increases CH_4 flux by approximately 0.2 t CO_2e ha^{-1} yr^{-1} (Liebig et al. 2010a; J. Derner, personal communication, March 2010). Further GHG mitigation can come from reductions in fuel use and upstream GHG costs (fertilizer and other inputs). By converting from cropland to pasture, the associated fuel use for tillage, harvesting, and planting can be brought close to zero, reducing GHG emissions by approximately 0.4 t CO_2e ha^{-1} yr^{-1}. Fertilizer use on pasture tends to be somewhat lower than on cropland, but because N fertilizer rates on pasture can range from occasional (Machado et al. 2006) to 600 kg N ha^{-1} yr^{-1} (intensively grazed pasture in New Zealand, Bhandral et al. 2007), the differences are difficult to assess. This assessment assume a conservative 25% reduction in total N fertilizer for a reduction in upstream emissions of 0.1 t CO_2e ha^{-1} yr^{-1}.[49] On the other hand, emissions leakage may occur due to displaced crop production. For landowners, the possibly lower agricultural productivity may make conversion to pasture feasible only on marginal cropland.

Table 29. Estimates of soil C sequestration potential for converting cropland to pasture, U.S.

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Lal (2003)	Cropland to pasture	United States, review, assumed 4.8 Mha	Expert estimate	*Low: 1.47 High: 4.40*
Murray et al. (2005)	Cropland to grassland	United States, rates from CRP	Expert estimate	*Low: 2.22 High: 4.70*
McPherson et al. (2006)	Cultivated soils to perennial grass cover	Colorado and Kansas; used Comet VR to generate potential at MLRA scale	Modeled	*4.58*
Franzluebbers et al. (2000)f	Georgia Piedmont, convert hay bermudagrass to grazed	Average of 16 yrs	1	1.58
Post and Kwan (2000)	United States, cropland to seeded grassland	Review, studies from Wyoming and South Dakota	6	1.12
Potter (2006)	Texas, cropland to pasture	39 yrs and 55 yrs	2	1.28
Franzluebbers (2010)	Southeastern United States, CT cropland to perennial pasture	Average of 25 yrs	17	3.02

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

49. Therefore, the assumption is that pasture receives only 75% of the fertilizer applied to cropland.

Set Aside Grazing Land

Available data suggest that grazing land set-aside is not generally a viable option for GHG mitigation. Annual forage productivity is often greater in grazed than in ungrazed grasslands and pasture (Franzluebbers et al. 2004; Haan et al. 2007), and land with appropriately managed grazing in most cases stores more soil C than ungrazed natural grassland (Table 30). From 28 field observations, the average change in soil C was a decrease of 0.5 t CO_2e ha^{-1} yr^{-1} (a range of -2.8 t CO_2e ha^{-1} yr^{-1} to 0.8 t CO_2e ha^{-1} yr^{-1}) after cessation of grazing.

Table 30. Estimates of soil C sequestration potential of setting grazing land aside, U.S. and Canada

Citation	Region	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Conant et al. (2001)	Global	Grazing sequesters soil C	Expert estimate, based on review	*-1.28*
Conant and Paustian (2002)	Canada and Great Plains	Review	4	-0.72
Liebig et al. (2005b)	Great Plains	Review	2	-0.20
Martens et al. (2005)	Western United States	Review, allow shrub (mesquite) encroachment on arid rangeland, high variability (-2.9 to 1.2)	14	-0.18
Derner and Schuman (2007)	Great Plains	Review, semi-arid grassland	4	-0.71
Smoliak et al. (1972)	Alberta		1	-0.31
Manley et al. (1995)	Wyoming	High variability (-6.9 to 0.3)	1*	-3.29
Reeder and Craft (1999), as cited in Franzluebbers (2005)	North Carolina	Coastal marshland, horse grazing reduced SOC	1	1.77
Reeder et al. (2004)	Colorado	More soil C in grazed area (66% of gain was inorganic)	1	-1.56

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Grazing activity increases soil C by stimulating shoot and root growth (Haan et al. 2007; Reeder et al. 2004) as well as organic acid root exudation, the latter of which can increase inorganic C in arid rangeland soils through carbonate precipitation (Reeder et al. 2004). Grazing activity also facilitates litter decomposition to SOC by removal of aboveground biomass and churning of surface soil by animal hooves. Removal of excess aboveground material regenerates root growth and hastens the onset of spring regrowth and photosynthesis (LeCain et al. 2000). Grazing returns the majority of nutrients back to the soil through excreta (Schnabel et al. 2001). On ungrazed pastures, vegetation breakdown may increase runoff and erosion (Webber et al. 2010).

However, unlike the response on the native grasslands of the Great Plains, eliminating grazing on coastal marshlands or on the arid rangeland of the Southwest may have a positive SOC impact. Reeder and Craft (1999, cited in Franzluebbers 2005) measured a soil C decrease of 1.8 t CO_2e ha^{-1} yr^{-1} on grazed coastal marshland in North Carolina. Shrub encroachment in arid areas may also store soil carbon. Of 11 studies comparing areas with mesquite and other leguminous woody plants to neighboring grassland, 9 found higher SOC in the shrub/mesquite area, and the authors concluded that the data suggest "an east to west gradient of C accumulation under shrubs across the southwestern USA" (Martens et al. 2005). In earlier work, Milchunas and Lauenroth (1993) reviewed more than 200 site observations in which soil C responses to grazing were almost equally positive and negative. Grazing activity may be a particular problem for soil C storage when a moisture deficit limits production (Schnabel et al. 2001).

Information on the N_2O and CH_4 flux effects of grazed versus ungrazed grazing land (i.e., grazing land that has been set aside) is lacking, although studies have shown that urine deposition from cattle can increase N_2O emissions (Liebig et al. 2005b). The soil of grazed grassland may capture more methane than that of ungrazed land (Franzluebbers 2005), but setting grazing land aside reduces enteric fermentation emissions, at least locally. Because any cattle moved from the pasture or rangeland will likely be grazed elsewhere, leakage may also need to be considered. Given that the net GHG impacts of grazed versus ungrazed land are so variable and regionally dependent, non-GHG considerations may dominate decisions to convert pasture or rangeland to ungrazed natural grassland. These considerations 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).

Set Aside Cropland or Plant Herbaceous Buffers

Setting cropland aside for unharvested perennial vegetation can provide multiple environmental benefits, including soil C sequestration, provision of wildlife habitat, erosion prevention, water quality protection, and aesthetics. Such set-aside can take the form of herbaceous buffers (grass strips) within a field or along a riparian area or consist of larger tracts of land. On the basis of 28 field comparison data points (Table 31), the average soil C sequestration rate is 2.0 t CO_2e ha^{-1} yr^{-1} (a range of -0.4 t CO_2e ha^{-1} yr^{-1} to 5.1 t CO_2e ha^{-1} yr^{-1}). With reduced N_2O and upstream and process emissions, the net GHG mitigation potential of setting cropland aside is 3.6 t CO_2e ha^{-1} yr^{-1}. A significant amount (13 Mha) of former cropland has already been taken out of production through the Conservation Reserve Program (CRP), and more than 1 Mha of land is enrolled in buffers through the Natural Resources Conservation Service and other state incentive programs. Experts estimate an additional 9–25 Mha of sensitive or marginal cropland could be beneficially set aside from agriculture (Bruce et al. 1999; Sperow et al. 2003). This assessment assumes that 17 Mha (the midpoint) of cropland could be for this retirement.

GHG Impact Summary

GHG category	Set aside cropland or plant herbaceous buffers	Restore wetlands
Number of observations	**28**	**70**[a]
Soil carbon, t CO_2e ha^{-1} yr^{-1}	**1.98** (-0.37–5.07)	**6.52** (-0.96–9.89)
N_2O, t CO_2e ha^{-1} yr^{-1}	**0.84**	**0.00**
CH_4, t CO_2e ha^{-1} yr^{-1}	**0.00**	**-3.33**
Process and upstream emissions, t CO_2e ha^{-1} yr^{-1}	**0.74**	**0.74**
Sum of GHGs, t CO_2e ha^{-1} yr^{-1}	**3.57** (1.22–6.66)	**3.94** (-3.54–7.31)
Maximum U.S. applicable area, Mha	**17**	**3.8**

Positive numbers depict removal of greenhouse gases from atmosphere or prevented emissions. Tables 33 and 34 on pages 50 and 51 compare all practices.
a. These experiments were not controlled side-by-side comparisons. Rather, they compared restored wetland sites with currently cropped or undrained wetlands, matching locations with similar characteristics.

Table 31. Estimates of soil C sequestration potential of setting cropland aside or planting herbaceous buffers, U.S. and Canada

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential (t CO_2e ha^{-1} yr^{-1})[a]
Bruce et al. (1999)	U.S. general		Expert estimate	*2.93*
Follett and Kimble (unpublished, as cited by Lal et al. 1999b)	CRP	5-yr average, 10-cm depth, 10 sites in 8 states	n/a	*2.93*
Sperow et al. (2003)	United States, convert highly erodible land to perennial grass	Modeled, assuming removal of 25.8 Mha cropland from production	Modeled	*1.49*
Lal et al. (2003)	United States, conservation buffers United States, additional CRP			*Low: 1.10* *High: 2.57* *Low: 2.20* *High: 3.30*
Gebhart et al. (1994)	Texas, Kansas, Nebraska; CRP	300-cm depth	5	3.34
Burke et al. (1995)	From cultivated to abandoned field	Colorado, 10 yrs	1	0.11
Reeder et al. (1998)	Wyoming, cropland to ungrazed "pasture"	6 yrs	2	1.26
Follett (2001b)	United States, convert to CRP	Great Plains, Rocky Mountains, Corn Belt, and Lake States locations, 8-yr average	8*	1.44
Gregorich et al. (2001)	Ontario, continuous bluegrass vs. corn	Grass not harvested, 35 yrs	1	4.74
Johnson et al. (2005)	U.S., cropland to grass in CRP	United States	5	2.06
Bailey et al. (2009)	Corn Belt, grass buffer strips	13 yrs, 10-cm depth	3	0.31
Franzluebbers and Stuedemann (2009)	Georgia Piedmont, unharvested land	150 cm depth, 12 yr study	3	2.87

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

The unharvested vegetation in set-aside land sequesters carbon in two ways: through retention of sediment from agricultural runoff and through capture and sequestration in biomass. More than 200 Mt of sediment is captured annually by buffers and vegetation planted for the CRP (USDA 2008). The physical potential of set-aside areas to sequester carbon depends on their size, vegetation, former land use, and structure, making generalizations difficult. The USDA estimates a national soil C gain of 48 Mt of CO_2e yr^{-1} through the CRP program alone; an additional 9 Mt CO_2 yr^{-1} could be offset through energy and fertilizer savings (USDA 2008). Planting herbaceous vegetation can be more appealing to farmers than planting trees due to the lower capital investment and labor entailed by the former. This vegetation is also easier to remove once a program ends, easing implementation, but also raising concerns about long-term C sequestration (permanence).

Kim et al. (2010) measured CH_4 flux in three types of buffer vegetation and adjacent cropland and found only small differences between the flux on cropland and that on adjacent riparian buffers. However, the N_2O emissions reduction from set-aside cropland can be significant (Kessavalou et al. 1998; Mummey et al. 1998), and buffers can also reduce N_2O emissions by capturing NO_3^- before it reaches surface water or groundwater and is denitrified off site (DeSimone et al. 2010). The extent to which this benefit can be achieved will depend on the characteristics of the buffer and nitrogen transfers. Different buffers have varying capacities to capture nitrogen and lose it as N_2O. Hefting et al. (2003) found that in conditions of high lateral nitrate loading (4,700 kg N ha^{-1} yr^{-1} in the Netherlands), forested buffers emitted 10 times more nitrous oxide (both in total quantity and as a proportion of total nitrogen) than grass buffers. Eliminating N fertilizer will also reduce land-based N_2O emissions and contribute to upstream GHG savings. The net N_2O impact will depend on baseline emissions in the land to be removed from production. Given this high variability and multiple influential factors, generalizing the N_2O emissions reductions for a typical buffer is difficult—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. As with any land-use change, the production decrease likely increases costs and may carry other nonprice disincentives for landowner participation.

Wetland Restoration

Often—but not always—comprised of organic soils (histosols), wetlands in North America contain large amounts of stored carbon and are estimated to sequester up to 180 Mt CO_2e yr^{-1} (Bridgham et al. 2006). Wetlands are highly variable with respect to amount—and characteristics—of organic matter, water level, vegetation, and other factors. Whether U.S. wetlands on the whole are net GHG sources or sinks is unknown, because uncertainties in all relevant GHG flux estimates are large (Bridgham et al. 2006). What is well understood is that draining wetlands—often for agricultural purposes—changes the balance of emissions so that CH_4 emissions nearly cease, while CO_2 emissions grow due to very high SOC oxidation rates. Restoration of wetlands can reverse this effect. Setting cropped histosol aside has been discussed above; this section focuses on nonhistosol wetlands.

The GHG impacts of wetland restoration can be determined by comparing the GHG balances of formerly cultivated land that has been restored with land still in cultivation. Experts have estimated that wetland restoration can generate soil C sequestration at a rate of approximately 1.5 t CO_2e ha^{-1} yr^{-1} (IPCC 2000; Lal et al. 2003). However, data from more recent studies suggest that the rate may be higher: an average of 6.5 t CO_2e ha^{-1} yr^{-1} (a range of -1.0 t CO_2e ha^{-1} yr^{-1} to 9.9 t CO_2e ha^{-1} yr^{-1}) in the Prairie Pothole region of southwest Minnesota, North and South Dakota, northwest Iowa, and northeast Montana (see Table 32). Rather than conducting side-by-side experiments, these studies compared restored wetlands with cropland (8 comparisons, Gleason et al. 2009) and with reference undrained wetlands (62 comparisons, Badiou et al. 2011; Euliss et al. 2006). Significant variability in wetland types leads to significant variability in soil C changes among sites. For example, Euliss et al. (2006) found high soil C sequestration rates in semipermanent wetlands but little to no accrual in seasonal wetlands. Lal et al. (2003) suggest that a 19 Mha of histosol plus wetland area is available for restoration. With histosols addressed earlier, this assessment makes the conservative assumption that the total U.S. land area available for wetland restoration is that of the prairie pothole region (3.8 Mha).

Table 32. Soil C sequestration potential of wetland restoration from agricultural land

Citation	Region and crop type	Comments or caveats	Number of comparisons[a]	Potential ($t\ CO_2e\ ha^{-1}\ yr^{-1}$)[a]
Restore wetlands				
IPCC (2000)	Global, wetland restoration		Expert estimate	*Low: 0.37* *High: 3.66*
Lal et al. (2003)	United States, wetland reserve program		Expert estimate	*Low: 0.73* *High: 1.10*
Euliss et al. (2006)	Wetland restoration	Compared restored wetlands to reference undisturbed wetlands	40	5.59
Badiou et al. (2010)	Canada, restore prairie pothole wetlands	Compared restored wetlands (2–8 yrs old) to reference upland sites	22	7.70
Gleason et al. (2009)	North Dakota, cropped wetland restored to grass in CRP	Prairie potholes	8	1.91

a. For explanation about notations and values in the last two columns, see footnote (a) from Table 1.

Wetland restoration likely has few implications for N_2O emissions (Badiou et al. 2010; Gleason et al. 2009), but may increase CH_4 emissions (Badiou et al. 2010) or have no effect (Gleason et al. 2009). The net GHG benefit is estimated to be 3.9 t CO_2e ha^{-1} yr^{-1}. However, in some cases, net GHG emissions increases may occur. One study found that 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).

Comparison of Mitigation Activities' Biophysical Potential

Tables 33 and 34 present a side-by-side comparison of the biophysical potential of the agricultural GHG-mitigation activities considered here. The estimates are U.S. averages, and variability is a result of regional, soil, climate, and crop differences as well as uncertainty in existing measurements and other determinations of soil carbon or GHG flux. The tables also indicate the maximum area in the United States to which each activity is applicable; limited land base and competing land uses make it probable that not all activities can achieve this total area. Therefore, it is not reasonable to calculate the maximum national GHG mitigation potential using these estimates. Economic analysis and assessment of co-effects and other modifying factors affect the competition among, and the prioritization of, agricultural land management activities for GHG mitigation. Any attempt to determine total mitigation potential should also consider these factors.

This assessment identified 20 agricultural land management activities with significant or moderate levels of research and that are likely beneficial for GHG mitigation (Table 33), although certain regions or issues may require further investigation. Four of these activities—convert cropland to pasture, plant SRWCs, set aside cropland, and restore wetlands—have relatively high mitigation potential (net > 3 t CO_2e ha^{-1} yr^{-1}) but have limited applicable area and require significant changes to cropping systems. The other 16 activities tend to have lower mitigation potential but are more widely applicable and often maintain the current cropping system. Table 33 itemizes the estimates of target and other GHG impacts (including soil C changes, N_2O emissions, CH_4 emissions, and upstream and process emissions) as well as net GHG mitigation potential.

The other 22 of the 42 activities investigated (Table 34) appear to have low or negative mitigation potential, lack supporting research or have inconsistent supporting data, or raise life-cycle GHG concerns. For these activities, a range of the target GHG effects from available field data, model estimates, and expert assessments are presented. The first eight activities (increase cropping intensity, introduce agroforestry on cropland and grazing land, improve irrigation, manage histosols or set them aside, improve manure application for N_2O emissions reduction, and introduce rotational grazing) appear to have significant GHG mitigation potential on the basis of the limited information available. Further research is needed to confirm this potential. One other activity, the application of biochar, merits special attention; its potential to sequester soil carbon and to offset fossil fuels makes it attractive, but the lack of field data and the high uncertainty regarding life-cycle greenhouse gases limit its implementation. For the thirteen remaining activities, mitigation potential is uncertain, low, or negative. The six uncertain activities, for which information is lacking or variability of mitigation potential is high, may deserve additional attention.

The interaction of multiple management activities implemented on one parcel of land may modify the biophysical GHG mitigation potential of each activity. The GHG implications of some such interactions—elimination of tillage with fallow reduction (Sainju et al. 2006a), conservation tillage with use of cover crops (Franzluebbers 2010; Parkin and Kaspar 2006), tillage reductions with crop diversification (Lal et al. 1999b; Sainju et al. 2006a), and crop diversification that includes winter cover crops (Liebig et al. 2010b)—have been documented. Some studies have examined numerous combined activities within complex systems. For example, Drinkwater et al. (1998) assessed the carbon balance in three systems with different crop rotations, N fertilizer sources, and chemical application rates, with and without cover crops. Wagner-Riddle et al. (2007) compared N_2O emissions from two systems that differed in tillage, N rate, N timing, and cover crop use. With input data from existing research, biogeochemical models can also provide estimates of GHG fluxes for numerous combinations of activities.

This assessment identifies several research and data gaps with implications for the incentivization of GHG-mitigating activities. The remaining gaps in the well-researched activities listed in Table 33 are top research priorities. These gaps include the response of soil C at depth to different tillage intensities in various regional, soil, or crop contexts; the soil C response to winter cover crops in different regions; the GHG implications of altering field activities to include a winter cover crop in crop rotations; and the baseline N management practices and the potential for N rate reductions or other activities to mitigate N_2O emissions without decreasing yield.

Of the activities lacking research, grazing management, rotational grazing, and other grazing land activities may deserve prioritization, especially given that the large land area on which they could be implemented could yield significant GHG mitigation potential. Manure and biochar application also warrant further attention, because they appear to have great soil C sequestration potential but uncertain life-cycle GHG implications. Research is needed to clearly assess the availability of "excess" manure and the soil C effect of removing residue for biochar and of not applying manure at the "source" location.

Table 33. Greenhouse gas mitigation potential of U.S. agricultural land management activities that have positive GHG mitigation potential and significant or moderate research coverage

Activity	Soil carbon	N_2O emissions	CH_4 emissions	Process & upstream emissions	National Total	Max area	Comments
	Mean (range); t CO_2e ha^{-1} yr^{-1}					Mha	
Significant research							
Switch to no-till	1.22* (-0.24–3.22)	0.12	0.01	0.12	1.47 (0.01–3.46)	94	N_2O emissions, which are well studied, depend on soil and climate.
Switch to other conservation tillage	0.44 (-0.54–1.38)	0.18	0.00**	0.08	0.70 (-0.29–1.63)	72	Soil C change varies by region.
Eliminate summer fallow†	0.60* (-0.22–1.20)	-0.03	0.00	-0.12	0.44 (-0.38–1.05)	20	Process and upstream emissions depend on N fertilizer rates for crop replacing fallow.
Use winter cover crops	1.34 (-0.07–3.22)	0.12	no data	0.46	1.92 (0.51–3.81)	66	This activity can reduce need for fertilizer N, but it may require timing changes for the main crop.
Diversify annual crop rotations	0.00* (-1.69–1.66)	0.17	0.00	0.00	0.17 (-1.52–1.83)	46	Net primary productivity is the key factor.
Include perennials in crop rotations	0.52 (-0.01–1.20)	0.03	0.00	0.17	0.71 (0.19–1.39)	56	
Switch to short-rotation woody crops††	2.51 (-7.34–13.26)	0.76	no data	0.65	3.92 (-5.93–14.67)	40	Upstream emissions do not include end use. Negative soil C results are limited to studies of less than six years.
Convert cropland to pasture††	2.39 (0.40–4.18)	0.46	-0.25	0.45	3.06 (1.07–4.85)	no data	The total area is uncertain.
Set aside cropland or plant herbaceous buffers††	1.98 (-0.37–5.07)	0.84	0.00	0.74	3.57 (1.22–6.66)	17	This activity excludes histosols. Differences in types of land for restoration result in a wide range of mitigation potential.
Reduce fertilizer N application rate by 15%††	no data	0.28 (0.03–0.82)	no data	0.06	0.33 (0.08–0.88)	68	
Adjust rice water management	-0.04	-0.79	1.97 (0.08–5.31)	no data	1.14 (-0.75–4.48)	1.3	U.S. studies are augmented with international data.
Moderate research							
Replace annuals with perennial crops††	0.67 (-0.86–2.00)	0.24	0.00	0.52	1.43 (-0.10–2.76)	13	
Restore wetlands††	6.52 (-0.96–9.89)	0.00	-3.33	0.74	3.94 (-3.54–7.31)	3.8	
Manage species composition on grazing land†	1.46 (0.18–3.12)	-0.86	-0.03	no data	0.57 (-0.71–2.23)	80	Emissions of N_2O and CH_4 are based on one study.
Switch fertilizer N source from ammonium-based to urea	no data	0.59 (0.03–1.47)	no data	no data	0.59 (0.03–1.47)	37	
Switch to slow-release fertilizer N source	no data	0.12 (0.04–0.21)	no data	0.06	0.18 (0.10–0.26)	93	Assuming less fertilizer N is used, upstream emissions will be reduced.
Change fertilizer N placement	no data	0.25 (0.00–0.69)	no data	no data	0.25 (0.00–0.69)	63	
Change fertilizer N timing	no data	0.18 (0.00–0.53)	no data	no data	0.18 (0.00–0.53)	53	
Use nitrification inhibitors	no data	0.41 (0.02–1.04)	no data	no data	0.41 (0.02–1.04)	92	
Plant rice cultivars that produce less CH_4	no data	0.00	0.97 (0.06–1.87)	0.00	0.97 (0.06–1.87)	1.3	U.S. studies are augmented with international data.

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

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

** Cells that are shaded indicate limited scientific data available (i.e., the estimate is based on expert opinion or on three or fewer field or laboratory comparisons).

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

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

Table 34. Greenhouse gas mitigation potential of U.S. agricultural land management activities that have significant research gaps, life-cycle GHG concerns, and low or negative GHG mitigation potential

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

Activity	Target	GHG benefits mean (range)	Max area	Comments
		t CO_2e ha^{-1} yr^{-1}	Mha	
Irrigate grazing land†	soil C	0.00–1.83	n/a	GHG costs of irrigation equipment and pumping may negate soil C gains. N_2O emissions are also higher with irrigated land.
Reduce rice area††	CH_4	2.32–10.26	1.3	Impact depends on subsequent land use and conditions for displaced rice production elsewhere.
Low or negative GHG mitigation potential for target GHG				
Reduce chemical use (other than N)	upstream/ process emissions	0.03–0.06	122	
Set aside grazing land††	soil C	-2.84–0.80**	unknown	Soil C response data are highly variable.

Note: The range indicates the minimum and maximum values for the target GHG from field comparisons, expert estimates, and model estimates, as available.
† These activities may increase agricultural productivity in the project/program area and thus result in positive leakage.
†† These activities may decrease productivity in the project/program area and thus result in negative leakage (production shifts elsewhere).
* The total area is not estimated for activities where net GHG effect is negative.
** The 80% range of 28 observations is presented. The mean is -0.53 t CO_2e ha^{-1} yr^{-1}.

Specialty Crops

U.S. farmers 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).[50] Specialty crops may also be viewed simply as 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 2007b). According to the 2007 Agricultural Census, 247,772 farms were growing specialty crops on a total harvested area of 3.9 Mha (2.0 Mha for orchards and 1.9 Mha for vegetables). This area equals 3.2% of total U.S. harvested cropland.

The farmgate value (cash receipts) of specialty crops forecasted for 2010 was approximately $83 billion—52% of a total U.S. crop value of $160 billion (USDA ERS 2010c). 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 2009b; Western Growers Association n.d.). The top five fruit, vegetable, or nut commodities produced in the United States are grapes, potatoes, lettuce, tomatoes, and almonds (Western Growers Association n.d.).

Much impetus for GHG mitigation action in specialty crops has come 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 could be applicable to specialty crops, including many that sequester soil C, reduce N_2O and CH_4 emissions, or both. However, mitigation potential values applicable to corn or wheat, for example, cannot be directly translated to specialty crops. Perhaps the most significant hurdle to overcome with specialty crops is that the mitigation potential of different activities can vary by crop, making determination of the optimal techniques for GHG mitigation difficult. Achieving a measurable soil C increase in specialty crops may also be challenging due to specialized field management practices requiring tillage, diverse rotations, and optimized timing for bringing crops to market (Morgan et al. 2010). For instance, the nature of the planting and harvesting of some vegetables, potatoes, and sugar beets results in frequent and intensive soil disturbance, which can increase N mineralization and possibly limit C sequestration opportunities (Freibauer et al. 2004).

Nevertheless, the limited research available suggests that some soil C storage potential exists in shifting practices for specialty crops. One study showed that 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 another study showed that

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

cover cropping and elimination of tillage increased soil C in California vineyards (Steenwerth and Belina 2008). In contrast, although cover cropping in a Mediterranean tomato-cotton rotation in California increased soil carbon, elimination of tillage did not (Veenstra et al. 2007); and in a tomato system in Georgia, elimination of tillage only increased soil C when combined with cover cropping and N fertilization (Sainju et al. 2002). Compost substitution for synthetic fertilizer may also have GHG mitigation potential, as shown in a maize-vegetable-wheat rotation in Pennsylvania where over nine years of compost application resulted in a 16% to 27% soil C increase compared with a soil C decrease with synthetic fertilizer application (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% to 95% after three years, increasing cropping efficiency and thereby creating potential for reverse leakage as well as increased soil C and reduced soil nitrate levels (which likely translates to N_2O emissions reduction). Reduced chemical use in such systems can also have a small but beneficial GHG impact, with little to no yield-reduction effect (Clark et al. 1998a).

Table 35. Nitrogen fertilizer applied on top specialty crops, 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, California	Smith et al. (2009a; 2009b); assuming 2 crops per calendar year
Head lettuce	76	289	California and Arizona	USDA NASS (2007a); 70,400 hectares
Lettuce, broccoli, celery		124–371	California	Burger et al. (2009)
Broccoli	84	242	California	USDA NASS (2007a); 52,000 hectares
Tomatoes, processing		124–297	California	Burger et al. (2009)
Tomatoes, fresh	29	242	California, Florida, Georgia, New Jersey, North Carolina, Ohio, Tennessee	USDA NASS (2007a); 42,700 hectares
Almonds		22–313	California	Freeman et al. (2008); higher rates are for producing years, lower for establishment years
Grapes		6–56	California	Vasquez et al. (2007); higher rates are for producing years, lower for establishment years
Fall potatoes	58	242	United States	USDA NASS (2007a); 138,000 hectares

Management practices such as irrigation and precision agriculture that are used for all crops—but more commonly for specialty crops—can also affect N_2O and other GHG emissions. Cover cropping may also decrease N_2O emissions in some systems; in a study of lettuce in a Midwestern sandy loam, N recovery was double in the cover-cropped system than in a winter bare-soil system (Wyland et al. 1995). Understanding of the GHG mitigation potential of specialty crops alone hinges in part on discovering how widespread these alternative management practices are for specialty crops.

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. For example, UC Cooperative Extension cost and return studies (Brittan et al. 2008; Frate et al. 2008; Smith et al. 2009a; Smith et al. 2009b) estimate N fertilizer application for lettuce at nearly 600 kg N ha^{-1} yr^{-1} compared with 280 kg N ha^{-1} yr^{-1} for corn, even though the N removal of lettuce is significantly below that of 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 United States and globally, fruits and vegetables use 4.4% and 15.6%, respectively, of total N fertilizer (Heffer 2009); application rates of up to 500 kg N ha^{-1} yr^{-1} are not uncommon for some crops (Table 35). Other nongrain/oilseed/cotton/sugar crops (which can include pasture and pulses, but also some specialty crops) use 24.2% and 16.0%, respectively, in the United States and globally. No evidence suggests that emissions factors for specialty crops vary significantly from those for corn, wheat, and other field crops; N_2O emissions are most likely affected by C substrate and N availability as well as soil moisture conditions. Thus, based on N fertilizer use alone, U.S. specialty crops could be responsible for 5% to 20% of fertilizer-related N_2O emissions from agriculture and perhaps a similar proportion of emissions from legume and manure- or compost-derived nitrogen. High fuel use rates for some specialty crops may provide scope for efficiency improvements to generate lower process and upstream emissions.

The sheer diversity of crops and differences in the extent to which alternative management practices affect GHG emissions make quantifying these emissions a huge challenge. Biogeochemical process-based models can be used for many crops and can track the GHG emissions effects of interactions among many management practices, but validating the models for each crop type in a variety of environments may be prohibitively expensive. However, modelers indicate that a significant amount of data on the GHG impacts of specialty crop systems and their management already exists and that these data are being incorporated into models at an accelerated rate.[51] Discussion and comparison of three representative biogeochemical models can be found in the supplemental T-AGG report *Selecting and Setting Up Process-based Models for Tier-2 or Tier-3 Quantification of Agricultural Greenhouse Gases.*

GHG Impacts of Plant Breeding and Biotechnology Advances

Biotechnology is defined as "any technological application that uses biological systems, living organisms, or derivatives thereof to make or modify products or processes for a specific use" (UNCBD (UN Convention on Biological Diversity) 2010). It 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 of agricultural efficiency and have fostered as much as 591 Gt CO_2e emissions avoidances since 1961 (Burney et al. 2010). However, due to ever-increasing demand, higher yields do not always correlate with reductions in agricultural land use or preclude agricultural expansion (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 the 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; plant breeding has contributed approximately 50% of the increase and improved management has resulted in the other 50% (Duvick 2005). Although much of the discussion about increased future yield potential centers around GE crops, some reports suggest that these crops 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 the 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 genetic engineering) has also improved lodging resistance in corn (Flint-Garcia et al. 2003). New GE crop varieties have also exhibited yield increases 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 ongoing (Flint-Garcia et al. 2003).

Bacillus thuringiensis (Bt) crop varieties have provided more effective pest control than conventional pesticide use and have reduced the environmental impact of agriculture by reducing the use of harmful pesticides (Pray et al. 2002; 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; developed countries are responsible for 50% of these benefits. The largest environmental gains were observed in cotton, but significant gains were also observed 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 and possibly to increased C sequestration potential (Coulter et al. 2010). Improved

51. S.J. Del Grosso, personal communication, 22 April 2010.

rooting structures in corn (from traditional breeding or genetic engineering) also enable better crop growth under NT systems,[52] extending the GHG mitigation impact beyond yield improvements to increase the feasibility of NT management, which garners a 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 increased use of glyphosate. Within the United States, the most rapid adoption of GM seeds has been in areas under NT management (GM cultivars comprised approximately 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 conversion to NT management, 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 affect 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 with characteristics that could help increase soil C storage, improve N use efficiency, or reduce irrigation requirements. Other relevant crop breeding and development activities include development of rice varieties with lower CH_4 emissions (Aulakh et al. 2001b; Wassmann et al. 2002) and genetic improvements 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 other N losses (leaching and runoff), and also demonstrate lower reliance on N fertilizer. The N use efficiency of corn crops in the United States has improved 36% over the past several decades (Gurian-Sherman and Gurwick 2009), and traditional and enhanced breeding has prompted a 42% gain in the N use efficiency of wheat in Mexico (Ortiz-Monasterio et al. 1997). Similar gains have been seen in other staple crops in other countries (Gurian-Sherman and Gurwick 2009). Genetic engineering of crops for improved N use efficiency involves gene insertion to increase nitrogen metabolism. While research in this field is still limited and the commercial potential of this technology is unknown, developments in transgenic canola, rice, maize, and wheat already demonstrate improved N use efficiency (Beatty et al. 2009). Canola varieties developed by Good et al. (2007) required 40% less N fertilizer to achieve yields that were equivalent to those of original varieties.

One additional biotechnology under development is optimization of crops for water use. Improved water use could have small GHG benefits resulting from increased yields in water-stressed areas or by reducing irrigation requirements—and thus avoiding the associated input emissions. Traditional plant breeding for yield increases has succeeded in improving water use efficiency by reducing the duration of crop growth. For instance, the modern “IRRI varieties” of rice have improved water use efficiency 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. 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 (e.g., the emergence of “superweeds” resistant to herbicides and the negative effects of monoculture cropping) and social or ethical resistance to advanced genetic manipulation may reduce or negate the value of the GHG mitigation potential and other positive effects.

Conclusion

The analyses assembled in this assessment can inform an evolving range of government and business policy and program options, from cap-and-trade laws to voluntary payment programs and corporate supply-chain requirements. By presenting data for a large number of agricultural land management activities in one place, the assessment can provide a starting point for prioritization of agricultural activities in GHG mitigation projects and programs. It can also help identify where research resources are most needed in order to achieve environmental goals.

52. F. Yoder, personal communication, 30 April 2010.

Of the 42 activities considered in the assessment, 20 appear promising for near-term implementation because research evidence supports the conclusion that they have positive net GHG mitigation potential. Many of these activities enhance soil C sequestration: reducing tillage, reducing fallow periods, increasing primary productivity through greater use of perennial crops, using short rotation woody crops, and converting cropland to pasture or setting it aside. Others reduce N_2O emissions: using nitrification inhibitors, reducing N fertilizer application rates, and changing the timing, placement, and source of fertilizer. Still other activities are aimed at reducing CH_4 emissions: rice water management and variety development. A few management practices on this early-action list have high mitigation potential but significant data gaps (data is lacking for some regions or some conditions have been unstudied). These activities—use of winter cover crops, various N management practices, conservation tillage, and crop rotation diversification—are recommended as top research priorities.

Eight of the remaining activities appear to have positive GHG mitigation potential, but the existing research is insufficient to support broad protocol or program development. These activities—histosol management or set aside, crop rotation intensification, irrigation management, agroforestry on cropland or pasture, manure management for N_2O emissions reduction, and rotational grazing on pasture—warrant research to clarify GHG and other implications. Biochar application also appears to have very high mitigation potential but uncertain life-cycle effects, and thus research on it is recommended. The remaining activities do not appear worth pursuing for GHG mitigation purposes at this time, because they have more significant data limitations, the evidence suggests that their GHG mitigation potential is very low or negative, or their life-cycle GHG effects serve to limit their potential.

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