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Greenhouse Gas Emissions from U.S. Agriculture and Forestry: A Review of Emission Sources, Controlling Factors, and Mitigation Potential

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Greenhouse gas emissions from U.S. agriculture and forestry: A review of emission sources, controlling factors, and mitigation potential

USDA GHG accounting guidelines project – Task 2 (Review of reviews)

Karolien Denef[1], Shawn Archibeque,[2] and Keith Paustian[1,3]
[1] Natural Resource Ecology Laboratory, Colorado State University
[2] Department of Animal Sciences, Colorado State University
[3] Department of Soil and Crop Sciences, Colorado State University

I. Introduction

Approximately 6% of all greenhouse gas (GHG) emissions originating in the United States come from agricultural activities (EPA, 2010). These gases are in the form of carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4). By employing improved management techniques, agricultural lands can both sequester carbon and reduce CO_2, CH_4, and N_2O emissions, thereby reducing **agriculture's** GHG footprint. Forestry activities in the U.S. are currently a net sink (CO_2 removal from the atmosphere) of carbon (C) as a result of net forest growth, increasing forest area and net accumulation of carbon stocks in harvested wood pools (EPA, 2010). Several options exist to maintain and further increase C sequestration on forested lands in the U.S.

In the past decade, a number of reviews and meta-analyses have been published addressing GHG emissions, C sequestration and GHG mitigation related to agricultural and forestry activities, comprising results from numerous empirical field studies. This report provides a synthesis of these review studies and presents the current state of the science regarding controlling factors and mitigation technologies for GHG emissions in U.S. agriculture and forestry, along with a summary of reported C sequestration and GHG emission reduction ranges for various mitigation practices. An overview of the meta-analyses and quantitative review studies used in this review can be found in Table 1. Other, more descriptive reviews included in the current assessment are cited in the text, with the most valuable ones summarized below each heading. The report also highlights current knowledge gaps and indicates further research needs to improve the accuracy and completeness of agricultural GHG accounting methods and resulting GHG emission estimates. The document is intended as a resource for the Working Groups involved in the development of U.S. entity-level GHG emission inventory guidelines for agriculture and forestry.

This report is organized as follows: Section II describes the main sources, processes and factors controlling soil and biomass C losses and emissions of CO_2, N_2O and CH_4 from agricultural and forest systems. It also discusses the effects of common land use and management practices on GHG emissions and provides a range of emission estimates from different empirical field studies. Section III provides a review and synthesis of the literature regarding mitigation technologies for GHG emissions relevant to U.S. agriculture and forestry, and provides quantitative emission reduction estimates for various practices based on empirical field studies. Section IV concludes by highlighting current knowledge gaps and further research needs to improve the accuracy and completeness of agricultural GHG accounting methods and resulting GHG emission estimates.

Table 1 Overview of quantitative review studies on GHG emission reduction potential of agricultural and forest mitigation practices used in this report.

Source category	**Practice category**	**Quantitative review studies**	**No. of studies included in review** [1]
Soil C stock changes	Conversion of cropland to grassland or forest	Ogle et al., 2005	17
		Ogle et al., 2010	49
		T-AGG, 2010[2]	18
		Post and Kwon, 2000	38
		Guo and Gifford, 2002	15
		Paul et al., 2002	43
		Laganière et al., 2010	33
	Cropland management	West and Post, 2002	93
		Franzluebbers et al., 2005	35
		Franzluebbers et al., 2010	52
		Johnson et al., 2005	19
		Martens et al., 2005	26
		Liebig et al., 2005	40
		T-AGG, 2010[2]	53
		Six et al., 2004	254[3]
		Ogle et al., 2005	111
		Ogle et al., 2010	126
		Campbell et al., 2005	20
	Grassland management	Conant et al., 2001	115
		T-AGG, 2010[2]	19
		Derner and Schuman, 2007	15
		Ogle et al., 2010	4
	Horticulture/(agro)forestry	T-AGG, 2010[2]	11
	Forest management	Johnson and Curtis, 2001	55
		Hoover, 2003	34
Soil N_2O emissions	Cropland management	Ogle et al., 2010	61[4]
		Liebig et al., 2005	13
		T-AGG, 2010[2]	20[5]
		Akiyama et al., 2010	35
		Six et al., 2004	44[3]
		Swan et al., in prep	19 (88[3])
	Grassland management/conversion	T-AGG, 2010[2]	unclear
		Swan et al., in prep	11 (44[3])
	Horticulture/(agro)forestry	T-AGG, 2010[2]	2
Soil CH_4 emissions	Rice cultivation	Ogle et al., 2010	15
		T-AGG, 2010[2]	5
Livestock emissions	Manure management	Kebreab et al., 2006	10
		T-AGG, 2010[2]	4
		Ogle et al., 2010	31
	Enteric emissions management	Ogle et al., 2010	31
		Ominski and Wittenberg, 2004	12
		Smith et al., 2007	35

[1] In some case, individual studies may include more than one field experiment.

[2] Several studies included in the meta-analysis of the T-AGG report are already meta-analysis-type studies, using data from several other field experiments. These underlying studies are not captured in the number presented in this table.

[3] Number of data points.

[4] Number of sites/locations.

[5] Includes grassland studies.

II. Sources and controlling factors of GHG emissions

Agricultural activities contribute directly to GHG emissions through crop and soil management, rice cultivation, livestock manure management, enteric fermentation in domestic livestock, and field burning of agricultural residues. The use of fossil fuels and electricity within each of these activities contribute to additional GHG emissions. The primary GHG emission sources in U.S. agriculture are N_2O emissions from soil management activities such as fertilizer application and other cropping practices, and CH_4 emissions from enteric fermentation by beef and dairy cattle and manure management. Carbon dioxide (CO_2) emissions contribute to a lesser extent to U.S. agricultural emissions and are mainly attributed to soil and crop management practices that reduce soil organic carbon stocks. Forest activities relate to GHG emissions and removals mainly via impacts on C storage, uptake and loss from woody biomass and soils, although CH_4 and N_2O emissions from biomass burning – and to a lesser extent N_2O emissions from intensively managed forest plantation – are other emission sources.

The following paragraphs briefly describe the main sources, processes and factors controlling the emissions of each of the dominant GHGs emitted in agricultural and forest systems and how these are affected by different management practices. The focus of this report is on GHG emissions from non-energy related sources.

II.a. CO_2 emissions

Carbon dioxide is produced in agricultural and forest systems through autotrophic or heterotrophic oxidation of carbon in organic compounds and chemical weathering of inorganic carbonate-containing minerals in soils. Carbon dioxide is also released through combustion. Organic carbon is present in plant vegetation and soil organic matter as a result of CO_2 uptake by photosynthesis, root turnover, litter decomposition and organic amendments (e.g., manure). In cropland and grassland (pastures, rangeland), soils constitute the dominant carbon stock, while in forests, a large proportion of organic carbon storage also occurs in live vegetation (30-65% of ecosystem C stock) (Conant et al., 2007). Soil inorganic carbon (SIC) is comprised of primary carbonate minerals, such as calcite ($CaCO_3$) or dolomite ($CaMg[CO_3]_2$), or secondary minerals formed when carbonate (CO_3^{2-}), derived from soil CO_2, combines with base cations (e.g., Ca^{2+}, Mg^{2+}) and precipitates within the soil profile in arid and semi-arid ecosystems.

II.a.1. Soil organic carbon

Agricultural soils can be a source of or a sink for CO_2 depending on current and historical land use and agricultural management. This source-or-sink behavior of agricultural soils is typically observed in field studies through a measured decrease or increase in soil organic carbon (SOC) stocks, respectively, and is governed by the balance between SOC losses induced by microbial decomposition of soil organic matter (SOM) and organic C inputs to soils as crop residues and manures (CAST, 2004; Paustian, 2009). When SOM decomposition is stimulated relative to the input of carbon, soils generally behave as net CO_2 sources with a decrease in SOC stocks as a result. Carbon inputs to soil are determined by the amount of primary productivity and organic amendments (manure, compost). Soil organic matter decomposition, on the other hand, is a much more complex process influenced by numerous physical, chemical and biological factors that can affect the activity of soil microorganisms and fauna. These factors include the physical environment (temperature, moisture, soil properties including soil pH, nutrients, aeration, soil texture and clay mineralogy, and soil physical disturbance), substrate quality and quantity, and microbial community composition and enzymatic capacity (Chapin et al., 2002). Land use and management of agricultural systems have substantially changed SOC stocks in agricultural land through variations in land use, tillage, cropping practices (intensity and types of crops), grazing practices, irrigation, fertilization, and other activities affecting the balance between C inputs and SOM decomposition.

Other processes such as soil erosion and deposition and dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC) runoff and leaching also influence the net soil C balance in a particular location. In most agricultural soils, fluxes of DOC and DIC out of the soil profile are minor and are typically not considered in soil C stock change estimates (note that C emissions from agricultural liming are considered as a separate process – see Section II.a.2 below). Soil erosion can dramatically reduce soil C stocks in upland portions of the landscape from the physical removal of topsoil which typically contains the highest concentration of organic carbon. However, the combined effect of erosion and deposition, changes in the decomposition rates of organic matter in eroded or deposited soil and subsequent impacts on plant C inputs on eroded or depositional areas, all of which impact the net soil C balance, is extremely complex. There is an active debate as to whether at larger scales (landscape, region, globe) erosion processes are a net source or sink of C with respect to the atmosphere (e.g., Harden et al., 2008; Kuhn et al., 2009; van Oost et al., 2007). While erosion processes are not included in Intergovernmental Panel on Climate Change (IPCC) methods for national GHG inventories for soils, erosion impacts are important to consider in soil C stock change measurements made from small research plots or fixed monitoring locations.

The following paragraphs summarize the main findings from recent review studies that have analyzed the results of a large number of empirical field studies on the effects of different land use and agricultural activities and present a range of GHG emission estimates reported for these different activities.

Soil and crop management

With cultivation of long-term native vegetation, loss of SOC can be rapid and extensive. Much of this loss in SOC can be attributed to reduced inputs of organic matter, increased decomposability of crop residues, and tillage effects that decrease the amount of physical protection to decomposition. A review by Franzluebbers (2005) on GHG emissions from agricultural activities in the southeastern U.S. estimated the loss of SOC within 50 years of soil disturbance to be 65% of that under native condition. This estimate is higher than what has been estimated for the rest of the U.S. possibly because of the warm/moist climate of the region which facilitates decomposition and erosion. For the relatively cold region of Canada, loss of SOC was 24 ± 6% of that under native vegetation (VandenBygaart et al., 2003). From a global set of data from 74 publications, Guo and Gifford (2002) reported a decline in soil carbon stocks following conversion of pasture to plantation forest (-10%), native forest to plantation forest (-13%), native forest to crop (-42%) and pasture to crop (-59%). One rather unexpected outcome of their analysis was that pine plantations tended to decrease SOC stocks following conversion of established pastures whereas broadleaf tree plantations did not. Ogle et al. (2005) analyzed the results from a global data set of field experiments (126 articles) addressing the change in SOC storage for a variety of agricultural activities including (1) long-term cultivation, (2) changing tillage management, (3) modifying C input to the soil by varying cropping practices, and (4) setting-aside land from crop production. Their study provided a number of 'response factors' for the change in SOC storage due to changing land use and management practices, which can be applied in the development of national GHG inventories using the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006). Long-term cultivation reduced SOC storage in almost every study in their analysis, with the exception of a few arid land sites. These trends were relatively consistent with other syntheses that estimate losses of SOC due to long-term cultivation at 20–30% (e.g., Murty et al., 2002). Stimulation of organic matter decomposition occurs during cultivation due to frequent tillage, which releases organic matter protected in aggregates and redistributes organic matter in the soil profile where environmental conditions are more favorable for decomposition (Six et al., 2002, 2004; Conant et al., 2007). Soil organic C has been shown to decline significantly following even one tillage event (1–11% of soil C lost) (Conant et al., 2007). Soil carbon losses are typically lower under less frequent and less intensive tillage management. Model analyses indicate that cultivating and ripping are less disruptive than moldboard plowing (Conant et al., 2007). No-tillage (NT) in particular has been shown in a majority of studies to increase soil carbon stocks in surface layers relative to plow tillage systems (Paustian et al., 1997; Smith et al., 1998; West and Post, 2002; Ogle et al., 2005) and as a consequence is being promoted as an alternative management practice in mitigating GHG emissions from agricultural systems (CAST, 2004; T-AGG, 2010) *(see further details in Section III on mitigation practices)*.

Cropping system and crop management can also influence CO_2 emissions by affecting the quantity of C inputs to the soil and the quality of crop residue returned to the soil. Management practices such as irrigation and fertilization are typically implemented with the purpose of increasing crop productivity, which directly controls the amount of C input to the soil. Other management practices increasing C inputs to the soil are the use of winter cover crops, green manures, organic amendments (manure), high residue crop varieties, or annual rotations that include one or more years of hay or pasture. On the other hand, inclusion of bare fallow or rest years in a rotation, planting low residue crops such as vegetables, or burning or removal of residues typically reduce C inputs to the soil. The meta-analysis by Ogle et al. (2005) produced input factors for moist and dry climates to estimate the relative effect of increasing or decreasing C input to the soil on SOC storage compared to 'medium' input cropping systems, defined as continuous cereal or row crop rotations with residues returned to the field after harvest. Low input rotations were estimated to decrease SOC storage in the top 30 cm after 20 years to 91-92% of the amount of SOC under medium input rotations in moist and dry climates, respectively.

SOC changes due to soil and crop management practices can differ widely across field studies, due to spatially varying factors such as climate and soil properties as well as the impacts of previous land use and management. Ogle et al. (2005) found that the impact on SOC storage of several management-related factors (e.g., tillage, C input levels, land set aside) tended to vary as a function of climate, with largest to smallest changes in SOC as follows: tropical moist>tropical dry>temperate moist>temperate dry.

Grassland management

Management of grasslands (pastures, rangeland) can modify SOC storage primarily by modifying C inputs to the soil, including net primary production, root turnover, and carbon allocation between roots and shoots (Conant et al., 2001). Soil organic carbon losses as a result of overgrazing and poor pasture management have been observed. In a review by Ogle et al. (2004), SOC storage in degraded managed grasslands (i.e., grasslands with reduced productivity and receiving no inputs) was reduced by 3–5% in tropical and temperate grasslands respectively, though the uncertainty around these estimates was quite large, most likely due to the impact of climate and other factors (e.g., biome type) on the response of grassland SOC storage to management practices (Derner and Schuman, 2007). In particular, the effects of grazing practices on SOC stocks are complex and may depend on many factors, including the influence of the grazing animals on grassland species composition, rooting patterns and litter quality, as well as climate and soil conditions (Pineiro et al., 2010; Ogle et al., 2010). Ogle et al. (2004) found a few studies where heavy grazing did not result in soil C losses. A notable example is from the shortgrass steppe in the western Great Plains, where heavy grazing regimes, in comparison to light or no grazing, favor the dominance of C4 grass species (*Bouteloua gracilis*, *Buchloe dactyloides*) which have a higher C allocation belowground compared to C3 grass species and forbs that are more abundant under light grazing. However, in other grassland systems heavy grazing can increase dominance of forbs with lower productivity and consequently decreased C inputs to the soil, which may lead to lower soil C stocks (Pineiro et al., 2010).

Forest management

Forests play a major role in the natural global C cycle by capturing C from the atmosphere through photosynthesis, converting photosynthate to forest biomass, and emitting C back into the atmosphere during respiration and decomposition. Forest soils can lose a significant amount of C as a result of clearing and cultivation, forest fires or forest management practices that reduce forest productivity and, thereby, reduce the C input to the soil, and by increasing soil disturbances, which stimulate high rates of decomposition or soil erosion losses. Harvesting and thinning for example remove biomass, disturb the soil and change the microclimate, which can lead to soil C losses. In the years following harvesting and replanting, soil C losses may even exceed C gains in the aboveground biomass. Johnson and Curtis (2001) reviewed the literature on forest management effects on soil C, including harvest effects, fire effects, fertilization and N fixation effects. Their meta-analysis showed different trends in soil C responses dependent on harvest and fire type, forest species and time. For example, whole tree harvesting (i.e., removal of all aboveground parts of the tree) significantly decreased soil C, while sawlog harvesting (i.e., stem harvesting with residue left on site) increased soil C; the latter being apparently a result of residues becoming incorporated into the soil, but was found to be restricted to coniferous species. There have been also instances reported where soil C increased after forest harvesting, in particular in the tropics, probably due to the incorporation of slash into the mineral soil (Johnson, 1992). In general, the long-term effects of conventional harvesting on soil C have been found to be rather small, to be mainly related to the degree of ground disturbance from site operations, and to diminish over time without a long lasting effect on soil C storage (Johnson, 1992; Hoover, 2003; Johnson and Curtis, 2001, Ryan et al., 2010).

The effects of forest fires on soil C have been found to be highly dependent on fire intensity and the invasion of N-fixing vegetation after fire (Johnson, 1992), as well as on the time since fire (Johnson and Curtis, 2001). The meta-analysis by Johnson and Curtis (2001) counter-intuitively found a decrease in soil C following prescribed fire whereas wildfires on average increased soil C, owing to increased abundance of charcoal and recalcitrant, hydrophobic organic matter produced during wildfires and the effects of naturally invading, post-fire, N-fixing vegetation. However, in contrast to the findings by Johnson and Curtis (2001), the studies reviewed by Hoover (2003) generally found only small and short-lived effects of prescribed burning and limited to SOC in the surface soil, substantial losses (also at deeper depth) of SOC (as high as 50%) as a result of wildfires, and no general effect of fire intensity. The authors concluded that the response of forest soils to wildfires is likely a result of interactions between forest type, climate, initial soil conditions, the presence/absence of vegetative recovery, elevation and other factors. In addition, the role of black carbon (charcoal) in carbon stabilization following forest fires remains to be better understood and needs to be elaborated in future studies looking at fire effects on forest carbon pools.

Forest soils can also loose significant amounts of C by soil erosion (Page-Dumroese et al., 2000). Soil erosion occurs when forests are disturbed either naturally (fires, extreme rainfall events) or by forest management (e.g., tillage, harvesting, thinning, burning, etc.) (Elliot, 2003). Typical soil carbon losses due to soil erosion in Northwestern U.S. forests range between 0.12 and 5 kg C (between 0.44 and 18 kg carbon dioxide equivalent (CO_2 eq.)) ha^{-1} yr^{-1}, depending on type of disturbance (Elliot, 2003). These rates can be minimized by proper forest management. For example, thinning and prescribed fire can reduce the risk of wildfires and their associated large erosion rates.

II.a.2. Inorganic carbon

Inorganic carbon is present in soils as primary or secondary carbonate minerals which can sequester or release CO_2 depending on climate, vegetation, groundwater dynamics, soil properties and land-use/management practices. Due to its dominant presence at deeper depths, SIC is less susceptible to physical disturbance from tillage and other cropping practices than soil organic carbon (SOC) (Lal, 2003). On an ecosystem level, weathering of primary carbonate minerals (e.g., limestone) in humid environments can result in a net loss in total soil C storage, although the amount of dissolved inorganic carbon (DIC) that is eventually released to the atmosphere will depend on whether it is leached and re-precipitated lower in the profile, retained in groundwater or released in surface waters. In contrast, in arid and semi-arid systems, formation of secondary (pedogenic) carbonate minerals can result in the sequestration of CO_2 derived from the atmosphere, if base cations are supplied from weathering of non-carbonate primary minerals (e.g., base-rich silicate minerals). Management practices such as irrigation or the use of acidifying fertilizers (ammonia, sulfur) in these alkaline environments can impact both dissolution and precipitation processes (Nordt et al., 2000). Irrigation can either result in a net storage or release of CO_2 from SIC depending on the amount and quality of irrigation water (i.e., concentration of Ca or Mg and bicarbonate ions) and the solution flow path. Literature is limited on the impacts of management practices on SIC dynamics in semi-arid lands, and CO_2 fluxes from the turnover of this pool remain highly uncertain. Changes in soil inorganic carbon stocks (excluding added agricultural lime) are not included in IPCC national greenhouse gas inventory methods.

Other land use and management activities that result in a release of CO_2 from inorganic carbon are related to external C additions, such as the addition of calcium carbonate to low pH soils (i.e., liming) and fertilizing cropland with urea. In the U.S., liming of agricultural soils and urea fertilization resulted in CO_2 emissions of 7.6 Tg CO_2 eq. in 2008 (EPA, 2010).

II.a.3. Tree biomass C

Tree biomass is the dominant carbon storage in most mature forests and can be an important carbon stock in non-forest land uses, such as orchards and agroforestry or mixed woody grasslands used for grazing. Biomass components include leaves, branches, stem and coarse roots of living trees as well as the woody components of dead trees.

Accumulation of biomass C following harvest or natural disturbances (fire, insects, disease) can occur over many decades (> 100 years) before net growth is offset by losses from respiration and mortality and a 'maximum biomass' level is achieved. The C accumulation rate during the growth cycle as well as the biomass C stock potential is dependent on the forest type (species/species mix), climatic regime, soil fertility and other biotic and abiotic factors.

Tree biomass (and dead wood) C stocks are subject to a variety of management controls. The most dramatic impacts on biomass C stocks are associated with land use change. In particular, with afforestation or reforestation, new biomass C stocks are created where they did not previously exist, and with deforestation and land use conversion, the tree biomass component is lost from the system. Management can also impact biomass C stocks on permanent forested land. In more intensively managed plantation forests, use of improved genetic stock, nutrient additions and weed control can increase growth and biomass C accumulation rates. In less intensively managed forests, management activities are largely related to harvest and thinning activities and stand protection from major disturbances (e.g., fire). In general terms, management activities to enhance forest C sequestration aim at increasing the average biomass C stocks over time and space – for example, by delaying harvest removals (longer forest rotation), reducing impact logging (reducing collateral tree damage/mortality), and restocking of under-stocked stands.

In contrast to other ecosystem stocks and GHG fluxes, the U.S. has an extensive and well-developed measurement-based inventory system that, together with associated models, provides the basis for estimating C stock dynamics on U.S. forest land. The Forest Inventory and Analysis (FIA) system consists of several hundred thousand inventory points nationwide which are monitored with remote sensing as well as periodic ground measurements. The empirical FIA data together with results from field studies on experimental forests and other published data have been used to develop C estimation systems such as the USDA Carbon Online Estimator (COLE) and the Forest Vegetation Simulator (FVS) that can be used at the stand- or entity-level, and the USDA FORCARB2 model (an updated version of the U.S. FORest CARBon Budget Model (FORCARB)), that is used for national and regional scale forest C inventory estimation. Documentation and access to these tools are available at http://nrs.fs.fed.us/carbon/tools/.

II.b. N_2O emissions

Agriculture is the largest source of N_2O emissions in the United States (73% of total N_2O emissions) (EPA, 2010). The main sources of agricultural N_2O emissions are (1) soil management activities, including the use of synthetic and organic fertilizers, production of nitrogen-fixing crops and cultivation of high organic content soils; (2) manure management, including storage and application of livestock manure to croplands and pasture; (3) urine and feces deposition in grazed pastures; and (3) crop residue management (e.g., residue burning). All of these practices directly add additional nitrogen to soils, some of which can then be converted to N_2O. Indirect additions of nitrogen to soils (e.g., from ammonia volatilization and nitrate leaching and their subsequent transport to off-farm environments) can also result in N_2O emissions (Del Grosso et al., 2006).

II.b.1. Soil management

Nitrous oxide is produced in soils primarily by denitrification and, though to a lesser degree, nitrification; two N transformation processes that are performed by soil microorganisms. N_2O is a by-product of nitrification and an intermediate product during denitrification. Denitrification takes place under anaerobic conditions and has been shown to occur in saturated environments, as well as in anaerobic microsites (e.g., soil aggregates or soil organic matter, SOM, particles) within aerated soils. In well-aerated soils with few anaerobic microsites, nitrification is likely to be the dominant source of N_2O. The rate of denitrification is mainly controlled by nitrate supply, the availability of organic compounds as an energy source for the microbes, and the frequency and extent to which soil microsites are anaerobic. Denitrification-derived N_2O formation is favored during periods of low soil temperature, high but not saturated soil moisture (i.e., moderately anaerobic conditions), high NO_3^- and low soil pH (Barton et al., 1999; Čuhel and Šimek, 2011).

Despite the large spatial and temporal variability of N_2O fluxes from soils, consistent differences in N_2O emissions among different individual cropping practices have been documented and are primarily related to differential inorganic N availability. Field studies in cropland agriculture have found positive correlations between N_2O emissions and N fertilizer rates (Bouwman and Boumans, 2002; T-AGG, 2010; Ogle et al., 2010). The IPCC Tier 1 method, using a global average derived from field studies, specifies 1% of the N applied from inorganic and organic N fertilizers to be directly lost as N_2O -N to the atmosphere, with an uncertainty range in the emission factor of 0.3 to 3% (IPCC, 2006). Much of this variability in N_2O emissions from fertilized fields is related to site-specific environmental conditions (soil, climate), and other management practices in place (T-AGG, 2010). Since denitrification requires anaerobic conditions, management and soil conditions that reduce soil aeration or drainage (e.g., irrigation, poor tile drainage, poorly-permeable soil layers) generally result in higher N_2O emissions (T-AGG, 2010). Tillage practices also interact with N_2O emissions, although the effects vary as a function of climate, soil type and time (Six et al., 2004; Swan et al., in prep; T-AGG, 2010). According to a meta-analysis by Six et al. (2004) and a more recent analysis by Swan et al. (2011), N_2O emissions under no-till compared to conventional tillage were higher in mesic regions, but lower in semi-arid regions. Also Six et al. (2004) found that higher emissions under no-till occurred mainly in the initial years following conversion to no-till and that differences between tillage systems tended to decrease over time.

Fertilizer type, placement and timing have also been shown to influence N_2O emissions, although the results are quite variable and dependent on region and climate (T-AGG, 2010; Snyder et al., 2007, 2009). Several field studies have also shown N_2O emission rates to rise in a nonlinear fashion after crop N needs have been met (Snyder et al., 2007, 2009; T-AGG, 2010), with much higher N_2O emission rates than those predicted by the IPCC Tier I factor. Nitrogen fertilizer applications in excess of uptake capacity would therefore need to be allocated larger N_2O emission factors (Snyder et al., 2007, 2009). However, a recent review by Ogle et al. (2010) including 160 field comparisons (31 site locations) in Europe and North America found no significant change in the proportion of N_2O emissions per unit of N addition to soils, and estimated a constant percentage of 1.19% of N added to be released as N_2O-N. This is slightly higher than the 1% value provided by the IPCC 2006 guidelines.

II.b.2. Manure management

Depending on how livestock manure is handled, stored or applied to the field, it can be a significant source of N_2O (as well as CH_4 – see Section II.c.3. below) emissions. Nitrous oxide is mainly produced during the aerobic storage and treatment of animal excreta, as well as after land spreading. Studies have found even higher N_2O emissions following equal N amendments of liquid manure compared to mineral N, due to greater C availability and/or lower oxygen availability with manure application (Saggar et al., 2004). Nitrous oxide emissions from manure application to fields vary with type of application, time since application, and supplemental water additions. For example, higher denitrification losses were found following cattle slurry injection compared to surface application to a grassland soil (Saggar et al., 2004). The increased denitrification rates after slurry injection were attributed to the presence of large quantities of inorganic N together with high organic C levels and increased soil water contents.

Manure storage can last for up to several months at livestock farms, during which time the manure decomposes, and gaseous by-products are released. N_2O emissions from manure storage are a result of nitrification and denitrification processes. These processes are largely controlled by manure water content, redox potential and available N, which differ depending on manure storage conditions (e.g., liquid versus solid) as well as livestock management, e.g., animal type (digestive properties) or diet (Brown et al., 2000; Kebreab et al., 2006). Nitrification requires ammonium and oxygen whereas denitrification requires nitrate and anaerobic conditions. In fresh manures, where nitrate levels are low, denitrification rates are minimal but will increase after ammonia is oxidized to nitrate (Kebreab et al., 2006). Hence, for direct N_2O emissions to occur, the manure must first be handled aerobically to stimulate nitrification, and then handled anaerobically to stimulate denitrification. These emissions are most likely to occur in dry manure handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to saturation. Storing manure in lagoons rather than in open-air mounds or pits can reduce N_2O emissions but generally stimulates CH_4 emissions. However, CH_4 can be captured for electricity generation by covering lagoons, reducing the GHG footprint of the manure storage system even further (Robertson and Vitousek, 2009). Other studies have shown increased N_2O emissions upon active turning of manure during composting, owing to the enhanced gas diffusion rates in the more aerated manure (e.g., Hao et al., 2001). N_2O emissions have also been found to be related to the amount of solids in manure due to their effect on oxygen availability (Kebreab et al., 2006). Incorporating straw into manure decreases bulk density and increases aeration of manure allowing for easier diffusion of oxygen into manure and N_2O out of it. Brown et al. (2000) reported much higher N_2O emissions upon addition of chopped straw to stored solid dairy manure. They generally found highest N_2O emissions at water contents between 55 and 70% and redox potentials between 150 and 250 mV. Such conditions are more likely to be met around the exterior of a manure pile. Therefore, the authors suggested that the surface area, rather than the volume of a manure pile, may be a more important determinant of the magnitude of N_2O emission from this source.

Finally, manure storage and application can also lead to N_2O emissions off-site, as result of ammonia volatilization and nitrate leaching and their subsequent transport to off-farm environments.

II.b.3. Grazing

In pastures, N_2O emissions are a direct result of nitrification and denitrification processes and can be enhanced upon increased N inputs to the grazed pasture land, e.g., from N fertilization, manure application, effluent irrigation, biological N fixation, as well as grazing animals. A number of studies have reported greater N_2O emissions in grazed compared to ungrazed pastures, owing to both increased C and N inputs from animal excreta (urine, feces) as well as the more anaerobic environments as a result of soil compaction caused by animal treading (Bolan et al., 2004; Saggar et al., 2004; Kebreab et al., 2006). High, but short-lived N_2O emissions have been found from urine patches possibly as a result of rapid changes in oxygen availability and soil pH stimulating denitrification (Bolan et al., 2004). High N_2O emission rates have also been observed from feces patches (Saggar et al., 2004). Fecal N can be influenced by the species of forage being consumed by the animal (Archibeque et al., 2001) and the N content of the forage consumed (Archibeque et al., 2002), which will influence the availability of N present in the fecal piles to undergo nitrification and denitrification.

Several studies have found large temporal variations in soil denitrification and N_2O emission rates in grazed pastures, generally associated with grazing and fertilizer application timings, as well as with variations in soil temperature and water content (Saggar et al., 2004). In addition, large spatial variation in N_2O emissions from grazed pastures has been documented (Saggar et al., 2004). Spatial variability in N_2O emissions is naturally large in most soils because of soil heterogeneity and the episodic nature of N_2O emissions, but tends to be enhanced in grazed pastures as a result of uneven distribution of excretal returns of grazing animals and the animal treading effect on soil.

II.b.4. Indirect fluxes

Indirect N_2O is defined as N_2O that was emitted as a result of other on-farm N losses (not as N_2O) that are transported offsite and deposited elsewhere in the landscape. In addition to direct losses as N_2O from nitrification/denitrification on site, N can be lost from agricultural systems by volatilization, leaching, runoff and through the consumption of agricultural products (CAST, 2004). Agricultural practices such as N fertilization, irrigation, and manure management and application to soils can increase N volatilization and/or nitrate runoff and leaching, promoting indirect N_2O emissions. Volatilized N (e.g., NH_3, NO_x) from applied/mineralized N in cropland, pastures or confined animal facilities can become deposited downwind on agricultural and nonagricultural soils and in water and be subjected to transformations that may result in N_2O emissions. Surface runoff and leaching of nitrate into groundwater or surface water can reemerge in riparian zones or aquatic ecosystems, where it may become emitted as N_2O. Some of the N contained in agricultural products eventually ends up in sewage treatment plants following human consumption, where it can become emitted as N_2O (CAST, 2004). In the U.S., indirect emissions contribute substantially to the total N_2O emissions from agricultural systems (21% as estimated in the 2008 U.S. GHG inventory by U.S. EPA, 2010). However, these indirect emissions are very uncertain and represent a major challenge for GHG emissions accounting (Ogle et al., 2010).

II.c. CH_4 emissions

Emissions of CH_4 from agriculture are linked to methanogenic processes of microorganisms in flooded soils under rice cultivation and from enteric fermentation that occurs in the digestive systems of livestock, particularly ruminants (e.g., cattle, sheep, goats). Methane emissions also occur during decomposition of animal manure in uncovered lagoons, from crop residues under very wet field conditions and from crop residue burning. While CH_4 emission from rice cultivation is an important source of GHGs worldwide, it is not a big source for the U.S. (only 1% of total CH_4 emissions in the U.S.). Much larger contributions come from enteric fermentation (25%) and manure management (8%) (EPA, 2010).

II.c.1. Rice cultivation

Rice soils emit CH_4 because microbial respiration in flooded conditions reduces oxygen potential, creating anaerobic conditions that lead to CH_4 production by methanogenic microorganisms (in the domain Archaea). Once formed, CH_4 is mainly released from the soil through the aerenchyma within the rice plants, though a small proportion can also move through the soil profile by diffusion and ebullition. Methane is consumed and oxidized to CO_2 by methanotrophic bacteria, where oxygen is not limiting. Therefore, the net CH_4 emissions from rice soils are determined by the difference between CH_4 formation where O_2 is absent, and CH_4 consumption, where O_2 is available, and is greatly affected by the water management system (depth of flooding, draining activities) under which rice is grown. Other factors that influence CH_4 emissions from flooded rice fields include fertilization and residue management practices (especially the use of organic amendments), soil temperature, soil type, rice variety, and cultivation practices (e.g., tillage, seeding, and weeding practices). The total amount of CH_4 released depends primarily on the amount of organic substrate available. The application of synthetic fertilizers has also been found to influence CH_4 emissions; in particular, both nitrate and sulfate fertilizers (e.g., ammonium nitrate and ammonium sulfate) appear to inhibit CH_4 formation (Snyder et al., 2007, 2009).

II.c.2. Enteric fermentation

Enteric CH_4 production arises principally from microbial fermentation of hydrolyzed dietary carbohydrates such as cellulose, hemicelluloses, pectin and starch. Methane represents loss of energy to the animal and varies between 2 to 12% of gross energy (GE) intake (Johnson and Johnson, 1995). The amount of CH_4 produced and emitted by an animal is influenced by dietary factors, environmental conditions and genetic factors (Kebreab et al., 2006). Ruminant animals (e.g., cattle, buffalo, sheep, goats) have the highest CH_4 emissions among all animal types because of their unique digestive system, composed of a rumen, or large "fore-stomach". In the U.S., beef and dairy cattle are the largest contributor of CH_4 emissions from enteric fermentation, accounting for 72% and 23% of enteric methane inventories respectively in 2008 (EPA, 2010). Kebreab et al. (2006) reported feed energy losses due to CH_4 outputs from dairy and beef cattle in North America of between 8.9 and 21.4 MJ d^{-1} $animal^{-1}$. Non-ruminant animals (e.g., swine, horses, and mules) also produce CH_4 emissions through enteric fermentation in the cecum and large intestine which has a much lower capacity to produce CH_4 due to shorter retention time and other digestive factors, relative to ruminant animals. Within ruminant animals, other digestive factors affecting CH_4 emissions are rumen pH and the presence of protozoa. CH_4 emissions have been found to decline with removal of protozoa and upon reduction of ruminal pH (Sejian et al., 2010). In addition to the type and conditions of the digestive system, an animal's feed intake also affects CH_4 emissions (Seijan et al., 2010). Feed intake is positively correlated to animal size, growth rate and production (e.g., milk), and therefore varies among animal types as well as among different management practices for individual animal types (e.g., animals in feedlots or grazing on pasture). Positive correlations have been reported between the level of CH_4 emission and dry matter intake (DMI), milk yield (MY) and feeding level. The composition (e.g., forage-to-concentrate ratio, carbohydrate type, grain source, lipid saturation, forage digestibility) and manipulation (e.g., feed supplements, forage grinding) of diet fed to the livestock is another important factor that influences CH_4 emission (Johnson and Johnson, 1995; Beauchemin and McGinn, 2005; Kebreab et al., 2006; Sejian et al., 2010). Lastly, there is a widespread consensus that increasing productivity of an animal through genetic selection or other factors (i.e., growth promoting implants, rBST, etc.) will also reduce the proportion of CH_4 per unit of product (Beauchemin et al., 2008; Boadi et al., 2004; Moss et al., 2000).

II.c.3. Manure management

Manure can emit CH_4 during storage, handling and with field application. Methane emissions are a direct result of anaerobic decomposition processes and are largely influenced by the composition of the manure (amount of degradable organic compounds) and storage conditions. When manure is stored under anaerobic conditions (e.g., as a liquid/slurry in lagoons, ponds, tanks, or pits), the decomposition of materials in the manure tends to produce CH_4. When manure is handled as a solid (e.g., in stacks or drylots) or deposited on pasture, range, or paddock lands, it tends to decompose aerobically and produce little or no CH_4. Air and manure temperature, moisture, pH, and manure storage or residency time affect the amount of CH_4 produced because they influence the growth of the bacteria responsible for CH_4 formation (Saggar et al., 2004). Manure composition, which varies by animal diet, growth rate, and type, including the animal's digestive system, also affects the amount of CH_4 produced. In particular, high contents of total solids, high NH_3 concentrations and the presence of sulfur in manure have been shown to negatively affect CH_4 emissions, due to their negative impact on methanogenic microorganisms (Kebreab et al., 2006). The potential amount of CH_4 production following manure application to fields will depend on manure type (solid, slurry, effluent), origin (type of animal), composition, time since application, as well as climatic and soil conditions (Chadwick et al., 2000; Saggar et al., 2004). For example, much lower CH_4 emissions have been found following field application of manure that was first treated by anaerobic digestion, as this significantly reduces the amount of easily degradable organic matter in the manure. Methane emissions appear to be greater under wetter and colder climatic conditions (Chadwick et al., 2000). Methane can also be produced during manure composting, though as with manure storage and field application, emissions are affected by several factors including aeration rate, water content, thermal insulation, weather conditions, and dung composition.

II.c.4. Grazed pastures

In grazed pastures, small amounts of methane can be produced from animal excreta deposited during grazing, with a majority produced in dung patches (Kebreab et al., 2006). At the time of deposition, dung has favorable conditions for methane production, i.e., a readily available supply of substrate C, methane-producing microbial communities, and warm and moist conditions. Methane emission rates from dung vary depending on the type of dung, time since deposition, and local climatic conditions at the time of deposition, while soil type appears to have only a limited effect (Saggar et al., 2004). However, CH_4 emissions from excreta from grazing animals are much smaller than what's produced during enteric fermentation or what would be emitted when the same dung would be stored as manure, in particular when stored in liquid form (Holter, 1997; Saggar et al., 2004; Kebreab et al., 2006). The much larger production of CH_4 during manure storage (especially in liquid form) can be attributed to the continued fermentation of the manure during storage compared with major aerobic decomposition of fecal excretions in the field.

III. Mitigation of GHG emissions in U.S. agriculture

Agricultural- and forestry-based mitigation of GHG concentrations in the atmosphere can be achieved by (1) decreasing emissions of CO_2, N_2O and CH_4, and/or (2) taking CO_2 from the atmosphere and sequestering it in biomass and soils. For CH_4, mitigation can also entail maximizing the methane oxidation potential of soils. EPA (2010) estimated that U.S. forests removed approximately 792 Tg CO_2eq in 2008, mainly due to C sequestration in aboveground and belowground tree biomass and harvested wood pools. Cropland and managed grassland soils in the U.S. form an additional but smaller C sink, with an overall GHG mitigation rate estimated at 75 Tg CO_2 eq. per year on mineral soils, due primarily to a long-term trend of increasing crop residue production, reductions in the tillage intensity, conversion of annual cropland to pasture/hay and set-aside of annual cropland in the Conservation Reserve Program (EPA, 2010). In contrast, organic soils (i.e., histosols, such as peat soils) used for cropland and managed grasslands are a net source of soil GHG emissions, estimated at 38 Tg CO_2 eq. per year (EPA, 2010).

The following paragraphs provide a review and synthesis of the literature regarding mitigation technologies for GHG emissions relevant to U.S. agriculture and forestry, and provide quantitative emission reduction estimates for various practices based on empirical field studies. Due attention is given to regional/local conditions, since climate, soil and landscape (texture, depth, drainage, landscape position) and land use history are known to greatly affect the response of any given soil to a particular change in management (Ogle et al., 2004; Paustian, 2009).

III.a. Agricultural land management

(T-AGG, 2010; Johnson et al., 2007; CAST, 2004; Paustian et al., 2006; Paustian 2009; Smith et al., 2006; Smith et al., 2007)

The following sections synthesize the most recent literature on the GHG mitigation potential of agricultural land management practices in the U.S., and draw primarily upon the findings of the recently published Technical working group on Agricultural Greenhouse Gases (T-AGG) report (T-AGG, 2010). This report provides quantitative estimates of the biophysical greenhouse gas mitigation potential of more than 40 individual agricultural land management activities in the U.S., based on results from field studies, or modeled estimates when field studies were not available.

III.a.1. Soil C sequestration

(Franzluebbers, 2005, 2010; Govaerts et al., 2009; Guo and Gifford, 2002; Morgan et al., 2010; Ogle et al., 2005; Paustian, 2000, 2009; Post and Kwon, 2000; Follet, 2001; West and Post, 2002; Morgan and Follett, 2009; Smith et al., 2006; Johnson et al., 2007)

Changes in agricultural practices for the purpose of sequestering C must either increase organic matter inputs to the soil, stabilize a larger proportion of C in longer-term C pools in the soil, slow down decomposition of soil organic matter, or a combination thereof. Soil C sequestration will therefore be favored under crop and soil management systems that maximize amounts of crop residue return to the soil and minimize soil disturbance and erosion (CAST, 2004). For cropland, these practices include reducing tillage intensity, decreasing or eliminating the fallow period, using a winter cover crop, changing from monoculture to rotation cropping, including perennial crops in rotations or converting cropland to pasture/hay, using organic amendments or altering soil inputs to increase primary production (fertilizers, pesticides and irrigation) and restoring highly degraded soils, e.g., by setting aside in the Conservation Reserve Program (CRP) (Smith et al., 2006; Paustian, 2009). In grasslands, C sequestering practices include improved grazing management (reducing disturbance and/or maximizing manure C returns), conversion from cultivation to native vegetation, sowing of legumes and grasses, earthworm introduction, fertilization and irrigation (Conant et al., 2001; Smith et al., 2006). Agroforestry practices (e.g., windbreaks, shelterbelts, riparian woodland) and other conservation structures with grass cover (e.g., filter strips, riparian buffers), that are integrated within agricultural fields, can contribute to C sequestration. However, very few studies have quantified the sequestration potential of these specific practices, which may differ significantly from large-scale land use conversions to grassland or forest vegetation (Ogle et al., 2010).

The following paragraphs provide a review of the literature on several options for mitigating GHG emissions through carbon sequestration in soils.

Conversion of cropland

(Ogle et al., 2005, 2010; Post and Kwon, 2000; Follett, 2001; T-AGG, 2010)

When agricultural land is no longer used for cultivation and allowed to revert to natural vegetation or replanted to perennial vegetation, soil organic carbon can accumulate, with the highest sequestration potential in sensitive or marginal agricultural land, prone to erosion or flooding. Setting aside cropland sequesters carbon in the soil due to reduced soil disturbance and a change in soil aggregate dynamics, similar to reducing tillage intensity (see below), and by increasing plant C inputs to soil. Converting sections of cropland to grass buffer strips can additionally sequester C by retention of sediment from agricultural runoff, and by sequestration in biomass (T-AGG, 2010). Setting-aside land for 20 years was estimated to restore SOC levels in the upper 30 cm within 0.82 ± 0.07% and 0.93 ± 0.05%, or 82–93%, of native stock levels in temperate moist and dry climates, respectively (Ogle et al., 2005). A global literature review by Post and Kwon (2000), including also subtropical systems, estimated average C sequestration rates for conversion of agricultural land to forest and grassland at 0.34 and 0.33 Mg C (1.2 and 1.3 Mg CO_2 eq.) ha^{-1} yr^{-1}, respectively, with a tendency for soil organic carbon (SOC) accumulation rates to increase from temperate regions to subtropical regions. In the U.S., the Conservation Reserve Program (CRP) has paid farmers during the past two decades to set-aside highly erodible cropland from production by planting grasses or trees, and numerous studies have shown that SOC content typically increases in these set-aside lands (e.g., Follet et al., 2001, who estimated that CRP sequesters 0.9 Mg C (3 Mg CO_2 eq.) ha^{-1} yr^{-1} in the 0-20 cm depth). The recent T-AGG (2010) report estimated an average C sequestration rate upon conversion of U.S. cropland to natural vegetation (including CRP) at 0.68 Mg C (2.5 Mg CO_2 eq.) ha^{-1} yr^{-1}, but the observed rates ranged from -0.04 to 1.29 (-0.15 to 4.73 Mg CO_2 eq.).The potential of set-aside land to sequester C greatly depends on the vegetation, former land use, soil type and climate. A meta-analysis on data from 49 studies in the U.S. estimated C sequestration rates up to 1 Mg C (3.7 Mg CO_2 eq.) ha^{-1} yr^{-1} during the first 20 years following conversion, but found large regional differences across the U.S. with higher sequestration rates in Pacific Northwest, Northern Plains, Upper Midwest and New England (Ogle et al., 2010). These authors also highlighted the uncertainty around the influence of land conversion on carbon storage at deeper depths in the profile (i.e., below 45 cm). An important factor also to consider is the time span of keeping cropland out of production. Set-aside lands are generally placed in a reserve status for short time spans, raising concerns about long-term C sequestration (Ogle et al., 2005; T-AGG, 2010).

Soil and crop management

(CAST, 2004; T-AGG, 2010; Follet, 2001; Conant et al., 2007; Six et al., 2004; Snyder et al., 2007, 2009; West and Post, 2002; Campbell et al., 2005; Franzluebbers et al., 2005, 2010; Ogle et al., 2005, 2010)

** Reduced tillage*

Adoption of conservation tillage has been promoted for C sequestration during the past two decades (e.g., CAST, 2004; T-AGG, 2010). Reduced tillage can lead to sequestration of carbon due to a reduced physical disturbance which slows down soil aggregate turnover and reduces the decomposition of soil organic matter (e.g., Six et al., 2002). Several studies have attempted to quantify C sequestration rates following conversion to reduced or no-till. More consistent soil C sequestration rates are typically found for conversions to no-till (NT) than to reduced till systems, probably due to the more straight-forward residue retention and physical disturbance limitations for NT than reduced tillage (Six et al., 2004; West and Post, 2002). Using a global dataset of 67 long-term agricultural experiments, West and Post (2002) reported SOC sequestration with NT of 0.48 ± 0.13 Mg C (1.76 ± 0.48 Mg CO_2 eq.) ha^{-1} yr^{-1} (mean of 15 years). For the southeastern U.S., SOC sequestration with NT was estimated at 0.42 ± 0.46 Mg C (1.54 ± 1.69 Mg CO_2 eq.) ha^{-1} yr^{-1} (10 ± 5 years) (Franzluebbers, 2005). A more recent review by the same author (Franzluebbers, 2010), which included an additional 51 comparisons of SOC on conventional- and conservation-tillage cropland in the southeastern U.S., estimated SOC sequestration with conservation tillage at 0.45 ± 0.04 Mg C (1.7 ± 0.15 Mg CO_2 eq.) ha^{-1} yr^{-1}. These rates are similar to those reported for the central U.S. (0.48 ± 0.59 Mg C (1.8 ± 2.2 Mg CO_2 eq.) ha^{-1} yr^{-1}, Johnson et al., 2005), but higher than rates reported for the southwestern U.S. (0.30 ± 0.21 Mg C (1.1 ± 0.77 Mg CO_2 eq.) ha^{-1} yr^{-1}, Martens et al., 2005), and the northwestern U.S. (0.27 ± 0.19 Mg C (0.99 ± 0.70 Mg CO_2 eq.) ha^{-1} yr^{-1}, Liebig et al., 2005). For the entire U.S., the recent T-AGG report (2010) estimated mean C sequestration rates of 0.3 Mg C (1.1 Mg CO_2 eq.) ha^{-1} yr^{-1} for conservation till and no till systems, which is similar to earlier estimates by West and Marland (2002) for no-till systems in the U.S. (0.34 Mg C (1.3 Mg CO_2 eq.) ha^{-1} yr^{-1}).

The impact of tillage practices on SOC stocks appears to heavily depend on climate (Ogle et al., 2005; Halvorson et al., 2002; Franzluebbers and Steiner, 2002; Six et al., 2004) as well as soil type (Dumanski et al., 1998) and initial soil organic C content (Franzluebbers, 2005). In a recent review by Ogle et al. (2010), C stock change rates were estimated for different regions in the U.S. for full tillage to no-till and reduced-till conversions. On average, an increase in soil C stocks was found for all U.S. regions with higher gains following conversion to no-till. The study also estimated a higher C sequestration potential in the eastern half of the U.S. and the upper Pacific Northwest. Estimated C sequestration rates upon no-till were especially high for non-sandy soils.

Other management practices such as crop complexity and fallow management (West and Post, 2002; Halvorson et al., 2002; Campbell et al., 2005) and cover crops (Franzluebbers, 2005), influence the effect of a ‘‘conservation tillage system’’ on SOC sequestration. Combining cover cropping with NT enhanced SOC sequestration (0.53 ± 0.45 Mg C (1.9 ± 1.7 Mg CO_2 eq.) ha^{-1} yr^{-1}) compared with NT and no cover cropping (0.28 ± 0.44 Mg C (1.0 ± 1.6 Mg CO_2 eq.) ha^{-1} yr^{-1}) in the southeastern U.S. (Franzluebbers, 2005). West and Post (2002) reported greater C sequestration rates with a change to NT for rotation cropping systems than for continuous monocultures. This study also did not observe a significant change in SOC upon conversion to NT for any of the wheat – fallow systems included in the study, while conversion to NT in systems with more continuous cropping was generally positive. Halvorson et al. (2002) also reported little impact of NT on SOC storage in dry climates when the cropping system had a year of bare summer-fallow, presumably due to enhanced decomposition during fallow that negated any benefit of reduced soil disturbance.

Reduced benefits for C storage with no-till adoption, or even losses relative to intensive tillage have also been reported for some cool, moist environments and on heavy textured soils (Anger et al., 2007). Reduced rates of residue C inputs under no-till for corn and wheat, predominately in cooler, wetter regions (upper Midwest), may explain part of the reduced C sequestration performance of no-till (Ogle et al., 2010). Also, soils in cool, wet environments often have already high surface organic matter contents and thus may be closer to a C ‘saturation-level’ and hence less of the residue added at the soil surface under no-till can be effectively stabilized (Stewart et al., 2007; Gregorich et al., 2009).

Other authors have highlighted the uncertainty around the impacts of tillage practices on soil C stocks at deeper depth (Baker et al., 2006) as well as confounding effects of erosion, transport and deposition of sediments, as well as the impact of tillage practices on N_2O emissions (Six et al., 2004). A recent paper by Kravchenko and Robertson (2011) points out that lower soil depths have lower and more variable carbon contents than surface horizons, making it extremely difficult to detect statistically valid changes with simple averaging over the whole profile. They recommend that assessments be based on the separate analysis of individual horizons or depth layers with inferences based on where statistically significant differences are observed. More research is needed to reduce these uncertainties and/or understand the conditions under which soils could gain or lose carbon at depth or during erosion events, transport and subsequent burials, as well as to better understand the trade-offs between C sequestration and N_2O emissions in reduced or no-till soils.

** Crop rotations*

Changes in crop rotations are done through crop rotation ‘diversification’ and crop rotation ‘intensification’; both have been found to affect soil C levels by affecting the amount of plant C input to the soil. Diversification most often involves changing from a monoculture or simple rotation to multiple crops over multiple years of a crop rotation. Intensification consists of adding another crop within an annual cycle, which adds to biomass inputs and can also reduce decomposition rates (Ogle et al., 2005; T-AGG, 2010). This also includes adding winter cover crops to, or eliminating summer fallow from the crop rotation. Another form of crop rotation change (both diversification and intensification) is by introducing perennials (e.g., alfalfa or grass hay) into the annual crop rotation, or the full conversion from annual crops to perennial crops. In the majority of studies reviewed by Ogle et al. (2005), high input rotations (use of winter cover crops, green manures, irrigation, high residue crop varieties, or rotations that have a year(s) of hay or pasture) increased SOC storage relative to medium input rotations (continuous cereal) by 7–11% after 20 years. The recent T-AGG report (2010) reported average C sequestration rates in the U.S. of 0.27 Mg C (0.99 Mg CO_2 eq.) ha^{-1} yr^{-1} for winter cover crops, 0.13 Mg C (0.48 Mg CO_2 eq.) ha^{-1} yr^{-1} for eliminating summer fallow, and 0.16 Mg C (0.59 Mg CO_2 eq.) ha^{-1} yr^{-1} for crop diversification. Including perennial crops in the rotation and full conversion to perennial crops resulted in average sequestration rates of 0.16 Mg C (0.59 Mg CO_2 eq.) and 0.62 Mg C (2.3 Mg CO_2 eq.) ha^{-1} yr^{-1}, respectively (see Table 2). However, the range of C sequestration rates from field observations included in this report was relatively large for each of the rotation practices, and often included negative values, e.g., for fallow management, crop diversification and perennial crops (see Table 2). Ogle et al. (2010) also found both positive and negative impact of winter cover crops on C uptake in U.S. cropland systems. Many factors can affect the impact of crop rotation practices on soil C levels, including soil and climate conditions, crop characteristics and interactions with other land management practices (e.g., tillage, fertilization, irrigation, etc.). For example, West and Post (2002) reported an average C sequestration rate of 0.2 ± 0.12 Mg C (0.7 ± 0.44 Mg CO_2 eq.) ha^{-1} yr^{-1} by enhancing rotation diversity and intensity in conventional tilled soils. This estimate excluded a change from continuous corn to corn-soybean, which did not result in a significant accumulation of SOC, probably as a result of much lower residue returns to the soil under soybean. However, the inclusion of legumes other than soybean has been found to increase SOC stocks (T-AGG, 2010). Enhancing crop complexity in soils already managed under NT did not increase SOC storage in the review by West and Post (2002), possibly due to the relatively high C contents of the NT soils included in this study. This would mean that soils with low C contents may have the greatest potential and efficiency to sequester C because they are further from their C saturation level (Stewart et al., 2007). On the other hand, the potential to sequester more C by eliminating or reducing summer fallow has been found in many studies to be greater in combination with NT (compared to conventional till) (T-AGG, 2010; Campbell et al., 2005). The key reason for this is that NT increases water storage and enables the frequency of summer fallowing to be reduced or eliminated, resulting in greater crop production over the rotation and hence greater C inputs to the soil. Studies that have reported SOC reductions after eliminating summer fallow have always been in combination with conventional tillage (T-AGG, 2010). The C sequestration potential of eliminating summer fallow in the U.S. has been estimated to range from 0.1 to 0.6 Mg C (0.4 to 2 Mg CO_2 eq.) ha^{-}1 yr^{-1} (Follett and McConkey, 2000; Follett, 2001; T-AGG, 2010).

* Fertilization and organic amendments

Fertilization and organic amendments are common practices in agricultural systems to stimulate crop biomass production. The resulting increase in residue C inputs to the soil can lead to soil C sequestration. Some studies have found increased SOC levels as a result of N fertilization (see Follett, 2001 and Snyder et al., 2007, 2009), however, other studies have found little to no changes in SOC stocks (see T-AGG report, 2010). These different trends may be attributed to differences in soil properties, in particular the initial C content of the soil (with C-rich soils closer to their C saturation level, Stewart et al., 2007), as well as differences in climate conditions, as decomposition tends to be stimulated at higher temperatures (Snyder et al., 2007, 2009; Johnson et al., 2007). Most U.S. cropland soils already receive N fertilizer additions to attain high crop yields and thus further increases in N addition, in the absence of other management changes, are not likely to substantially increase crop residue inputs to soil. In addition, the benefits of N fertilization in terms of C sequestration can be largely offset by N_2O emissions generated by nitrification and denitrification of applied fertilizer N, as well as by the C-cost of fertilizer manufacturing, distribution and application (T-AGG, 2010). A review by Franzluebbers (2005) took into account these C costs and found an optimum fertilization rate to maximize net C offset between 24 to 37 kg N ha^{-1} yr^{-1} to achieve SOC sequestration of 0.07 to 0.11 Mg C (0.3 to 0.40 Mg CO_2 eq.) ha^{-1} yr^{-1}. Organic amendments, on the other hand, may displace N fertilizer use and can, depending on how manure is treated and applied, reduce N_2O emissions. Land applications of manure can also decrease CH_4 emissions when compared to alternative handling of manure, such as the long-term storage in anaerobic lagoons or stockpiles (Johnson et al., 2007).

The impacts on SOC from application of animal waste (e.g., farmyard manure, poultry litter), and to a lesser degree compost, have been studied extensively in long-term field experiments (see Paustian et al. 1997; Ogle et al. 2005; T-AGG, 2010). Soil organic matter invariably increases with manure/compost additions and typically at a higher rate per unit of organic matter added than for additions as crop residues or green manure. This is not surprising given the fact that the original plant matter has already been partially decomposed (i.e., in animal digestive tract and manure storage or composting system) and therefore is stabilized to a greater degree than fresh plant material (Paustian et al., 1997). In a global meta-analysis, Ogle et al. (2005) found that the cropland management activity that increased SOC levels the most was high inputs of organic amendments, usually manures, with an average increase of around 35% in SOC in the top 30 cm compared to reference cropping systems with no manure addition. One issue in considering organic amendments (manure, compost) as a soil carbon sequestration activity is that the organic matter in the manure often originates from off-site locations (i.e., as purchased livestock feed), and thus a balanced accounting of the impact of manure additions requires an estimate of effects of the removal of plant C where the feedstocks were produced (Smith and Powlson, 2000).

Animal waste is the most commonly applied organic amendment in agricultural systems. Recently, *biochar* has received attention as a promising soil amendment with benefits both for soil fertility, plant productivity and GHG mitigation (Lehmann et al., 2006; Lehmann, 2007; De Gryze et al., 2010). Biochar application in agricultural and forest management systems is being considered as a potential technology for generating C offset credits under several U.S. voluntary offset programs. Biochar is the product of pyrolysis, which is the incomplete combustion of biomass into charred and highly recalcitrant organic matter. Its GHG mitigation potential lays in its ability to sequester C by adding recalcitrant C to the soil, stabilizing existing soil C and increasing crop production, as well as potential impacts on N_2O and CH_4 emissions (Johnson et al., 2007; Gaunt and Driver, 2010). Biochar recalcitrance and its potential to stabilize C in soils appears to differ among biochar types and seems to depend on factors such as biomass source (e.g., crop residues, urban waste, manure, wood, etc.) and the conditions of the pyrolysis process (temperature, time) (Gaunt and Driver, 2010). In addition, the impacts of biochar on soil C, N_2O and CH_4 emissions may vary across soil types. However, the contribution of different soil conditions, such as hydrology, texture, microbial community structure, etc. to the impact of biochar on soil C sequestration and GHG emissions needs further investigation (T-AGG, 2010). Other research needs are to evaluate the GHG mitigation potential of different types of biochar upon large-scale application in different agricultural systems in the U.S.

** Irrigation*

In semi-arid climates, irrigation can significantly increase crop production and, as a consequence, return more residue C inputs to soils. Carbon sequestration rates through irrigation have been indirectly estimated at 50 to 150 kg C (180 to 550 kg CO_2 eq.) ha^{-1} yr^{-1} based on increased plant C inputs (Lal et al., 1998; Follett, 2001). However, irrigation may also decrease SOC storage by stimulating organic matter decomposition as a result of enhanced microbial activity and soil water content (Gillabel et al., 2007; Sainju et al., 2008). Moreover, irrigated fields often undergo more intensive tillage, and crop residues are not always returned to the soil, which could further reduce C storage (Follett, 2001). Finally, irrigation could also lead to greater losses of C from soil erosion (Follett, 2001). Therefore, broad estimates of soil C accumulation based solely on C input may overestimate the SOC accumulation via irrigation. Yet, similar or even much higher C sequestration rates than those estimated by Lal et al. (1998) have been documented in the literature, based on soil C measurements, with rates as high as 0.82 Mg C (3.0 Mg CO_2 eq.) ha^{-1} yr^{-1} (T-AGG, 2010), indicating a clear potential of irrigation for sequestering atmospheric C. However, this C sequestration potential of irrigation needs to be weighed against increased emissions of N_2O and CH_4 likely to occur due to the increased anaerobic conditions in irrigated soils (T-AGG, 2010).

Research on C sequestration under irrigation is scarce, is mostly limited to C changes in the soil surface (< 30 cm) and frequently disregards the soil inorganic carbon (SIC) pool. The few studies that have looked at soil inorganic carbon (SIC) dynamics under irrigation have shown contrasting results (e.g., Entry et al., 2004; Denef et al., 2008; Blanco-Canqui et al., 2010). Therefore, more research is needed on the factors controlling SIC dynamics in irrigated systems, to develop more accurate estimates of C fluxes from this pool. Moreover, the majority of studies looking at irrigation effects on soil C have primarily focused on the comparison of irrigated vs. non-irrigated fields. More studies are needed to address the impacts of variable levels of irrigation, in particular in semiarid regions facing reduced water supplies.

Pasture/Rangeland management

Grazing land can be divided into rangeland (i.e., uncultivated land with minimal inputs and extensively grazed) and pastures (i.e., more intensively managed with periodic agronomic inputs). Most of the GHG mitigation potential of grazing land is related to soil C sequestration. For rangeland, the most promising soil C sequestration practices include improved grazing management (intensity or stocking rates, timing and frequency), destocking during drought, introduction of legumes, control of undesirable species, and in some cases, addition of fertilizer N (Follett and Reed, 2010; Morgan et al., 2010). Pastures can sequester C mostly by implementing improved management practices such as fertilization, irrigation, altering grazing practices or fire regimes, introducing legumes, and planting more productive varieties of grasses (Conant et al., 2001; Ogle et al., 2004). Most of these practices primarily affect SOC storage by modifying C inputs to the soil through changes in net primary production, root turnover, and carbon allocation between roots and shoots, but their potential may vary depending on climate, biome, management and environmental conditions (Morgan and Follett, 2009; Morgan et al., 2010). Ogle et al. (2004) estimated the increase in SOC storage in the top 30 cm and over a 20-year period with a single grassland improvement activity at 14% ± 0.06 and 17% ± 0.05 in temperate and tropical regions, respectively, and an additional 11% ± 0.04 with additional improvement(s). A meta-analysis by Conant et al. (2001), which analyzed the results of 115 studies worldwide, estimated the C sequestration potential by type of improvement between 0.11 and 3.04 Mg C (between 0.40 and 11.2 Mg CO_2 eq.) ha^{-1} yr^{-1}, with a mean of 0.54 Mg C (2.0 Mg CO_2 eq.) ha^{-1} yr^{-1}. Their estimated C sequestration rates appeared also to be highly influenced by management improvement system, biome type and climate. The influence of climate on the response of grazing lands to management practices in terms of their function as a C source or sink was also highlighted by Derner and Shuman (2007) and Zhang et al. (2010). One of the observations in their studies was that drought conditions may change rangelands from sinks to sources of atmospheric CO_2 because limiting soil water proportionally affects photosynthetic rates more than total respiration. The recent T-AGG report (2010) reported different C sequestration potentials of improved grazing management for the different climate regions in the U.S., with increasing rates from the Southwest to the Northeast.

** Grazing and improved grazing*

Grassland soil C stocks can increase as a result of sustainable grazing when compared to ungrazed land. Grazing-induced changes in plant productivity and plant community composition are likely responsible for most of the changes in C sequestration observed with grazing practices. For the majority of studies included in the review by Conant et al. (2001), the increase in soil C upon grazing was related to increased forage production, and was most pronounced in warm dry regions, and in locations that had already a long history of grazing and low productivity. However, in a few studies manipulation of the stocking rate led to changes in plant community structure. On rangeland, higher soil C stocks were observed, while forage production was negatively affected. In these studies, decreased C inputs from aboveground biomass may have been offset by manure inputs from grazing animals, and by changes in species composition that can result in greater root: shoot ratios and increased belowground inputs (Ogle et al., 2004; Derner and Schuman, 2007). Soil C stocks can also increase when grazing results in lower standing stocks of biomass, but increased production and turnover. Other factors that may affect the impact of grazing on soil C stocks are the type of livestock and stocking rates or frequency of grazing.

Conant et al. (2001) reported a decrease in soil C in overgrazed sites by 0.19 Mg C (0.70 Mg CO_2 eq.) ha^{-1} yr^{-1} compared to moderately grazed sites. A number of studies have reported soil C sequestration with rotational grazing on pastures in mesic climates (T-AGG, 2010). In these studies, rotational grazing led to higher productivity. In addition, research has suggested that rotational grazing may also lead to lower enteric methane emissions from grazing animals due to its impact on forage quality (T-AGG, 2010). However, a recent literature review and analysis by Ogle et al. (2010) found only one study with a relatively large SOC gain of more than 1 Mg C (4 Mg CO_2 eq.) ha^{-1} yr^{-1} with rotational grazing, while all other studies showed a minimal (both positive and negative) change in soil organic C stocks. In a literature review of publications on rotational grazing on rangeland, Briske et al. (2008) found that plant production was equal or greater in continuous compared to rotational grazed plots in 87% of the experiments. Similarly, animal production per head was equal or greater in 92% of the experiments comparing continuously and rotationally grazing animals. However, both the T-AGG and Ogle et al. reviews included very few data due to a lack of published studies that address grazing intensity in pastures in relation to soil C sequestration. More research is needed in order to derive generalizable estimates of the C sequestration potential of improved grazing practices.

* Other (fertilization, irrigation, legumes)

Most management practices that are implemented for the purpose of increasing forage production in pasture systems tend to increase soil C stocks by increasing the amount of C returned to the soil. Studies have shown that grassland irrigation in dryland conditions as well as the addition of manure or mineral N fertilizer as well as other nutrients to grassland soils can lead to increased production and increased C sequestration (e.g., Conant et al., 2001; Derner and Schuman, 2007). However, the benefits of increased SOC sequestration with irrigation and N-fertilization may be offset by enhanced soil N_2O losses from denitrification, and by enhanced CO_2 and N_2O emissions during fertilizer production. The introduction of N-fixing legumes has also been shown to promote C sequestration in grassland soils and may therefore provide an alternative to N-fertilization with a lower overall GHG footprint (Conant et al., 2001). The review by Conant et al. (2001) also included two studies that evaluated the impact of earthworms in grazing lands. They reported an increase of nearly 4% per year in SOC upon introduction of earthworms. However, both studies took place on allophanic soils which have a high capacity for organic matter stabilization. More research is needed on different soil types to obtain a better estimate of the impact of this practice on soil organic C. Other research needs include quantifying C sequestration rates in improved systems across the entire soil profile. Most studies evaluating the impact of grassland management on soil C stocks generally sample only the upper 20-30 cm depth, and may greatly underestimate the full C sequestration potential of these practices.

Orchards, Agroforestry and Woody Crops

* Horticulture

Relatively little attention has been paid to soil C sequestration and GHG emissions in vegetable, vineyard, and especially orchards lands. Various agricultural land management activities with GHG mitigation potential can be applied to field management practices in horticultural systems including many that sequester C (e.g., cover crops, reduced tillage) and reduce N_2O and CH_4 emissions (e.g., improved fertilizer and irrigation management). However, efforts to sequester soil C or reduce GHG emissions have been limited so far because of the specialized management required for these high-value cash crops. For example, the planting and harvesting of many vegetables causes substantial physical disturbance to the soil, which can limit the C sequestration potential of practices such as conservation tillage (T-AGG, 2010). In orchards, herbicide-treated tree rows with grass alleys are standard orchard floor management practices in the U.S. Cover crops are often avoided in conventional orchard systems to limit competition for N (Sanchez et al., 2003). Nevertheless, limited research in vegetable and vineyard systems suggests that there is some soil C storage potential of a number of management practices, including cover cropping, adding grain crops in rotation with vegetables, compost additions, and no-tillage (e.g., Al-Sheikh et al., 2005; Steenwerth and Belina, 2008; see also reviews by Morgan and Follett, 2009 and T-AGG, 2010), although the latter has shown mixed results or only positive results in combination with cover crops and N fertilization (e.g., Sainju et al., 2002). One of the only orchard studies on soil C (Sanchez et al., 2003) found increases in soil C and N, including their mineralizable forms, in soils under cherry trees by using mulches, cover crops and composted manure applications compared to a conventional orchard system with herbicide-treated tree rows and high N fertilizer additions. Ground cover including natural weeds increased the active (mineralizable) N pool (25%), though the increase was much larger (60%) in combination with composted manure. This indicates that including living ground cover under orchard trees can both sequester C and enhance N supply to plants, in particular in combination with manure additions.

More research is needed in horticultural systems to evaluate the effect of different horticulture management practices on soil C stocks, and to quantify C sequestration in promising horticultural systems (Morgan and Follett, 2009; Morgan et al., 2010). Another pressing research need is to better quantify C storage in orchard C pools, both above- and below-ground. While the U.S. Forest Service's Forest Inventory and Analysis (FIA) Program has established reliable methods to quantify C stocks in forest biomass, little effort has been done for orchard trees. The few studies that do exist have tried to develop allometric equations to estimate biomass (whole tree, or different parts of the tree) based on trunk diameter or cross-sectional area, for apple trees (Westwood and Roberts, 1970; Barden et al., 2002) and pecan trees (Smith and Wood, 2006).

** Agroforestry/silvopasture*

Integrated production systems like agroforestry or silvopastoral systems, where woody plants are integrated into crop or pasture animal production systems, represents a significant opportunity for sequestering C on agricultural lands in that a substantial proportion of the C is sequestered in woody biomass, thus creating a system that sequesters a large amount of C per unit area and for a longer duration than many other practices (Montagnini and Nair, 2004; Schoeneberger, 2009; Morgan and Follett, 2009). The soil C sequestration potential can be also significant. The recent T-AGG review (2010) reported an average soil C accumulation of 0.74 t C (2.7 t CO_2 eq.) ha^{-1} yr^{-1} from agroforestry practices[1]. The C sequestration potential of agroforestry is however greatly dependent on the specific practice (alley cropping, silvopasture, windbreaks, etc.), individual site characteristics and the time frame. In

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[1] In this report, "t" represents tonnes or metric tons.

general, the more intensively planted with trees, the greater the C sequestration potential (T-AGG, 2010).

Critical research needs to better assess the C sequestration potential of agroforestry include developing effective strategies for measuring and monitoring C sequestration in soil and woody components, quantifying C dynamics in agroforestry systems, and developing biomass allometric equations for agroforestry trees (e.g., Zhou et al., 2007) (Morgan et al., 2010, Morgan and Follett, 2007).

** Short-rotation forestry*

The rising demand on biomass for energy purposes has increased the interest in woody biomass from short-rotation forestry (SRF) on former arable soils. Besides the high potential of sequestering C in biomass (and subsequent mitigation of GHG from bioenergy production), the minimal physical disturbance in these managed tree systems may also enhance C storage in the soil. Most SRF systems tend to be managed under no-till (Kahle et al., 2010). Estimates of the soil C sequestration potential of SRF (e.g., poplar, willow, etc.) compared to annual crops vary greatly across studies and could be influenced by tree species and climate conditions (T-AGG, 2010). Most studies reviewed in the T-AGG report (2010) found either an increase or no change in soil C stocks with growing woody crops (Table 2).

Table 2 Estimates of soil C sequestration rates (t CO_2 eq. ha^{-1} yr^{-1}) for agricultural and (agro)forestry practices in the U.S.

GHG mitigation practice	**Soil C (t CO_2 eq. ha^{-1} y^{-1})**	**Citation**
1. Land conversions		
Land conversion to perennial	2.49 (-0.15 – 4.74)	T-AGG, 2010
	Low: 1.58 (0.99 – 2.13) High: 3.45 (2.68 – 4.18)	Ogle et al., 2010[1]
2. Cropland		
Full till to reduced till	1.09 (0.00 – 1.82)	T-AGG, 2010
	Low: 0.24 (-0.27 - 0.74) High: 0.52 (0.13 – 0.92)	Ogle et al., 2010[1]
	Southeastern U.S.: 1.65 ± 0.15	Franzluebbers, 2010
	Central U.S.: 1.76 ± 2.16	Johnson et al., 2005
	Southwestern U.S.: 1.1 ± 0.77	Martens et al., 2005
	Northwestern U.S.: 0.99 ± 0.70	Liebig et al., 2005
Full till to no-till	1.09 (-0.26 – 2.60)	T-AGG, 2010
	Sandy Low: 0.04 (-1.03 – 1.11) High: 0.93 (0.10 – 1.76) Non-sandy Low: 0.15 (-0.59 – 0.92) High: 1.06 (0.26 – 1.83)	Ogle et al., 2010[1]
	Dry: 0.36 Humid: 0.81	Six et al., 2004
	0.37 – 2.56 *Average U.S.: 1.25*	CAST, 2004 *(West and Marland, 2002)*
Annual crop rotations	Eliminating fallow: 0.48 (-0.88 – 2.35) Diversification: 0.58 (-2.50 – 3.01)	T-AGG, 2010

	Winter cover crops: 0.83 (0.37 – 3.24)	
	Winter cover crops: -0.55 – 2.57	Ogle et al., 2010
Perennial crops	Include perennials in crop rotation: 0.57 (-1.75 – 2.20) Shift from annual to perennial: 2.26 (0.00 – 4.67)	T-AGG, 2010
N fertilization	0.18 – 0.55 (estimated for the U.S.)	Follett, 2001 (based on Lal et al., 1998 estimates)
	0.51 – 0.67 (for fertilization rates of 67 and 134 kg N ha-1 yr-1)	Halvorson et al., 1999
	9.17 kg/kg N applied	Franzluebbers and Steiner, 2002
	7.81 – 11.18 kg/kg N applied (U.S. and Canada)	Franzluebbers, 2005
	5.5 – 15.03/kg N applied (Canada)	VandenBygaert et al., 2003
Organic amendments	Manure:1.53 (0.18 – 5.10) Biochar:3.37 (0.13 – 8.92)	T-AGG, 2010
Cropland irrigation	Switch to irrigation: 1.46 (1.14 – 4.77) Irrigation improvements: 0.36 (0.18 – 0.58)	T-AGG, 2010
	0.65-0.70 (0-20 cm) 1.10 (0-75 cm)	Gillabel et al., 2007; Denef et al., 2008
	0.025 per mm increase in irrigation water applied	Blanco-Canqui et al., 2010
3. Pasture/rangeland		
Best management practices	Rangeland: 0.26 – 1.1 Pastures: 1.1 – 5.13	Morgan et al., 2010; Morgan and Follett, 2009
Improved grazing	1.28 2.94 (0.55 – 5.87) (pastures) 1.01 (-0.10 – 4.99) (rangeland) 0.00 – 1.1	Conant et al., 2001 T-AGG, 2010 T-AGG, 2010 Derner and Schuman, 2007
Rotational grazing (pastures)	2.20 (1.50 – 2.90) -0.73 – 5.5	T-AGG, 2010 Ogle et al., 2010
Fertilization	1.1 1.05 (0.37 – 5.87) 1.5 – 34.1	Conant et al., 2001 T-AGG, 2010 Derner and Schuman, 2007
Irrigation	0.40 1.04 (0.00 – 1.83)	Conant et al., 2001 T-AGG, 2010
Species management	2.75 - 11.15 2.44 (0.18 - 4.84) 1.21 to 5.72	Conant et al., 2001 T-AGG, 2010 Derner and Schuman, 2007
Introduction of earthworms	8.62	Conant et al., 2001
4. Planting trees		
Agroforestry	2.27 (0.84 – 4.23)	T-AGG, 2010
Woody crops	2.71 (0.00-10.22)	T-AGG, 2010

[1] Soil C stock changes were estimated over 20 years for the top 30 cm for different U.S. Land Resource Regions (LRR). The lowest and highest estimated mean and their 95% confidence interval (CI) are provided.

III.a.2. Soil N_2O emissions

(T-AGG, 2010; CAST, 2004; Snyder et al., 2007, 2009; Robertson and Vitousek, 2009; Akyiama et al., 2010; Swan et al., in prep; Ogle et al., 2010)

Cropland

Most agricultural management options that have been found to reduce N_2O emissions are related to controlling N inputs and minimizing N losses. Several major reviews on this subject have pointed out that the most effective strategies to reduce N_2O emissions from cropland consist of reducing N fertilization rates, placing fertilizers deeper in the soil profile, applying N fertilizer in spring rather than fall, and using nitrification inhibitors and slow-release fertilizers (CAST, 2004; Snyder et al., 2007, 2009; Robertson and Vitousek, 2009; Akiyama et al., 2010; Ogle et al., 2010; T-AGG, 2010).

** N fertilizer rate*

Nitrous oxide emissions are directly related to the amount of N added to soils. The latest IPCC guidelines for GHG inventories assume that 1% of the mineral N added to soils is lost as direct emission of N_2O. A slightly higher value (1.19%) was estimated by Ogle et al. (2010) based on published data from 31 sites in Europe and North America. N_2O emissions can therefore be reduced by lowering N application rates, though this practice should only be promoted to a level where crop yields are not compromised. The objective should therefore be to reduce N application rates in conjunction with N use efficiency gains (i.e., productivity per unit of N applied). Where producers over fertilize their crops, there is substantial potential to reduce N application rates without causing a decrease in crop yield (McSwiny et al., 2005; T-AGG, 2010). McSwiny and Robertson (2005) observed a threshold N fertilizer uptake capacity by the ecosystem, above which N_2O emissions rapidly increase without a concomitant increase in crop yield. This N uptake threshold could be estimated from pre-planting measurements of residual N, accounting for the normal N mineralization potential from SOM for soils in a field, and then filling in the deficit between the sum of these two N-inputs and reasonable crop yields with an appropriate amount of timely and well-placed N fertilizer (Snyder et al., 2007, 2009). Soil N tests, in particular the pre-sidedress soil nitrate test (PSNT) performed at planting, can decrease over fertilization, and consequently N_2O emissions, significantly, by allowing farmers to adjust N application rates accurately to yield goals (Snyder et al., 2007, 2009; Robertson and Vitousek, 2009). However, prefertilization soil tests have not always been effective predictors of future N needs in many areas where they have been tested. An alternative approach now being adopted in much of the U.S. Corn Belt is based on the site-specific N rate at which the economic value from increased grain yield just matches the cost of added N (Robertson and Vitousek, 2009).

The optimal overall strategy to reduce N application rates and associated N_2O emissions in conjunction with N use efficiency gains is through adoption of the right combination of fertilizer rate, placement, timing and type (e.g., enhanced-efficiency or slow-release fertilizers) (Roberts, 2007). The potential of each of these strategies to reduce N_2O emissions is described in the following paragraphs.

** N fertilizer placement*

Nitrogen fertilizer can be applied on the soil surface through broadcast applications, injected into the soil, or incorporated into the soil through tillage after surface application. Nitrogen losses by leaching or volatilization, and consequently N_2O emissions, can be minimized by placing N fertilizer near the zone of active root uptake. Injecting anhydrous ammonia into soil near rows has been found to decrease N leaching and volatilization by as much as 35% (CAST, 2004). Decreased N_2O emissions have also been found when comparing surface-applied urea to urea placed in a band below and to the side of the seed-row (Hultgreen and Leduc, 2003). Ogle et al. (2010) concluded that in order to decrease N_2O emissions, N fertilizer needs to be injected or incorporated deep enough in the soil profile. Drury et al. (2006) on the other hand reported lowest N_2O emissions with shallower placement of N fertilizer, though some of this could have been attributed to potential NH_3 losses (Snyder et al., 2007, 2009). The optimal depth of injection or incorporation may greatly depend on crop, climate and soil type, and should be further investigated before promoting deep N placement as a mitigation strategy for reducing N_2O emissions (T-AGG, 2010).

** N fertilizer timing*

Synchronizing N fertilizer application with crop growth will optimize N uptake and reduce losses of N from soil including N_2O emissions. Several studies have found lower N_2O emissions from spring compared to fall N fertilizer applications (Snyder et al., 2007, 2009; T-AGG, 2010). Reduced N_2O emissions have also been found when applying N in split applications, with a starter rate applied at planting and the remaining N applied once the crop has germinated and entered a rapid growth phase, especially in areas with greater rainfall or irrigation (see T-AGG report, 2010). However, other studies have not found reduced N_2O emissions with split versus one-time application of N fertilizer (Snyder et al., 2007). A review by Ogle et al. (2010) did not find a generalizable effect of fertilizer application timing on N_2O emissions, but their evaluation was limited by the number of studies included (only five). Optimizing timing of fertilizer application to crop uptake for minimizing N losses and N_2O emissions therefore remains to be further researched in different agricultural systems.

** Enhanced-efficiency fertilizers*

Enhanced-efficiency fertilizers, such as those containing nitrification or urease inhibitors, and slow- or controlled release fertilizers have been developed to increase the efficiency of fertilizer use by crops (Snyder et al. 2007, 2009). By delaying the enrichment of the soil mineral N pool until crop growth occurs, these compounds have the potential to decrease N_2O emissions (CAST, 2004; Akiyama et al., 2010). Nitrification inhibitors delay bacterial oxidation of NH_4^+ to NO_3^-, to help match the timing of NO_3^- supply with peak demand. Limiting the rate of nitrification could also reduce NO_3^- leaching which is a source of indirect N_2O emissions. Urease inhibitors delay the hydrolysis of urea fertilizer to NH_4^+, which depends on the enzyme urease. Slow-release fertilizers slow the rate of nutrient release through coating or chemical modification of the fertilizers. A recent meta-analysis by Akiyama et al. (2010) using field experiment data from 35 studies, reported N_2O emissions to decrease on average by 35% (95% CI: 14-58%) through slow-release fertilizer use, and by 38% (95% CI: 31-44%) upon the use of nitrification inhibitors. The latter was in the same range as what has been found by Ogle et al. (2010), where nitrification inhibitors reduced N_2O emissions on average by 32% ± 9%. Urease inhibitors did not effectively reduce N_2O emissions in Akyiama's meta-analysis. The combination of nitrification inhibitors with slow-release fertilizers has also been found to reduce N_2O losses from soils (Zaman et al., 2008). The use of nitrification inhibitors could be of particular interest in no-tilled soils, for the purpose of reversing the potential increase in N_2O emissions upon conversion to no-tillage (Bhatia et al., 2010). However, the effectiveness of enhanced-efficiency fertilizers in reducing N_2O emissions has varied across studies. For example, Parkin and Hatfield (2010) reported only a change in the N_2O flux in the near term when using nitrification inhibitors with ammonium fertilizers, while the annual flux was unaffected. Therefore, more research is needed to better understand the interactions of these compounds with fertilization (source, timing, placement, etc.) and soil conditions (temperature, pH, etc.) as well as with other management practices (T-AGG, 2010).

* Other

Some of the earlier described mitigation strategies that enhance soil C sequestration in agricultural land can alter soil conditions (moisture, N availability, etc.) that may enhance or reduce denitrification rates. Changing from full tillage to no-tillage for example can significantly increase soil C stocks depending on the region and cropping system, but at the same time may create more anaerobic conditions in the soil promoting denitrification. A meta-analysis by Six et al. (2004), comparing no-till with conventional till systems, found higher N_2O emissions following the adoption of no-till in mesic regions, but lower emissions in semi-arid regions. A more updated meta-analysis by Swan et al. (in prep), found similar variation with climate. They predicted increases in N_2O emissions with no-till adoption in humid climates between 0.27 and 0.88 kg N (between 130 and 430 kg CO_2 eq.) ha^{-1} $yr^{-}1$ on low to moderate clay soils, respectively; but decreases in N_2O emissions with no-till adoption in drier climates by as much as 1 kg N (500 kg CO_2 eq.) ha^{-1} yr^{-1} on similar soil types.[2] No-till systems can therefore lead to greater GHG emission reductions when implemented in semi-arid cropland regions when both C sequestration rates and N_2O emissions are included in the emission estimate, but conversely, the C sequestration offset will tend to be reduced in mesic cropland regions (Ogle et al., 2010).

The GHG mitigation potential of irrigation practices needs also to consider the potential increase in N_2O emissions due to the wet and anaerobic conditions in irrigated fields. The recent T-AGG study (2010) reported an average increase in N_2O and CH_4 emissions of 0.42 t CO_2 eq. ha^{-1} yr^{-1} when dryland is converted to irrigated land. On the other hand, improving the efficiency of existing irrigation practices has the potential to decrease N_2O emissions. Decreased N_2O emissions in the range of 0.14-0.94 t CO_2 eq. ha^{-1} yr^{-1} have been reported upon reduction of irrigation intensity and upon a switch from furrow irrigation to drip irrigation (see T-AGG, 2010 for a review).

Growing winter or fallow cover crops may reduce N_2O emissions from soils by preventing the buildup of residual mineral N in the soil, or indirectly by reducing the amount of mineral nitrogen fertilizer applied to fields to maintain crop productivity (CAST, 2004; McSwiney et al., 2010; T-AGG, 2010). In addition, cover crops can effectively reduce NO_3^- leaching in the winter months and subsequent drainage losses of NO_3^- (Snyder et al., 2007, 2009). Reductions in N leaching/drainage losses are likely to reduce N_2O emissions off-site. Using N-fixing cover crops can be especially beneficial by further reducing the need for mineral N fertilizer. The overall GHG mitigation potential of using cover crops is even larger when

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[2] The values for CO_2 eq. were calculated using a GWP of 310.

also considering the reduced energy use for fertilizer production and the large potential to sequester soil C.

There is currently a growing interest in the GHG mitigation potential of biochar, a highly recalcitrant product formed during the incomplete combustion of biomass, e.g., naturally by fire or as product of industrial pyrolysis). Research has suggested that biochar not only sequesters large amounts of carbon upon addition to soils, but it also has the potential to reduce N_2O (as well as CH_4) emissions (Spokas et al., 2009). However, the effectiveness of biochar applications to mitigate GHG emissions needs further research. On the one hand, more basic process-oriented research is needed to better understand how biochar influences C and N cycling in soils. In addition, more long-term field studies applying different types of biochar to soils in different regions and soil types are needed to develop accurate estimates of the GHG mitigation potential of biochar amendments.

** Indirect fluxes*

The most effective strategy for decreasing indirect N_2O emissions is by minimizing the losses of excess N to off-site environments, or by capturing the transported N from off-site environments. The CAST report (2004) lists a few mitigation strategies that aim at maximizing crop-N use efficiency on-site, maximizing N catchment by plants off-site (e.g., buffer strips), reducing NH_3 volatilization by optimizing manure management and N fertilization, and removing N in wastewater treatment plants. The earlier described strategies to reduce direct N_2O emissions from N fertilizer application also apply for indirect N_2O emissions. These strategies consist of better matching N supply with crop demand, e.g., by using soil/plant tests to determine fertilizer N needs, minimizing fallow periods and including cover crops to limit mineral N accumulation, and using the right combination of fertilizer rate, placement, timing and type (Snyder et al., 2007, 2009; Robertson and Vitousek, 2009; T-AGG, 2010). Other practices include better integration of animal and crop production systems in terms of manure re-use and plant production, and optimized tillage, irrigation and drainage practices (Mosier et al., 1996). The impact of these mitigation strategies on indirect N_2O emissions is still highly uncertain and requires further research in order to develop better estimates for GHG emissions accounting in systems where these practices are being introduced.

Pasture/Rangeland management

Nitrous oxide emissions from managed grasslands are largely tied to fertilizer application, nitrogen-fixing plants such as legumes, and emissions from dung and urine deposited by grazing animals. In grazed pastures, denitrification and N_2O emissions can also be stimulated as a result of increased soil compaction by treading animals. Emissions of N_2O will be promoted when animals are grazed in fertilized pastures or concentrated in small pastures as more N will be added than what can be taken up by pasture plants (CAST, 2004). The most effective strategy to reduce N_2O emissions from grazing land is by avoiding overfertilization with N either from mineral fertilizer or by the grazing animals themselves, and by maximizing plant uptake of added N. Therefore, reductions can be accomplished by reducing N fertilizer applications, using nitrification inhibitors, improving N nutrient management, reducing N content of forages, e.g., by species management, as well as by improved (e.g., lower intensity, rotation) grazing management (T-AGG, 2010; CAST, 2004; Snyder et al., 2007, 2009; Smith et al., 2008; Robertson and Vitousek, 2009). For example, in a review by Snyder et al. (2007, 2009), N_2O emissions from pastures were reported to be reduced by 40-92% when using nitrification inhibitors.

Conversions from cropland to natural grassland (e.g., setting aside under the CRP program) will typically result in lower N_2O emissions due to the elimination of fertilizer N use. A recent review by Swan et al. (in prep) reported decreased N_2O emissions with conversion of cropland to grassland, by as much as 4.5 kg N (2.2 t CO_2 eq.) ha^{-1} yr^{-1} if the amount of N fertilizer application was also reduced. In the case of no change in N fertilizer, N_2O emission changes were estimated to be near zero, while N_2O emissions were estimated to increase by as much as 6.9 kg N (3.4 t CO_2 eq.) ha^{-1} yr^{-1} if fertilization rates were higher on the grassland. Their analysis suggested that conversion of cropland to grassland can change soil N_2O emissions, but the amount of change depends greatly on N fertilizer management. Planting herbaceous buffers within a field or along a riparian area can further reduce N_2O emissions by capturing NO_3^- runoff from agricultural land before it becomes denitrified off-site (T-AGG, 2010).

Orchards, Agroforestry, Woody Crops

N_2O emissions tend to be reduced in agroforestry systems or in woody crop cultivations as these systems tend to rely less on N fertilization. However, results vary widely across studies. Machefert et al. (2002) found much lower N_2O emissions in forested versus agricultural land in Europe. Others have found little differences in N_2O emissions between annual crops and woody crop plantations (e.g., poplar) but much lower emissions from natural forests (Scheer et al., 2008). On the other hand, conventional horticulture systems tend to be heavily fertilized to maximize crop yields. Much higher fertilization rates were reported for example for lettuce compared to corn (T-AGG, 2010) while N removal by lettuce is much smaller than for corn. Combined with irrigation practices, horticulture land may be a significant source of N_2O emissions. N_2O emissions from horticulture land may be reduced by using the same mitigation strategies as promoted for annual cropland (e.g., timing and placement of N fertilizer, enhanced-efficiency fertilizers, etc.). Management practices such as cover crops or manure applications might reduce the need for N fertilizer and decrease N_2O emissions (e.g., Wyland et al., 1995). Further research needs include evaluating N_2O mitigation strategies for the different types of horticulture crops, and quantifying N_2O emissions from different management practices. In particular, research is lacking in orchard systems regarding both soil C sequestration and N_2O and CH_4 emissions.

Table 3 Estimates of N_2O emission reduction potential (t CO_2 eq. ha^{-1} yr^{-1}) for agricultural practices in the U.S.

GHG mitigation practice	**N_2O emission reduction (t CO_2 eq. ha^{-1} yr^{-1})**	**Citation**
N fertilizer rate	0.40 (0.14 – 1.32)	T-AGG, 2010
	0.019 t CO_2 eq./kg N applied	Ogle et al., 2010
N fertilizer placement	0.33 (0.12 – 0.47)	T-AGG, 2010
	ca. 0.002 t CO_2 eq./kg N applied at or below 10 cm depth	Ogle et al., 2010
N fertilizer timing	0.35 (0.01 – 0.52)	T-AGG, 2010
Enhanced efficiency fertilizer	SR: 35% (95% CI: 58-14%) NI: 38% (95% CI: 31-44%)	Akiyama et al., 2010
	NI: 32% ± 9%	Ogle et al., 2010
	NI: 33-96% [1]	Snyder et al., 2007, 2009 (adapted from Weiske, 2006)
	NI: 9-95%	T-AGG, 2010
	SR: 0.46 (0.00 – 1.43) NI: 0.49 (0.00 – 1.65)	T-AGG, 2010
Cover crops	0.25 (0.00 – 1.05)	T-AGG, 2010
Irrigation improvements	0.66 (0.14 – 0.94)	T-AGG, 2010
No-tillage	Humid climate: -0.26 to -0.86 Semiarid climate: 0.97	Swan et al., in prep
Biochar	1.14 (0.82 – 2.93) [2]	T-AGG, 2010
Grassland conversion	With reduced N-fertilization: 4.4	Swan et al., in prep
Agroforestry and woody crops	0.76 (0.00 – 1.52)	T-AGG, 2010

NI=nitrification inhibitors; SR=slow release fertilizers

[1] Includes also N_2O emission reductions estimated in pastures

[2] Includes also the reduction in CH_4 emissions

III.a.3. Soil CH_4 emissions

(T-AGG, 2010; Ogle et al., 2010; CAST, 2004)

Most of the CH_4 emitted from soils in the U.S. as well as globally come from rice fields (Robertson, 2004). A much smaller contribution to the CH_4 flux from cropland and grazing land arises from residue burning. The greatest potential to mitigate CH_4 emissions from U.S. agricultural land should therefore be found in rice cultivation management practices (Smith et al., 2006). Management options for limiting CH_4 emissions from rice include water, crop residue and nutrient management, and potentially also cultivation of specific rice cultivars (CAST, 2004; Ogle et al., 2010). Draining rice fields during the growing season has been shown to decrease CH_4 emissions, though this may, in certain cases, also increase N_2O emissions, in particular for high C soils (T-AGG, 2010). Water management during the non-growing season (winter, fallow periods) has shown more variable effects on CH_4 emissions, in particular with regard to winter flooding (T-AGG, 2010). Winter water management therefore remains questionable as a mitigation practice and may have to be further evaluated under different soil and climate conditions. Inorganic N fertilizer additions tend to decrease CH_4 emissions, whereas organic amendments (straw, compost) have been found to increase CH_4 emissions. Again, the impact on N_2O emissions needs to be better understood before promoting N fertilization as a GHG mitigation option in rice fields. Emissions from rice residue can be reduced by burning the residue instead of incorporating it into the soil before flooding. Ogle et al. (2010) estimated an average reduction rate of 101 ± 31 kg CH_4-C (2.83 ± 0.87 t CO_2 eq.) ha^{-1} yr^{-1} by burning, though a portion of this reduction will be offset when taking into account the emissions from the burning process. Alternative residue management practices such as residue removal and fermentation, should be further investigated (Ogle et al., 2010). Different but inconsistent CH_4 emission rates have been found for different rice cultivars (T-AGG, 2010). High yield rice crops can have significantly lower CH_4 emissions due to more C being allocated to the grain than to the rhizosphere where it can undergo methanogenesis (Smith et al., 2006; T-AGG, 2010). However, more region-specific research is needed before cultivar breeding becomes fully promoted as a mitigation practice in rice systems (CAST, 2004).

Table 4 Estimates of CH_4 emission reduction potential (t CO_2 eq. ha^{-1} yr^{-1}) for rice cultivation practices in the U.S.

GHG mitigation practice	**CH_4 emission reductions (t CO_2 eq. ha^{-1} yr^{-}1)**	**Citation**
Water management (draining)	1.56 (-0.88 – 5.22)	T-AGG, 2010[1]
Burning vs. incorporating straw	2.83 ± 0.87	Ogle et al., 2010
Rice cultivar selection	1.17 (0.00 – 2.71)	T-AGG, 2010

[1] Includes also studies in Asia, due to lack of U.S. studies

III.b. Manure management

(Kebreab et al., 2006; Robertson and Vitousek, 2009; T-AGG, 2010; Ogle et al., 2010; CAST, 2004; Saggar et al., 2004)

Livestock manure can produce N_2O and CH_4 emissions during manure storage and treatment, and upon application or deposition (grazing animals) to fields. Emissions are mainly affected by storage conditions (aeration, temperature, pH), land application conditions (timing, amount, method) and manure composition (e.g., nitrogen, sulfur, solids content). The latter varies by animal type, growth rate and diet (Kebreab et al., 2006). Reductions in N_2O and CH_4 emissions can therefore potentially be achieved by altering manure storage conditions, the animal's diet and the method of manure application to fields (Robertson and Vitousek, 2009). The impact of improved practices on these GHG emissions should always be evaluated for both gases combined as some N_2O mitigation practices may enhance CH_4 emissions and vice versa. Storing manure in aerobic conditions (e.g., solid storage in open-air mounds or pits, land deposition) or increasing aerobic conditions e.g., by adding straw or by compost stirring, generally reduces CH_4 emissions but may increase the N_2O flux (e.g., Brown et al., 2000; Hao et al., 2001). Storing manure in anaerobic conditions (e.g., as liquid/slurry) generally suppresses N_2O emissions but may stimulate CH_4 emissions. However, CH_4 emissions from anaerobic storage systems can be minimized by implementing technologies that capture and destroy emitted CH_4 for renewable energy production such as covered lagoons and anaerobic digesters (Clemens et al., 2006). Other practices that have been shown to reduce CH_4 emissions from manure storage include separating solids from slurry, covering and cooling of manure, and manure treatment with additives (Monteny et al., 2006; Amon et al., 2006, 2007; Hansen et al., 2006; Berg et al., 2006; Kebreab et al., 2006), although the results from experimental studies on these practices are not very consistent. Managing manure as a solid rather than liquid, or aerating the manure during composting can also reduce CH_4 emissions, though N_2O formation is likely to increase as a result. Finally, reducing manure storage time and applying manure to land as soon as possible will also reduce CH_4 emissions, but needs to be in synchrony with crop demand for N (to avoid N_2O emissions). Applying manure to land can also displace mineral N fertilizer use and potentially lead to lower N_2O emissions, in particular when manure is composted prior to field application (T-AGG, 2010).

The impact of aeration on CH_4 and N_2O emissions during manure storage does not always follow the same trend and appears to be influenced by animal type. A review of 31 studies addressing CH_4 and/or N_2O emissions found highest CH_4 emissions from liquid/slurry storage of dairy cattle, but for swine manure, highest CH_4 emissions were found when stored as solids (Ogle et al., 2010). Also for N_2O, the results were not always consistent with the general expectation of lower N_2O emissions under more anaerobic conditions. However, the reported data used by Ogle et al. (2010) are partly confounded by how the data in the different studies were normalized (on a mass or volume of manure basis). Other factors that limited data comparability in this review were the high variability in duration of emission measurements, differences in when (i.e., which season) emission measurements were performed, and the lack of information on the chemical composition of the manures which is known to affect both CH_4 and N_2O emissions (Kebreab et al., 2006). In addition, no information was provided on feeding regimes and diet composition. In fact, very little is known about the effects of diet on emissions from stored manure. Dietary changes may impact GHG emissions from manure by modifying the chemical properties of the manure, such as the C and N content, pH, C/N ratio, soluble C, inorganic versus organic forms and proportions, and volatile fatty acids (VFS) (Cardenas et al., 2007). Dietary changes that have been investigated in terms of their impact on GHG emissions from manure include forage type (e.g., Cardenas et al., 2007; Kulling et al., 2003), crude protein contents (Kulling et al., 2001, 2003) and feed supplements and additives (e.g., Hao et al., 2007, 2011). Increased corn ethanol production in the U.S. has led to an increased availability and use of high N containing co-products for livestock feed, such as dried distillers' grains with solubles (DDGS). However, increased use of DDGS in feedlot diets was found to result in greater N_2O emissions during manure composting (Hao et al., 2011).

A few studies have investigated the potential of 'precision feeding' for minimizing N excretion rates and subsequently N_2O emissions from manure. Reductions in N excretion have been achieved by lowering protein contents in the diet of non-ruminants, supplemented with specific amino acids; by feeding lactating cows on the basis of milk production and by adjusting metabolizable protein components (see Robertson and Vitousek, 2009). 'Phase feeding', in which rations are adjusted to animal age, can further decrease N excretion rates for all livestock (Robertson and Vitousek, 2009).

Improved manure application practices that can reduce N_2O emissions generally aim to maximize the likelihood for plant uptake of manure-derived N and avoid overfertilizing (CAST, 2004). These include (1) applying manure to fields only during or immediately prior to periods of active plant growth to promote manure-N immobilization by the crop; (2) adjusting the amount of manure based on crop N needs to prevent overfertilization with its associated N losses; (3) adjusting commercial N application rates to account for N addition in the manure; and (4) injecting liquid manure below the soil surface rather than surface application to prevent ammonia loss and reduce indirect N_2O emissions. Other studies have indicated the importance of the 'form' of manure that is being applied to the field. Reduced N_2O emissions were found upon application of solid manure compared to liquid manure (e.g., Gregorich et al., 2005). The use of nitrification inhibitors can also reduce N_2O losses following manure application to land (Saggar et al., 2004).

CH_4 emissions from feces and urine of grazing animals are less than 1% of enteric CH_4 emissions and much smaller than emissions from liquid manure storage. However, dung and urine deposits in grazed pastures have been shown to be rather important sources of N_2O. The most important management option to minimize N_2O emissions from grazing animals is by ensuring that pastures are not overfertilized with N added by the animals themselves and by maximizing the likelihood for plant uptake of added N (CAST, 2004).

Other useful papers on manure management:

<u>Facility type:</u> *Philippe et al., 2007; Amon et al., 2007; Fabbri et al., 2007*

<u>Liquid manure management:</u> *Sharpe et al., 2002; Berg et al., 2006; Amon et al., 2006, 2007; Martinez et al., 2003; Dinuccio et al., 2008*

<u>Composting:</u> *Fukimoto et al., 2003, 2006; Yamulki, 2006; Hao et al., 2001, 2004, 2005; Lopez-Real and Baptista, 1996; Sommer and Moller, 2000*

III.c. Enteric fermentation

(Boadi et al., 2004; Johnson and Johnson, 1995; Ominski and Wittenberg, 2004; Kebreab et al., 2006, 2008; Sejian et al., 2010; Ogle et al., 2010; CAST, 2004; Smith et al., 2007)

Livestock, in particular ruminants, lose a significant proportion of their ingested energy as CH_4 (2-12 % of GE intake) due to methanogenic activity within their digestive tract (Johnson and Johnson, 1995). Research on mitigating CH_4 emissions from livestock has been mainly driven by the need to reduce this loss of energy to animals, but has recently also been spurred by the important contribution of the livestock sector to global greenhouse gas emissions and the large opportunities in this sector for climate change mitigation (e.g., Steinfeld et al., 2006). Several reviews have been published in the last decade on the mechanisms controlling CH_4 emissions and mitigation strategies to reduce enteric CH_4 emissions, with a focus on ruminants (e.g., cattle, sheep, goat) (Boadi et al., 2004; Ominski and Wittenberg, 2004; Kebreab et al., 2006; Ogle et al., 2010). The dominant factors that control enteric CH_4 production are the availability of hydrogen (H_2) and the proportion of volatile fatty acids (VFA), specifically acetate: propionate produced in the rumen as a result of microbial fermentation. Dietary factors that influence the ratio of acetate:propionate include the residence time of the feed in the rumen, the type of carbohydrates consumed by the animals and diet digestibility (Ominski and Wittenberg, 2004). Hydrogen availability for CH_4 production can be reduced through the addition of fats, in particular unsaturated fatty acids, to the animal's diet, which can either reduce the H_2 supply or provide an alternative hydrogen sink through bio-hydrogenation (Kebreab et al., 2008). Level of intake also influences enteric methane emissions. Johnson and Johnson (1995) concluded that feeding highly available carbohydrates at limited intakes, results in high fractional methane losses while feeding highly available carbohydrates at high intakes leads to low fractional methane losses. Using data from 31 studies in North America and Europe, Ogle et al. (2010) found a strong correlation between enteric CH_4 emissions (log basis) and feed intake for different classes of cattle, but with a threshold of CH_4 emissions around 10-12 kg dry matter intake per head per day.

There are several management strategies that may be employed in the U.S. livestock sector to reduce enteric methane emissions via the mechanisms described above. These management strategies can be categorized as follows: diet quality (e.g., forage type, forage-to-grain), feed additives (e.g., ionophores, probiotics, essential oils, inhibitors, etc.), and improved production efficiencies (Kebreab et al., 2006; Ominski and Wittenberg, 2004; Boadi et al., 2004; Ogle et al., 2010). Methane can be reduced with higher quality diets by providing higher levels of easily digestible nonstructural carbohydrates, minimizing the use of fibrous grasses and hays, incorporating more grain and earlier harvesting/grazing of higher quality forages. Feed additives such as ionophores or edible oils have been found to decrease the formation of CH_4 by inhibiting methanogenic activity. Newer CH_4 mitigation options include the addition of probiotics, acetogens, bacteriocins, archaeal viruses, organic acids, plant extracts (e.g., essential oils) to the diet, as well as immunization, and genetic selection of cows. These new strategies are promising, but more research is needed to validate these approaches and to assess in vivo their effectiveness in reducing CH_4 production (Boadi et al., 2004). A cluster analysis by Ogle et al. (2010), including 31 studies on different diet modifications and feed additives, found the highest CH_4 reduction potential to be associated with manipulation of feed quality, in particular the amount and type of carbohydrates. Finally, optimizing animal performance, e.g., by improving water quality and providing adequate nutrient supplementation, is another suggested strategy by which the livestock industry may realize a net reduction in enteric methane emissions (Ominski and Wittenberg, 2004).

Longer term future technologies may develop methods to (genetically) alter the microbial population in the rumen in a way that would provide an alternative hydrogen sink (instead of CH_4) of more usefulness to the animal and less damage to the environment, or by altering the metabolic pathways within the microorganisms themselves (CAST, 2004; Kebreab et al., 2006; Ogle et al., 2010). Genetic selection of cattle based on residual feed intake (RFI) (i.e., the difference between energy intake and energy required for maintenance and weight gain) may also be considered as a mitigation strategy, in particular as this could also result in lower manure production because of reduced dry matter intake by low RFI cattle (Nkrumah et al., 2006; Kebreab et al., 2006).

Research has shown large variability in the observed effects from dietary changes on enteric methane emissions, and very few studies have been able to produce quantitative estimates for mitigation potential of individual practices (Ogle et al., 2010). This is largely attributed to the wide range of dietary manipulation experiments involving different livestock production systems and animal types, performed in different climate regions. Another factor that can explain this variability is the technique used to measure CH_4 emissions from livestock (Kebreab et al., 2008). While the most accurate method of assessing enteric production of methane is via whole body indirect calorimeters, this process requires highly specific equipment and expertise and is generally only used for individual or very small groups of animals. The commonly used SF_6 tracer technique generally underestimates emissions by about 8% mainly because emissions through the rectum are not accounted for in the measurements (Kebreab et al., 2006; Grainger et al., 2007). However, these techniques can be used with reasonable accuracy for inventory purposes. Also, very few reliable CH_4 data have been produced for pasture-based dairy and grazing beef animals due to questionable measurement techniques and the complexity of variables that affect their emissions such as dry matter intake and diet composition (Kebreab et al., 2008). In terms of future research, standardized measurements of enteric CH_4 emissions and more region- and production-specific evaluations would be useful to develop more reliable estimates of the CH_4 reduction potential of enteric mitigation strategies per livestock class.

It is equally important to evaluate CH_4 mitigation strategies in terms of the total greenhouse gas budget (Boadi et al., 2004). For example, in the case of CH_4 emissions, very few studies have evaluated the combined effect of dietary changes to both enteric and manure methane emissions. Diet manipulations that can reduce enteric methane emissions, such as by supplementing feed with concentrate, have been found to result in higher manure slurry methanogenesis because this is associated with extra amounts of undigested fiber, which may be a substrate for fecal microbes (Hindrichsen et al., 2006). Therefore, combining enteric and slurry methane measurements are essential in future research to quantify the true effect of methane mitigation strategies.

Table 5 presents the mitigation potentials in livestock management for reducing GHG emissions via improved manure handling, animal feeding practices, specific dietary additives and longer term structural and management changes/animal breeding. The majority of emission reduction values for manure management are derived from individual studies on specific manure handling systems, as only a limited amount of reviews have been published in this field.

Table 5 Estimates of GHG emission reduction potential (t CO_2 eq. ha^{-1} yr^{-1}) for livestock management practices in the U.S.

GHG mitigation practice	**CH_4 and N_2O emission reductions (t CO_2 eq. ha^{-1} yr^{-1} or %)**	**Citation**
1. Manure management	**CH_4 and N_2O emission reductions**	
Diet manipulation	Forage quality: * Up to 70% reduction in N_2O * Up to 83% reduction in CH_4 * No effect of forage type Feed additives: * No changes (lignite-coal extract additions) * 117% increase in N_2O; 7% decrease in CH_4 (DDGS additions) * 18% decrease in CH_4 (concentrate additions)	Cardenas et al., 2007 (laboratory incubations) Kulling et al., 2003 Hao et al., 2007 Hao et al., 2011 Hindrichsen et al., 2006
Manure storage/treatment	* Slurry separation: 37% * Anaerobic digestion: 59% * Slurry aeration: 42% * Covering: 48-52% * Covering manure solids: 88% (CH_4), 99% (N_2O)	Amon et al., 2006[1] Amon et al., 2006[1] Amon et al., 2006[1] Amon et al., 2007 Hansen et al., 2006
Composting	Composting: 31-78% (compared to untreated manure storage) Compost manipulations: * Straw additions: 32% * Phosphogypsum amendment: >58% * Decreased pile size: 20% (N_2O), 47% (CH_4)	T-AGG, 2010 Yamulki et al., 2006 Hao et al., 2005 Fukumoto et al., 2003
Land application *(timing, amount, injection, form)*	* 0.00 (-0.17 – 1.30) t CO_2 eq. ha^{-1} yr^{-1} * 0.89 (0.37 – 1.22) t CO_2 eq. ha^{-1} yr^{-1}	Smith et al., 2007 T-AGG, 2010
2. Enteric fermentation	**CH_4 emission reductions**	
Diet quality	* 10 (4 –16) % * 9-40% * Up to 43% (forage quality) * 29-44% (early grazing)	Smith et al., 2007[2] Boadi et al., 2004 Ominski and Wittenberg, 2004 Ominski and Wittenberg, 2004
Feed additives	* 6.8 (0.4-11) % * 5-71% * Up to 33% (canola oil) * 30% (sunflower seed)	Smith et al., 2007[2] Boadi et al., 2004 Ominski and Wittenberg, 2004 Ominski and Wittenberg, 2004
Genetic selection	21%	Boadi et al., 2004
Production efficiency	* 2.1 (0.3-3.0) % * 20-30%	Smith et al., 2007[2] Boadi et al., 2004

[1] Include N_2O and CH_4 emissions during storage and field application

[2] Reduction potential estimates for North America (IPCC Fourth Assessment Report)

III.d. Forests and Forest Management

Much effort has been made in the U.S. to develop reliable estimates of forest C pools by the U.S. Forest Service's Forest Inventory and Analysis (FIA) Program, the U.S. authority for estimates of forest carbon stocks in the United States. This program continuously has been collecting data on trees, soils, and forest land area in private and public lands in all 50 States and has provided annual estimates of aboveground and belowground forest carbon stocks at the national level to EPA for international reporting since 1993 (EPA, 2010). Recently, FIA scientists have developed new methods for breaking these national estimates down into regional and state-level estimates as well as for individual forest types. Some limitations include the large spatial scale at which the FIA inventories are conducted limiting the reliability of estimates of carbon stocks at smaller spatial scales, and the much greater uncertainty around estimates for soil C pools compared to aboveground C pools. Measuring changes in soil C is more difficult because its spatial variability is high and soil C accumulation is a slow process (Conen et al., 2004). Moreover, FIA estimates do not provide any information on management or land-use effects on forest C stocks or N_2O and CH_4 emissions. Therefore, the following paragraphs focus on strategies that can enhance soil C sequestration, or minimize GHG emissions from forested land. These strategies can be categorized as *(1) land conversion or forest preservation strategies*, including afforestation, reforestation and avoided deforestation, and *(2) forest management strategies,* such as harvesting, thinning, fertilization, drainage, tree species selection, and control of natural disturbances. While some of these strategies have great potential to increase forest C stocks, they may lead to C losses or GHG emissions elsewhere due to relocation of activities (i.e., C leakage). It is therefore important to recognize these trade-offs before promoting forest management practices for C sequestration (Ryan et al., 2010).

III.d.1. Afforestation/reforestation

Afforestation and reforestation involve the introduction of trees on land which has been without trees for some time. The conversion of treeless land to forest has been cited as an effective method for reducing the atmospheric CO_2 concentration because of the ability to sequester C in vegetation and soil (e.g., Jandl et al., 2007; Laganière et al., 2010). Studies have reported wide ranges in the rate of C sequestration upon afforestation/reforestation. For the U.S., estimated ranges were 1.48 to 6.4 t C (5.43 to 23 t CO_2 eq.) ha^{-1} yr^{-1} for afforestation (including both biomass and soil C) and 0.74 to 5.18 t C (2.7 to 19.0 t CO_2 eq.) ha^{-1} yr^{-1} for reforestation (including only biomass C) (EPA, 2005; based on Birdsey, 1996). Wide ranges in C accumulation rates in the soil upon forest establishments have also been documented. Jenkinson (1971) found a soil C increase of 0.596 t C (2.19 t CO_2 eq.) ha^{-1} yr^{-1} after conversion of an old field to an oak forest, while Robertson & Vitousek (1981) reported a value of only 0.094 t C (0.34 t CO_2 eq.) ha^{-1} yr^{-1} for the same vegetation conversion type. Other studies have reported decreases or no changes upon afforestation (Laganière et al., 2010). A review by Post and Kwon (2000) came up with an average soil C sequestration rate of 0.338 t C (1.24 t CO_2 eq.) ha^{-1} yr^{-1}, but values ranged from -0.51 to 3.1 t C (-1.9 to 11 t CO_2 eq.) ha^{-1} yr^{-1}, with a tendency for C sequestration rates to increase from temperate regions to subtropical regions. From these observations, the authors suggested that soil C sequestration by forest establishment is mainly driven by plant productivity, which increases with temperature and moisture. A number of literature reviews, including the one by Post and Kwon (2000) have evaluated the impact of afforestation on soil organic C storage across a wide range of field studies, in an attempt to better understand the mechanisms and controlling factors that influence the recovery of SOC stocks following afforestation (e.g., Post and Kwon, 2000; Paul et al., 2002; Guo and Gifford, 2002; Laganière et al., 2010). Paul et al. (2002) identified previous land use as the main determining factor, followed by climate and species planted, while Post & Kwon (2000) identified plant productivity, soil physical and biological properties, the history of C inputs and physical soil disturbance. Guo & Gifford (2002) concluded that afforestation of pastureland does not affect soil C stocks when deciduous species are planted, and that soil C stocks decline when pine is planted. However, when afforestation takes place on cropland, restoration of SOC stocks does occur. The recent meta-analysis by Laganière et al. (2010) found the most contributing factors to be land use history, tree species planted, soil clay content and preplanting disturbance, and, to a lesser extent, climatic zone. In particular, their meta-analysis found greater soil C sequestration rates following afforestation (1) on cropland compared to pastures or natural grasslands; (2) with broadleaf tree species compared to coniferous species; (3) in clay-rich soils (>33%) compared to soils with a lower clay content (<33%); and (4) with lower preplanting disturbances. The authors also highlighted that much of the variability in observed SOC responses to afforestation across field studies may be explained by methodological factors, e.g., the inclusion or exclusion of the organic layer in the calculation of SOC stocks, plantation age, and study design.

III.d.2. Reducing/Avoiding deforestation

The GHG benefits of reducing or avoiding deforestation are in many ways very similar as those from afforestation or reforestation. Most of the GHG mitigation potential lays in its avoidance of significant losses of forest C stocks upon forest clearing. However, the process of deforestation—clearing forest land for another use —releases carbon into the atmosphere much more rapidly upon the time of harvest than what would be restored following afforestation or reforestation. The rate of avoided GHG emissions by avoiding deforestation in the U.S. was estimated by EPA (2005) to range between 207 and 425 t CO_2 eq. ha^{-1} yr^{-1}. These estimates were based on acres deforested land from National Resource Inventory (NRI) data and carbon stock declines estimated by the FORCARB model, from 1990 to 1997.

III.d.3. Forest management

The management of existing forests can also sequester C, either by reducing vegetation removal (e.g., from harvesting, thinning and prescribed burning) or soil carbon losses (from activities causing soil disturbance), or by increasing forest productivity and C inputs to soil (Jandl et al., 2007; Ryan et al., 2010). For the U.S., the EPA (2005) reported ranges between 1.41 and 2.09 t C (between 5.17 and 7.66 t CO_2 eq.) ha^{-1} yr^{-1} sequestered due to changes in forest management, based on data from Row (1996).

Harvesting at an older forest age or reducing the amount of biomass removed in a harvest event can store more carbon in the forest (see Figure 1 from Ryan et al., 2010). However, there is still some skepticism with respect to the role of the elongation of the harvest interval on C sequestration (Jandl et al., 2007). Forests beyond a certain age are not that productive anymore and become also more susceptible to disturbances (Ryan et al., 2004).

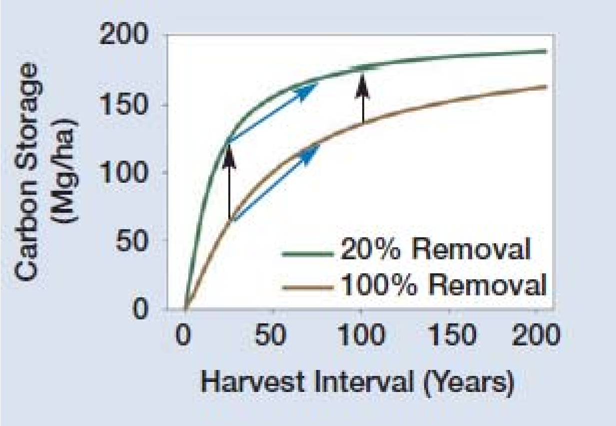

Figure 1 Impact of time between harvest age of forests (harvest interval) and how much biomass is removed each harvest on forest C storage (from Ryan et al., 2010).

Forest management can also affect soil C stocks. A meta-analysis by Johnson and Curtis (2001) found that while forest harvesting often had little or no effect on soil C stocks, significant effects of harvest type and species were noted. In particular, sawlog harvesting was found to increase (+18%) soil C, while whole tree harvesting caused a decrease (-6%). The positive effect of sawlog harvesting was restricted to coniferous species. Similarly, Hoover (2003) found limited impacts of forest harvesting on soil C storage in their literature review, while also indicating that effects likely depend on climate, forest type, and soil properties. They also suggested that effects may occur over a longer time frame than is traditionally studied.

Optimized forest management with regard to soil C sequestration should aim to secure a high productivity of the forest to maximize C inputs, and avoid soil disturbances as much as possible to maximize C stabilization in the soil (Jandl et al., 2007). Depending on the forest type, management options to increase productivity may include controlled burning, thinning, controlling competing vegetation, fertilizing, liming, planting N-fixing trees, planting genetically improved trees, and selecting species for superior productivity (Ryan et al., 2010). Both fertilization and N-fixing vegetation have been found to increase soil C (Johnson, 1992; Johnson and Curtis, 2002; Jandl et al., 2007). However, inefficient use of fertilizer N by the trees may lead to the formation of N_2O. Methane uptake and oxidation in the soil may also be reduced following N fertilization. Nitrogen fertilization as a mitigation practice for sequestering C in forest soils should therefore be treated with caution and should always be evaluated based on complete GHG budgets (Hoover, 2003). Liming on the other hand may have the opposite effect of fertilization, increasing CH_4 uptake and decreasing emissions of N_2O. Liming of forest soils is much more common in Northern and Central Europe than in the U.S. and is mainly done to mitigate acidification. The effects of liming on soil C stocks are still uncertain, mainly due to the lack of studies on this subject. The few existing studies have found liming to reduce the C content of the forest-floor as a result of higher microbial activity as well as increased leakage of dissolved organic C (e.g., Lundström et al., 2003), while no significant differences were found in the mineral soil C pool (Hoover, 2003).

Many forest systems, in particular in the Western U.S., are overloaded with down dead wood that has accumulated from long-term fire suppression management. This situation provides a large amount of fuel for wildfires which have dramatic consequences for soil C stocks (increasing risk for erosional losses, potentially large soil C losses, Elliot, 2003; Hoover, 2003; Page-Dumroese et al., 2003). Frequent controlled burning can potentially prevent some of the larger impacts expected from intense wildfires. The extent and impact of soil changes under controlled burning conditions are still not fully understood. Several reviews have found mixed results in terms of the immediate effects following prescribed fires, but generally found these effects to be transitory (Johnson and Curtis, 2001; Page-Dumroese et al., 2003; Hoover, 2003). However, the effects of fire on the chemistry of carbon (in particular on black C formation and its properties) as well as on forest species composition needs further investigation to better understand the effects of fire on forest C pools (Hoover, 2003). There have been cases reported where wildfires were found to increase soil C stocks (Johnson and Curtis, 2001), owing to the introduction of charcoal, hydrophobic organic matter and to the invasion of N-fixing species.

IV. Current research gaps

The table below summarizes critical research needs to better estimate the C sequestration and GHG mitigation potential of different strategies applicable to U.S. agriculture and forestry.

Table 6 Summary of critical research needs.

Topics	Research needs
General	Conduct more field studies in different regions, soil types, management practices in order to clarify management and environmental interactions on C sequestration and GHG emissions.
	Conduct simultaneous measurements of all three GHGs (CO_2, N_2O, and CH_4) from agricultural practices over extended time periods.
	Validate and expand experimental findings from experimental field sites to real-time situations, with inventory and monitoring of soil C stocks over time on working farms, especially linkages to existing land use inventories (e.g., National Resource Inventory (NRI) system.
	Evaluate specific mitigation strategies in terms of the total greenhouse gas budget (i.e., emissions of all GHG's, and impacts of mitigation strategies on emissions up/downstream) and consider the cost associated with the various strategies.
Methodologies/data reporting	Standardize SOC and GHG emission assessment methods in order to improve estimates of GHG emissions and C sequestration rates from land-use changes.
	Improve consistency in methodologies/standardized measurements for quantifying GHG emissions from livestock operations (enteric, manure storage) and more standardized ways of expressing emission rates or conversion approaches, to improve data comparability.

	In particular for enteric CH_4 emissions, there is a need for standardized CH_4 measurement methods and within specific production systems to produce better estimates of CH_4 emissions per livestock category and for the different mitigation practices.
Indirect GHG emissions/ C sequestration	Evaluate the fate of eroded C and N losses from nitrate leaching/runoff or volatilization.
C sequestration	Include deeper depths in C assessments to resolve which types of management changes are likely to have impacts deeper in the profile than the plow layer, and the mechanisms involved.
	Clarify management impacts (e.g., irrigation) on soil inorganic C dynamics and quantify contribution to C stock changes.
	Quantify soil C changes with cultivation of woody crops, agroforestry, and management of orchards (almost no data presently) in U.S. agriculture. Develop allometric functions and biomass estimates for agroforestry, orchards and other non-forest ecosystems containing woody species.
	More studies on processes/factors controlling SOC dynamics in pastures and rangelands. Critical research needs include quantifying C sequestration in arid and semi-arid shrublands (almost no data presently), evaluating impact of different forage species mixtures, and quantifying interactions of management with climate.
Non-CO_2 GHG emissions	Conduct more research on CH_4 and N_2O fluxes under almost any agricultural operation in the region; conduct more basic research on factors affecting microbial CH_4 consumption; enhance understanding of spatial variability of N_2O and CH_4.
	Evaluate interaction of different management practices (N fertilization, tillage, fertilizer type, rate, timing, application method) and impact on N_2O emissions.
Promising strategies needing further research	Conduct more studies to clarify impacts of including perennials in annual crop rotations.
	Quantify impact of biochar amendments on C sequestration and GHG emissions at larger scale and evaluate interaction with environmental conditions. Implement more field applications of biochar to better understand and predict its GHG mitigation potential in different soil types/regions/etc.

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