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Quantifying Greenhouse Gas Fluxes in Agriculture and Forestry:
Methods for Entity-Scale Inventory

Chapter 3
Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems

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Acronyms, Chemical Formulae, and Units

C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
CRP	Conservation Reserve Program
EPA	U.S. Environmental Protection Agency
GHG	Greenhouse gas
H_2CO_3	Carbonic acid
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
K	Potassium
LRR	Land Resource Region
m	Meter
Mg	Megagrams
N	Nitrogen
N_2	Nitrogen gas
N_2O	Nitrous Oxide
NH_4^+	Ammonium
NO	Nitric oxide
NO_3^-	Nitrate
NO_x	Mono-nitrous oxides
NRCS	Natural Resources Conservation Service
NUE	Nitrogen use efficiency
O_2	Oxygen
Pg	Petagram
PRISM	Parameter-Elevation Regressions on Independent Slopes Model
SOC	Soil organic carbon
SOM	Soil organic matter
SSURGO	Soil Survey Geographic Database
Tg	Teragrams
USDA	U.S. Department of Agriculture

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3 Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems

This chapter provides methodologies and guidance for reporting greenhouse gas (GHG) emissions and sinks at the entity scale for cropland and grazing land systems. More specifically, it focuses on methods for land used for the production of crops and livestock (i.e., grazing lands). Section 3.1 provides an overview of cropland and grazing land systems management practices and resulting GHG emissions, system boundaries and temporal scale, a summary of the selected methods/models, sources of data, and a roadmap for the chapter. Section 3.2 presents the various management practices that influence GHG emissions in upland and wetland cropping systems and land-use change to cropland. Section 3.3 provides a similar discussion for grazing land systems and land-use change to grazing systems. Section 3.4 discusses agroforestry, and Section 3.5 provides the estimation methods. Finally, Section 3.6 includes a summary of research gaps with additional information on the nitrous oxide (N_2O) methodology and supplemental methodology guidance in the Appendices.

3.1 Overview

Cropland and grazing land systems are managed in a variety of ways, which results in varying degrees of GHG emissions or sinks. Table 3-1 provides a description of the sources of emissions or sinks and the section in which methodologies are provided along with the corresponding GHGs.

Table 3-1: Overview of Cropland and Grazing Land Systems Sources and Associated Greenhouse Gases

Source	Method for GHG Estimation			Description
	CO_2	N_2O	CH_4	
Biomass and litter carbon stock changes	✓			Estimating herbaceous biomass carbon stock during changes in land use is necessary to account for the influence of herbaceous plants on carbon dioxide (CO_2) uptake from the atmosphere and storage in the terrestrial biosphere for at least a portion of the year relative to the biomass carbon and associated CO_2 uptake in the previous land use system. Agroforestry systems also have a longer term gain or loss of carbon based on the management of trees in these systems.
Soil organic carbon stocks for mineral soils	✓			Soil organic carbon stocks are influenced by land use and management in cropland and grazing land systems, as well as conversion from other land uses into these systems (Aalde et al., 2006). Soil organic carbon pools can be modified due to changes in carbon inputs and outputs (Paustian et al., 1997).
Soil organic carbon stocks for organic soils	✓			Emissions occur in organic soils following drainage due to the conversion of an anaerobic environment with a high water table to aerobic conditions (Armentano and Menges, 1986), resulting in a significant loss of carbon to the atmosphere (Ogle et al., 2003).
Direct and indirect N_2O emissions from mineral soils		✓		N_2O is emitted from cropland both directly and indirectly. Direct emissions are fluxes from cropland or grazing lands where there are nitrogen additions or nitrogen mineralized from soil organic matter. Indirect emissions occur when reactive nitrogen is volatilized as ammonia (NH_3) or nitrogen oxide (NOx), or transported via surface runoff or leaching in soluble forms from cropland or grazing lands, leading to N_2O emissions in another location.

Source	Method for GHG Estimation			Description
	CO_2	N_2O	CH_4	
Direct N_2O emissions from drainage of organic soils		✓		Organic soils (i.e., histosols) are a special case in which drainage leads to high rates of nitrogen mineralization and increased N_2O emissions. The method assumes that organic soils have a significant organic horizon in the soil, and therefore, the main inputs of nitrogen are from oxidation of organic matter.
Methane uptake by soils			✓	Agronomic activity universally reduces methanotrophy in arable soils by 70% or more (Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000). Recovery of methane (CH_4) oxidation upon abandonment from agriculture is slow, taking 50 to 100 years for the development of even 50% of former (original) rates (Levine et al., 2011).
Methane and N_2O emissions from rice cultivation		✓	✓	There are a number of management practices that affect CH_4 and N_2O emissions from rice systems. The method addresses key practices including the influence of water management, residue management and organic amendments on CH_4 emissions from rice (Lasco et al., 2006; Yan et al., 2005) and associated impacts on N_2O emissions.
CO_2 from liming	✓			Addition of lime to soils is typically thought to generate CO_2 emissions to the atmosphere (de Klein et al., 2006). However, prevailing conditions in U.S. agricultural lands lead to CO_2 uptake because the majority of lime is dissolved in the presence of carbonic acid (H_2CO_3). Therefore, the addition of lime will lead to a carbon sink in the majority of U.S. cropland and grazing land systems.
Non-CO_2 emissions from biomass burning		✓	✓	Biomass burning leads to emissions of CO_2 as well as other GHGs or precursors to GHGs that are formed later through additional chemical reactions. Note: CO_2 emissions are not addressed for crop residues or grassland burning, because the carbon is re-absorbed from the atmosphere in new growth of crops or grasses within an annual cycle.
CO_2 from urea fertilizer application	✓			Urea fertilizer application to soils contributes CO_2 emissions to the atmosphere. The CO_2 emitted is incorporated into the urea during the manufacturing process. In the United States, the source of the CO_2 is fossil fuel used for NH_3 production. The CO_2 captured during NH_3 production is included in the manufacturer's reporting so its release via urea fertilization is an additional CO_2 emission to the atmosphere and is included in the farm-scale entity reporting.

3.1.1   Overview of Management Practices and Resulting GHG Emissions

Guidance is provided in this section for reporting of GHG emissions associated with entity-level fluxes from farm and/or livestock operations. The guidance focuses on methods for estimating the influence of land use and management practices on GHG emissions (and sinks) in crop and grazing land systems. Methods are described for estimating biomass and soil carbon stock changes, soil N_2O emissions, CH_4 emissions from flooded rice, CH_4 sinks from methanotrophic activity, CO_2 emissions or sinks from liming, biomass burning non-CO_2 GHG emissions, and CO_2 emissions from urea fertilizer application (see Table 3-2).

Table 3-2: Overview of Cropland and Grazing Land Systems Sources, Method and Section

Section	Source	Method
3.5.1-3.5.2	Biomass carbon stock changes	Herbaceous biomass is estimated with an IPCC Tier 2 method using entity specific data as input into the IPCC equations developed by Lasco et al. (2006) and Verchot et al. (2006). Woody plant growth and losses in agroforestry or perennial tree crops are estimated with an IPCC Tier 3 method, using a simulation model approach with entity input.
3.5.3	Soil organic carbon stocks for mineral soils	An IPCC Tier 3 method is used to estimate the SOC at the beginning and end of the year for mineral soils with the DAYCENT process-based model. The stocks are entered into the IPCC equations developed by Lasco et al. (2006), Verchot et al. (2006) to estimate carbon stock changes.
3.5.3	Soil organic carbon stocks for organic soils	CO_2 emissions from drainage of organic soils (i.e., Histosols) are estimated with an IPCC Tier 2 method using the IPCC equation developed by Aalde et al. (2006) and region specific emission factors from Ogle et al. (2003).
3.5.4	Direct N_2O emissions from mineral soils	The direct N_2O methods are estimated with an IPCC Tier 3 method. For major commodity crops, a combination of experimental data and process-based modeling using the DAYCENT[1] model and DNDC[2] (denitrification-decomposition) are used to derive expected base emission rates for different soil texture classes in each U.S. Department of Agriculture Land Resource Region. For minor commodity crops and in cases where there are insufficient empirical data to derive a base emission rate, the base emission rate is based on the IPCC default factor multiplied by the nitrogen input (de Klein et al., 2006). These emission rates are scaled with practice-based scaling factors to estimate the influence of management changes such as application of nitrification inhibitors or slow-release fertilizers.
3.5.4	Direct N_2O emissions from drainage of organic soils	Direct N_2O emissions from drainage of organic soils, i.e., Histosols, are estimated with the IPCC Tier 1 method (de Klein et al., 2006).
3.5.4	Indirect N_2O emissions	Indirect soil N_2O emissions are estimated with the IPCC Tier 1 method (de Klein et al., 2006).
3.5.5	Methane uptake by soils	Methane uptake by soil is estimated with an equation that uses average values for methane oxidation in natural vegetation—whether grassland, coniferous forest, or deciduous forest—attenuated by current land use practices. This approach is an IPCC Tier 3 method.
3.5.6	Methane and N_2O emissions from flooded rice cultivation	IPCC Tier 1 methods are used to estimate CH_4 and N_2O emissions from flooded rice production (de Klein et al., 2006; Lasco et al., 2006).

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[1] The version of DAYCENT coded and parameterized for the most recent U.S. national GHG inventory (U.S. EPA, 2013) was used to derive expected base emission rates.
[2] DNDC 9.5 compiled on Feb 25, 2013 was used to derive expected base emission rates.

Section	Source	Method
3.5.7	CO_2 from liming	An IPCC Tier 2 method is used to estimate CO_2 emissions from application of carbonate limes (de Klein et al., 2006) with U.S.-specific emissions factors (adapted from West and McBride, 2005).
3.5.8	Non-CO_2 emissions from biomass burning	Non-CO_2 GHG emissions from biomass burning of grazing land vegetation or crop residues are estimated with the IPCC Tier 2 method (Aalde et al., 2006).
3.5.9	CO_2 from urea fertilizer application	CO_2 emissions from application of urea or urea-based fertilizers to soils are estimated with the IPCC Tier 1 method (de Klein et al., 2006).

3.1.1.1   Description of Sector

Croplands include all systems used to produce food, feed, and fiber commodities, in addition to feedstocks for bioenergy production. Croplands are used for the production of adapted crops for harvest and include both cultivated and non-cultivated crops (U.S. EPA, 2013). Cultivated crops are typically categorized as row or close-grown crops, such as corn, soybeans, and wheat. Non-cultivated crops (or those occasionally cultivated to replenish the crop) include hay, perennial crops (e.g., orchards and vineyards), and horticultural crops. The majority of U.S. cropland is in upland systems outside of wetlands as defined in Section 4.1.1, Wetlands, and upland cropping systems (i.e., dry land) may or may not be irrigated. Rice can be grown on natural or constructed wetlands, but we will refer to these systems as flooded rice to avoid confusion with Chapter 4. In addition, wetlands can also be drained for crop production, which again is considered a cropland because the principal use is crop production. Some croplands are set aside in reserve, such as lands enrolled in the Conservation Reserve Program (CRP). Croplands also include agroforestry systems that are a mixture of crops and trees, such as alley cropping, shelterbelts, and riparian buffers.

Grazing lands are systems that are used for livestock production, and occur primarily on grasslands. Grasslands are composed principally of grasses, grass-like plants, forbs, or shrubs suitable for grazing and browsing, and include both pastures and native rangelands (U.S. EPA, 2013). Furthermore, savannas, some wetlands and deserts, and tundra can be considered grazing lands in the United States if used for livestock production. Grazing land systems include: (1) managed pastures that may require periodic clearing, burning, chaining, and/or chemicals to maintain the grass vegetation; and (2) native rangelands that typically require limited management to maintain but may be degraded if overstocked or otherwise overused.

Crop and grazing land management influences GHG emissions (Smith et al., 2008b), which can be reduced by adopting conservation practices (CAST, 2004; 2011). Operators of cropland systems use a variety of practices that have implications for emissions, such as nutrient additions, irrigation, liming applications, tillage practices, residue management, fallowing fields, forage and crop selection, set-asides of lands in reserve programs, erosion control practices, water table management in wetlands, and drainage of wetlands. Operators of grazing systems also have a variety of management options that influence GHG emissions, such as stocking rate, forage selection, use of prescribed fires, nutrient applications, wetland drainage, irrigation, liming applications, and silvopastoral practices.

3.1.1.2   Resulting GHG Emissions

Cropland and grazing lands are sources of N_2O and CH_4 emissions and have a large potential to sequester carbon with changes in management (Smith et al., 2008b). In fact, N_2O emissions from

management of agricultural soils are a key source of GHG emissions in the United States (U.S. EPA, 2013). N_2O emissions result from the processes of nitrification and denitrification, which are influenced by land use and management activity. Land use and management can also influence carbon stocks in biomass, dead biomass, and soil pools. Carbon stocks can be enhanced or reduced depending on land use and management practices (CAST, 2004; IPCC, 2000; Smith et al., 2008b). Consequently, crop and grazing land systems can be either a source or sink for CO_2, depending on the net changes in biomass, dead biomass, and soil carbon. Burning biomass is a practice that can initially reduce biomass carbon stock but can provide sufficient stimulus to enhance ensuing ecosystem carbon storage. In general though, burning causes a decline in soil organic carbon stocks due to loss of carbon input from plant litter and roots. Burning will also lead to non-CO_2 GHG emissions—CH_4, N_2O, and other aerosol gases (CO, NO_x)—that can be later converted to GHGs in the atmosphere or once deposited onto soil.

Soils in crop and grazing land systems can also be a source or sink for CH_4 depending on the conditions and management of soil. CH_4 can be removed from the atmosphere through the process of methanotrophy in soils. Methanotrophy occurs under aerobic conditions and is common in most soils that do not have standing water. In contrast, CH_4 is produced in soils through the process of methanogenesis, which occurs under anaerobic conditions (e.g., soils with standing water such as soils used for flooded rice production). Both of these processes are driven by the activity of microorganisms in soils, and their rate of activity is influenced by land use and management.

3.1.1.3 Management interactions

The influence of crop and grazing land management on GHG emissions is not typically the simple sum of each practice's effect. The influence of one practice can depend on another practice. For example, the influence of tillage on soil carbon will depend on residue management. The influence of nitrogen fertilization rates can depend on the application of nitrification inhibitors. A variety of examples is given in Section 3.2 and Section 3.3. Because of these synergies, estimating GHG emissions from crop and grazing land systems will depend on a complete description of the practices used in the operation, including past management to capture legacy effects on GHG emissions, as well as ancillary variables such as soil characteristics and weather or climate conditions.

3.1.1.4 Risk of Reversals

Any trend in GHG emissions associated with a change in crop and grazing land management can be reversed if the operator reverts to the original practice. Reversals will not negate the GHG mitigation for CH_4 or N_2O that occurred prior to the reversion. If emissions are reduced for CH_4 or N_2O, the emission reduction is permanent and cannot be changed by subsequent management decisions.

Reversals can occur with carbon sequestration in biomass and soils. CO_2 can be removed from the atmosphere through crop and forage production and sequestered in biomass or soils following the adoption of a conservation practice, such as no-till (CAST, 2004; USDA, 2011). If carbon is increasing in the biomass or soils, then the practice effectively reduces the amount of CO_2 in the atmosphere. However, net CO_2 can be returned to the atmosphere if there is a reversion in management to the previous practice that causes a decline in the biomass or soil carbon stocks. For example, enrollment of land in the CRP has increased the amount of carbon in soils (i.e., increase in soil carbon stock), and thus mitigates CO_2 emissions to the atmosphere associated with other emissions sources, such as fossil fuel combustion (USDA, 2011). However, tilling former CRP lands will lead to a decline in soil carbon stocks, thereby reversing the trend for CO_2 uptake from the atmosphere and leading to CO_2 emission to the atmosphere. In general, GHG emissions involving

carbon stocks in biomass, dead biomass, or soils can be considered reversible, depending on future management decisions. Consequently, reversals involving carbon stocks not only affect future emission trends, but also have consequences on past mitigation efforts by returning previously sequestered CO_2 to the atmosphere.

3.1.2   System Boundaries and Temporal Scale

System boundaries are defined by the coverage, extent, and resolution of the estimation methods. The coverage of methods in this guidance can be used to estimate GHG emission sources that occur on farm and ranch operations, including emissions associated with biomass carbon, litter carbon, and soils carbon stock changes; CH_4 and N_2O fluxes from soils; emissions from burning of biomass; and CO_2 fluxes associated with urea fertilization and addition of carbonate limes. GHG emissions also occur with production of management inputs, such as synthetic fertilizers and pesticides, and the processing of food, feed, fiber, and bioenergy feedstock products following harvest; but methods are not provided to estimate these emissions. Moreover, emissions from energy use, including those occurring on the entity's operation, are not addressed in the methods.

The methods provided for crop and grazing land systems have a resolution of an individual parcel of land or field and include the spatial extent of all fields in the entity's operation. Fields are areas used to produce a single crop or rotation of crops, or to raise livestock (i.e., pasture, rangeland). Fields are often, but not always, divided by fences. Emissions are estimated for each individual field that is used for cropland and grazing land on the operation, and then the emissions are added together to estimate the total emissions from the crop and grazing land systems in the entity's operation. The totals are then combined with emissions from forest and livestock to determine the overall emissions from the operation based on the methods provided in this guidance. Emissions are estimated on an annual basis for as many years as needed for GHG emissions reporting.

3.1.3   Summary of Selected Methods/Models Sources of Data

The Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2006) has developed a system of methodological tiers related to the complexity of different approaches for estimating GHG emissions. Tier 1 represents the simplest methods, using default equations and emission factors provided in the IPCC guidance. Tier 2 uses default methods, but emission factors that are specific to different regions. Tier 3 uses country-specific estimation methods, such as a process-based model. The methods provided in this report range from the simple Tier 1 approaches to the most complex Tier 3 approaches. Higher-tier methods are expected to reduce uncertainties in the emission estimates, if sufficient activity data and testing are available.

Tier 1 methods are used for estimating CO_2 emissions from urea fertilization, CH_4 emissions from flooded rice, indirect soil N_2O emissions, and direct soil N_2O emissions from drained organic soils. These methods are the most generalized globally, and lack ability to capture specific conditions at local sites, and consequently have more uncertainty for estimating emissions from an entity's operation. Soil N_2O emissions, CO_2 emissions or sinks from liming, biomass carbon stock changes, soil carbon stock changes for drained organic soils, and biomass burning non-CO_2 GHG emissions all have elements of Tier 2 methods, but may rely partly on emission factors provided by the IPCC (2006). These methods incorporate some information about conditions specific to U.S. agricultural systems and the influence on emission rates, but again lack specificity for local site conditions in many cases. Soil carbon stock changes for mineral soils are estimated using a Tier 3 method with a process-based simulation model (i.e., DAYCENT). CH_4 sinks from methanotrophic activity are also estimated with a Tier 3 method, due to the absence of IPCC guidance for estimating land use and management effects on CH_4 uptake in soils. The Tier 3 method associated with soil carbon stock changes in mineral soils has the greatest potential for estimating the influence of local conditions on

GHG emissions. The application has a general set of parameters that have been calibrated across a national set of experiments. However, the model does incorporate drivers associated with local conditions, including specific management practices, soil characteristics, and weather patterns, providing estimates of GHG emissions that are more specific to the entity's operation. Future research and refinements of the cropland and grazing land methods will likely incorporate more Tier 3 methods in the future, and thus provide a more accurate estimation of GHG emissions for entity reporting.

All methods include a range of data sources from varying levels of specificity on operation-specific data to national datasets. Operation-specific data will need to be collected by the entity, and generally are activity data related to the farm and livestock management practices (e.g., tillage practices, grazing practices, fertilizer usage). National datasets are recommended for ancillary data requirements that are used in methods, such as climate data and soil characteristics. However, the entity does have the option to use operation-specific data for climate (i.e., weather data) and soils.

3.1.4   Organization of Chapter/Roadmap

The croplands/grazing lands portion of this report is organized into four primary sections. Sections 3.2 and 3.3 provide a description of management impacts on GHG emissions in crop and grazing land systems. Section 3.2 is further subdivided into sections focused on upland agriculture, flooded management for crop production, and the influence of land-use change. Section 3.3 is subdivided into a general description of management practices and the influence of land-use change. The first two sections provide the scientific basis for how management practices influence GHG emissions. These two sections also discuss management options that require further study. Section 3.4 provides an overview of agroforestry systems. A general description of the various GHG emissions and sinks that result from management practices and potential management interactions is provided in this section.

Section 3.5 describes the methods. Each method includes a general description (including equations and factors if appropriate), activity data requirements, ancillary data requirements, limitations of the method, and uncertainties associated with the estimation. A single method is provided for each of the GHG emission sources (and sinks), based on the best available method for application in an operational system for entity-scale reporting. A single method was selected to ensure consistency in emission estimation by all reporting entities. More advanced approaches may be adopted in the future as the methods mature.

Section 3.6 provides a summary of research gaps. The gaps highlight key research areas that require further study for one of two reasons. The first reason is that a practice lacks sufficient evidence or a clear impact on GHG emissions based on existing research. This gap is most often related to a lack of mechanistic understanding of the processes influenced by the practice. These practices may be included in future revisions to the methods if further study leads to a consensus that the practice has an impact on emissions. The second reason for identifying the need for further study is that the practice is included in estimation methods, but there is need for further research to reduce uncertainty. This second gap may involve further mechanistic study, but could also require further methods of development or refinement.

Finally, Appendix 3-A provides a more comprehensive description of the soil N_2O modeling framework specifications. This appendix includes a discussion of the process-based models used in the methodology; the empirical scalars for the base emission rates; and the practice-based scaling factors. Appendix 3-B provides alternative methodologies in cases where an entity is managing crops not included in the DAYCENT model.

3.2 Cropland Management

How cropland is managed can have a significant effect on GHG emissions and removals. This section provides a summary of the current state of the science and describes how management practices drive GHG emissions or sinks in upland cropland systems.

3.2.1 Management Influencing GHG Emissions in Upland Systems

The cropland management practices presented below focus primarily on mitigation potential for soil N_2O, CH_4 emissions, and carbon sequestration. Each subsection describes the practice and the underlying GHG phenomenon that influence mitigation potential.

3.2.1.1 Nutrient Management (Manufactured and Organic)

Nutrient management refers to the addition and management of synthetic and organic fertilizers to cropland soils, primarily to augment the supply of nutrients to the crop. Nitrogen is generally the most important nutrient from an agronomic standpoint, because it is usually the primary nutrient limiting crop yields and often must be added more frequently and in greater amounts than other nutrients such as phosphorus and potassium (ERS, 2011; Robertson and Vitousek, 2009). Nitrogen is also the primary nutrient of concern with regard to GHG emissions, because once fertilizer nitrogen enters the soil it can be directly converted to N_2O by soil biological processes and, in some cases, chemical reactions (Firestone and Davidson, 1989; Kool et al., 2011; Venterea, 2007). While relatively little of the fertilizer nitrogen applied is converted to N_2O, these emissions are generally a large component of the total GHG budget of croplands (e.g., Mosier et al., 2005; Robertson et al., 2000) because N_2O has 310 times the global warming potential of CO_2 (IPCC, 2007). Other forms of nitrogen originating from fertilizers may also be lost to the environment, including NH_3, nitric oxide (NO), and nitrate (NO_3^-). Once transported to downwind or downstream ecosystems, these other nitrogen species can be converted to N_2O; such emissions are referred to as "indirect" N_2O emissions (Beaulieu et al., 2011; de Klein et al., 2006).

Nutrient management can also affect GHG emissions other than N_2O, most notably the sequestration of carbon upon manure addition and crop residue retention or addition. The addition of organic carbon amendments, such as manure or residues, can increase soil carbon within the boundaries of the land parcel receiving the amendment (Ogle et al., 2005). However, soil carbon losses may occur from the source field (Schlesinger, 2000) depending on the management (Izaurralde et al., 2001). Manufactured nitrogen additions can also lead to carbon sequestration (Ladha et al., 2011) where additions lead to increased residue return to soil.

Fertilizer rate, timing, placement, and formulation strongly affect N_2O fluxes. In general, any practice that increases crop nitrogen use efficiency (NUE) would be expected to reduce N_2O emissions, because applied nitrogen that is taken up by crops or cover crops is not available to the soil processes that generate N_2O, at least in the short term; this also may prevent nitrogen leaching. Thus, strategies to reduce N_2O emissions can also reduce the loss of NO_3^- and other forms of reactive nitrogen from cropping systems.

However, practices that improve NUE will not always reduce N_2O emissions. Different fertilizer formulations, for example, can result in different N_2O emissions irrespective of NUE effects (e.g., Gagnon and Ziadi, 2010; Gagnon et al., 2011). Likewise, banded fertilizer placement can increase NUE (e.g., Yadvinder-Singh et al., 1994) but also can increase rather than decrease N_2O emissions (e.g., Engel et al., 2010), and tillage management can also increase NUE without reducing N_2O emissions (Grandy et al., 2006). Thus, NUE is generally important but not by itself sufficient to predict or manage N_2O emissions. Fertilizer rate, timing, placement, and formulation can affect NUE and N_2O emissions independently.

Fertilizer Rate: More than any other factor, the amount of nitrogen fertilizer applied to soil affects the amount of N_2O emitted; in many cases other nitrogen-use strategies (timing, placement, and formulation) provide their benefit by effectively reducing fertilizer nitrogen available in the soil. In this sense, fertilizer rate integrates the effects of multiple practices and is the basis for the IPCC Tier 1 N_2O accounting method (de Klein et al., 2006), whereby N_2O emissions are assumed to be a simple fraction of nitrogen inputs.

Irrespective of other practices, however, fertilizer rate itself can be refined to reduce N_2O emissions so long as rates are not reduced to the point that yields decline. Otherwise market leakage—the need to make up yields elsewhere with more intensive fertilizer use and concomitant N_2O loss—may limit the benefit of reducing local fertilizer rates. The question then becomes whether nitrogen fertilizer rates can be reduced without reducing yields in a particular field. At least for corn, recent changes in recommended fertilizer rates for many Midwest States suggest that there is latitude for reducing fertilizer nitrogen rates for some farmers. Since the 1970s, most fertilizer nitrogen recommendations have been based on yield goals, which use expected maximum yield multiplied by nitrogen yield factors to calculate fertilizer recommendations (Stanford, 1973). Preceding legume crops, manure inputs, and soil nitrogen tests are then used to further refine or reduce recommended nitrogen application rates (Andraski and Bundy, 2002).

An alternative to the yield-goal approach is the Maximum Return to Nitrogen approach (Sawyer et al., 2006), whereby the rate of nitrogen fertilizer applied is based on the maximum fertilizer rate that generates sufficient additional yield to justify the fertilizer cost. The rates are determined from crop nitrogen response curves. Typically (but not always) this rate is significantly less than that recommended by the yield goal approach. Maximum Return to Nitrogen calculators for corn have been adopted in at least seven States in the Midwest. This calculator and similar decision support tools have the potential for reducing the amount of fertilizer nitrogen applied to crops and more precisely match crop nitrogen requirements, without affecting the net returns (Archer et al., 2008), and with concomitant decreases in N_2O emissions (Millar et al., 2010).

Hundreds of fertilizer addition experiments worldwide have shown that typically 0.5 to 3 percent of nitrogen added to soil is emitted as N_2O (Bouwman et al., 2002; Linquist et al., 2011; Stehfest and Bouwman, 2006). Site-to-site variation is well recognized and is to be expected based on soils, climate, and fertilizer practices—including rate. Recent evidence suggests that emission rates may be even higher at nitrogen input levels that exceed crop demand (Hoben et al., 2011; Ma et al., 2010; McSwiney and Robertson, 2005; Van Groenigen et al., 2010).

Fertilizer Timing: A major challenge in managing nitrogen fertilizer for crop production is synchronizing nitrogen availability in the soil with the crop's demand for nitrogen. In general, crop demand for nitrogen is minimal early in the growing season and increases several weeks after planting.

In many cases, it may be most convenient and/or cost-effective for the producer to apply nitrogen fertilizer prior to planting or soon after plant emergence. In many parts of the U.S. Corn Belt, however, application of nitrogen fertilizer commonly occurs in the fall prior to the growing season (Bierman et al., 2011; Ribaudo et al., 2011). In the absence of an active and well-developed root system to utilize the fertilizer nitrogen, these practices increase the potential for soil microbial and chemical processes to transform the applied nitrogen into N_2O and other mobile forms such as NO_3, which can contribute to indirect N_2O emissions.

Improving the synchrony between soil nitrogen availability and crop nitrogen demand can be achieved by switching from fall to spring nitrogen application; applying nitrogen several weeks after planting with "sidedress" fertilizer applications that are timed to coincide with plant growth stages; and using multiple "split" applications distributed in time over the growing season. Each of

these strategies has the potential to reduce N_2O emissions, but this is not always the case. Switching from fall to spring nitrogen fertilizer, for example, has been shown to reduce N_2O emissions in some cases (Burton et al., 2008a; Hao et al., 2001) but not always (Burton et al., 2008a). Similarly, switching from pre-plant to post-plant applications has been shown to reduce N_2O emissions in some studies (Matson et al., 1998), but only part of the time or not at all in other studies (Burton et al., 2008b; Phillips et al., 2009; Zebarth et al., 2008b). Some studies have found reduced nitrate leaching, which implies reduced indirect N_2O emissions, with fertilizer application later in the season (e.g., Errebhi et al., 1998).

Fertilizer Placement: The manner in which nitrogen fertilizer is applied to soil can affect its availability for crop uptake and therefore its susceptibility to soil transformation and N_2O production. Three aspects of fertilizer placement are significant to N_2O emissions: (1) broadcast application versus banding within the crop row; (2) the soil depth to which nitrogen is applied; and (3) adding fertilizer uniformly across a field versus applying at a spatially variable rate.

There is some evidence that applying nitrogen fertilizer in narrow bands can improve crop NUE (Malhi and Nyborg, 1985). However, banding also creates zones of highly concentrated soil nitrogen, which can increase N_2O production compared with broadcast applications (Engel et al., 2010). Other studies have found no differences in N_2O emissions in broadcast versus banded applications (Burton et al., 2008a; Sehy et al., 2003). Direct comparisons of application depth effects on N_2O emissions have also shown inconsistent results (e.g., Breitenbeck and Bremner, 1986b; Drury et al., 2006; Fujinuma et al., 2011; Hosen et al., 2002; Liu et al., 2006). However, variable rate application uses different nitrogen rates for different areas of field, based on expected variations in crop nitrogen demand. This is a new technique that appears promising based on its ability to substantially improve fertilizer use efficiency at the field scale (Mamo et al., 2003; Scharf et al., 2005), and at least one early study has shown reduced N_2O emissions when nitrogen rate was varied to match crop yield potential (Sehy et al., 2003).

Fertilizer Formulation and Additives: The most commonly used forms of synthetic nitrogen fertilizer in the United States include anhydrous ammonia (35 percent of total use), urea (24 percent), and liquid solutions, including urea ammonium nitrate (29 percent) (ERS, 2011). Available evidence suggests that N_2O emissions following applications of anhydrous ammonia are greater than emissions following broadcast urea, although in some studies this may be partly due to fertilizer placement. In five studies, anhydrous ammonia resulted in 40 to 200 percent greater N_2O emissions compared with broadcast urea (Breitenbeck and Bremner, 1986a; Fujinuma et al., 2011; Thornton et al., 1996; Venterea et al., 2005). One study (Burton et al., 2008a) found no difference in N_2O emissions between anhydrous ammonia and broadcast urea when both were applied at a lower rate (80 kg N ha^{-1} year^{-1}) compared with the other studies ($\geq$ 120 kg N ha^{-1}). Consequently, there may be a threshold in the application rate before there is a significant effect on emissions.

The chemical form of nitrogen fertilizer influences losses of nitrogen from three major pathways: surface volatilization, soil microbial processes, and NO_3^- leaching. All fertilizers are susceptible to denitrification once nitrified to (or applied as) NO_3^-. Ammonium-based fertilizers, including anhydrous ammonia, urea, and organic sources such as manure, are also susceptible to N_2O loss during nitrification. Urea, anhydrous ammonia, and manure are additionally susceptible to surface volatilization as NH_3 under some conditions. Volatilized NH_3 and leached NO_3^- contribute to indirect N_2O loss.

Chemical additives have been developed to release fertilizer nitrogen into the soil more gradually and to delay the nitrification of nitrogen from ammonium (NH_4^+) to NO_3^- in order to improve the synchrony between crop nitrogen demand and soil nitrogen availability. Polymer-coated urea slowly releases nitrogen with increasing soil temperature and water, and is intended to make

nitrogen supply more synchronous with plant nitrogen demand and reduce nitrogen losses. Effects on N_2O production, however, appear mixed, with some studies showing reduced N_2O for polymer-coated urea (e.g., Hyatt et al., 2010) and others showing no impact or even higher emissions (Venterea et al., 2011a). A recent meta-analysis of 13 studies of mostly volcanic and wetland-derived soils found that polymer-coated urea reduced N_2O emissions by 35 percent on average compared with conventional fertilizers, but results are difficult to generalize because most of the soils included in the analysis were not typical for U.S. cropping systems (Akiyama et al., 2010).

Fertilizers formulated with nitrification inhibitors can potentially reduce emissions from nitrification and denitrification, as well as NO_3^- leaching. Some U.S. field studies show substantial reductions in N_2O emissions when fertilizers with nitrification inhibitors are added compared with conventional fertilizers (e.g., Halvorson et al., 2010a), while others show little or no impact (e.g., Parkin and Hatfield, 2010a). A meta-analysis of some 28 studies worldwide reported an average reduction of 38 percent (Akiyama et al., 2010), but again results are difficult to generalize due to the small sample size and soils that are not typical of U.S. cropping systems.

One reason the impacts of fertilizers designed to reduce emissions are inconsistent is that the form of nitrogen applied interacts with other factors to control nitrogen losses. Among these factors is weather, which directly affects the processes that lead to gaseous nitrogen losses and NO_3^- leaching, and indirectly affects these processes by controlling plant nitrogen uptake. Soil properties such as texture and hydraulic status are also important. In general, nitrification is important in well-aerated soils, while denitrification is more important in poorly drained soils. The nitrogen source also interacts with other management practices. For example, polymer-coated urea substantially reduced N_2O emissions under no-till but not full till cultivation for irrigated corn in Colorado (Halvorson et al., 2010a).

Organic Fertilizer Effects on N_2O Emissions: Land application of animal manure has been related to N_2O emissions. Mosier et al. (1998) and Petersen (1999) measured increases in N_2O emissions with manure application. Kaiser and Ruser (2000) measured annual emissions of the added nitrogen in slurry ranging from 0.74 to 2.86 percent, and De Klein et al. (2001) found that annual N_2O-N losses ranged from zero to five percent of the organic nitrogen applied to soils. Others (e.g., Barton and Schipper, 2001) found N_2O emissions following the addition of manure slurries exceeded emissions from an equivalent amount of manufactured N, likely due to the slurry's creating enhanced conditions for denitrification. However, GHG emissions also occur if manure is managed in pits, lagoons, or solid storage.

Injection of manure is a common practice to avoid surface runoff and reduce objectionable odors from manure application. Both Flessa and Besse (2000)and Wulf et al. (2002) suggested that injection of swine manure would create more favorable conditions for N_2O and CH_4 formation because of the reduced aeration within the soil. However, Dendooven et al. (1998) did not find differences in either N_2O or CH_4 emissions from injected or surface-applied swine slurry onto a loamy soil. These findings suggest that the rate, timing, placement, and formulation of manure is important to N_2O production, similar to manufactured nitrogen fertilizer, but there is a need for additional research.

CO_2 Emissions Generated from Urea Fertilizer Applications: Unlike other nitrogen fertilizers, urea results in the direct production of CO_2 in addition to whatever N_2O might be subsequently produced by microbes (de Klein et al., 2006). Since urea is 20 percent C, every metric ton of urea applied to soil results in the direct emission of 20 kg CO_2-C; alternatively, every kilogram of nitrogen applied as urea results in the direct emissions of 0.43 kg CO_2-C. Urea is manufactured by reacting NH_3 and CO_2 to form ammonium carbamate, which is then dehydrated to form urea prills. In the United

States the CO_2 in urea is captured from the fossil fuel used to manufacture NH_3, so the soil CO_2 produced represents a fossil fuel emission.

Management System Interactions: Nitrogen management practices can interact with other cropland management components in regulating GHG emissions. As emphasized above, any factor that affects crop NUE has the potential to affect N_2O emissions. Therefore, optimizing other practices—including tillage and the management of soil pH, pests, irrigation, drainage, and other factors—will tend to increase nitrogen fertilizer uptake by the crop and therefore reduce N_2O emissions. For this reason, nutrient management effects on GHG emissions should be considered in the context of the entire set of cropland management practices. For example, there is evidence that fertilizer placement can interact with tillage management in controlling N_2O emissions (Venterea et al., 2005), and that inadequate management of other nutrients (e.g., phosphorus and potassium) can reduce NUE (Snyder et al., 2009). Efforts to minimize or remediate water quality impacts of nitrate transport from farm to aquatic systems may also reduce indirect N_2O emissions. For example, the use of subsurface bioreactors to remove nitrate from drainage water has beneficial impacts on indirect N_2O. However, to date these bioreactors have not been implemented at large (field) scales and there are also questions about release of N_2O and CH_4 during the treatment process that need to be answered before their net effect on GHGs can be assessed (Elgood et al., 2010). Also, environmental and climate factors, which are generally not under management control, may affect N_2O emissions; for example, nitrogen fertilizer applied just before large rainfall events can stimulate increased emissions (Li et al., 1992).

3.2.1.2   Tillage Practices

Different tillage practices are generally classified into one of three categories: full tillage, reduced tillage, or no tillage. Tillage intensity is based on implements, number of passes, and the percentage of surface and depth of tillage disturbance. Tools are available to determine tillage intensity (e.g., the STIR Model; see USDA NRCS, 2008). No-tillage practices are characterized by the use of seed drills and fertilizer or pesticide applicators with no additional tillage events or implements. Surface residues are not incorporated into the soil when following no-tillage practices, and there is limited disturbance to the soil profile; consequently no-tillage management increases soil cover and improves aggregate stability (Six et al., 2000). In contrast, examples of full tillage (often referred to as conventional tillage) include one or more passes with the following tillage implements: moldboard plow, disk plow, disk chisel, twisted point chisel plow, heavy duty offset disk, subsoil chisel plow, and bedder or disk ripper. Systems are also classified as full tillage if there are two or more passes with one of the following implements: chisel plow, single disk, tandem disk, offset disk-light duty, one-way disk, heavy-duty cultivator, ridge till, or rototiller. Systems with other tillage practices, such as a single pass with a ridge till implement, mulch till, or chisel plow, lead to intermediate disturbance of the soil and are classified as reduced tillage.

Changes in tillage practices can influence vertical distribution of carbon in the soil profile and total soil carbon stocks (Paustian et al., 1997). Historically, full tillage has resulted in the reduction of soil carbon stocks (Lal et al., 2004). A synthesis of previous analyses estimated that long-term full tillage can decrease soil carbon stocks by 30 percent (Ogle et al., 2005; West et al., 2004). Changing from full tillage to no tillage can reverse historic losses of soil C. No-tillage practices can lead to accumulation of soil carbon in the upper soil profile (0 to 30 cm), with little to no change in the lower soil profile (30 to 60 cm) (Syswerda et al., 2011). The opposite, a decrease in the upper soil horizon with an increase in the lower soil horizon, can sometimes occur with a change from no tillage to full tillage (Baker et al., 2007). However, changes in the lower soil profile tend to be more variable, thereby requiring a larger sample size to detect significant differences (Kravchenko and Robertson, 2011). A reduction in carbon input associated with the influence of no-till management on crop production may also lead to losses of soil carbon, particularly in cooler and wetter climates

(Ogle et al., 2012). However, while differences in the response of soil carbon stocks to tillage occur among field experiments, comprehensive analyses of available field data indicate that, on average, soil carbon stocks increase with a change from full tillage to reduced tillage or no tillage, especially with long-term adoption of no tillage (Ogle et al., 2005; Six et al., 2004; West and Marland, 2002).

Decreased tillage intensity increases soil carbon because of reduced disturbance to soil aggregates, reduced exposure of soil organic matter to weathering processes, and decreased decomposition rates (Paustian et al., 2000). The extent to which soil carbon accumulation occurs after a reduction in tillage intensity is determined by the history of land management, soil attributes, regional climate, and current carbon stocks (West and Six, 2007). In general, greater soil carbon accumulation will be observed in C-poor soils (i.e., due to long-term cultivation) with a clayey texture under high biomass cropping systems in temperate humid and warm climates (Franzluebbers and Steiner, 2002; Plante et al., 2006; Six et al., 2004). In some cases, intermittent tillage, during long-term reduced or no tillage, is needed to reduce soil compaction, for weed control, or to reduce pests or pathogens. While intermittent tillage can cause a decrease in soil stocks, up to 80 percent of soil gains from no-tillage practices can be maintained when implementing no tillage with intermittent tillage (Conant et al., 2007; Venterea et al., 2006).

The effect of tillage management changes on soil N_2O emissions is variable and not fully understood. Increases (Rochette, 2008), decreases (Mosier et al., 2006), and no changes (Grandy et al., 2006; Lemke et al., 1998) in soil N_2O emissions have been observed. However, those differences are not totally random and past meta-analyses have concluded that climate regime, duration of practice, and nitrogen fertilizer placement have influenced tillage effects on N_2O emissions (Six et al., 2004; van Kessel et al., 2012). Other variables such as soil texture may also be important.

Regional climate has also been identified as a major driver for the change in N_2O emissions with adoption of no-tillage practices, with emissions increasing in humid climates and decreasing in dry climates (Six et al., 2004). However, time since adoption of no tillage might also play a role with higher emissions initially after adoption of no tillage in both humid and dry climates, but over time emissions from no-tillage systems may decline in humid climates relative to previous emissions from full tillage systems. Nevertheless, various field studies have shown mixed results, both supporting and contradicting the finding. Studies in drier climates of the Great Plains have shown a decrease in emissions even when no-tillage practices had been adopted for less than 10 years (Kessavalou et al., 1998; Mosier et al., 2006). Long-term no tillage in moist climates of Minnesota and Canada led to both higher and lower emissions of N_2O (Drury et al., 2006; Venterea et al., 2005).

Another important factor influencing N_2O emissions under no tillage, and one that farmers can actively manage, is fertilizer placement (van Kessel et al., 2012). Venterea et al. (2005) found that when nitrogen fertilizer was placed on the surface, N_2O emissions were greater under no tillage than full tillage, but the reverse was found when nitrogen fertilizer was placed below 10 centimeters. Fertilizer placement in general has been found to have differing results on N_2O emissions, as discussed in Section 3.2.1.1. However, the findings of Venterea et al. (2005) as well as other studies (e.g., Groffman, 1985; Venterea and Stanenas, 2008) indicate that deeper nitrogen placement tends to decrease N_2O emissions when accompanying no-till or reduced-tillage practices, at least relative to full tillage cropping systems at the same location. The conflicting results associated with N_2O emissions from fertilizer applications may be partly explained by the tillage practice.

In addition, Lemke et al. (1998) determined that soil clay content explained 92 percent of the variation in N_2O emissions between full tillage and no tillage across multiple sites in Alberta. Similarly, Burford et al. (1981) found that emissions from no-tillage practices were greater than

from full tillage on soils with higher clay contents at a study site in the United Kingdom. It is argued that soils with higher clay contents have higher moisture content and therefore have a greater potential for increased N_2O emissions under no tillage. Indeed, Rochette (2008) attributed higher rates of N_2O flux from minimum versus standard tillage to greater soil compaction, poor soil drainage, reduced gas diffusivity, and air-filled porosity in high clay soils.

3.2.1.3   Crop Rotations, Cover Crops, and Cropping Intensity

Crop rotation refers to the sequence of crops planted in a field, within or across years. Crop rotations vary by location and growing region, and may be practiced for a variety of reasons such as improved economic returns, pest management, disease control, nutrient management and water availability. A simple rotation may be a sequence of corn and soybeans that is repeated over time, while more complex rotations might include perennial crops such as alfalfa with corn and sunflower rotation over five years, with three years of alfalfa and one year each of corn and sunflower. The actual rotations can also vary from a strict order to the sequence, particularly in response to market demand, i.e., opportunistic rotations. Rotations with high biomass-yielding crops or perennial hay crops or grass cover can increase soil carbon stocks (Ogle et al., 2005).

Cropping intensity can vary across years, due to variations in fallow frequency and use of multiple growing seasons with more than one crop planted and harvested in a single year. For example, in semi-arid environments, crop rotations often include a year-long fallow period in order to increase the amount of water stored in the soil profile for the subsequent crop. This limits the amount of organic matter input to the soil, and with the severe water limitation, these cropping systems produce small amounts of biomass, leading to a reduction in soil carbon stocks (Doran et al., 1998). Consequently, intensifying crop production by reducing fallow frequency, which will generally involve adoption of no-tillage practices, will increase carbon input across the whole rotation and possibly the amount of soil organic carbon (Sherrod et al., 2003; 2005).

Winter cover crops can also be used to provide plant cover outside of the normal growing season. Prior to planting the following summer crop, the cover crop is either left to decompose as a green cover or harvested for forage. In general, the inclusion of a cover crop in a crop rotation will lead to an increase in soil carbon due to the increased carbon input derived from the cover crop (Kong et al., 2005), especially cover crop roots (Kong and Six, 2010). Cover crops can also be used effectively for nitrogen management. In the fall and spring they can capture soil nitrogen that would otherwise be transformed directly to N_2O by soil microbes or leach to groundwater and contribute to indirect N_2O emissions (i.e., offsite emissions due to nitrogen losses from the site). Additionally, when killed prior to planting the main crop, their decomposition can provide nitrogen that will displace some portion of crop fertilization requirements (whether manufactured or organic). Therefore, cover crops can reduce indirect N_2O emissions and possibly offset fertilization rates. However, there are no studies demonstrating that adding nitrogen to soils in cover crops rather than through fertilization will reduce direct N_2O emissions. In the future, cover crop biomass may also be harvested for cellulosic ethanol feedstock, leaving roots to enhance soil carbon stocks similar to perennial plants grown in rotation (Ogle et al., 2005).

The effects of crop rotation and intensity on soil organic carbon can also interact with other management practices, such as residue management, tillage, and irrigation (Eghball et al., 1994). Consequently, management interactions among practices including tillage and irrigation will be important in determining the influence of crop rotations on GHG emissions. Additionally, crop selection as a component of crop rotation can have a major effect on N_2O emissions (Cavigelli and Parkin, 2012) insofar as crops can vary in their nitrogen use efficiencies and nitrogen fertilizer needs. This is particularly the case when long-lived perennial crops are substituted for annual crops in forage or cellulosic biofuel cropping systems (Robertson et al., 2011).

3.2.1.4   Irrigation

Types of irrigation systems include surface or flood irrigation, (micro-) sprinkler irrigation, subsurface drip irrigation, and subirrigation. In general, irrigation increases soil water content, evapotranspiration rates, and relative humidity; decreases soil and air temperatures; and can lead to increased regional precipitation (Lobell and Bonfils, 2008; Pielke et al., 2007). These changes affect important processes such as plant growth rates and soil microbial activity that control net GHG fluxes.

As soil water content approaches saturation, oxygen (O_2) diffusion is inhibited, resulting in anaerobic conditions that can enhance CH_4 emissions (Chan and Parkin, 2001; Delgado et al., 1996), or at least reduce the CH_4 sink strength of otherwise aerobic soils (Livesley et al., 2010). Saturated conditions also enhance denitrification rates and potentially N_2O emissions (Delgado et al., 1996; Jambert et al., 1997; Livesley et al., 2010), but note that peak N_2O emissions from denitrification often occur at water contents lower than saturation because when O_2 is extremely limiting, N_2O is likely to be further reduced to N_2 before diffusing from the soil surface to the atmosphere (Davidson, 1991; Dunfield et al., 1995). Furthermore, nitrification rates peak at approximately 50 percent of saturation, and water contents close to field capacity (60 to 70 percent of saturation) are expected to support maximum total N_2O emission rates (Davidson, 1991). In addition, irrigation can increase indirect N_2O emissions by enhancing NO_3^- leaching and runoff if more water is added than is evaporated (Gehl et al., 2005; Spalding et al., 2001).

Wetting of dry soils typically increases CO_2 emissions (Fierer and Schimel, 2002). However, irrigation also increases plant growth rates and, therefore, soil organic carbon levels typically increase after upland cropping is converted to irrigated cropping, although loss of soil carbon from erosion can also increase under irrigation (Follett, 2001; Lal et al., 1998). Furthermore, irrigation can affect inorganic carbon levels, but current available data show contrasting results (Blanco-Canqui et al., 2010; Denef et al., 2008; Entry et al., 2004).

Flood and Surface Irrigation: Flood irrigation involves flooding the entire field with water. Under continuously flooded conditions, soils are highly anoxic, thus facilitating high methanogenesis and denitrification rates (Mosier et al., 2004). However, high denitrification rates do not necessarily imply high N_2O emissions because the extremely anoxic conditions facilitate further reduction of N_2O to N_2 before it is emitted from the soil (Mahmood et al., 2008). This is supported by observations showing higher N_2O emissions from intermittent compared to continuously flooded rice systems (Katayanagi et al., 2012; Xu et al., 2012), although it remains difficult to predict the relative portion of denitrified nitrogen that is emitted as N_2O relative to N_2.

Surface irrigation also involves supplying large amounts of water to the surface of soils, but in this case the water is added through furrows adjacent to crop beds. These systems are often not very efficient, because water losses from evaporation and seepage can be large. The impact of furrow irrigation on GHG emissions depends on how often and the extent to which furrows are filled with water. Wetting and drying cycles are likely to emit large pulses of NO and N_2O (Davidson, 1992), as well as CO_2 (Fierer and Schimel, 2002). Spatial variability can also be high, such as the higher N_2O emissions from furrows compared with beds that have been observed for irrigated cotton cropping (Grace et al., 2010). In addition, micro to landscape scale heterogeneity in environmental conditions, due to topography and other factors, contribute to multiscale variability in N_2O emissions (Hénault et al., 2012; Yates et al., 2006). This spatial and temporal heterogeneity in environmental conditions and flux rates makes it very difficult to quantify GHG fluxes from these types of systems with high levels of accuracy and precision.

Sprinkler Systems: Sprinkler systems deliver water to vegetation and the soil from above the surface using overhead sprinklers or guns. This is usually more efficient than surface irrigation, but

evaporative losses from water intercepted by vegetation, litter, and the soil surface can still be substantial. During and shortly after irrigation events, soil may become saturated and emit pulses of N_2O, but because the soil is not continuously saturated, N_2O emissions are expected to be lower compared with surface irrigation (Nelson and Terry, 1996). Both N_2O emissions and soil carbon levels are expected to increase with sprinkler irrigation compared with upland cropping.

Surface and Subsurface Drip Irrigation: Surface drip irrigation supplies water from drip lines placed adjacent to crop rows. Evaporative losses should be less compared with above-surface sprinkler systems, because less water is intercepted by growing vegetation. However, evaporative losses can still occur to the extent that surface litter and soil layers absorb water from the drip sprinkler. The impacts of surface drip irrigation on GHG fluxes are expected to be similar to those of sprinkler systems, although there is early evidence that both surface and subsurface drip irrigation leads to less emissions of CH_4 and N_2O (Kallenbach et al., 2010; Kennedy et al., 2013).

Subsurface drip irrigation targets water delivery to the root zone using buried pipes and tubing. These systems can be very efficient, because water is concentrated in the root zone at a slow, steady rate, hence minimizing or eliminating evaporation losses and avoiding saturation of the whole soil profile. Consequently, these systems are not expected to be large CH_4 sources (Del Grosso et al., 2000a). Soil water content has less temporal variation with subsurface drip irrigation compared with sprinkler and surface systems, so pulses of N_2O and CO_2 emissions are also expected to be of smaller magnitude (Kallenbach et al., 2010). Similarly, subsurface drip irrigation/fertigation of high values crops, such as tomatoes, has been shown to reduce N_2O emissions compared with furrow irrigation (Kennedy et al., 2013).

Subirrigation: Subirrigation is used in areas with relatively high water tables and involves artificially raising the water table to allow the soil to be moistened from below the root zone. Because water is supplied to roots from below, evaporation losses are not enhanced as they would be with surface irrigation systems. This system can decrease NO_3^- leaching (Elmi et al., 2003) but may increase N_2O losses from denitrification (Munoz et al., 2005).

Management Interactions: Irrigation systems interact with other crop management strategies such as changes in crop rotation, cropping intensity, tillage, and fertilizer amount to control net GHG fluxes. Irrigation tends to amplify the effects of these factors on N_2O and CH_4 emissions at the same time as the practices increase crop yields and soil carbon stocks. However, the response of soil carbon to irrigation is complex and driven by interacting factors. When water and nutrient stress are reduced through irrigation and fertilization, the portion of total plant production allocated below ground can decrease, but absolute below ground production and soil organic carbon can increase (Bhat et al., 2007). However not all experiments show increased soil carbon with irrigation (Denef et al., 2008). Consequently, the irrigation benefits of increased yields and potential carbon storage may be counter-balanced with the increased N_2O and CH_4 fluxes.

However, there are also options for limiting emissions, particularly with fertilization. Fertigation adds nutrients to the irrigation system to deliver water along with soluble nutrients to the root zone. These systems have the potential to be very efficient from both nutrient and water use perspectives (Spalding et al., 2001), because the slow and timed supply of nutrients and water is more synchronous with plant demand and they are concentrated in the root zone. Consequently, N_2O and other nitrogen losses are minimized while plant growth, carbon inputs, and carbon sequestration can be maximized. Similarly, CH_4 emissions are minimized because soil saturation is avoided.

3.2.1.5   Erosion Control

Soil erosion processes include soil detachment, transport, and deposition. Soil erosion can potentially reduce soil carbon stocks and increase net carbon flux to the atmosphere through decreased plant productivity and subsequent decreased organic matter input to soil and increased decomposition of the eroded soil fraction (Lal, 2003). However, soil erosion can also potentially increase net soil carbon stocks and decrease net carbon flux to the atmosphere through dynamic replacement of soil carbon on eroded landscapes and decreased decomposition rates in zones of soil deposition (Harden et al., 1999; Stallard, 1998).

Lal (2003) estimated that 20 percent of carbon in eroded soil is emitted to the atmosphere, due to oxidation of soil organic carbon following the disruption of soil aggregates caused by detachment and transport. However, in an analysis of 1,400 soil profiles, Van Oost et al. (2007) found negligible carbon loss as a direct result of soil detachment and transport. At sites where the transported soil was deposited, there was a slight (~one percent) decrease in soil carbon decomposition rates, resulting in slightly higher soil carbon accumulation. More importantly, it was found that on average, 25 percent of eroded carbon was replaced on the eroded sites over a 50-year period (Harden et al., 2008). The combination of these findings supports an approximate 26 percent sink capacity of eroded soil (Van Oost et al., 2007).

The accumulation of soil carbon on eroded locations within landscapes is referred to as dynamic replacement (Harden et al., 1999). Dynamic replacement occurs as a result of soil carbon building toward a steady state of soil carbon content, constrained by soil type and climate (West and Six, 2007). Steady state occurs when soil carbon accumulation equals soil carbon losses. Both Van Oost et al. (2007) and Lal and Pimentel (2008) note that the dynamic replacement rate may be low in areas with lower cropland production inputs. For example, dynamic replacement may be low in crop systems with low residue production, such as cotton and tobacco in the United States, which have lower carbon accumulation rates than high residue inputs crops (Ogle et al., 2005).

Note that while water erosion can generate a small carbon sink, the benefit of a carbon sink is offset by other negative impacts from soil erosion. For example, soil erosion can result in water pollution due to sediment loading, air pollution from airborne particulate matter (PM10), and decreased soil fertility resulting in subsequent yield declines.

3.2.1.6   Management of Drained Wetlands

Drainage of wetlands effectively creates an upland cropping system by lowering water tables with tiles or ditches to produce annual crops. The most obvious effect of wetland drainage is increased oxidation and tillage of soils. For example, conversion of native wetlands and grasslands into cropland has been shown to deplete native soil carbon stocks by 20 to more than 50 percent (Blank and Fosberg, 1989; Euliss et al., 2006; Mann, 1986). In turn, CO_2 emissions increase with higher decomposition rates, particularly in organic soils, i.e., Histosols (Allen, 2012; Armentano and Menges, 1986). Loss of the organic layer has caused tremendous subsidence in U.S. croplands (Stephens et al., 1984) such as the Florida Everglades (Shih et al. 1998) and the California Delta region (Broadbent, 1960; Weir, 1950), where rates vary from 0.46 to 2.3 cm year^{-1} (Deverel and Rojstaczer, 1996; Deverel et al., 1998; Rojstaczer and Deverel, 1995). Similar subsidence rates have also occurred in other regions such as the Florida Everglades.

Manipulation of water levels can have multiple effects on nutrient cycling in wetlands. Drainage also may result in more optimal soil moisture conditions (e.g., 40 to 60% water-filled pore space) that enhance formation of N_2O as a byproduct of nitrification and denitrification reactions (Davidson et al., 2000). Drainage increases nitrogen mineralization rates with conversion from anaerobic to aerobic conditions and enhances N_2O emissions (Duxbury et al., 1982; Kasimir-

Klemedtsson et al., 1997). In contrast, drainage decreases CH_4 emissions by reducing the frequency and duration of soil saturation required for CH_4 production as well as enhancing frequency of methanotrophic activity (Dorr et al., 1993; Gleason et al., 2009; Phillips and Beeri, 2008). However, in situations where wetlands are in a crop production, but not directly drained, CH_4 production can actually be enhanced due to increased runoff from adjacent croplands or consolidation drainage, which increases water depth and hydroperiods (Gleason et al., 2009).

Managing the water table by raising the depth of drainage to the extent possible has been an effective measure to reduce loss of CO_2 and other GHGs from drained organic soils (Jongedyk et al., 1950; Shih et al., 1998). Recent research suggests that even periodic flooding of organic soils that are drained for crop production may be effective in reducing CO_2 emissions (Morris et al., 2004). There is limited information on the effect of drainage in mineral soils with a high water table (i.e., hydric soils), but the influence on GHG emissions is likely less significant than in drained organic soils. It is important to note that wetlands are afforded some protection by laws (e.g., Clean Water Act) and conservation programs that recognize the importance of wetlands, such as for wildlife habitat, and provide agricultural producers incentives to avoid draining wetlands (e.g., the "Swampbuster" provision of the Food Security Act).

3.2.1.7 Lime Amendments

Agricultural lime consists primarily of crushed limestone ($CaCO_3$) and dolomite ($CaMg(CO_3)_2$) in varying proportions. Agricultural lime, hereinafter referred to as lime, is applied to soils to decrease soil acidity. Lime is commonly applied to agricultural lands where nitrogenous fertilizers are continuously used and where precipitation exceeds evapotranspiration.

The application of lime to soils can create a sink or source of CO_2 to the atmosphere (Hamilton et al., 2007), depending on the strength of the weathering agent. Weathering of lime by carbonic acid (H_2CO_3), formed when CO_2 is dissolved in water, results in the uptake of one mole of CO_2 for every mole of lime-derived carbon dissolved (Eq. 1). Carbonic acid weathering produces bicarbonate (HCO_3-) that contributes to alkalinity in groundwater, streams, and rivers (Oh and Raymond, 2006; Raymond et al., 2008). Alternatively, when lime reacts with the stronger nitric acid (HNO_3), which is produced when nitrifying bacteria convert NH_4^+ based fertilizer and other sources of NH_4^+ to nitrate (NO_3^-), carbon in lime is dissolved and released directly to the atmosphere (Eq. 2).

$CaCO_3 + H_2O + CO_2 = Ca^{2+} + 2HCO_3^-$ Eq. 1

$CaCO_3 + 2HNO_3 = Ca^{2+} + 2NO_3^- + H_2O + CO_2$ Eq. 2
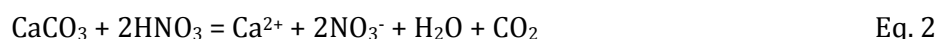
Field measurements and modeling analyses indicate that more lime is dissolved by carbonic acid than by nitric acid. For example, West and McBride (2005) estimated that 62 percent of lime was dissolved by carbonic acid weathering, Hamilton et al. (2007) estimated 75 to 88 percent, and Oh and Raymond (2006) estimated 66 percent. Biasi et al. (2008) used chamber flux measurements to estimate 15 percent loss of lime-derived carbon by dissolution with strong acids and inferred that 85 percent is dissolved by carbonic acid.

West and McBride (2005) also estimated the precipitation of HCO_3^- back to $CaCO_3$ once HCO_3^- reaches the ocean, thereby releasing CO_2 to the atmosphere. However, the long time period (many decades to centuries) over which precipitation would occur in the ocean (Hamilton et al., 2007) effectively results in carbon sequestration for annual accounting purposes.

Current consensus of leached drainage samples, stream gauge data, and mass balance modeling indicates that about 66 percent of carbon in applied lime is essentially transferred from one long-lived pool ($CaCO_3$ in geologic formations) to another (HCO_3^- in oceans), and is therefore not counted as new sequestration. However, the atmospheric CO_2 newly captured by this process does

represent sequestration when corrected for the 33 percent released to the atmosphere as CO_2; this results in a net 33 percent sink strength per carbon in lime. This estimate is similar to that of Oh and Raymond (2006) and West and McBride (2005), and is within the range of Hamilton et al. (2007). While lime can increase soil carbon via effects on soil microbial activity (Fornara et al., 2011), in most soils liming has no direct carbon effect (Page et al., 2009).

3.2.1.8   Residue Management

Crop residues are the residual remaining after harvest of the economic part of the crop. The amount of crop residue varies with the crop and the harvest operation method. For example, cotton harvest contributes very little aboveground residue to the soil due to the plant's low leaf area index and small amount of plant material after leaf drop. Soybean and other legume crops also have small amounts of aboveground residue that rapidly decompose because of low C:N ratios. In contrast, crops like corn can leave substantial amounts of residue on the soil surface unless the whole plant is harvested for silage or the residue is collected for bedding or other purposes.

Aboveground residue management entails five potential strategies: (1) leave the residue on the soil surface to decay and be incorporated into the soil (requires no-till management); (2) incorporate the residue into the soil via tillage; (3) remove the residue through a harvesting operation (i.e., silage or cellulosic biomass harvest); (4) allow livestock to graze on the residue; or (5) burn the residue. Each of these management practices has the potential to affect GHG emissions. Leaving crop residue on the surface and incorporating it into the soil after decay by microorganisms affects CO_2 release from the soil due to the enhanced biological activity, and potentially increases N_2O emissions through an alteration of the nitrogen balance in the soil. A similar process occurs when residue is incorporated into the soil via tillage. Note that tillage also causes reductions in soil carbon stocks, and additional CO_2 is released through burning fuel to run tillage equipment. Harvesting the residue releases CO_2 from burning fuel in the engines linked with the harvesting process, although residue harvested for biofuel production may create net fossil fuel offset credits. Burning crop residues in the field releases CO_2, CH_4, and N_2O (as well as CO and NO_x) emissions to the atmosphere. In general, but not always, residue removal reduces soil carbon stocks (Gregg and Izaurralde, 2010; Wilhelm et al., 2007).

Management interactions are also important when considering the influence of residue management on GHG emissions. For example, the influence of residue management on soil organic carbon will be affected by the tillage practices (Malhi et al., 2006).

3.2.1.9   Set-Aside/Reserve Cropland

The 1985 Farm Bill established the Conservation Reserve Program (CRP) to pay producers to convert highly erodible cropland or other environmentally sensitive agricultural areas into vegetative cover. These areas could be converted into grassland, native bunchgrasses, pollinator habitat, shelterbelts, filter or buffer strips, or riparian buffers. Areas are removed from production and seeded with annual and perennial species to form a cover that would be undisturbed for a minimum of 10 years. In return, producers or landowners received a payment for enrolling these land areas into the CRP. Throughout the agricultural history of the United States, there have been times in which agricultural lands were set aside to reduce agricultural surpluses; however, the time period of removal was typically short-term (one to two years) and maintained in a weed-free state.

The primary aims of CRP are to decrease erosion, restore wildlife habitat, and safeguard ground and surface water quality. An important ancillary aim is carbon capture: CRP lands sequester carbon in soil and long-lived plants, and thus represent a valuable mitigation opportunity. In a meta-analysis of paired soils, Ogle et al. (2005) found that 20 years of set-aside resulted in temperate region soils' accumulating 82 to 93 percent of the carbon levels under original native

vegetation, on average. Post and Kwon (2000) concluded from a global meta-analysis that, on average, soil carbon sequestration rates on land converted from agricultural production to grassland is 33 g C m^{-2} year^{-1}. At 39 paired CRP-crop sites in Wisconsin, Kucharik (2007) found sequestration rates of 50 g C m^{-2} year^{-1} on Mollisols and 44 g C m^{-2} y^{-1} on Alfisols. Follett et al. (2009) estimate that CRP soils sequester ~50 g C m^{-2} year^{-1} on average. The Council for Agricultural Science and Technology (2011) estimates that CRP lands are currently responsible for 6.3 Tg of soil carbon sequestration per year. Gebhart et al. (1994) reported a mean 18.8 percent increase on five CRP sites during a six-year period. However, there are studies showing little or no increase in C, leading to uncertainty in the effect of set-aside land in a reserve program (Jelinski and Kucharik, 2009; Karlen et al., 1999; Reeder et al., 1998). For example, Karlen et al. (1999) compared CRP land with perennial grasses to cropland across five States and found that only one site of the five showed a significant difference in total organic carbon content in the soil after being in CRP.

Increases in soil carbon resulting from setting aside cropland in CRP can be reversed by converting these lands back into production. Gilley et al. (1997) found that the positive changes in CRP land disappeared immediately when the soils were tilled upon conversion back into crop production. However, many studies indicate that if land under CRP is returned to cultivation, some or all of the soil carbon can potentially be retained if the land is cultivated with no-till practices (Bowman and Anderson, 2002; Dao et al., 2002; Olson et al., 2005). In addition to changes in soil carbon stocks, changes will also occur in N$_2$O emissions depending on the nutrient management practices. Gelfand et al. (2011) measured a net carbon cost of 10.6 Mg CO$_2$-eq ha^{-1} (289 g C-eq m^{-2}) for the first year of no-till soybeans following 20 years of CRP grassland, and a significant portion of the net emission was due to N$_2$O produced in the conversion year.

3.2.1.10 Biochar

Biochar is a soil amendment that is promoted for its ability to improve crop production and sequester carbon in soils (Atkinson et al., 2010; Lehmann, 2007a; 2007b). Biochar is charcoal produced when wood or other plant biomass is burned under low-oxygen conditions, known as pyrolysis. When applied to soils, biochar can persist for long periods of time; its chemical structure makes it resistant to microbial attack under most soil conditions. However, its persistence can vary greatly for reasons not yet completely understood. Biochar is a common component of most U.S. agricultural soils (Skjemstad et al., 2002), left from fires that occurred prior to conversion of the original forest or prairie. Adding biochar to soils has been proposed as a way to sequester carbon (Lehmann, 2007a) because of this potential to persist for centuries (Kimetu and Lehmann, 2010; Nguyen et al., 2008). But biochar's longevity in soil depends on a number of factors including pyrolysis conditions (e.g., pyrolysis temperature) and the chemical composition of the biochar feedstock (Spokas, 2010). Climate and soil factors such as mineralogy and pre-existing organic matter content also affect biochar's persistence in soil.

An additional benefit of biochar is its positive effects on agricultural soil fertility (Atkinson et al., 2010; Laird et al., 2010), largely by providing advantages similar to other forms of soil organic matter: improved soil structure, water holding capacity, and cation-exchange capacity. Biochar has also been shown to reduce soil N$_2$O emissions in some laboratory studies, but the small number of field trials so far reported have documented no significant effects under field conditions (e.g., Scheer et al., 2011).

It is too early to know if promising results from laboratory and short-term field experiments can be generalized to long-term field conditions. Biochar soil additions may be a future source of carbon credits for pyrolysis waste if long-term field experiments confirm results from shorter term studies. The climate advantage of adding biochar to soil is less clear, however, relative to other potential uses of plant biomass. Life cycle analyses (e.g., Roberts et al., 2010) suggest that biochar may

increase or decrease net emissions depending on alternative uses of the original biomass and life cycle system boundaries. Furthermore, if the biomass (or biochar) was burned directly for energy then the source of displaced energy must also be considered (Roberts et al., 2010). Nevertheless, both the sequestration and N_2O suppression potential of biochar merit further study.

3.2.2 Management Influencing GHG Emissions in Flooded Cropping Systems

There are a variety of flooded cropping systems in the United States, including systems for rice, wild rice, cranberries, and taro. Apart from rice, these systems are relatively minor (specialty crops) and there is little to no research or information on their GHG emissions. Rice systems emit both CH_4 and N_2O; however, many reports show an inverse relationship between CH_4 and N_2O during the rice cropping season, with CH_4 occurring under anaerobic conditions and N_2O emissions occurring under aerobic conditions (Zou et al., 2005). Therefore, to accurately determine a mitigation strategy one needs to consider the net cumulative effect of GHG emissions by evaluating both CH_4 and N_2O. Water and residue management have received the most attention in terms of offering possibilities for mitigating CH_4 emissions. Other mitigation options have also been examined and show promise (e.g., Feng et al., 2013; Linquist et al., 2012; Majumdar, 2003; Wassmann and Pathak, 2007; Yagi et al., 1997) and further research is required in many areas before these options can be scaled up. The intent here is not to provide a review of the literature but to provide a brief overview of some factors affecting GHG emissions from flooded rice systems.

3.2.2.1 Water Management in Flooded Rice

In the United States, rice is planted in one of two ways: (1) water seeded, where seeds are sown by airplane in flooded fields; or (2) dry-seeded, where seeds are drilled or broadcast (then incorporated) into dry fields. Water seeding is the predominant practice in California and parts of Louisiana, while dry seeding is predominant in much of the southern United States (e.g., Arkansas, Mississippi, Missouri, and Texas). Water management varies between these two established practices. In water-seeded rice, the fields are typically flooded for the entire season. However, in Louisiana, the field may be drained with a pinpoint flood system (three to five days) or with a delayed flood (up to 20 days) after seeding. In dry-seeded rice, rainfall or flush irrigation events are relied upon during the first three to five weeks of establishment and then flooded for the rest of the season. In all cases, fields are typically drained a few weeks before harvest to allow the soil to dry out enough to support harvest equipment. Further details of U.S. rice production systems can be found in Snyder and Slaton (2001) and Street and Bollich (2003).

Midseason drain or intermittent irrigation is a strategy to mitigate CH_4 emissions. This practice results in aerobic conditions that are unfavorable for methanogens. However, such conditions are favorable for N_2O emissions (e.g., Zou et al., 2005). Most studies report that midseason drains significantly decrease CH_4 emissions but increase N_2O emissions relative to continuous flooding. Regardless, net GHG emissions in rice systems are usually decreased with midseason drain despite the increase in N_2O. Wassman et al. (2000) reported that CH_4 emission reductions ranged from seven percent to 80 percent. The reduction in CH_4 emissions depends on the number of drainage events during the cropping season and on other management factors and soil properties. Yan et al. (2005) reported that CH_4 fluxes from rice fields with single and multiple drainage events were reduced by 60 percent and 52 percent compared to continuously flooded rice fields. This practice has not been widely evaluated in the United States, and it may be difficult to drain and re-flood the large relatively flat parcels of land that are commonly used for rice production in the United States. Furthermore, such practices can lead to increased weed and disease pressure along with lower yields and grain quality.

Soil carbon stocks are also influenced by water management. For example, carbon stocks in Chinese rice systems are higher than in upland crops, presumably due to the accumulation of carbon under the flooded conditions (Pan et al., 2010; Wu, 2011). It remains unknown if efforts to mitigate CH_4 emissions in the United States using intermittent flooding will lead to a reduction in soil carbon stocks.

The use of midseason drainage has been shown to delay harvest in California. Therefore, in climates with a short growing season, the use of a midseason drain will increase risk of crop failure, and therefore will be a less appealing alternative to growers.

3.2.2.2   Residue Management

Straw management has a large impact on CH_4 production. Straw additions, particularly those with a high carbon to nitrogen ratio, increase CH_4 emissions but have the potential to reduce N_2O emissions (e.g., Zou et al., 2005). This reduction in N_2O may be due to increased nitrogen immobilization or more effective conversion to N_2. Low carbon to nitrogen organic materials tend to increase N_2O emissions (Kaewpradit et al., 2008). Yan et al. (2005) reported that the timing of straw application is also an important factor. For example, applying rice straw before transplanting increased CH_4 emissions by 2.1 times, while applying rice straw in the previous season increased CH_4 emissions by 0.8 times. Several studies have demonstrated that composting rice straw prior to incorporation reduces CH_4 emissions (Wassmann et al., 2000); however, this requires additional energy to collect the straw and then spread it back on the field after composting.

In contrast to the potential for reducing CH_4 emissions with removal of rice straw, there is also the potential to reduce soil carbon stocks due to less carbon input to soils. Other nutrients (particularly K) are removed in large amounts with residues, and these need to be replaced to maintain the productivity of the system.

3.2.2.3   Organic Amendments

Various organic amendments can be applied to rice fields, including farmyard manure specialty mixes of organic fertilizers, and green manures (e.g., cover crops). Based on a meta-analysis by Linquist et al. (2012), livestock manure increases CH_4 emissions by 26 percent and green manures increased CH_4 by 192 percent. Neither manure source had a significant effect on N_2O emissions. Few studies have evaluated the influence of different manure storage and processing techniques on CH_4 emissions. One example is a study by Wassman et al. (2000), who found that fermentation of farmyard manure prior to application can reduce CH_4 emissions. Farmyard manure will also influence soil carbon stock and soil N_2O emissions.

3.2.2.4   Varieties, Ratoon Cropping, and Fallow Management

Seasonal CH_4 (Lindau et al., 1995) and N_2O (Chen-Ching, 1996) emissions are affected by rice variety. The cause of varietal differences vary but may be due to gas transport through arenchyma cells, different rooting structures, or differences among varieties in terms of root exudates (Wassmann and Aulakh, 2000). Identifying the mechanisms for varietal differences may enable breeding programs to select varieties that have lower CH_4 emissions.

In some States, the climate allows re-sprouting of a second, or ratoon crop, that grows from the stubble of the first crop after harvesting. Ratoon crop yields are smaller than the first crop, but can add substantially to the overall annual yield, thereby reducing costs of production per unit. In addition, it takes fewer resources and less time to grow a ratoon crop than to grow the first crop. However, ratooning has higher CH_4 emission rates (about two to three times higher) than the first crop, because the straw from the first crop remains in the field under anaerobic conditions during the ratoon period rather than the field being drained so that the stubble can decay aerobically

(Lindau et al., 1995). Therefore, the amount of CH_4 producing organic material (i.e., material available for anaerobic decomposition) is considerably higher than with the primary crop.

Management of rice fields during the winter has a significant effect on annual GHG emissions. For example, in California, legislation in the 1990s has limited the burning of rice straw to a maximum of 25 percent of an area, although in reality only about 10 percent of rice production fields are burned. Currently, rice straw is incorporated after harvest on about 85 percent of the rice production fields in California, and in these fields about half are intentionally flooded to facilitate straw decomposition, although this value can vary widely from year to year. Winter flooding has increased annual CH_4 emissions (Devito et al., 2000), but it has also increased the quality of habitat for overwintering waterfowl in the Pacific Flyway. Rice straw is baled and removed on about five percent of the area.

3.2.2.5   Nitrification and Urease Inhibitors in Flooded Rice

Nitrification inhibitors prevent or slow the conversion of NH_4^+ to NO_3^- and thus reduce N_2O emissions from nitrification and subsequent denitrification. In a meta-analysis of these products, Akiyama et al. (2010) found that in rice systems the use of nitrification inhibitors on average reduced N_2O emissions by 30 percent, although some products were more effective than others. Certain nitrification inhibitors (i.e., dicyandiamide, thiosulfate, and encapsulated calcium carbide) can mitigate both CH_4 and N_2O emissions. Reduced CH_4 emissions using dicyandiamide was attributed to a higher redox potential, lower pH, lower Fe^{2+}, and lower readily mineralizable carbon content (Bharati et al., 2000).

Urease inhibitors, such as hydroquinone, slow the microbial conversion of urea to NH_4^+, thus reducing the amount of nitrogen available for nitrification and dentrification. Both CH_4 and N_2O emissions were reduced with the use of hydroquinone (Boeckx et al., 2005). It is suggested that urease inhibitors mitigate CH_4 emission by inhibiting the methanogenic fermentation of acetate (Wang et al., 1991). Furthermore, a combination of a urease inhibitor (hydroquinone) and a nitrification inhibitor (dicyandiamide) was shown to result in lower GHG emissions compared with using only one of the products (Boeckx et al., 2005). See Section 3.2.1.1 for more information on nitrification and urease inhibitors.

3.2.2.6   Fertilizer Placement in Flooded Rice

Incorporating/injecting or placing fertilizer deep into the soil has been shown in some studies to reduce both CH_4 (Wassmann et al., 2000) and N_2O (Keerthisinghe et al., 1995) emissions. While much of a flooded rice field's soil is anaerobic, the floodwater and top few centimeters of soil typically remain aerobic while soil below five centimeters exists in an anaerobic, reduced state (Keeney and Sahrawat, 1986). Thus mineral nitrogen in the top few centimeters of soil may undergo nitrification and denitrification, which can lead to N_2O emissions; but mineral nitrogen in lower soil depths will remain as ammonium. In contrast, nitrogen fertilizer that is applied to the soil surface (either preseason or midseason) tends be more susceptible to losses either from ammonia volatilization or more rapid nitrification-denitrification processes (Griggs et al., 2007). By placing nitrogen into anaerobic soil layers, it is better protected from losses and remains available for crop nitrogen uptake (Linquist et al., 2009). The effect of deep fertilizer placement on CH_4 reduction remains uncertain. See Section 3.2.1.1 for more information on fertilizer placement.

3.2.2.7   Sulfur Products

Sulfur-containing fertilizers (i.e., ammonium sulfate, calcium sulfate, phosphogypsum, and single super phosphate) reduce CH_4 emissions (Lindau et al., 1998). The magnitude of CH_4 reduction is dependent on fertilization rate with averages between 208 and 992 kg S ha^{-1}, reducing CH_4

emissions by 28 percent and 53 percent, respectively (Linquist et al., 2012). At low levels of sulfur fertilization, which are common in recommended rates, the effect on CH_4 emissions will be limited (Linquist et al., 2012). Sulfur mitigates CH_4 emissions in two ways. First, SO_4 additions to soil add electron acceptors, thus slowing soil reduction (Majumdar, 2003). Second, the product of SO_4 reduction (H_2S) may inhibit methanogenic bacteria and thus depress CH_4 production. Unfortunately, most studies have not examined the effect on N_2O emissions.

3.2.3 Land-Use Change to Cropland

Conversion from one land-use category (e.g., forestland, wetlands) to cropland can have significant effects on the GHG emissions and removals associated with the land under conversion. When land is converted to cropland, there is often a loss of carbon, an increase in N_2O and CH_4 emissions, a reduction in CH_4 oxidation, and if biomass is burned, an increase in non-CO_2 GHG emissions. A number of variables influence the direction and magnitude of the emissions and sinks including prior land use, climate, and management. The influence of land-use change on carbon, nitrogen, methane, and non-CO_2 GHGs are discussed below.

3.2.3.1 *Influence on Carbon Stocks*

Land-use conversion to cropland can have significant effects on biomass, litter, and soil carbon (IPCC, 2000). Houghton et al. (1999) estimated that land clearance in the United States has led to a loss of 27 Pg C to the atmosphere since the 1700s, although recently some carbon has been restored with conversion of cropland back to other uses and also improved soil management (U.S. EPA, 2010). Clearing forest leads to a large loss of aboveground and belowground biomass and litter C; grassland conversion can also reduce the amount of carbon in these pools, but to a lesser extent than forest conversion because grasslands have less biomass. Soil carbon losses can be significant with conversion to cultivated crop management (Davidson and Ackerman, 1993), with relative losses in temperate regions from 20 to 30 percent on average (Ogle et al., 2005).

Ultimately, the net influence of land conversion will depend on the previous land use, vegetation composition, and management, and the resulting cropland system and its associated vegetation composition and management. For example, conversion of grassland to tree crops, such as orchards, may lead to gains in carbon relative to the grassland due to accumulation of carbon in woody biomass.

3.2.3.2 *Influence on Soil Nitrous Oxide*

The conversion of land to cropland generally accelerates nitrogen cycling, with subsequent effects on N_2O and CH_4 fluxes. Soil nitrogen availability is the factor that most often limits soil N_2O emissions (see Section 3.2.1.1), so any practice that increases the concentration of inorganic nitrogen in soil is likely to also accelerate N_2O emissions. As noted above, land-use change typically results in faster soil organic matter turnover and associated nitrogen mineralization, which means that even in the absence of nitrogen fertilizer, soil N_2O fluxes will be higher on converted land. Additional nitrogen from fertilizers, whether synthetic or organic, or from planted legumes will further enhance N_2O fluxes, as will tillage—insofar as tillage stimulates nitrogen mineralization.

The conversion of unmanaged land to cellulosic biofuel production may avoid additional GHG loading if care is taken to avoid soil carbon oxidation and excess soil nitrogen availability (Robertson et al., 2011). This might occur, for example, if existing perennial vegetation were harvested for feedstock or when new perennial grasses were direct-seeded into an otherwise undisturbed soil profile, and when no or minimal nitrogen inputs are used. Although the current market for cellulosic biomass is nascent at best, as it develops in response to legislative mandates and energy demand there will be pressure to convert lands now unmanaged into biofuel cropping

systems. Minimizing the GHG impact of these conversions will be crucial for avoiding long-term carbon debt that will otherwise lead to carbon sources rather than carbon sinks, irrespective of their capacity to generate fossil fuel offset credits (Fargione et al., 2008; Gelfand et al., 2011; Pineiro et al., 2009).

3.2.3.3 *Influence on Methanotrophic Activity*

Methanotrophic bacteria capable of consuming (oxidizing) atmospheric CH_4 are found in most aerobic soils. CH_4 uptake in soils is globally important; the size of the soil sink is the same magnitude as the atmospheric increase in CH_4 (IPCC, 2001), suggesting that significant changes in the strength of the soil sink could significantly affect atmospheric CH_4 concentrations if uptake declines due to land use and management. In unmanaged upland ecosystems, CH_4 uptake is controlled largely by the rate at which it diffuses to the soil microsites inhabited by active methanotrophs. Diffusion is regulated by physical factors—principally moisture but also temperature, soil structure, and the concentration of CH_4 in the soil.

Agricultural management typically diminishes soil CH_4 oxidation by 70 percent or more (Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000) for at least as long as the soil is farmed. The mechanism for this suppression is not well understood; likely it is related to nitrogen availability as affected by enhanced nitrogen mineralization, fertilizer, and other nitrogen inputs (Steudler et al., 1989; Suwanwaree and Robertson, 2005). NH_4^+ is known to competitively inhibit methane monooxygenase, the principal enzyme responsible for oxidation at atmospheric concentrations. Microbial diversity also seems to play an important role (Levine et al., 2011).

There are no known agronomic practices that promote soil CH_4 oxidation; although a better understanding of the mechanisms responsible for its suppression may eventually suggest mitigation opportunities. To date, recovery of significant CH_4 oxidation capacity following agricultural management has only been documented decades after conversion to forest or grassland; complete recovery appears to take a century or longer (Robertson et al., 2000; Smith et al., 2000).

3.2.3.4 *Non-CO$_2$ GHG Emissions from Burning*

Burning can be conducted on lands in preparation for cultivation to facilitate access for equipment, remove standing accumulated biomass, and provide organic material (ash) for incorporation into soils. Burning of the biomass can be an important source of non-CO_2 GHGs (N_2O, CH_4) as well as precursors to GHG formation (CO, NO_x) following additional chemical reactions in the atmosphere or soils. More information on burning of grazing lands vegetation can be found in Section 3.3.1.5, and burning of the remaining biomass with clearing of forest can be found in Section 6.4.1.9.

3.3 Grazing Land Management

Rangelands are defined as land on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species managed for grazing and browsing. Conversely, pasturelands represent land managed primarily for the production of introduced forage plants for livestock grazing, with management consisting of fertilization, weed control, irrigation, reseeding or renovation, and control of grazing (USDA, 2009). How grazing lands are managed influences the potential for carbon sequestration or GHG emissions. The paragraphs below highlight some of the key management practices and their associated GHG emissions and removals summarizing the current state of the science.

3.3.1 Management Activity Influencing GHG Emissions

Soil organic carbon dominates the terrestrial carbon pool in grazing lands. Aboveground carbon is < five percent of the total ecosystem carbon pool in most non-woody plant dominated ecosystems, but up to 25 percent in grassland-shrubland ecosystems. Grazing lands can be carbon sinks, with rates of soil organic carbon sequestration up to 0.5 Mg C ha^{-1} year^{-1} for rangelands (Derner and Schuman, 2007; Liebig et al., 2010) and 1.4 Mg C ha^{-1} year^{-1} for pastures (Franzluebbers, 2005; 2010a). Actual rates are often less than these apparent maximal rates of soil organic carbon sequestration due to management, climate, weather, and other environmental constraints. Potentially high rates of soil organic carbon accumulation are predicted in newly established pastures and restoration of degraded rangelands, while improper management and drought can result in significant carbon releases. Due to the large land area, the movement of carbon into and out of the soil reservoir in grazing land can be an important component of the global carbon cycle. In addition to soil organic C, a large pool of soil inorganic carbon occurs as carbonates in semi-arid and arid rangeland soils that can lead to either sequestration or release of CO_2 (Emmerich, 2003). However, the direction and magnitude of soil inorganic carbon stocks are currently poorly understood (Follett et al., 2001; Liebig et al., 2006; Svejcar et al., 2008).

Two important management factors that control the fate of soil organic carbon in grazing lands are long-term changes in production and quality of aboveground and belowground biomass that can alter the quantity of nitrogen available and the C-to-N ratio of soil organic matter (Pineiro et al., 2010), and grazing-induced effects on vegetation composition, which can be as important as the direct impact of grazing (e.g., grazing intensity) on soil organic carbon sequestration (Derner and Schuman, 2007). The rate of soil organic carbon sequestration can be linear for decades (Franzluebbers et al., 2012), but eventually diminishes to a steady-state level with no further change in the stock following several decades of a management practice (Derner and Schuman, 2007). Additional positive changes in management or inputs are often needed to sequester additional soil organic carbon (Conant et al., 2001), but negative changes in management causing loss of soil structure and surface litter cover can lead to erosion and loss of productivity resulting in a decline in soil organic carbon (Pineiro et al., 2010).

Methane flux from grazing lands is controlled by the balance of enteric and manure emissions from ruminant animals and uptake of CH_4 by soil. (Emissions and methods for estimating CH_4 emissions from ruminants are discussed further in Section 5.3). In the western United States, grasslands have greater CH_4 uptake by soil than do neighboring croplands (Liebig et al., 2005), probably due to greater surface soil organic matter that promotes the growth of methanotrophic bacteria. In an assessment of GHG emissions from three grazing land systems in North Dakota, enteric emissions of CH_4 from grazing cattle were three to nine times greater (on a CO_2 equivalent basis) than CH_4 uptake by soil (Liebig et al., 2010). With CH_4 emissions directly tied to number of cattle, fertilized grasslands are often a net carbon source due to enhanced CH_4 emission from cattle and potentially greater N_2O emissions, while unfertilized grasslands are often a net carbon sink (Luo et al., 2010; Tunney et al., 2010).

3.3.1.1 Livestock Grazing Practices

Livestock grazing practices (i.e., stocking rate and grazing method) are summarized below along with data on the influence these practices have on GHG emissions and removals.

Stocking Rate: Stocking rate is the number of animals per management unit utilized over a specified time period, e.g., number of steers per acre per month. Based on published studies, responses of soil organic carbon to stocking rate and grazing intensity have been variable, despite grazing either causing an increase or having little effect on the more commonly measured property of soil bulk density (Greenwood and McKenzie, 2001; Schuman et al., 1999). In northern mixed-grass prairie,

soil organic carbon has increased in grazed compared with ungrazed areas, partly resulting from increasing dominance of shallow-rooted, grazing-resistant species, such as blue grama (*Bouteloua gracilis*), which incorporates a larger amount of root mass in the upper soil profile than the mid-grass species that it replaces during grazing (Derner et al., 2006). Further research is needed to determine the extent of different root distributions on total carbon storage in an entire soil profile. Increasing stocking rate beyond an optimum for achieving maximum livestock production per unit land area (Bement, 1969; Dunn et al., 2010) would be expected to result in a loss of soil organic carbon due to reduced plant vigor and root distribution in the soil profile. With suboptimal stocking rate, vigor of pasture forages may decline as plant residues develop a thick litter layer at the soil surface. However, in semi-arid regions, the high UV light intensity may significantly reduce litter on the soil surface through photochemical decomposition processes, regardless of grazing intensity (Brandt et al., 2010). Vegetation composition shifts that change the quantity and quality of plant material produced can influence the amount of carbon inputs to soils. In managed pastures, it has been shown that soil organic carbon can be optimized with a moderate stocking rate compared with no grazing or heavy, continuous grazing (Franzluebbers, 2010b). An optimized stocking rate for a particular region (climatic conditions), vegetation composition, and soil type is thought to maximize the amount of soil organic carbon sequestered.

Limited evidence shows that grazing at moderate levels can further increase environmental benefits over those of grassland establishment alone, in addition to providing an important economic return to producers. If soil organic carbon were to decline with overgrazing, there would also be a decline in animal productivity due to lack of forage. Therefore, a negative relationship between soil organic carbon storage and animal productivity is likely when grazing intensity exceeds a moderate level. This response is likely modified under moderate grazing pressure due to the fact that greater animal product per head can be achieved with lower GHG emissions. Limiting the effect of high stocking rate on soil organic carbon levels may be achievable with high nitrogen fertilizer inputs, an outcome with an uncertain carbon footprint relative to GHG intensity. Stocking rate and fertilizer nitrogen input interactions need to be quantified to accurately assess total GHG intensity. Some evidence in the humid United States suggests that overgrazing can lead to increased soil erosion and a reduction in soil quality. Literature from other regions has also shown increasing soil erosion and declining soil quality with excessive stocking rates. While evidence is lacking, an assumption is that soil organic carbon follows this same positive response to moderate grazing and negative response to overgrazing.

Emissions of N_2O from grazing lands are affected by grazing, but net flux can be increased or decreased, depending on stocking rate, grazing system, and season (Allard et al., 2007). Stocking rate had little influence on N_2O emissions from mixed-grass prairie in North Dakota (Liebig et al., 2010). While elevated N_2O emissions may be expected under increased stocking rate, Wolf et al. (2010) suggested that grazing can counteract potential N-induced emissions on rangelands by reducing surface biomass, resulting in more extreme soil temperatures, lower soil moisture, and corresponding inhibition of microbial activity responsible for N_2O emissions. If grazing intensity on pastures were viewed as a fertilizer effect with increasing animal manure deposition, then N_2O flux from a grazing effect does not behave in the same manner as manufactured nitrogen fertilizer inputs. Interactions between stocking rate and nitrogen fertilizer inputs have not been quantified, despite such diversity in management likely occurs among producers. Stocking rate and manure and fertilizer nitrogen inputs are areas requiring further research to better understand the complex set of controlling factors in addition to soil texture and environmental conditions on N_2O emissions in grazing lands. On rangelands, the abundance of N-fixing legumes in the plant community becomes more critical for increasing SOC, particularly since fertilizer additions and manure are not as significant for returning nitrogen to the soil compared to pasture systems. This is an area requiring further research to better understand the controlling factors on N_2O emissions.

Grazing Method: Grazing methods vary based on producer goals and the type of forage available (Scheaffer et al., 2009). Two distinct grazing methods, continuous and rotational grazing, represent the prevalent methods employed on grazing lands in the United States to manage the livestock. Continuous grazing allows animals to freely move and have full access to a grazing area, whereas rotational grazing is more controlled, involving movement of animals based on monitoring forage condition, such as plant height, between two or more paddocks subdivided from a larger grazing area. Rotational grazing terminology has been confused with terms such as holistic grazing, planned grazing, prescribed grazing, and management-intensive grazing, which continue to be used with multiple and ambiguous meanings despite attempts to standardize definitions (SRM, 1998). Terms to define intentions of rotational grazing systems include rest-rotation, deferred-rotation, high-intensity-short-duration, and season-long grazing (Briske et al., 2008; Briske et al., 2011). Here we define rotational grazing as the movement of livestock between two or more subunits of grazing land such that alternating periods of grazing and no grazing ('rest') occur within a single growing season (Heitschmidt and Taylor, 1991).

Rotational grazing limits plants from reaching reproductive stages in which forage quality rapidly declines. This contrasts with continuous grazing in which there is more selective grazing of the highest quality forages. As such, forage quality may be maintained at a high level longer into the growing season. Therefore, rotational stocking in the humid United States could provide more uniform forage consumption across pastures and allow sufficient rest to forage species between grazing events to promote greater production. Pastures with greater plant production via an improved stocking method would be expected to have lower soil erosion and greater soil organic carbon storage. Although these expectations seem intuitive, there are limited data in the scientific literature to support them. Two studies have suggested an increase in soil organic carbon with rotational grazing compared with continual season-long grazing (Conant et al., 2003; Teague et al., 2010), and another study found no difference between systems (Manley et al., 1995). Since rotational grazing data are mostly available for rangeland and few studies conducted on pastures, there is not enough evidence to evaluate how rotational grazing might affect soil organic carbon in pastures. Given that the preponderance of evidence suggests that rotational grazing does not influence vegetation production in rangelands (Briske et al., 2008), changes in soil organic carbon with rotational grazing would be expected only if substantial vegetation change occurred independently from stocking rate. Rangelands typically have a much higher diversity and multiple growth patterns of forbs, cool-season and warm-season grasses, which would result in a smaller influence of stocking method on vegetation phenology (i.e., keeping forage in a vegetative rather than a reproductive state) than would occur in monoculture or simple mixtures of forages in pastures. Much more research on grazing method is needed, due to the high adoption rate and promotion of the benefits of improved grazing methods for soil organic carbon sequestration by producers and agricultural advisors (Beetz and Rhinehart, 2010).

3.3.1.2 Forage Options

Cool- and warm-season forages have growth activity at different times of the year, thereby affecting when root and litter carbon inputs are supplied to soil. Depending on environmental growing conditions (i.e., relatively short, cool, and wet summer with long, cold winter versus long, hot, and dry summer with mild, wet winter), the performance of cool- versus warm-season forages will vary across regions. In the southeastern United States, perennial cool-season forages (e.g., tall fescue) have produced greater soil organic carbon than warm-season forage (e.g., bermudagrass) in grazing land systems, despite the more vigorous growing habit of bermudagrass (Franzluebbers et al., 2000). This result is likely due to the opportunities of forages for growth and the balance of water in soil that remains for microbial decomposition of organic matter.

Timing of forage grazing can affect plant productivity, wildlife habitat, and compaction of soil. Each of these effects can, in turn, affect soil organic carbon sequestration and GHG emissions. The capacity of soil to withstand compaction forces of animal treading, resulting in significant deformation, destabilization, loss of infiltration capacity, and soil organic carbon sequestration, can be exceeded—especially under wet conditions (Bilotta et al., 2007). Soil saturation during winter and spring lead to severe effects from animal trampling. In northern latitudes and rangelands of the western United States subject to freeze-thaw cycles, sandy and loamy soils are less likely to be affected by the negative impacts of compaction. Intuitively, deferring grazing to periods of limited active forage growth (e.g., winter and spring) might contribute to increased soil compaction. However, allowing forage to accumulate to full canopy prior to grazing might be beneficial to controlling erosion by providing a longer period of forage and residue cover. Grazing of winter cover crops may also be an effective farm-diversity strategy, but the effects on soil erosion control and soil condition need to be quantified. Wildlife management guidelines on rangeland suggest longer-term (> one year) rest to accumulate vegetation structure for certain birds needing habitat. Timing of grazing could be a critical factor in controlling compaction, susceptibility to erosion, and soil organic carbon sequestration, so the sequence of when pastures are grazed should be rotated among years to ensure that plant communities are not always grazed at the same time to ensure greater community sustainability.

Organic matter-rich surface soil absorbs compactive forces of grazing much like a sponge, in which soil often rebounds in volume once forces are removed. However, effects of winter grazing of deferred growth may be different in colder than in warmer regions: frozen soil may avoid compaction, but nutrient runoff may become more important (Clark et al., 2004). In the southern United States, perennial cool-season grasses are often grazed during late winter and throughout spring during typically wet conditions, but due to active forage growth, soil can also dry quickly and trampling may not always cause damage. In Georgia, soil organic carbon was greater under long-term stands of cool-season tall fescue (typically grazed in spring and autumn) than under warm-season bermudagrass (typically grazed in summer) (Franzluebbers et al., 2000).

In the southeastern United States, annual cool-season forages are often planted as a cover crop following summer crops or sod-seeded into perennial grass pastures. This practice can enhance forage production and should increase soil organic C, although limited data are available to support this conclusion. In an integrated crop/livestock system in the southeastern United States, there was a limited effect of grazing annual cover crops on soil organic C, either in the summer or winter compared with ungrazed cover crops (Franzluebbers and Stuedemann, 2009).

3.3.1.3   Irrigation

Water is a limiting factor in the ability of plants to fix carbon and subsequently produce the carbon input necessary to accumulate soil organic C. It is also a factor limiting decomposition of soil organic C. While the extent of irrigation in grazing lands is limited, where it occurs there are consequences for soil organic carbon storage. For example, some productive meadows in the western United States are irrigated. How irrigation affects soil organic carbon will depend on the quantity, frequency, and timing of irrigation events. Irrigation only at peak plant growth stages will likely cause a much greater positive impact on forage carbon fixation than a negative impact on soil organic carbon decomposition. In the same manner, irrigation quantity, frequency, and timing will likely affect N_2O and CH_4 emissions, although pulsed responses of these GHGs could likely be much more dramatic. Unfortunately, there are only limited studies on these potential impacts. See Section 3.2.1.4 for more information on irrigation methods.

In a comparison of agricultural systems with surrounding arid and semi-arid natural vegetation, Entry et al. (2002) found that soil organic carbon was greater in irrigated agricultural systems due

to enhanced productivity. Emission of N_2O from irrigated systems occurs following closely timed irrigation and nitrogen fertilizer applications in cropland conditions, and this would be expected under grazing lands as well, but there are few data available (Liebig et al., 2006; Liebig et al., 2012).

3.3.1.4   Nutrient Management (Synthetic and Organic)

Fertilizers are often applied to pastures, due to the high yield response with adequate precipitation, but are less common in western rangelands due to inconsistent yield response and risky cost-effectiveness with limited and variable precipitation. Nitrogen availability in soil determines to a large extent the emissions of N_2O. Grazing lands typically have lower nitrogen availability in soil than croplands, and therefore have lower N_2O emissions (Liebig et al., 2005). However, application of fertilizer nitrogen to rangeland has been found to consistently stimulate N_2O emissions (Flechard et al., 2007). Liebig et al. (2010) observed two-fold greater N_2O emissions from fertilized crested wheatgrass compared with unfertilized mixed-grass prairie. Addition of fertilizer nitrogen to pasture in Michigan had a negligible effect on N_2O emissions (Ambus and Robertson, 2006), whereas application of poultry manure on a bermudagrass pasture in Arkansas increased N_2O emissions by 45 percent compared with pasture without manure; N_2O flux and soil nitrate dynamics were positively associated (Sauer et al., 2009). A strategy to reduce soil nitrate by interseeding annual ryegrass on manure-amended soil decreased N_2O emissions by 50 percent. Similar to cropland, reducing soil nitrate to low levels during periods of low root activity and high levels during periods of high root activity will generally enhance plant nitrogen uptake and reduce N_2O emissions. Application of composted green waste could sequester C, but this research topic has not been fully evaluated. A significant increase in soil organic carbon has only been demonstrated at one of two sites in California (Ryals et al., 2014). From model simulations, compost application has been shown to reduce the overall GHG emission on CO_2 equivalent basis, by sequestering carbon and reducing N_2O emissions, while manure slurry and inorganic fertilizer applications led to net GHG emissions on CO_2 equivalent basis (DeLonge et al., 2013). For more information on management options associated with fertilization practices, see Section 3.2.1.1.

3.3.1.5   Prescribed Fires

Burning has the potential to alter soil organic carbon through effects on photosynthesis, soil, and canopy respiration, and through species changes, in addition to stabilizing or increasing livestock gains, improving habitat diversity, and reducing fuel loads (Boutton et al., 2009; Toombs et al., 2010). Although carbon loss from burning grazing lands is a minor component of the annual carbon emissions, burning rangelands with a significant woody aboveground plant biomass can result in substantial immediate ecosystem carbon loss (Bremer and Ham, 2010; Rau et al., 2010). However, prescribed burning of grazing lands could also affect long-lived char that accumulates in soil, and therefore would influence soil carbon stocks. Burning also leads to non-CO_2 GHG emissions, which can be significant due to the higher global warming potential of these gases compared with CO_2 (IPCC, 2006). For more information on non-CO_2 GHG emissions from burning, see Section 3.2.3.4.

3.3.1.6   Erosion Control

Riparian buffers can be a significant sink for excess nutrients running off neighboring grazing lands. The fate of nutrients is dependent on the flow characteristics and type of vegetation. Excess nitrate in saturated soil of riparian areas can lead to significant N_2O emissions—although these emissions are typically treated as indirect, with the emissions associated with the field or livestock facility that is contributing the excess nutrients (See Section 3.2.1.1). Transport of soluble carbon into riparian areas could also enhance CH_4 emissions from saturated soil.

3.3.1.7   Management of Drained Wetlands

Drainage of wetland or hydric soils that are used for grazing has implications for soil organic carbon and GHG emissions, similar to drainage for crop production. The water regime and plant communities are significantly altered and soils are modified from anaerobic to aerobic conditions. Increasing oxygen in soil will cause organic matter to decompose more rapidly than under saturated conditions, resulting in release of CO_2 (Eagle et al., 2010; Franzluebbers and Steiner, 2002; IPCC, 2006; Liebig et al., 2012). Large emissions of CO_2 result from drainage of wetlands (Allen, 2007; 2012), and drainage can also increase nitrogen mineralization and enhance N_2O emissions directly (IPCC, 2006). Emissions of CH_4 are reduced considerably with drainage, but this impact is often not considered in estimation of GHG emissions (IPCC, 2006). A large proportion of grassland wetlands have been directly drained or modified to enhance agricultural production (Dahl and Johnson, 1991), and many other wetlands are indirectly affected by subsurface tile drains and agricultural practices in surrounding catchments. See Section 3.2.1.6 for more information about management of drained soils.

3.3.1.8   Lime Amendments

Lime amendments are needed when soil pH is low (e.g., pH<5) to enhance productivity and support balanced nutrient levels in grazing land soils. Typical liming materials in grazing lands are calcitic limestone ($CaCO_3$), dolomitic limestone ($CaMg(CO_3)_2$), and confined livestock manure, particularly poultry litter, which has liming activity from lime additive to the feed ration. When carbonate lime is applied to soil it dissolves in solution over time, with the cation and carbonate dissociating. There is potential for releasing CO_2 to the atmosphere depending on whether the lime reacts with carbonic or nitric acid in the soil solution. The enhanced plant nutrient offered by liming can have a net positive effect on the carbon balance for an extended period of time. See Section 3.2.1.7 for more information on lime and the consequences for GHG emissions.

3.3.1.9   Woody Plant Encroachment

Woody plant encroachment[3] leads to carbon accumulation in above-ground and root biomass and may increase overall ecosystem carbon storage, but can degrade agricultural productivity of grazing land (McClaran et al., 2008). Over the past century in western rangelands, soil organic carbon has increased in near-surface soils with woody plant encroachment (Boutton et al., 2009; Creamer et al., 2011; Liao et al., 2006; Liebig et al., 2012). Removal of woody plants by fire or other mechanisms depletes these shallow, relatively susceptible soil organic carbon stores associated with encroachment (Neff et al., 2009; Rau et al., 2010); but does not have an effect on SOC or total nitrogen stocks at depths of >20 cm (Dai et al., 2006). Regardless, removal of the woody plants will cause a decline in aboveground biomass carbon stocks (Rau et al., 2010).

In a summary of research on CH_4 emissions from grazing lands, Liebig et al. (2012) reported CH_4 uptake under mesquite, but net CH_4 production under grassland and dead mesquite stumps. Methane uptake under mesquite was associated with reduced soil bulk density and increased soil moisture (McLain and Martens, 2006), as well as greater nitrogen accrual/accumulation associated in the area around mesquite plants (10 meters) (Boutton and Liao, 2010; Liao et al., 2006; Liu et al., 2010). Methane uptake under mesquite was also associated with altered soil microbial communities (Hollister et al., 2010; Liao and Boutton, 2008), which can affect NO_x and N_2O rates, while CH_4 production from grassland and woody detritus was likely caused by termite activity. The

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[3] Woody encroachment will eventually lead to a transition from grazing land to a forest. See Chapter 7: Land Use Change for definition of forestland to determine when woody encroachment will lead to a transition to forestland.

role of mesquite to fix N, thereby altering nitrogen dynamics, resulted in N_2O emissions under mesquite canopy four-fold greater than under grasses or woody detritus (McLain et al., 2008).

3.3.2   Land-Use Change to Grazing Lands

Land-use conversion to grazing lands influences the carbon stocks and GHG emissions of a parcel. Prior land use, climate, soil type, and management practices are just a few of the factors influencing the magnitude and direction of GHG emissions and removals resulting from a land-use conversion to grazing lands. The paragraphs below summarize the current state of the science on the influence of a land-use conversion on carbon stocks, soil N_2O, CH_4, and non-CO_2 GHGs resulting from biomass burning.

3.3.2.1   Influence on Carbon Stocks

Establishment of pastures on previous cropland helps reduce soil erosion and improves soil quality (Singer et al., 2009). There is substantial evidence that establishment of pastures leads to significant soil organic carbon sequestration. The rate of accumulation across a number of studies averaged 0.84 Mg C ha^{-1} $year^{-1}$ (Franzluebbers, 2010a). Literature is inadequate to determine whether forage composition or soil type have a discernible influence on soil organic carbon stock (see Section 3.3.1.2). The quantity of forage produced and the quantity of residues from surface litter and root biomass are likely key determinants of soil organic carbon accumulation. These quantities can be influenced by factors such as forage mixture, climatic conditions, soil type, inherent soil fertility, fertilizer application, and liming.

3.3.2.2   Influence on Soil Nitrous Oxide

Depending upon previous land use, grassland establishment may or may not affect net N_2O emissions during land-use change. In general, emissions of N_2O are controlled by soil nitrogen availability with additional influence of soil oxygen and soluble carbon availability. If the previous land use was for example, a nutrient-limited forest, converted subsequently to high-fertility pasture, then N_2O emissions would likely increase. If the previous land use was nutrient-rich cropland converted to pasture, then N_2O emissions would likely decline due to greater opportunity for perennial forage species to assimilate available soil nitrogen and thus reduce opportunities for soil nitrogen transformations to N_2O. This is an area requiring further research to obtain quantitative responses, however.

3.3.2.3   Influence on Methanotrophic Activity

Land-use change to grazing land, particularly from forestland, may involve fertilization to enhance forage production. Nitrogen fertilization causes a reduction of methanotrophic activity in soils and therefore reduces the uptake of CH_4 from the atmosphere (Ambus and Robertson, 2006). See Section 3.2.3.3 for more information on the impact of land-use change on methanotrophic activity.

3.3.2.4   Non-CO_2 GHG Emissions from Burning

Biomass burning in grazing land can be an important source of GHGs (CO_2, N_2O, CH_4) (Aalde et al., 2006; Andreae and Merlet, 2001; Badarinath et al., 2009; IPCC, 2006). While conversion of cropland to grazing land rarely involves burning, conversion of forest to grazing land can involve burning of the wood and/or slash left from land clearing. The effect on GHG emissions from biomass burning is discussed further in the cropland section (Section 3.2.3.4) and in the forestland section (Section 6.4.1.9).

3.4 Agroforestry

Agroforestry represents a unique case within GHG accounting, encompassing both forest and agricultural components, along with many combinations of their respective management activities (Table 3-1 and Table 3-2). Agroforestry is defined within the United States as an "intensive land-use management that optimizes the benefits (physical, biological, ecological, economic, and social) from biophysical interactions created when trees and/or shrubs are deliberately combined with crops and/or livestock" (Gold and Garrett, 2009). Another way of looking at agroforestry is as a set of tree-based[4] conservation/production practices combined into bigger agricultural operations, providing forest-derived functions and interacting with agriculture-derived functions in support of agricultural land use. While providing many other services (see Table 3-3), agroforestry can contribute to carbon sequestration, GHG mitigation, and adaptation to shifting climate (CAST, 2011; IPCC, 2000; Morgan et al., 2010; Verchot et al., 2007).

Table 3-3: Six Categories of Agroforestry Practices Practiced in the United States

Practice	Description[a]	Benefits[b]
Alley cropping	Trees or shrubs planted in sets of single or multiple rows with agronomic, horticultural crops, or forages produced in the alleys between the sets of woody plants that produce additional products	▪ Produce annual and higher-value but longer-term crops for diversification of income ▪ Enhance microclimate conditions to improve crop or forage quality and quantity ▪ Reduce surface water runoff and erosion ▪ Improve soil quality by increasing utilization and cycling of nutrients ▪ Alter subsurface water quantity or water table depths ▪ Enhance wildlife and beneficial insect habitat ▪ Decrease offsite movement of nutrients or chemicals ▪ Increase carbon storage in plant biomass and soils ▪ Improve air quality
Forest farming (also called multi-story cropping)	Existing or planted stands of trees or shrubs that are managed as an overstory with an understory of woody and/or non-woody plants that are grown for a variety for products	▪ Improve crop diversity by growing mixed but compatible crops having different heights on the same area ▪ Improve soil quality by increasing utilization and cycling of nutrient and maintaining or increasing soil organic matter ▪ Increase net carbon storage in plant biomass and soil
Riparian forest buffers[c] (combines Natural Resources Conservation Service Practice Standards: Riparian Forest Buffer and Filter Strip)	A combination of trees, shrubs, and grasses established on the banks of streams, rivers, wetlands, and lakes	▪ Decrease offsite movement of nutrients or chemicals ▪ Stabilize streambanks ▪ Enhance aquatic and terrestrial habitats ▪ Provide economic diversification either through plant production or recreational fees ▪ Increase carbon storage in plant biomass and soils

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[4] Also referred to as trees-outside-forests, the term "tree" here includes both tree and shrubs (Bellefontaine et al., 2002).

Practice	Description[a]	Benefits[b]
Silvopasture	Trees combined with pasture and livestock production	▪ Provide diversification of crops in time and space ▪ Produce annual and higher-value but longer-term crops ▪ Decrease offsite movement of nutrients or chemicals
Windbreaks (also referred to as shelterbelts)	Linear plantings of trees and shrubs to form barriers to reduce wind speed (may be specifically referred to as crop or field windbreak, livestock windbreak, living snowfence, or farmstead windbreak, depending on the primary use)	▪ Control wind erosion ▪ Protect wind-sensitive crops ▪ Enhance crop yields ▪ Reduce animal stress and mortality ▪ Serve as a barrier to dust, odor, and pesticide drift ▪ Conserve energy ▪ Provide snow management benefits to keep roads open or harvest moisture
Special applications	Use of agroforestry technologies to help solve special concerns, such as disposal of animal wastes or filtering irrigation tailwater, while producing a short- or long-rotation woody crop	▪ Treat municipal and agricultural wastes ▪ Treat stormwater ▪ Use in center pivot corner plantings ▪ Produce biofeedstock ▪ Reduce impacts of flooding ▪ Decrease offsite movement of nutrients or chemicals

Source: USDA Natural Resources Conservation Service (2012).
[a] Descriptions follow USDA Natural Resources Conservation Service Conservation Practices Standards.
[b] All agroforestry plantings add increased diversity within the agricultural landscape. As such, they will improve wildlife habitat and generally are designed or managed with this as a secondary benefit.
[c] Riparian forest buffer refers to the planted practice. This category does not include naturally established riparian forests.

In the United States, five main categories of agroforestry practices are recognized: alley cropping, forest farming, riparian forest buffers, silvopasture, and windbreaks. There is an emerging sixth category of special applications or adaptations of these practices (Table 3-3). These practices are treated within the cropland and grazing land system section with the exception of forest farming. Forest farming (also referred to as multi-story cropping within USDA Natural Resources Conservation Service Practice Standards) involves the manipulation of existing forest canopy cover in order to produce high-value non-timber (i.e., food, floral, medicinal, and craft) products in the understory, thus maintaining land use as forest. As such, GHG accounting in forest farming practices will need to be treated within the methods and approaches presented in Section 6.2 and Section 6.4.

The many services derived from agroforestry practices can extend well beyond the small parcel or amount of land they physically occupy within the agricultural landscape (Bellefontaine et al., 2002; Garrett, 2009). The use of agroforestry technologies are important components at the rural/community interface, as well as within urban settings to address emerging needs such as stormwater treatment, recreation or green space, and feedstock production (Schoeneberger et al., 2001). Although agroforestry is categorized into these practices, each agroforestry planting, even within a practice, potentially represents a unique case of species selection, arrangement, placement within other practices and the larger landscape, and use of management activities, depending on landowner objectives. Agroforestry plantings are therefore more of a "designer landscape feature" than a standardized and easily described practice (Mize et al., 2008) within GHG accounting activities.

Silvopasture provides a good illustration of this complexity in agroforestry systems. Silvopasture is the deliberate combination of three components— trees, forage, and livestock—along with the range of their respective management activities. Studies demonstrate a higher carbon sequestration potential in silvopasture compared with forest or pasture alone (Haile et al., 2010; Nair et al., 2007;

Sharrow and Ismail, 2004). Much of this new carbon is in the woody biomass, but soil carbon also has the potential to increase as a consequence of carbon inputs from the trees, which over time extend further into the forage component (Peichl et al., 2006), as well as management of the forage and of the livestock (see Franzluebbers and Stuedemann, 2009; Karki et al., 2009). Management activities within a silvopasture may include fertilization, liming, cultivation, and harvesting of the forage crop (in some years); periodic harvesting of pine needles for pine straw; incorporation of pruned woody material into the forage component; and different grazing intensities and rotations. The frequency and intensity of management activities and inputs from all three components can vary significantly from year to year, which makes accounting for the sequestered carbon in a silvopasture operation challenging.

Rates and amounts of GHG emissions within each agroforestry planting will vary depending on prior land management and current conditions (i.e., site, climate), as well as by stand development. These rates and amounts will also be dependent on landowners' decisions that determine planting design, as well as management activities—agricultural, forestry, and grazing—used over the lifetime of an agroforestry system (Table 3-4).

Table 3-4: Management Activities[5] and Other Factors Within Agroforestry Practices That May Alter Carbon Sequestration and GHG Emission Amounts

Practice	Management Activities
Windbreaks	• Establishment disturbance to soil during site preparation • Deposition of wind- and water-transported sediments, nutrients, and other agricultural chemicals into the planting • Windbreak renovation (removal of dead and dying trees over time)
Riparian forest buffers	• Establishment disturbance to soil during site preparation • Deposition of wind- and water-transported sediments, nutrients, and other agricultural chemicals into the planting • Harvesting of herbaceous materials planted in Zone 3 (zone closest to crop/grazing system) and of woody materials planted in Zone 2 (middle zone)
Alley cropping	• Establishment disturbance to soil during site preparation • Weed control (mechanical or chemical) • Pruning, thinning, and harvesting of woody material (amount and frequency vary greatly depending on short- and long-term objective of practice) • Fertilization for alley crop and occasionally needed for trees in rows • Tillage in alleys (frequency and intensity) • Crop species used in alley production • Complex harvesting schedules stratified in space and time
Silvopasture	• Establishment disturbance to soil during site preparation • Weed control (mechanical or chemical) • Pruning, thinning, and harvesting of woody material (amount and frequency vary greatly depending on short- and long-term objective of practice) • Fertilization of forage component • Tillage in forage component (frequency and intensity) • Crop species used in forage component • Grazing management (timing, intensity, frequency) • Complex harvesting schedules stratified in space and time

3.4.1   Carbon Stocks

Agroforestry's potential for sequestering large amounts of carbon per unit area is well recognized (Dixon et al., 1994; Kumar and Nair, 2011; Nair et al., 2010), with sequestration rates being greater

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[5] Forest Farming is not included in these considerations.

than many of the other agricultural options (IPCC, 2000). Carbon is sequestered directly into the woody biomass and soil. Indirectly, agroforestry practices can alter carbon cycling by enhancing crop and forage production (up to 15 H—height of trees—distance from the windbreak) and trapping wind-blown and runoff erosion (Brandle et al., 2009). Lack of data limits accounting of these other carbon fluxes impacted by the addition of trees and is beyond the scope of this effort.

Woody Biomass: The majority of new carbon contributed to a site by agroforestry will be from the production of woody biomass, with the larger contribution being from the aboveground woody biomass, as generally observed in forest establishment plantings (Nui and Duiker, 2006). The more open environment created in agroforestry plantings results in the trees having different growth forms than encountered under forest conditions—e.g., greater branch production (Zhou, 1999) and specific gravity (Zhou et al., 2011)—which will need to be taken into account when estimating the aboveground woody biomass.

The belowground biomass pool in agroforestry plantings will also be a significant portion of new carbon added to the site. However, measuring, estimating, and/or verifying this component is very difficult and expensive. The contributions from root biomass can be estimated using various approaches that rely on knowing the aboveground portion.

Forest Products and Other Removed Materials: Windbreaks and riparian forest buffers are planted for purposes that require the trees to be in place for the targeted function(s) (i.e., alteration of microclimate; interception of sediments, nutrients, and chemicals). Windbreak renovation (removal of dead trees and replanting) is recommended to maintain microclimate benefits (Brandle et al., 2009). Periodic harvesting of plant materials in the herbaceous zone (adjacent to crop field) and middle woody zone is also recommended in riparian forest buffers to maintain higher rates of nutrient uptake and therefore water quality services (Dosskey et al., 2010). More innovative and diversified planting designs that incorporate bioenergy feedstocks are being considered for both of these practices, which would increase levels of harvesting within these systems. In the case of riparian forest buffers, harvesting of the herbaceous and woody middle zone for bioenergy feedstocks would serve to replenish a higher nutrient uptake rate and thus water quality services, as well as provide an additional income stream (Schoeneberger et al., 2008). Many alley cropping and silvopasture systems are managed for high-value veneer and saw-timber. These trees, along with some special applications of agroforestry technologies, are also being investigated for their use in producing bioenergy feedstocks. For these plantings, removal or harvesting of aboveground woody material can occur as early as three years to 75 years or more, depending on the product. Harvested materials can also include stem-pruning, generally up to 15 feet over several years to attain a clean bole, to periodic thinning in order to maintain a canopy cover that is optimal for the growth of the tree as well as the crop being grown in the alleys. The material may be left onsite to create wildlife habitat, chopped and incorporated into the soil, or taken off-site and burned.

Soil: Studies have documented that U.S. agroforestry practices generally have greater soil carbon stocks (under the whole practice, which may vary from just under a windbreak to under the whole tree/crop system, such as alley cropping) when compared with that in conventional agricultural and grazing practices (Nair et al., 2010). However, estimating change or flux in soil carbon stocks in agroforestry plantings is challenging due to its inherently high spatial and temporal variability. For instance, Sharrow and Ismail (2004) found variability of soil carbon to be two to three times greater in a non-grazed silvopasture system than in the adjacent forest or pasture alone.

Soil carbon can increase in agroforestry systems due to added carbon inputs from the trees, the elimination of carbon loss due to annual cropping activities (i.e., conservation tillage), and potentially the addition of carbon through other agricultural management activities, such as incorporation of different crops, cover crops, residue management, and fertilization regimes.

Changes in soil carbon stocks have been estimated in a number of forest establishment plots from the Midwest, and were found to vary from -0.07 to 0.58 Mg C ha^{-1} year^{-1} and -0.85 to 0.56 Mg C ha^{-1} year^{-1} in deciduous and coniferous plots, respectively. Paul et al. (2003) attributed the variation to the impact and variable recovery from tree planting, but also mentioned the possibility that variation may be due to the use of present-day cropping fields as the carbon baseline for comparison. Many agroforestry studies are reporting comparable rates of soil sequestration (see Nair et al., 2010). Results from temperate agroforestry studies indicate, especially for alleys receiving high level of organic matter input from the trees, that it may be several years before significantly measurable carbon differences are detectable between the agroforestry planting and traditional sole cropping system (Peichl et al., 2006; Udawatta et al., 2009). The amount and duration of soil organic matter accumulation in agricultural soils with agroforestry management will depend on the degree to which prior soil carbon stocks are depleted. In addition, it will depend on the soils in general, climate, placement within a landscape, type of vegetation, and most importantly, by the additional management activities employed in the mixed tree/agricultural system (Table 3-4).

Note that carbon increases from nitrogen inputs may be offset through enhanced N_2O emissions, depending on a number of factors (see Section 6.4.1.6). Many agroforestry plantings, such as windbreaks and riparian forest buffers, are purposefully designed to intercept soil in wind erosion and surface runoff, which is another addition of carbon to this pool (Sauer et al., 2007). Deposition of sediment will influence cycling of both elements and therefore net GHG values (McCarty and Ritchie, 2002; Sudmeyer and Scott, 2002). We currently lack the understanding and data needed for adequately modeling and therefore predicting these intra- and inter-soil carbon transfers from erosion and deposition.

3.4.2   Nitrous Oxide

Data on direct N_2O emissions in agroforestry plantings are sparse. The few studies to-date found reduced N_2O emissions in afforested plots that were older than five years (Allen et al., 2009), under windbreaks (Ryskowski and Kedziora, 2007) and riparian forest buffers (Kim, 2008). Alley cropping systems reduced N_2O emissions by 0.7 kg ha^{-1} year^{-1} compared with the annual cropping systems with no tree cover (Thevathasan and Gordon, 2004). These studies suggest the trees can act as a "nitrogen-safety net" in the system, taking up the "extra" nitrogen that might otherwise result in N_2O emissions. In addition, reduced nitrogen leaching has been documented within agroforestry plantings compared with the annual cropping system with no tree cover (Allen et al., 2004; Lopez-Diaz et al., 2011; Nair et al., 2007). The reduced leaching implies that less nitrogen is available for indirect soil N_2O emissions, which could be beneficial in those agroforestry plantings requiring fertilization (i.e., alley cropping and silvopasture systems) or that receive large inputs of nitrogen through surface and subsurface runoff (i.e., riparian forest buffers). As many agroforestry plantings are purposefully designed and planted to provide tighter nutrient cycling capabilities as a means to protect water quality (Olson et al., 2000), the capability and capacity of these systems to reduce N_2O emissions in agricultural systems warrants further study to determine whether and how it should be accounted for in GHG accounting methods.

3.4.3   Methane

Very little research has been done to determine whether the establishment of agroforestry plantings can lead to any change in CH_4 sinks or sources in soils due to changes in methanotrophy or methanogenesis, respectively. Kim et al. (2010) did not find any evidence in established riparian forest buffers in Iowa (seven to 17 years old) that CH_4 flux differed from neighboring crop fields. Riparian forest buffers could potentially serve as a CH_4 emitter given the periodic flooding that may occur within these plantings. However, riparian forest buffers established on agricultural lands may

not be significant emitters of CH_4 because the hydrological connections within these landscapes have been decoupled. This indicates use of riparian forest (naturally occurring) derived data may result in overestimating sink/source capacity of riparian forest buffers. In general, there is insufficient data to model and predict methane fluxes in agroforestry at this time.

3.4.4 Management Interactions

Agroforestry practices can indirectly alter carbon cycling by enhancing crop and forage production and trapping windblown and surface runoff sediments. Examining the carbon potential of windbreaks in the Great Plains, Brandle et al. (1992) estimated indirect carbon benefits could potentially be double the amount of the carbon sequestered in the wood. Although projects to examine indirect carbon benefits from several of the agroforestry practices are ongoing, we currently lack the ability to model or predict these impacts.

3.5 Estimation Methods

This section provides methods for estimating GHG emissions from cropland and grazing land systems on an entity's land. The methods are applied for both land remaining in cropland or grazing lands, as well as land-use change to cropland or grazing lands. The methods provided are for estimating the emission levels for a given year on a parcel of land. A parcel is a field in the entity's operation with uniform management. If management varies across the field, then the field should be subdivided into separate parcels for estimating the emissions.

Trends across years or comparisons to baselines can be made using the annual emission estimates. Guidance is not given here on how to develop baselines or subsequent trends for emission estimation. The level of emissions for carbon stocks is based on estimating the change in stock from the beginning and end of the year, while the level of emissions for N_2O and CH_4 are based on estimating the total annual emissions. Methods are also provided for estimating total emissions of precursor gases emitted during biomass burning, as well as nitrogen compounds that are volatilized or subject to leaching and runoff from an entity's cropland or grazing land that are later converted into GHGs.

The methods range in complexity for the different emission source categories according to the state of the science and prior method development. Simple methods are selected for several of the emission or carbon stock change source categories; because the more complex methods are not fully developed for operational accounting of emissions or the simple methods provide a reasonably accurate and precise result. Although simplicity may be preferred for transparency in estimation, some of the methods use more complex approaches, such as process-based simulation models, because these methods greatly improve the accuracy and/or precision of the result.

3.5.1 Biomass Carbon Stock Changes

Method for Estimating Biomass Carbon Stock Changes

- A modified version of the methodology developed by the IPCC (Lasco et al., 2006; Verchot et al., 2006) has been adopted for entity-scale estimation of herbaceous and woody biomass stock changes associated with land use.
- The DAYCENT process-based simulation model or the traditional forest inventory approaches are used to estimate carbon for aboveground biomass for agroforestry.
- U.S. specific default values (West et al., 2010) are used for estimating biomass carbon for annual crops and grazing lands. The IPCC default is used for estimating the carbon fraction value. Yield in units of dry matter can be estimated by the entity or average values from USDA-National Agricultural Statistics Service statistics can be used.
- This method was chosen because it captures the influence of land-use change on crop or forage species on biomass carbon stocks by using U.S. specific default values where entity specific activity data are not available and a process-based simulation model for agroforestry systems.

3.5.1.1 Rationale for Selected Method

Both IPCC (2006) and the U.S. Environmental Protection Agency (2011) consider herbaceous biomass carbon stocks to be ephemeral, and recognize that there are no net emissions to the atmosphere following crop growth and senescence during one annual crop cycle (West et al., 2011). However, with respect to changes in land use (e.g., forest to cropland), the IPCC (Lasco et al., 2006) recommends that cropland biomass be counted in the year that land conversion occurs, and the same assumption also applies for grassland (Verchot et al., 2006). According to the IPCC, accounting for the herbaceous biomass carbon stock during changes in land use is necessary to account for the influence of herbaceous plants on CO_2 uptake from the atmosphere and storage in the terrestrial biosphere. However, this method does not recognize changes in herbaceous biomass that occur with changes in crop rotations, nor does it recognize long-term increases in annual crop yields. The method is a considered a Tier 2 method as defined by the IPCC because it incorporates factors that are based on U.S. specific data.

Agroforestry, along with other woody vegetation in croplands, such as orchards and vineyards, sequester significant amounts of new carbon within long-lived biomass over time with tree growth. Methods for estimating the aboveground woody and whole tree biomass for trees growing under forest conditions are described in the Forestry Section of this report. However, these methods, developed from forest-derived (i.e., greater canopy closure) conditions, do not accurately reflect conditions encountered in agroforestry or woody crops. Trees growing under windbreak and other linear-type plantings have been documented to differ from forest-grown trees in terms of architecture and properties, such as crown:trunk allocation (Zhou, 1999), specific gravity (Zhou et al., 2011), and taper (Zhou et al., in review). Moreover, the Forest Inventory and Analysis program of the USDA Forest Service and National Resource Inventory of the USDA Natural Resources Conservation Service do not collect agroforestry or woody crop data through their surveys (Perry et al., 2005). Therefore, a Tier 3 method using process-based models is a viable alternative for estimating the carbon stock changes associated with agroforestry and woody crops without direct measurement through a survey. Specifically, the DAYCENT model has been parameterized to simulate tree growth and has been adopted for estimating woody biomass carbon for agroforestry and woody crops.

3.5.1.2 Description of Method

A modified version of the methodology developed by the IPCC (Lasco et al., 2006; Verchot et al., 2006) has been adopted for entity-scale reporting in the United States of herbaceous and woody biomass stock changes associated with land use change. The method consists of estimating the mean annual biomass stock for a cropland or grazing lands following a land use change, which can be averaged across years for a crop or rotation. This method only addresses a change in the herbaceous biomass carbon stocks in the year following a land-use change, consistent with the IPCC methods (Lasco et al., 2006; Verchot et al., 2006). In contrast, carbon stock change in woody biomass is estimated every year.

Use Equation 3-1 to estimate the total biomass carbon stock change for a land parcel over a year:

Equation 3-1: Total Biomass Carbon Stock Change

$$\Delta C_{Biomass} = (H_t + W_t) - (H_{t-1} + W_{t-1})$$

Where:

$\Delta C_{Biomass}$ = Total change in biomass carbon stock (metric tons CO_2-eq year^{-1})

H $\quad$ = Mean annual herbaceous biomass (metric tons CO_2-eq year^{-1})

W $\quad$ = Mean annual woody biomass (metric tons CO_2-eq year^{-1})

t $\quad$ = Current year stocks

t-1 $\quad$ = Previous year's stocks

Herbaceous Biomass: Estimate the mean annual herbaceous biomass stock in a land parcel for cropland or grazing land following a land use change with the following equation:

Equation 3-2: Mean Annual Herbaceous Biomass Carbon Stock

$$H = [H_{peak} + (H_{peak} \times R{:}S)] \times A \times CO_2MW / Y_f$$

Where:

H $\quad$ = Mean annual herbaceous biomass carbon stock (metric tons CO_2-eq year^{-1})

H_{peak} $\quad$ = Annual peak aboveground biomass (metric tons C ha^{-1} year^{-1})

R:S $\quad$ = Root-shoot ratio (unitless)

A $\quad$ = Area of land parcel (ha)

CO_2MW = Ratio of molecular weight of CO_2 to carbon = 44/12
$\quad\quad\quad$ (metric tons CO_2 (metric tons C)$^{-1}$)

Y_f $\quad$ = Approximate fraction of calendar year representing the growing season (unitless)

The mean annual biomass stock is intended to represent the time period following harvest where no crop exists and both litter and roots are decomposing quickly (Gill et al., 2002), and the time period during the growing season where biomass continues to grow until it reaches peak annual biomass. The average of zero biomass and peak biomass (e.g., peak biomass divided by two) is considered representative of the mean annual carbon stock (i.e., $Y_f = 0.5$).

Equation 3-3 is used to estimate the peak aboveground biomass in a land parcel from harvest yield data in croplands or peak forage yields in grazing lands.

> **Equation 3-3: Aboveground Herbaceous Biomass Carbon Stock**
>
> $$H_{peak} = (Y_{dm} / HI) \times C$$
>
> Where:
>
> H_{peak} = Annual peak aboveground herbaceous biomass carbon stock
> (metric tons C ha^{-1} year^{-1})
>
> Y_{dm} = Crop harvest or forage yield, corrected for dry matter content
> (metric tons biomass ha^{-1} year^{-1})
>
> = Y x DM
>
> Y = Crop harvest or forage yield (metric tons biomass ha^{-1} year^{-1})
>
> DM = Dry matter content of harvested crop biomass or forage (dimensionless)
>
> HI = Harvest Index (dimensionless)
>
> C = Carbon fraction of aboveground biomass (dimensionless)

This method captures the influence of land-use change and changes in crop or forage species on biomass carbon stocks. Therefore, crop harvest or peak forage yields should be averaged across years as long as the same forage species, crop or rotation of crops are grown. The harvest index is set to one for grazing lands.

Peak forage estimates for grazing lands can be estimated using the biomass clipping method.[6] This method is destructive with the removal of forage samples from the field. Non-destructive methods can also be used including the comparative yield method for rangelands[7], or the robel pole method on rangelands or pastures (Harmoney et al., 1997; Vermeire et al., 2002). Any sampling that is done, whether destructive or non-destructive, should occur at locations that are representative of the land parcel. If sampling the forage is not feasible, default forage production values are provided by the Natural Resources Conservation Service in Ecological Site Descriptions (ESDs).[8] After identifying the appropriate ESD, the entity would select the plant community that is representative of the parcel. These values represent total production for the site so Y_f in Equation 3-2 would be set to 1 if the aboveground forage production is obtained from an ESD.

Woody Biomass: The largest amount of carbon captured by agroforestry systems is in woody biomass, with the majority occurring in the aboveground biomass. Woody crops also gain carbon as they grow. This method also addresses carbon removals through harvest or other events that remove tree biomass.

The methods to estimate biomass carbon in a land parcel for the more-open growth of agroforestry systems and woody crops (W_t and W_{t-1} in Equation 3-1) are based on DAYCENT model simulations and growth functions for agroforestry. Agroforestry practices are based on the Natural Resources Conservation Service agroforestry practice standards, which are provided in a pick list. For woody crops, the DAYCENT model simulates the influence of common management practices on biomass stocks, including irrigation, fertilization, organic matter amendments, groundcover management,

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[6] See section 15, "Standing Biomass"
http://www.nrisurvey.org/nrcs/Grazingland/2011/instructions/instruction.htm
[7] See section 13, "Dry Weight Rank"
http://www.nrisurvey.org/nrcs/Grazingland/2011/instructions/instruction.htm
[8] See ESDs https://esis.sc.egov.usda.gov/

pruning of branches, thinning of young fruit, and harvest and removal of mature fruit. Given the practice, DAYCENT simulates changes in woody biomass carbon stocks for the reporting period.

For agroforestry systems where the entity has measured tree parameters, an empirical model is provided to more precisely estimate woody biomass carbon growth increment for the year (Merwin and Townsend, 2007; Merwin et al., 2009). The empirical model uses an individual tree growth equations based on Lessard (2000) and Lessard et al. (2001). Carbon pools are then derived from diameter-based allometric equations that predict total aboveground biomass components for 10 broad species groups in the United States. (Jenkins et al., 2003; 2004). Both published and unpublished data for the U.S. Forest Service Forest Inventory and Analysis program were used to develop the growth increment model.

In addition, harvested woody products associated with agroforestry are estimated using the approaches described in the Forestry Chapter (Section 6.5). Woody products may be harvested from silvopasture, alley cropping, and other agroforestry practices, providing a variety of products such as veneer, saw timber, and bioenergy feedstocks.

3.5.1.3 Activity Data

Activity and related data needed to estimate biomass carbon for annual crops and grazing lands (as applicable) include:

- Crop type, cropland area, and harvest indices;
- Type of forage, grazing area, and peak forage yield data;
- Total aboveground yield of crop or peak forage yield for grazing lands (metric tons biomass per ha);
- Root:shoot ratios;
- Carbon fractions; and
- Dry matter content of forage and harvested crop biomass to estimate dry matter content.

If the entity does not provide values, default values for moisture content, residue-yield ratios, and root:shoot ratios are provided in Table 3-5. A general default value for crop carbon fraction is 0.45. In some years, the entity may not harvest the crop due to drought, pest outbreaks or other reasons for crop failure. In those cases, the entity should provide the average yield that they have harvested in the past, and an approximate percentage of average crop growth that occurred in the year. The yield is estimated based on multiplying the average crop yield by the percentage of crop growth obtained prior to crop loss. Peak forage yields will vary from year to year, but can be based on a five-year average.

Table 3-5: Representative Dry Matter Content of Harvested Crop Biomass, Harvest Index, and Root:Shoot Ratios for Various Crops [.a]

Crop	Dry Matter Content	Harvest Index	Root:Shoot Ratio
Food crops			
Barley	0.865 (3.8%)	0.46 (18.7%)	0.11 (90.7%)
Beans	0.84 (3.3%)	0.46 (18.7%)	0.08 (89.7%)
Corn grain	0.86 (1.9%)	0.53 (15.0%)	0.18 (97.3%)
Corn silage	0.74 (1.9%)	0.95 (3.3%)	0.18 (97.1%)
Cotton	0.92 (1.4%)	0.40 (20.0%)	0.17 (44.0%)
Millet	0.90 (1.9%)	0.46 (17.6%)	0.25 (91.1%)
Oats	0.865 (1.9%)	0.52 (18.7%)	0.40 (90.9%)
Peanuts	0.91 (1.9%)	0.40 (16.6%)	0.07 (12.4%)
Potatoes	0.20 (9.3%)	0.50 (20.0%)	0.07 (44.1%)

Crop	Dry Matter Content	Harvest Index	Root:Shoot Ratio
Rice	0.91 (1.6%)	0.42 (28.1%)	0.22 (13.2%)
Rye	0.90 (1.9%)	0.50 (18.7%)	0.14 (90.1%)
Sorghum grain	0.86 (1.9%)	0.44 (14.8%)	0.18 (97.2%)
Sorghum silage	0.74 (1.9%)	0.95 (3.3%)	0.18(97.2%)
Soybean	0.875 (1.7%)	0.42 (16.7%)	0.19 (89.8%)
Sugarbeets	0.15 (12.4%)	0.40 (24.1%)	0.43 (43.9%)
Sugarcane	0.258 (11.6%)	0.75 (6.4%)	0.18 (37.4%)
Sunflower	0.91 (1.9%)	0.27 (11.1%)	0.06 (44.0%)
Tobacco	0.80 (1.9%)	0.60 (3.3%)	0.80 (44.0%)
Wheat	0.865 (3.8%)	0.39 (17.7%)	0.20 (86.2%)
Forage and Fodder crops			
Alfalfa hay	0.87 (1.8%)	0.95 (3.3%)	0.87 (21.8%)
Non-legume hay	0.87 (1.8%)	0.95 (3.3%)	0.87 (21.8%)
Nitrogen-fixing forages	0.35 (3.3%)	0.95 (3.3%)	1.1 (21.2%)
Non-nitrogen-fixing forages	0.35 (3.3%)	0.95 (3.3%)	1.5 (21.2%)
Perennial grasses	0.35 (3.3%)	0.95 (3.3%)	1.5 (21.2%)
Grass-clover mixtures	0.35 (3.3%)	0.95 (3.3%)	1.5 (21.2%)

Source: Revised from West et al. (2010).

[a] Uncertainty is expressed on a percentage basis as half of the 95% confidence interval.

Activity data for estimating carbon in aboveground biomass for agroforestry will entail the collection of some level of inventory of trees associated with the agroforestry practice. Simplified inventory approaches requiring a minimum of work by the landowner have been developed by the USDA Natural Resources Conservation Service and the Colorado State University Natural Resource Ecological Laboratory (USDA, 2012), which are largely based on methods described in the Natural Resources Conservation Service National Forest Handbook (USDA NRCS, 2004). The specific activity data requirements include:

- Species of trees and number by age of diameter class for each agroforestry practice; and
- Diameter at breast height for a subsample of trees using one of three sampling methods that capture the spacing arrangements and densities within the different practices (i.e., row type plantings, woodlot-like plantings, and riparian forest buffers).

3.5.1.4   Ancillary Data

No ancillary data are needed for this method.

3.5.1.5   Model Output

Model output is generated for the change in biomass carbon stocks. This change is determined based on subtracting the total biomass carbon stock in the previous year from the total stock in the current year, which will include both herbaceous and woody biomass. The herbaceous stocks will represent mean estimates over years if the same forages, crop, or rotation of crops are grown, and is only estimated for a land use change. The approach for estimating biomass carbon for wetlands and forestlands are described in Sections 4.3.1 and 6.2.1, respectively.

Emissions intensity is also estimated based on the amount of emissions per unit of yield for crops in cropland systems, or of animal products in grazing systems. Note that the biomass change is based solely on woody plant growth except in a year following a land-use change.

The emissions intensity is estimated with the following equation:

Equation 3-4: Emissions Intensity of Biomass Carbon Stock Change

$$EI_{BiomassC} = \Delta C_{Biomass}/Y$$

Where:

$EI_{BiomassC}$ = Emissions intensity (metric tons CO_2 per metric ton dry matter crop yield, metric tons CO_2 per kg carcass yield, or metric tons CO_2 per kg fluid milk yield)

$\Delta C_{Biomass}$ = Change in biomass stock in CO_2 equivalents (metric tons CO_2-eq year^{-1})

Y = Total yield of crop (metric tons dry matter crop yield), meat (kg carcass yield) or milk production (kg fluid milk yield)

3.5.1.6   Limitations and Uncertainty

Uncertainty in herbaceous carbon stock changes will result from lack of precision in crop or forage yields, residue-yield ratios, root-shoot ratios, and carbon fractions, as well as the uncertainties associated with estimating the biomass carbon stocks for the other land uses. Emissions intensity will also include uncertainty in the total yield for the crop, meat, or milk product. This herbaceous biomass method is based on the assumption that half of the crop harvest yields or peak forage amounts provide an accurate estimate of the mean annual carbon stock in cropland or grazing lands. This assumption warrants further study, and the method may need to be refined in the future.

Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity. Table 3-6 provides the relative uncertainty for the DAYCENT model and the carbon fraction of biomass.

Table 3-6: Available Uncertainty Data for Biomass Carbon Stock Changes

Parameter	Mean	Units	Relative Uncertainty Low (%)	High (%)	Distribution	Data Source
DAYCENT (empirical uncertainty)	NS	Various	NS	NS	Normal	Ogle et al. (2007); EPA (2013)
Carbon fraction of aboveground biomass	0.45	Fraction	11	11	Normal	IPCC (1997)

NS = Not Shown. Data are not shown for parameters that have 100's to 1000's of values (denoted as NS).

The uncertainty differs whether it is herbaceous biomass or trees. Uncertainty associated with estimating carbon in live trees is influenced by a number of factors, including sampling and measurement error and error associated with regression models (see Melson et al. 2011; further discussion in Forestry Section). Estimating carbon in agroforestry trees, especially for young seedlings and saplings (up to 10 years or so depending on species and growing conditions) remains highly uncertain particularly since traditional forestry-derived equations have been shown to underestimate whole-tree biomass in agroforestry systems and requires additional field work to further document biomass carbon allocation differences. Melson et al.(2011) noted in their forest-based work that estimation of live-tree carbon was sensitive to model selection (with model-selection error of potentially 20 to 40 percent), and that model selection could be improved by matching tree form to existing equations for use in the models. On-going work comparing agroforestry-derived equations with a variety of forest-derived equations in the Great Plains region indicate uncertainty could be reduced through use of a correction factor. Currently belowground biomass/C estimates are calculated using two approaches: root:shoot ratios (see Birdsey, 1992),

and aboveground density allometry (Cairns et al., 1997), both with large uncertainties due to lack of data. The full set of probability distributions have not been developed for the agroforestry method, and so will require further research before uncertainty can be estimated. See Chapter 6, Forestry, for further discussion of uncertainty of tree volume and biomass equations.

3.5.2 Litter Carbon Stock Changes

Litter in herbaceous biomass decomposes mostly over a one-year period. However the influence of litter carbon stocks on atmospheric CO_2 is assumed to be insignificant after addressing the changes in biomass and subsequent influence on soil carbon stocks. Further methods development may be possible in the future, given this potential limitation to the methods in this report. For cropland or grazing land systems with trees, coarse woody debris and litter carbon should be estimated based on forest methods (See Section 6.2.2.4 and 6.2.2.5). The loss of litter and coarse woody debris with conversion from forestland to cropland and grazing land is also addressed in Section 6. 3.

3.5.3 Soil Carbon Stock Changes

Method for Estimating Soil Carbon Stock Changes

Mineral soils:
- The DAYCENT process-based simulation model estimates the soil organic carbon (SOC) at the beginning and end of the year. These inputs are entered into the IPCC equation to estimate carbon stock changes in mineral soils developed by Lasco et al. (2006), and Verchot et al. (2006).
- This method was chosen because the DAYCENT model has been demonstrated to represent the dynamics of soil organic carbon and estimate soil organic carbon stock change in U.S. cropland and grasslands (Parton et al., 1993), and uncertainties have been quantified (Ogle et al. (2007). The model captures soil moisture dynamics, plant production, and thermal controls on net primary production and decomposition with a time step of a month or less.

Organic Soils:
- IPCC equation developed by Aalde et al. (2006; USDA, 2011) using region specific emission factors from Ogle et al. (2003).
- This method was chosen because it is the only readily available model for estimating soil carbon stock changes from organic soils.

3.5.3.1 Rationale for Selected Method

SOC stocks are influenced by land use and management in cropland and grazing land systems, as well as conversion from other land uses into these systems (Aalde et al., 2006). SOC pools can be modified due to changes in carbon inputs and outputs (Paustian et al., 1997). Carbon inputs will change over time due to interannual variability and longer term trends in net primary production, as well as differences in carbon removals from harvesting and residue management practices. External carbon inputs will also have an influence on the SOC stocks, such as manure, compost, sewage sludge, wood chips, and biochar amendments. Carbon outputs will change due to interannual variability and longer term trends in microbial decomposition rates. In addition, erosion and deposition contribute to changes in SOC stocks associated with crop and grazing land soils. Recent studies (Harden et al., 2008; Van Oost et al., 2007) provide evidence that the majority of carbon in eroded soils is dynamically replaced, compensating for the losses, and at least some of the carbon transported from the site is deposited at the edge of fields, downslope, or in rivers. In all cases, SOC is moved from one location to another under the assumption that only a portion of the

carbon in transport is lost to the atmosphere. This assumption may have significant variation due to the diversity of environmental conditions in which eroded carbon is transported and subsequently resides. Other environmental drivers will also influence carbon dynamics in soils, particularly weather and soil characteristics.

Process-based models, which are considered an IPCC Tier 3 methodology, have been developed and sufficiently evaluated for application in an operational tool to estimate SOC stock changes in mineral soils. The DAYCENT process-based model (Parton et al., 1987; Parton, 1998) has been selected because it is well-tested for estimating soil carbon dynamics in cropland and grazing land systems (Parton et al., 1993) and is also used in the U.S. national GHG inventory (Ogle et al., 2010; U.S. EPA, 2011). Del Grosso et al. (2011) demonstrated the reduction in uncertainty associated with the more advanced approach using the DAYCENT model compared to the lower tier methods. The DAYCENT model simulates plant production by representing long-term effects of land use and management on net primary production (NPP), as influenced by selection of crops and forage grasses. The influence of management practices on NPP are also simulated, including mineral fertilization, organic amendments, irrigation and fertigation, liming, green manures and cover crops, cropping intensity, hay or pasture in rotation with annual crops, grazing intensity and stocking rate, and bare fallow. Nutrient and moisture dynamics are influenced by soil characteristics, such as soil texture. The method addresses interannual variability due to annual changes in management and the effect of weather on NPP.

In the DAYCENT model, three soil organic carbon pools are included representing active, slow, and passive soil organic matter, which have different turnover times. It is generally considered that the active carbon pool is microbial biomass and associated metabolites having a rapid turnover (months to years), the slow carbon pool has intermediate stability and turnover times (decades), and the passive carbon pool represents highly processed and humified decomposition products with longer turnover times (centuries). However, these pools are kinetically defined and do not necessarily represent explicit fractions of soil organic carbon that can be isolated. Soil texture, temperature, moisture availability, aeration, burning, and other factors are represented in the simulations that influence the decomposition and loss of carbon from these pools.

The model simulates management practices influencing soil organic carbon pools. These practices include addition of carbon in manure and other organic amendments, such as compost, wood chips, and biochar; tillage intensity; residue management (retention of residues in field without incorporation, retention in the field with incorporation, and removal with harvest, burning, or grazing). The influence of bare and vegetated fallows is represented, in addition to irrigation effects on decomposition in cropland and grazing land systems. The model can also simulate setting-aside cropland from production; the influence of fire on oxidation of soil organic matter; and woody plant encroachment, agroforestry, and silvopasture effects on carbon inputs and outputs.

A water/soil moisture submodel (e.g., Parton et al., 1987) is used to represent the influence of weather, irrigation, crop type, and management on soil moisture dynamics. This impact is particularly important because moisture tends to be a more proximal factor controlling soil organic carbon dynamics, which, in turn, is influenced by land use and management activity. For example, irrigation influences plant production and carbon inputs because of the modification to the moisture regime.

The modeled estimates from DAYCENT are combined with measurement data from a monitoring network to formally evaluate uncertainty. This approach leverages the scalability of the model while providing an underlying measurement-basis for the method (Conant et al., 2011; Ogle et al., 2007).

Erosion and deposition influence soil organic carbon stocks (Izaurralde et al., 2007) and therefore are represented in the method, although there is uncertainty in the net effect on CO_2 exchange between the biosphere and atmosphere. Moreover, there is also some risk of double-counting carbon as it is transferred across ownership boundaries, in terms of who receives credit for the eroded carbon in their accounting. Regardless, erosion clearly has an impact on carbon stocks in a field, which can be estimated with reasonable accuracy using erosion calculators, such as the Revised Universal Soil Loss Equation, Version 2 (RUSLE2) for water erosion (USDA, 2003) and Wind Erosion Prediction System (WEPS) for wind erosion (USDA, 2004). Therefore, the current method will include an estimate of erosion-related carbon loss from a field, but neither the fate of eroded C, nor the deposition of carbon from other areas onto a land parcel, will be estimated. As more studies are conducted, carbon transport and deposition can be incorporated in future versions of the method.

Drainage of organic soils for crop production leads to net annual emissions due to increased decomposition of the organic matter after lowering the water table and creating aerobic conditions in the upper layers of the soil (Allen, 2012; Armentano and Menges, 1986). There has been less evaluation of process-based models for organic soils, particularly the simulation of water table dynamics throughout the year, which will influence the emission rate. Consequently, the approach is based on more simplistic emission factor approach developed by the IPCC (Aalde et al., 2006). The method incorporates U.S. emission rates associated with region-specific drainage patterns (Ogle et al., 2003), so it is a Tier 2 method as defined by the IPCC.

3.5.3.2   Description of Method

The method representing the influence of land use and management on SOC and associated CO_2 flux to the atmosphere is estimated with a carbon stock change approach (Aalde et al., 2006). For mineral soils, the method will require estimates of carbon stocks at the beginning and end of the year in order to estimate the annual change using the equation below. In contrast, carbon stock changes in organic soils (i.e., Histosols) will address only the emissions occurring with drainage, which is the typical situation in cropland. Emissions occur in organic soils following drainage due to the conversion of an anaerobic environment with a high water table to aerobic conditions (Armentano and Menges, 1986), resulting in a significant loss of carbon to the atmosphere (Ogle et al., 2003). Recent data on subsidence were used to derive these estimates (e.g., Shih et al., 1998).

Mineral Soils: The model to estimate changes in SOC stocks for mineral soils has been adapted from the method developed by IPCC (Aalde et al., 2006). The annual change in stocks to a 30 centimeter depth for a land parcel is estimated using the following equation:

Equation 3-5: Change in Soil Organic Carbon Stocks for Mineral Soils

$$\Delta C_{Mineral} = [(SOC_t - SOC_{t-1}) / t] \times A \times CO_2MW$$

Where:

$\Delta C_{Mineral}$ = Annual change in mineral soil organic carbon stock (metric tons CO_2-eq year^{-1})

SOC_t = Soil organic carbon stock at the end of the year (metric tons C ha^{-1})

SOC_{t-1} = Soil organic carbon stock at the beginning of the year (metric tons C ha^{-1})

t = 1 year

A = Area of parcel (ha)

CO_2MW = Ratio of molecular weight of CO_2 to carbon
= 44/12 (metric tons CO_2 (metric tons C)$^{-1}$)

The DAYCENT model is used to simulate the SOC stocks at the beginning and end of each year for Equation 3-5 based on recent management practices for a land parcel. Initial values for DAYCENT are needed for the SOC_{t-1} and are based on a simulation of historical management to provide accurate stocks and distribution of organic carbon among the pools represented in the model (active, slow, and passive soil organic matter pools). Each pool has a different turnover rate (representing the heterogeneous nature of soil organic matter), and the amount of carbon in each pool at any point in time influences the forward trajectory of the total soil organic carbon storage (Parton et al., 1987). By simulating the historical land use, the distributions of carbon in active, slow, and passive pools are estimated in an unbiased way.

Three steps are required to estimate the initial values. The first step involves running the model to a steady-state condition (e.g., equilibrium) under native vegetation, historical climate data, and the soil physical attributes for the land parcel. The second step is to simulate period of time from the 1800's to 1980 and 1980 to 2000. The entity is provided a list of options for selecting the practices that best match the land management for the parcel. From 2000 to the initial year for reporting, the entity enters more specific data on crops planted, tillage practices, fertilization practices, irrigation, and other management activity (See Section 3.5.3.3 for more information). The simulated carbon stock at the end of the simulation provides the initial baseline value (SOC_{t-1}).

The stock at the end of a year (SOC_t) is estimated by the DAYCENT model based on simulating management activity during the specific year. The entity provides the management activity for the land parcel, including crops planted, tillage practices, fertilization practices, irrigation and other management activity data (See Section 3.5.3.3 for more information). The change in SOC stocks are estimated for additional years by using the ending stock from the previous year as the initial SOC stock (SOC_{t-1}) and then simulating the management for another year to produce the stock at the end of the next year (SOC_t).

Eroded carbon is estimated with the RUSLE2 for water erosion (USDA, 2003) and WEPS for wind erosion (USDA, 2004). Neither the deposition of carbon on the site nor the fate of eroded carbon is in this version of the USDA methods. The eroded carbon estimate is reported separately to account for uncertainty associated with the potential effect of erosion on SOC stocks, and may be used as a discount for the SOC stock changes estimate with Equation 3-5.

The DAYCENT model is not able to estimate soil organic carbon stocks in mineral soils for all crops. In instances where a crop is not estimated by the DAYCENT model, the method developed by the IPCC (2006) (i.e., a Tier 1 methodology) may be used (See Appendix 3-B).

Organic Soils: The methodology for estimating soil carbon stock changes in drained organic soils has been adopted from IPCC (Aalde et al., 2006). The method applies to Histosols and soils that have high organic matter content and developed under saturated, anaerobic conditions for at least part of the year, which includes Histels, Historthels, and Histoturbels. The following equation is used to estimate emissions from a land parcel:

Equation 3-6: Change in Soil Organic Carbon Stocks for Organic Soils

$$\Delta C_{Organic} = A \times EF \times CO_2MW$$

Where:

$\Delta C_{Organic}$ = Annual CO_2 emissions from drained organic soils in crop and grazing lands (metric tons CO_2-eq year^{-1})

A = Area of drained organic soils (ha)

EF = Emission factor (metric tons C ha^{-1} year^{-1})

CO_2MW = Ratio of molecular weight of CO_2 to C (= 44/12) (metric tons CO_2 (metric tons C)$^{-1}$)

Emission factors have been adopted from Ogle et al. (2003) and are region-specific, based on typical drainage patterns and climatic controls on decomposition rates; these rates are also used in the U.S. national GHG inventory (U.S. EPA, 2011). Drained cropland soils lose carbon at a rate of 11±2.5 metric tons C ha^{-1} year^{-1} in cool temperate regions, 14±2.5 metric tons C ha^{-1} year^{-1} in warm temperate regions, and 14±3.3 metric tons C ha^{-1} year^{-1} in subtropical climate regions. Organic soils in grazing lands are typically not drained to the depth of cropland systems, and therefore the emission factors are only 25 percent of the cropland values (Ogle et al., 2003).

3.5.3.3 Activity Data

The activity data requirements vary between mineral soils and organic soils. Mineral soils require the following activity data for croplands:

- Area of land parcel (i.e., field);
- Crop selection and rotation sequence;
- Planting and harvesting dates;
- Residue management, including amount harvested, burned, grazed, or left in the field;
- Irrigation method, application rate, and timing of water applications;
- Mineral fertilizer type, application rate, and timing of application(s);
- Lime amendment type, application rate, and timing of application(s);
- Organic amendment type, application rate, and timing of application(s);
- Tillage implements, dates of operation, and number of passes in each operation (which can be used to determine tillage intensity with the STIR Model (USDA NRCS, 2008));
- Use of drainage practices and depth of drainage (commonly in hydric soils); and
- Cover crop types, planting, and harvesting dates (if applicable).

The method for grazing land on mineral soils requires the following management activity data:

- Area of land parcel (i.e., field);
- Plant species composition;
- Periods of grazing during the year;
- Animal type, class, and size used for grazing;
- Stocking rates and methods;
- Irrigation method, application rate, and timing of water applications;
- Mineral fertilizer type, application rate, and timing of application(s);
- Lime amendment type, application rate, and timing of application(s);
- Organic amendment type, application rate, and timing of application(s);
- Pasture/Range/Paddock (PRP) N excreted directly onto land by livestock (i.e., manure that is not managed);

- Use of drainage practices and depth of drainage (commonly in hydric soils);
- Level of woody plant encroachment; and
- Total yield of crop (metric tons dry matter crop yield year[-1]), meat (kg carcass yield year[-1]) or milk (kg fluid milk year[-1]).

Longer-term history of site management will be used to simulate initial soil organic carbon stocks for the crop or grazing system. In order to estimate the initial values, the entity will need to provide management activity data for the past three decades. A list of management systems will be provided. The entity will also provide the previous land use and year of conversion if a land-use change occurred during the past three decades. Historical data for activity from more than three decades in the past will be represented based on national agricultural statistics using enterprise budgets and census data for various regions in the country. However, an entity can provide the longer term history if it is known. Data on the carbon and nitrogen content of organic amendments will also be needed from the entity, although defaults are provided below if the entity does not have this information. Pasture/Range/Paddock (PRP) manure N input is the N excreted directly onto land by livestock, and the manure is not collected or managed (de Klein et al., 2006). The amount of PRP manure N is estimated with the livestock methods (See Chapter 5, Section 5.3.2 *Enteric Fermentation and Housing Emissions from Beef Production Systems*) and assumed to be split with 50% of the N in urine and the other 50% of the N in solids.

Table 3-7: Nitrogen and Carbon Fractions of Common Organic Fertilizers – Midpoint and Range (Percent by Weight)

Organic Fertilizer	% N[a]	% C
Poultry manure	2.25% (1.5-3)	8.75% (7-10.5)[b]
Pig, horse, cow manure	0.45% (0.3-0.6)	5.1% (3.4–6.8)[c]
Green manure	3.25% (1.5-5)	42%[d]
Compost	1.25% (0.5-2)	16% (12-20)[e]
Seaweed meal	2.5% (2-3)	27%[f]
Sewage sludge	3% (1-5)	11.7% (3.9-19.5)[b]
Fish waste	7% (4-10)	24.3% (14.6-34)[g]
Blood	11% (10-12)	35.2% (32-38.4)[h]
Human urine/night soil	1.25% (1-1.5)	9.5% (9-10)[i]

[a] Hue, N.V. *Organic Fertilizers in Sustainable Agriculture* Retrieved from http://www.ctahr.hawaii.edu/huen/hue_organic.htm.
[b] USDA. 1992. Agricultural Waste Characteristics. Chapter 4. In Animal Waste Management Field Handbook: Natural Resources Conservation Service, United States Department of Agriculture.
[c] EPA, 2013. Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2011. Weighted U.S. average carbon:nitrogen ratio for manure available for application.
[d] Assumes dry matter is 42% carbon.
[e] A1 Organics. *Compost Classification, Specification and Resource Manual.* http://www.a1organics.com/CLSP/CLASS%20MANUAL%20-%20COLORADO.pdf
[f] http://www.naorganics.com/en/science_analysis.asp. North Atlantic Organics.
[g] Hartz, T.K. and P.R. Johnstone. 2006. Nitrogen available from high-nitrogen-containing organic fertilizers. HortTechnology 16:39-42.
[h] Sonon, D, et al. 2012. Mineralization of high-N organic fertilizers. Clemson University.
[i] Polprasert, C. 2007. Organic Waste Recycling: Technology and Management. IWA Publishing.

The method for organic soils requires the following activity data for croplands and grazing lands:

- Area of land parcel (i.e., field); and
- Total yield of crop (metric tons dry matter crop yield year[-1]), meat (kg carcass yield year[-1]) or milk (kg fluid milk year[-1]).

3.5.3.4 Ancillary Data

Ancillary data for the mineral soil method include historical weather patterns and soil characteristics. Weather data may be based on national datasets such as the Parameter-Elevation Regressions on Independent Slopes Model (PRISM) data (Daly et al., 2008). Soil characteristics may also be based on national datasets such as the Soil Survey Geographic Database (SSURGO) (Soil Survey Staff, 2011). However, there will also be an option for entities to substitute soils data collected from the specific field. The erosion model will also require ancillary data on topography (i.e., slope), length of field and row orientation, crop canopy height, diversions, surface residue cover, and soil texture.

No ancillary data are needed for the method to estimate emissions from drainage of organic soils.

3.5.3.5 Model Output

Model output is generated for the quantity of emissions and emissions intensity. The change in mineral soil organic carbon stocks is estimated based on stock changes over five-year time periods in order to manage uncertainty. Uncertainties in the model-based estimates are about three times larger for annual estimates in change rate compared with five-year blocks (Compare U.S. Environmental Protection Agency (2009) and (2010)). Uncertainties are larger at the finer time scale because there is large variability in measurements of soil carbon stock changes at annual time scales, and this variability is incorporated into the model uncertainty using the empirically based method (Ogle et al., 2007). In addition, trends in soil organic carbon will be estimated for the 30 previous years of history and the reporting period.

Emissions intensity is based on the amount of emissions per unit of yield for crops in cropland systems or animal products in grazing systems. The emissions intensity is estimated with the following equation:

Equation 3-7: Emissions Intensity of Soil Organic Carbon Stock Change

$$EI_{SoilC} = (\Delta C_{Mineral} + \Delta C_{Organic})/Y$$

Where:

EI_{SoilC} = Emissions intensity (metric tons CO_2 per metric ton dry matter crop yield, metric tons CO_2 per kg carcass yield, metric tons CO_2 per kg fluid milk yield)

$\Delta C_{Mineral}$ = Annual CO_2 equivalent emissions from soil organic carbon change in mineral soils (metric tons CO_2-eq year^{-1})

$\Delta C_{Organic}$ = Annual CO_2 equivalent emissions from soil organic carbon change in organic soils, Histosols (metric tons CO_2-eq year^{-1})

Y = Total yield of crop (metric tons dry matter crop yield year^{-1}), meat (kg carcass yield year^{-1}) or milk production (kg fluid milk yield year^{-1})

3.5.3.6 Limitations and Uncertainty

Uncertainties in the mineral soil methods include imprecision and bias in the process-based model parameters and algorithms, in addition to uncertainties in the activity and ancillary data. Uncertainty in the parameterization and algorithms will be quantified with an empirically based approach, as used in the U.S. national GHG inventory (Ogle et al., 2007; U.S. EPA, 2011). The method combines modeling and measurements to provide an estimate of SOC stock changes for entity scale reporting (Conant et al., 2011). Measurements of carbon stock changes are expected to be based on

a national soil monitoring network (Spencer et al., 2011). The network should include samples from different regions of the country and soil types that are used for crop production or grazing, and a random sampling of the management systems in each of the regions. The sampling plots will need to be designed for resampling over time in order to evaluate the modeled changes in SOC stocks (Conant et al., 2003). Uncertainties in national datasets for weather will be based on information included with the dataset, while uncertainties in the SSURGO should be quantified using the underlying field data that form the basis for the mapping exercise, or an independent accuracy assessment of the map product. Other input data is assumed to be known by the entity, such as the crop plants, yields, tillage, and residue management practices.

The limitations of the mineral soil carbon method include no assessment of the effect of land use and management in sub-surface layers of the soil profile (below 30 centimeters), no assessment of the location of transport and deposition of eroded C, and limited data to assess uncertainty in the parameters and algorithms using the empirically based method. For agroforestry, the DAYCENT model has been used in the COMET-Farm voluntary carbon reporting tool to simulate soil organic carbon stock changes. However, there are several unknowns with the use of the DAYCENT model for estimating soil organic carbon stock changes in agroforestry, including whether the model is able to take into account the interactions occurring between woody and herbaceous vegetation and respective management activities. Oelbermann and Voroney (2011) evaluated the use of the Century model, the monthly time-step version of the DAYCENT model, to predict soil organic carbon in temperate and tropical alley cropping systems that were 13 and 19 years old, respectively. They found that the model underestimated the levels of soil organic carbon compared with measured values. With more testing, the methods may be revised in the future to use the DAYCENT model for the purposes of estimating soil organic carbon stock changes in agroforestry systems.

Biochar research has been an area of rapid development over the past few years, but there are still uncertainties. Biochar is a product of combusted biomass that has a variety of chemical structures depending on the biomass and pyrolysis method, and the variation has implications for the stability of the carbon in the soil (Spokas, 2010). Biochar can have concomitant impacts on emissions of other GHGs such as CH_4 and N_2O (Cayuela et al., 2010; Malghani et al., 2013; Yu et al., 2013), although some studies have shown no effect (Case et al., 2013; Clough et al., 2010). Soil amendments with biochar may also prime the decomposition of the native soil organic matter although the CO_2 emissions from priming appear to be considerably smaller than the carbon added in the biochar (Stewart et al., 2013; Woolf and Lehmann, 2012). Other research suggests that there may even be "negative" priming leading to a reduction in heterotrophic respiration (Case et al., 2013). Furthermore, the temporal duration of the GHG mitigation potential of biochar is also uncertain but appears to be of a short term nature (Spokas, 2013). The influence of biochar on emissions and priming needs more research before the full effect of biochar on carbon sequestration and GHG emissions can be incorporated into models and GHG reporting frameworks. Microbial degradation of biochar can occur over time scales ranging from as little as a few decades to 1000s of years (Spokas, 2010). In the technical methods, biochar is treated as a high carbon to low nitrogen amendment in the DAYCENT model framework, but with a conservative residence time of the carbon from decades to a century. These methods can be further refined in the future as the different types and residence times of biochar are further resolved.

The method for organic soils also has limitations, particularly the inability to estimate the effect of mitigation measures such as water table management because emission factors are set for each climate region (i.e., currently scaling factors are not available to revise the emission factors for water table management). Only complete restoration of the wetland with no further drainage can be addressed with the method (i.e., assumes no further emissions of CO_2). However, if crop

production is maintained on the land parcel, the most practical method for reducing emissions is to raise the water table to near the rooting depth of the crop during the growing season and then not draining the soil during the non-growing season (Jongedyk et al., 1950; Shih et al., 1998), or possibly managing the system with periodic flooding (Morris et al., 2004).

For all systems there is additional uncertainty associated with climate change. Modeled output for any given location assumes temperature and precipitation similar to that of the past 30 years, the period for which historical weather is used to simulate soil organic carbon dynamics. Expected changes in temperature, precipitation, and extreme events such as droughts, floods, and heat waves will add further uncertainty to estimates of soil organic carbon stock change.

While there is considerable evidence and mechanistic understanding about the influence of land use and management on SOC, there is less known about the effect on soil inorganic C. Consequently, there is uncertainty associated with land use and management impacts on soil inorganic carbon stocks, which cannot be quantified. Current methods do not include impacts on inorganic C, but this may be added in the future as more studies are conducted and methods are developed.

Uncertainties in model parameters and structure are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity. Table 3-8 provides the probability distribution functions associated with the mineral and organic soils methods.

Table 3-8: Available Uncertainty Data for Soil Organic Carbon Stock Change

Parameter	Mean	Units	Relative Uncertainty		Distribution	Data Source
			Low (%)	High (%)		
DAYCENT (empirical uncertainty)	NS	Various	NS	NS	Normal	Ogle et al. (2007); EPA (2013)
Emission factor for cropland in cool temperate regions	11	metric tons C ha^{-1} year^{-1}	45	45	Normal	Ogle et al. (2003)
Emission factor for cropland in warm temperate regions	14	metric tons C ha^{-1} year^{-1}	35	35	Normal	Ogle et al. (2003)
Emission factor for cropland in subtropical regions	14	metric tons C ha^{-1} year^{-1}	46	46	Normal	Ogle et al. (2003)
Emission factor for grazing land in cool temperate regions	2.8	metric tons C ha^{-1} year^{-1}	45	45	Normal	Ogle et al. (2003)
Emission factor for grazing land in warm temperate regions	3.5	metric tons C ha^{-1} year^{-1}	35	35	Normal	Ogle et al. (2003)
Emission factor for grazing land in subtropical regions	3.5	metric tons C ha^{-1} year^{-1}	46	46	Normal	Ogle et al. (2003)

NS = Not Shown. Data are not shown for parameters that have 100's to 1000's of values (denoted as NS).

3.5.4   Soil Nitrous Oxide

Method for Estimating Soil Direct N$_2$O Emissions

Mineral Soils

- The method is based on using results from process-based models and measured N2O emissions in combination with scaling factors based on U.S. specific empirical data on a seasonal timescale.
- Process-based modeling (an ensemble approach using DAYCENT and DNDC) combined with field data analysis are used to derive base emission rates for the major cropping systems and dominant soil texture classes in each USDA Land Resource Region. In cases where there are insufficient empirical data to derive a base emission rate, the base emission rate is based on the IPCC default factor.
  The base emission factors are adjusted by scaling factors related to specific crop management practices that are derived from experimental data.

Organic Soils

- Direct N$_2$O emissions from drainage of organic soils uses the IPCC equations developed in de Klein et al., (2006). The method for organic soils assumes that there is still a significant organic horizon in the soil, and therefore, there are substantial inputs of nitrogen from oxidation of organic matter.
- The emission rate for drained organic soils is based on IPCC Tier 1 emission factor (0.008 metric tons N$_2$O-N ha^{-1} year^{-1}).
- This method relies on entity specific activity data as input into the equations.

Method for Estimating Soil Indirect N$_2$O Emissions

- This method uses the IPCC equation for indirect soil N$_2$O (de Klein et al., 2006).
- IPCC defaults are used for estimating the proportion of nitrogen that is subject to leaching, runoff, and volatilization. In land parcels where the precipitation plus irrigation water input is less than 80 percent of the potential evapotranspiration, nitrogen leaching and runoff are considered negligible and no indirect N$_2$O emissions are estimated from leaching and runoff.
- This method uses entity specific seasonal data on nitrogen management practices.

3.5.4.1   Rationale for Selected Method

N$_2$O fluxes are notoriously difficult to measure because of the labor required to sample emissions, combined with high spatial and temporal variability. Agronomic practices that affect N$_2$O fluxes in one soil, climate, or site-year may have little or no measurable effect in others. Consequently, considerable care is required to ensure that methods to estimate changes in emissions for a particular cropping practice are accurate and robust for the geographic region for which they are proposed, or are sufficiently generalizable to be accurate in aggregate.

De Klein et al. (2006) provide three estimation strategies for direct N$_2$O emissions from cropland. Two are based on emission factors, the proportion of nitrogen added to a crop that becomes N$_2$O. Tier 1 is based on a near-universal emission factor, applicable globally without regard to geography, cropping practice, or fertilizer placement, timing, or formulation. Tier 2 methods utilize

geographic, crop, or practice-specific emission factors where field tests show that a factor different from the one percent Tier 1 factor is warranted. At present there is only one Tier 2 example in the primary literature that is specific to conditions in the United States, and it is for corn in the North Central region (Millar et al., 2010). This method has been incorporated into several N_2O reduction protocols (Verified Carbon Standard, American Carbon Registry, and Climate Action Reserve). The third option for estimating direct N_2O emissions, or Tier 3, is a measurement or process-based modeling approach. In this case, emissions are monitored specifically for the entity's field by deploying instruments in a measurement system or by gathering the information specific to the field conditions to simulate N_2O emissions with a process-based model. This third option is the most precise, but requires more resources and sufficient testing prior to implementation.

In Section 3.2.1.1, several practices are discussed that have been shown to reduce N_2O emissions in field experiments. However, many of the experiments have been conducted for a limited number of specific cropping systems and regions. Consequently, there are no mitigation practices for which emission reductions have been quantified under all conditions in the United States. Nevertheless, for many practices there is sufficient knowledge at the cropping system and regional levels to establish that adoption will reduce soil N_2O emissions.

Process-based simulation models use knowledge of C, N, and water processes (among others) to predict ecosystem responses to climate and other environmental factors, including crop and grazing land management (see soil carbon methodology in Section 3.5.3). N_2O fluxes can be predicted using simulation models (Chen et al., 2008; Del Grosso et al., 2010). A key advantage of simulation models is that they are generalizable to a wide variety of soils, climates, and cropping systems, allowing factors to interact in complex ways that may be difficult to predict with less sophisticated approaches. However, a disadvantage is that complexity can limit their transparency, and at present there are still substantial data gaps that limit our ability to fully test available models for their sensitivity to different management practices across various regions and crops in the United States.

To overcome these challenges, a hybrid approach that utilizes process-based simulation models and field data was developed to estimate N_2O emissions. The method uses a base emission rate associated with the typical amount of nitrogen applied, and then adjustments are applied via scaling factors to account for management practices that affect N_2O emissions. This approach is a Tier 3 method as defined by the IPCC.

Base emission rates are estimated for each dominant crop and three soil texture classes (coarse, medium, fine) within a climatic region using process-based simulation modeling. The factors are developed at the scale of USDA Land Resource Regions (LRR). Field data indicate that N_2O emissions generally increase as the amount of applied nitrogen increases, especially when nitrogen application rates exceed crop uptake rates (Hoben et al., 2011; Kim et al., 2013; McSwiney and Robertson, 2005; Shcherbak et al., in press) Research data from field experiments were compiled and used to adjust the emission rates for nitrogen fertilizer application rates that exceeded the typical nitrogen application rate for the crop in a land resource region. For crops where sufficient data are not available to simulate the base emission rate with a process-based model, the standard IPCC Tier 1 emission factor is applied. In addition, for land parcels that have a mix of crops where only some can be simulated, the standard IPCC Tier 1 approach should also be applied.

Emissions are affected by specific farm management practices such as reducing tillage intensity; adding nitrification inhibitors, or changing how, when and where nitrogen fertilizers are applied. To account for the effect of management practices on N_2O emission, scaling factors were developed to adjust the base emission rates. The scaling factors were estimated from available research data (See Appendix 3-A for more information). Management practices other than those included in the

equation may also mitigate N_2O emissions, but there are not currently sufficient data to create generalized scaling factors. Additional data may lead to their inclusion in future updates to the method.

This method incorporates more information than a method based solely on the IPCC model. It provides a transparent and science-based means of estimating annualized N_2O emissions from crop and grazing lands, and it facilitates the estimation of uncertainty. For N_2O emissions from crop and grazing lands, an IPCC Tier 1 approach is only sensitive to nitrogen application rate, and therefore does not reflect the full suite of factors that are known to influence N_2O emissions including climate, soils, crops, and management practices that range from tillage to cover crops to fertilizer timing, placement, formulation, and additives. Dynamic process models as embodied in the IPCC Tier 3 approach can, in concept, account for most of these factors but to date have not been sufficiently evaluated for many U.S. locations, crops, and management practices. This report takes a hybrid approach that represents the best available science at the time of publication: dynamic process models to estimate baseline N_2O emissions for those crops and locations sufficiently evaluated, then scaled by management practices to the extent supported by available research results. Initial testing indicates that this method is more sensitive to U.S. nutrient management practices than the IPCC Tier 1 approach. The authors anticipate publication of an addendum that will provide test results and suggest further tuning of the method. Over time, as dynamic process models are further developed and tested. The method will likely migrate towards an exclusive Tier 3 approach to better account for management effects given the local variables and conditions. In the interim, in addition to providing best-available and reliable estimates of N_2O emissions from crop and grazing lands, the method outlined here is expected to set a research agenda that provides for broader evaluation of environmental conditions and management practices influencing N_2O emissions as well as further development of models to more accurately estimate emissions.

Offsite or indirect N_2O emissions, which occur when reactive nitrogen escapes to downwind or downstream ecosystems where favorable conditions for N_2O production exist, are even more difficult to estimate than direct emissions because there is uncertainty in both the amount of reactive nitrogen that escapes and the portion of this nitrogen that is converted to N_2O. Ideally, fluxes of volatile and soluble reactive nitrogen leaving the entity's parcel of land would be combined with atmospheric transport and hydrologic models to simulate the fate of reactive N. At present there are no linked modeling approaches sufficiently tested to be used in an operational framework. Consequently, the indirect N_2O emissions are based on the IPCC Tier 1 method (de Klein et al., 2006).

Similarly, direct N_2O emissions from drainage of organic soils are based on the IPCC Tier 1 methods (de Klein et al., 2006). Although research is ongoing to provide improved emission factors and methods for estimating N_2O emissions from drainage of organic soils (Allen, 2012), more testing will be needed before incorporating them into an operational method. Future revisions to these methods will need to consider advancements and revise the methods accordingly.

3.5.4.2 Description of Method

N_2O is emitted from cropland both directly and indirectly. Direct emissions are fluxes from cropland or grazing lands where there are nitrogen additions or nitrogen mineralized from soil organic matter. Indirect emissions occur when reactive nitrogen is volatilized as NH_3 or NO_x or transported via surface runoff or leaching in soluble forms from cropland or grazing lands, leading to N_2O emissions in another location.

Direct N_2O Emissions

Mineral Soils: Total direct N_2O emissions from mineral soils are estimated for a land parcel using Equation 3-8.

Equation 3-8: Direct Soil N_2O Emissions from Mineral Soils

$$N_2O_{Direct} = ER_p \times A \times N_2O_{MW} \times N_2O_{GWP}$$

Where:

N_2O_{Direct} = Total direct soil N_2O emission for parcel of land (metric tons CO_2-eq year^{-1})

ER_p = Practice-scaled emission rate for land parcel (metric tons N_2O-N ha^{-1} year^{-1})

A = Area of parcel of land (ha)

N_2O_{MW} = Ratio of molecular weights of N_2O to N_2O-N
= 44/28 (metric tons N_2O (metric tons N_2O-N)$^{-1}$)

N_2O_{GWP} = Global warming potential for N_2O (metric tons CO_2-eq (metric tons N_2O)$^{-1}$)

The practice-scaled emission rate for the parcel of land (ER_p) is estimated using Equation 3-9.

Equation 3-9: Practice-Scaled Soil N_2O Emission Rate for Mineral Soils

$$ER_p = [ER_b + (\Delta N_{prp} * EF_{prp})] \times \{1 + [S_{sr} \times (N_{sr}/N_i)]\} \times \{1 + [S_{inh} \times (N_{inh}/N_i)]\} \times (1 + S_{till}) \times \{1 - [N_{residr}/(N_i + N_{residr})]\}$$

Where:

ER_p = Practice-scaled emission rate for land parcel (metric tons N_2O-N ha^{-1} year^{-1})

ER_b = Base emission rate for crop or grazing land that varies based on nitrogen input rate from mineral fertilizer, organic amendments, residues, and additional mineralization with land-use change or tillage change (metric tons N_2O-N ha^{-1} year^{-1})

ΔN_{prp} = Difference in PRP manure N excretion[a] between the PRP manure N excretion based on entity activity data (N_{PRPe}) and PRP manure N excretion for the base emission rate (N_{PRPb}) (metric tons N)

= N_{PRPe} - N_{PRPb}

EF_{prp} = Emission factor for PRP manure N input to soils, 0.02 metric tons N_2O-N ha^{-1} year^{-1} (metric tons N)$^{-1}$ for cattle, poultry and swine, and 0.01 metric tons N_2O-N (metric tons N)$^{-1}$ for other livestock[b]

N_i = Nitrogen inputs, including mineral fertilizer, organic amendments, PRP manure N, residues, and SOM mineralization (See Equation 3-11) (metric tons N ha^{-1} year^{-1})

S_{sr} = Scaling factor for slow-release fertilizers, 0 where no effect (dimensionless)

N_{sr} = Nitrogen in slow-release nitrogen fertilizer applied to the parcel of land (metric tons N ha^{-1} year^{-1})

S_{inh} = Scaling factor for nitrification inhibitors, 0 where no effect (dimensionless)

N_{inh} = Nitrogen in nitrogen fertilizer with inhibitor applied to the parcel of land (metric tons N ha^{-1} year^{-1})

S_{till} = Scaling factor for no-tillage, 0 except for NT (dimensionless)

N_{residr} = N removed through collection, grazing, harvesting or burning of aboveground residues (metric tons N ha^{-1} year^{-1}). Estimate using Equation 3-10 for results generated with DAYCENT and DNDC models with the exception of hay crops. No calculation is needed for results generated by the IPCC method or for results associated with hay crops generated by DAYCENT and DNDC (set value to 0).

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[a] A difference arises in the ER_b estimation of PRP manure N input and the actual PRP manure N input because a typical rate of N input was assumed in the DAYCENT and DNDC simulations for the ER_b calculation (See Textbox 3-1 and Appendix 3-A).
[b] Emission factors from de Klein et al. (2006).

In this equation, the base emission rate (ER_b) varies by the amount of nitrogen input to the soil. The rate may also vary for different crop and grazing land systems by LRR to capture variation in climate, and by texture class in order to represent the influence of soil heterogeneity on N_2O emissions. More information about base emission rates is given in Text box 3-1.

Practice-based emission scaling factors (0 to 1) are used to adjust the portion of the emission rate associated with slow release fertilizers (S_{sr}), nitrification inhibitors (S_{inh}), and pasture/range/paddock (PRP) manure nitrogen additions ($S_{prp,cps}$). The slow-release fertilizer,

nitrification inhibitor and PRP manure scaling factors are weighted so that their effect is only on the amount of nitrogen influenced by these practices relative to the entire pool of nitrogen (i.e., the amount of slow-release fertilizer, fertilizer with nitrification inhibitor or PRP manure nitrogen added to the soil). In contrast, scaling factors for tillage (S_{till}) are used to scale the entire emission rate under the assumption that this practice influences the entire pool of mineral nitrogen inputs (i.e., N_i).

Equation 3-10: Aboveground Residue N Removal

For Crops:

$$N_{residr} = [((Y_{dm} / HI) - Y_{dm}) \times R_r) \times N_a]$$

For Grazing Forage:

$$N_{residr} = [Y_{dm} \times (F_r + R_r) \times N_a]$$

Where:

N_{residr} = N removed through collection, grazing, harvesting or burning of aboveground residues (metric tons N ha^{-1} year^{-1})

Y_{dm} = Crop harvest or forage yield, corrected for moisture content (metric tons biomass ha^{-1} year $^{-1}$)
= Y x DM

Y = Crop harvest or total forage yield (metric tons biomass ha^{-1} year $^{-1}$)

DM = Dry matter content of harvested biomass (dimensionless)

HI = Harvest Index (dimensionless)

F_r = Proportion of live forage removed by grazing animals (dimensionless)

R_r = Proportion of crop/forage residue removed due to harvest, burning or grazing (dimensionless)

N_a = Nitrogen fraction of aboveground residue biomass for the crop or forage (metric tons N (metric tons biomass)$^{-1}$)

Table 3-9: Scaling Factors for Nitrogen Management Practices

Management Practice	Nitrogen Management Factor	Factor (Proportional Change in Emissions)	Source
Slow-release fertilizer use	S_{sr}	-0.21 (-0.12 to -0.30)	See Appendix 3-A
Manure nitrogen directly deposited on pasture/range/paddock	$S_{prp,cps}$	+0.5 (0.33 to 0.67)	IPCC (2006)
Nitrification inhibitor use	S_{inh} – semi arid/arid climate	-0.38 (-0.21 to -0.51)	See Appendix 3-A
Nitrification inhibitor use	S_{inh} – mesic/wet climate	-0.40 (-0.24 to -0.52)	See Appendix 3-A
Tillage	S_{till} – semi arid/arid climate (< 10 years following no-till adoption)	0.38 (0.04 to 0.72)	van Kessel et al. (2012), Six et al. (2004)
Tillage	S_{till} – semi arid/arid climate (≥ 10 years following no-till adoption)	-0.33 (-0.16 to -0.5)	van Kessel et al. (2012), Six et al. (2004)

Management Practice	Nitrogen Management Factor	Factor (Proportional Change in Emissions)	Source
Tillage	S_{till} – mesic/wet climate (< 10 years following no-till adoption)	-0.015 (-0.16 to 0.16)	van Kessel et al. (2012), Six et al. (2004)
Tillage	S_{till} – mesic/wet climate (≥ 10 years following no-till adoption)	-0.09 (-0.19 to 0.01)	van Kessel et al. (2012), Six et al. (2004)

Note: See Appendix 3-A for further explanation on the practices included in the soil N_2O method and the sources of data that were used to derive the base emission rates and scaling factors for the management practices.

Text box 3-1: Base Emission Rate for Direct Soil N_2O Emissions from Mineral Soils

The base emission rate is a crop or grazing land specific estimate that varies based on the total mineral nitrogen input to the soil. There are two methods used to estimate the base emission rate. The first method uses a combination of process-based modeling and measurement data to estimates N_2O base emission rates by land resource region, major crop type, and soil texture class. The second method uses the default IPCC emission factor of one percent (de Klein et al., 2006), multiplying this value by the total nitrogen input (See Equation 3-11) to estimate the base emission rate. The second approach is used for crops that are not included in the process-based modeling analysis.

The remainder of this box describes the first method. The equation for the first method, combining the modeling and measurement data, is given below:

$$ER_b = ER_0 + (EF_{typical} + (S_{EF} \times \Delta N_f)) \times N_f$$

ER_b = Base emission rate (metric tons N_2O-N ha^{-1} year^{-1})

ER_0 = Emission rate modeled at 0 level of nitrogen input (N_t =0)
        (metric tons N_2O-N ha^{-1} year^{-1})

$EF_{typical}$ = Emission factor for the typical fertilization rate
        (metric tons N_2O-N (metric tons N)$^{-1}$)
        = $(ER_{typical} - ER_0)/ N_{tf}$

$ER_{typical}$ = Emission rate for the typical case modeled (metric tons N_2O-N ha^{-1} year^{-1})

S_{EF} = Base EF scalar;
        for ΔN_f > zero: S_{EF} = 0.0274 for all non-grassland crops,
        S_{EF} = 0.117 for grasslands;
        for $\Delta N_f \leq$ zero (less than or the same as typical fertilizer rates): S_{EF} = 0;
        ((metric tons N_2O-N (metric tons N)$^{-2}$) ha year)

ΔN_f = N_f - N_{tf} (metric tons N ha^{-1} year^{-1})

N_f = Actual nitrogen fertilizer rate, including synthetic and organic
        (metric tons N ha^{-1} year^{-1})

N_{tf} = Typical nitrogen fertilizer rate (metric tons N ha^{-1} year^{-1})

Process-based models were used to simulate N_2O emissions at the typical nitrogen fertilization rate for major commodity crops according to the USDA Agricultural Resource Management Survey data ($ER_{typical}$), in addition to a zero rate application (ER_0). The N_2O emission at the typical rate of fertilization for major commodity crops are produced for coarse, medium, and fine textured soils in each land resource region. The emission factor ($EF_{typical}$) for fertilization rates greater than the typical rate for the crop or grass are scaled according to the trend in measured soil N_2O data across a range of fertilization rates based on experimental data. The change in the emission factor between the typical nitrogen fertilization rate and a higher rate was averaged to derive an emission factor scalar or rate of change per unit of additional N. The scalar is multiplied by the additional nitrogen to derive an adjustment to the emission factor ($S_{EF} \times \Delta N_f$) that is then added to the emission factor derived for the typical fertilizer rate ($EF_{typical}$). No scaling is done for the case where $\Delta N_f \leq$ zero, i.e., where the fertilization rate is equal to or less than the typical rate of nitrogen application. In this case S_{EF} = 0 such that $S_{EF} \times \Delta N_f$ = 0. The resulting emission factor is multiplied by the actual fertilizer rate (N_f) and added to the emission rate at the 0 level of nitrogen fertilization (ER_0) to derive the base emission rate (ER_b).

Nitrogen inputs are estimated with the following equation:

Equation 3-11: Nitrogen Inputs[a]

$$N_i = N_{sfert} + N_{man} + N_{comp} + N_{resid} + N_{smin} + N_{prp}$$

Where:

N_i = Nitrogen inputs, including mineral fertilizer, organic amendments, PRP manure N, residues, and SOM mineralization (metric tons N ha^{-1} year^{-1})

N_{sfert} = Nitrogen in synthetic fertilizer applied to a parcel of land (metric tons N ha^{-1} year^{-1})

N_{man} = Nitrogen mineralization from manure amendments (or sewage sludge) applied to a parcel of land (metric tons N ha^{-1} year^{-1})

N_{comp} = Nitrogen mineralization from compost applied to a parcel of land (metric tons N ha^{-1} year^{-1})

N_{resid} = Nitrogen mineralization from crop and cover crop residues above and belowground that are left on the parcel of land following senescence (i.e., not collected, grazed, or burned) (metric tons N ha^{-1} year^{-1})

N_{smin} = Nitrogen inputs from soil organic matter mineralization as estimated by the DAYCENT mineral soil C method (See Section 3.5.3.2) (metric tons N ha^{-1} year^{-1}). Value set to 0 for crops that are not estimated with the DAYCENT mineral soil C method.

N_{prp} = Nitrogen in urine and mineralization from solids associated with manure in pasture/range/paddock (PRP) (metric tons N ha^{-1} year^{-1})[b]

[a] The approach for estimating nitrogen mineralization inputs is consistent with the U.S. National Inventory (U.S. EPA, 2013).
[b] Pasture/Range/Paddock (PRP) manure N is a term utilized by the IPCC (de Klein et al., 2006) for the N excreted directly onto land by livestock, and the manure is not collected or managed. The total PRP manure N is estimated with methods from Chapter 5, and assumed to be split with 50% of the N in urine and 50% of the N in solids.

The total N mineralization is estimated from the DAYCENT mineral soil C method in aggregate for manure amendments (N_{man}), compost (N_{com}), residues (N_{res}), soil organic matter (N_{smin}) and solids associated with PRP manure, and is used to approximate these N inputs in Equation 3-11. This approach creates a linkage between the mineral soil C method (See Section 3.5.3.2) and the N_2O method, ensuring consistency in treatment of N. In instances where crops cannot be estimated by the DAYCENT mineral soil C method, the method from the IPCC guidelines (Aalde et al., 2006) can be used to estimate the N inputs from mineralization with the exception of N_{smin}, which is set to 0 (See Appendix 3-B).

Organic Soils: The method for organic soils includes Histosols and soils that have high organic matter content and developed under saturated, anaerobic conditions for at least part of the year, which includes Histels, Historthels, Histoturbels. The method assumes that there is a significant organic horizon in the soil, and therefore, major inputs of nitrogen are from oxidation of organic matter rather than external inputs from synthetic and organic fertilizers. If these assumptions are not true, then the entity should use the mineral soil method to estimate the N_2O emissions. Total direct N_2O emissions from drained organic soils are estimated for individual parcels of land (i.e., fields) with the following equation:

Equation 3-12: Direct N_2O Emissions from Drainage of Organic Soils (Histosols)

$$N_2O_{ORGANIC} = A_{OS} \times ER_{OS}$$

Where:

$N_2O_{ORGANIC}$ = Direct soil N_2O emission from drainage of organic soils
(metric tons N_2O-N year^{-1})

A_{os} = Area of organic soils drained on a parcel of land (ha)

ER_{OS} = Emission rate for cropped Histosols,
IPCC Tier 1 ER_{OS} = 0.008 metric tons N_2O-N ha^{-1} year^{-1}

Indirect N_2O Emissions: The method to estimate indirect N_2O emissions for mineral soils has been adopted from the approach developed by IPCC (de Klein et al., 2006). The following equation is used to estimate the total indirect N_2O emissions associated with nitrogen volatilization and nitrogen leaching and runoff from the land parcel:

Equation 3-13: Total Indirect Soil N_2O Emissions from Mineral Soils

$$N_2O_{Indirect} = (N_2O_{Vol} + N_2O_{Leach}) \times N_2O_{MW} \times N_2O_{GWP}$$

Where:

$N_2O_{Indirect}$ = Indirect soil N_2O emission (metric tons CO_2-eq year^{-1})

N_2O_{Vol} = N_2O emitted by ecosystem receiving volatilized nitrogen
(metric tons N_2O-N year^{-1})

N_2O_{Leach} = N_2O emitted by ecosystem receiving leached and runoff nitrogen
(metric tons N_2O-N year^{-1})

N_2O_{MW} = Ratio of molecular weights of N_2O to N_2O-N = 44/28
(metric tons N_2O (metric tons N_2O-N)$^{-1}$)

N_2O_{GWP} = Global warming potential for N_2O (metric tons CO_2-eq (metric tons N_2O)$^{-1}$)

The following equation is used to estimate the indirect emissions associated with nitrogen volatilization from the land parcel:

Equation 3-14: Indirect Soil N$_2$O Emissions from Mineral Soils —Volatilization

$$N_2O_{Vol} = [(F_{SN} \times FR_{SN}) + (F_{ON} \times FR_{ON})] \times EF_{VOL}$$

Where:

N_2O_{Vol} = Indirect soil N$_2$O emitted by ecosystem receiving volatilized nitrogen (metric tons N$_2$O-N year^{-1})

F_{SN} = Synthetic nitrogen fertilizer applied (metric tons N year^{-1})

FR_{SN} = Fraction of N$_{SN}$ that volatilizes as NH$_3$ and NO$_x$. IPCC default Tier 1 = 0.10 (metric tons N (metric ton N$_{sfert}$)$^{-1}$)

F_{ON} = Nitrogen fertilizer applied of organic origin including manure, sewage sludge, compost and other organic amendments (metric tons N year^{-1})

FR_{ON} = Fraction or proportion of F$_{ON}$ that volatilizes as NH$_3$ and NO$_x$. IPCC default Tier 1 = 0.2 (metric tons N (metric ton N$_{ON}$)$^{-1}$)

EF_{VOL} = Emission factor for volatilized nitrogen or proportion of nitrogen volatilized as NH$_3$ and NO$_x$ that is transformed to N$_2$O in receiving ecosystem; IPCC Tier 1 EF = 0.01 (metric tons N$_2$O-N (metric ton N)$^{-1}$)

The IPCC defaults are used for *FR$_{SN}$ and FR$_{ON}$.*

The following equation is used to estimate the indirect N$_2$O emissions associated with leaching or overland flow of reactive nitrogen that is transported from the land parcel (i.e., field):

Equation 3-15: Indirect Soil N$_2$O Emissions from Mineral Soils —Leaching and Runoff

$$N_2O_{leach} = (N_i \times FR_{leach}) \times EF_{leach}$$

Where:

N_2O_{leach} = Indirect soil N$_2$O emitted by ecosystem receiving leached and runoff nitrogen (metric tons N$_2$O-N year^{-1})

N_i = Nitrogen inputs, including mineral fertilizer, organic amendments, PRP manure N, residues, and SOM mineralization (metric tons N ha^{-1} year^{-1}) (See Equation 3-11)

FR_{leach} = Fraction or proportion of N$_i$ that leaches or runs off. IPCC default Tier 1 = 0.30 except a) where irrigation+precipitation is less than 80% of potential evapotranspiration (metric tons N (metric ton N)$^{-1}$) FR$_{leach}$ = 0; and b) cropping systems with leguminous or non-leguminous winter cover crops, for leguminous cover crops, FR$_{leach}$ = 0.18, and for non-leguminous cover crops, FR$_{leach}$ = 0.09.

EF_{leach} = Emission factor for leached and runoff nitrogen or proportion of leached and runoff nitrogen that is transformed to N$_2$O in receiving ecosystem; IPCC Tier 1 EF = 0.0075 (metric tons N$_2$O-N (metric ton N)$^{-1}$)

The fraction of nitrogen that is leached from a profile will vary depending on the level of precipitation and irrigation water applied in the field. In land parcels (i.e., fields) where the precipitation plus irrigation water input is less than 80 percent of the potential evapotranspiration, nitrogen leaching and runoff are considered negligible and no indirect N$_2$O emissions are estimated (U.S. EPA, 2011). IPCC default fractions are used for *EF$_{leach}$* and *FR$_{leach}$* where no cover crops are present. Where winter cover crops precede the cash crop, *FR$_{leach}$* is further adjusted to account for cover crop effects on nitrate leaching. In a meta-analysis of 36 geographically distributed field

studies, Tonitto et al. (2006) found a 40 percent and 70 percent reduction in nitrate leaching with the use of legume and non-legume cover crops, respectively. Accordingly, FR_{leach}, is reduced to 0.18 for legume cover crops (0.3 × (1-0.4); or 18% of total nitrogen inputs) and 0.09 for non-legume cover crops (0.3 × (1-0.7); or nine percent of total nitrogen inputs).

3.5.4.3 Activity Data

Calculating emissions requires the following activity data for croplands:

- Area of land parcel (i.e., field);
- Prior-year crop type, dry matter yields, and residue-yield ratios to calculate crop residue nitrogen input, including cover crop (if present);
- Residue management, including amount harvested, burned, grazed, or left in the field;
- Synthetic fertilizer type (chemical formulation) and coatings (if present);
- Synthetic and organic fertilizer application rate, application method (broadcast, banded, or injected, including depth of injection), timing of application(s);
- Type of nitrification inhibitor applications (if used);
- Tillage implements, dates of operation, and number of passes in each operation (which can be used to determine tillage intensity with the STIR Model), (USDA NRCS, 2008);
- Irrigation method, application rate and timing of applications;
- Total dry matter yield of crop (metric tons dry matter year^{-1}), dry matter content of yield, and harvest index; and
- Cover crop types, planting, and harvesting dates (if applicable).

The method for grazing land requires the following management activity data:

- Area of land parcel (i.e., field);
- Prior-year grass type and dry matter production to calculate grass nitrogen input;
- Synthetic fertilizer type (chemical formulation) and coatings (if present);
- Organic amendment types and timing;
- Synthetic and organic amendment application rate, application method (broadcast, banded, or injected, including depth of injection), timing of application(s);
- Pasture/range/paddock (PRP) N excreted directly onto land by livestock (i.e., manure that is not managed);
- Type of nitrification inhibitor applications (if used);
- Tillage implements, dates of operation, and number of passes in each operation which can be used to determine tillage intensity with the STIR Model, (USDA NRCS, 2008);
- Irrigation method, application rate, and timing of applications;
- Periods of grazing during the year;
- Animal type, class, and size used for grazing;
- Stocking rates and methods; and
- Total yield of meat (kg carcass yield year^{-1}) or milk (kg fluid milk year^{-1}).

Crop yields are provided by the entity for the crop system, or peak forage amounts for grazing systems. In some years, the entity may not harvest the crop due to drought, pest outbreaks, or other reasons for crop failure. In those cases, the entity should provide the average yield that they have harvested in the past five years, and an approximate percentage of crop growth that occurred prior to crop failure. The yield is estimated based on multiplying the average crop yield by the percentage of crop growth obtained prior to failure.

To calculate the amount of synthetic fertilizer nitrogen applied to soils, the type of fertilizer applied and its nitrogen content are required. Table 3-10 provides nitrogen content information for common types of synthetic fertilizers.

Pasture/range/paddock (PRP) manure N input is the N excreted directly onto land by livestock, and the manure is not collected or managed (de Klein et al., 2006). The amount of PRP manure N is estimated with the livestock methods (See Chapter 5), and assumed to be split with 50% of the N in urine and the other 50% of the N in solids.

3.5.4.4   Ancillary Data

Ancillary data for estimating direct soil N_2O emissions from mineral soils include land resource region, soil texture, and climate variables. Land resource region can be identified based on the geographic coordinates of the field. Soil data are available from national datasets such as SSURGO (Soil Survey Staff, 2011), and average growing season precipitation and evapotranspiration data are available from national weather datasets such as PRISM (Daly et al., 2008). These data are used by the models to determine base emission rates.

Table 3-10: Nitrogen Fraction of Common Synthetic Fertilizers (percent by weight)

Synthetic Fertilizer	% N
Ammonium nitrate (NH_4NO_3)	33.5%
Ammonium nitrate limestone	20.5%
Ammonium sulfate	20.75%
Anhydrous ammonia	82%
Aqua ammonia	22.5%
Calcium cyanamide ($CaCN_2$)	21%
Calcium ammonia nitrate	27.0%
Diammonium phosphate	18%
Monoammonium phosphate	11%
Potassium nitrate (KNO_3)	13%
Sodium nitrate ($NaNO_3$)	16%
Urea $CO(NH_2)_2$	45%

Source: Fertilizer 101 (2011).

3.5.4.5   Model Output

N_2O emissions are expressed both as the quantity of emissions and as emissions intensity—emissions per unit yield, e.g., g N_2O per Mg grain or animal product. Reducing the emissions intensity can be assumed to avoid emissions from indirect land-use change. In contrast, if the emissions intensity increases due to a loss of yield, then there is potential for additional land to be converted into agriculture to make up for a yield loss.

Equation 3-16: Soil N_2O Emissions Intensity

$$EI_{N2O} = (N_2O_{Direct} + N_2O_{Indirect}) / Y$$

Where:

EI_{N2O} = N_2O emissions intensity (metric tons CO_2-eq per metric ton dry matter crop yield or kg carcass or kg fluid milk)

N_2O_{Direct} = Total direct soil N_2O emission (metric tons CO_2-eq year^{-1}) (See Equation 3-8)

$N_2O_{Indirect}$ = Total indirect soil N_2O emission (metric tons CO_2-eq year^{-1}) (See Equation 3-13)

Y   = Total yield of crop (metric tons dry matter crop yield year^{-1}), meat (kg carcass yield year^{-1}), or milk production (kg fluid milk yield year^{-1})

3.5.4.6   Limitations and Uncertainty

The primary limitation of N_2O estimation models is that they depend on surrogate measures that will not allow fluxes for a particular location or time to be predicted precisely. Nevertheless, while it may be decades, if ever, before annual rates of N_2O emissions from a specific field can be measured with great certainty and for low cost, average estimates for similar cropping systems and landscapes will converge as estimates aggregate to larger areas.

Limitations in the method also occur due to:

- Lack of knowledge of how different practices affect fluxes in some regions and cropping systems.
- Lack of knowledge about how some of the management practices interact with each other and with soil and climate factors to affect the fundamental processes driving N_2O emissions—e.g., nitrification, denitrification, gas diffusion, etc.—and incorporation of these effects into process models.
- Limited number of data sets currently available to test the efficacy of practices to mitigate fluxes and to evaluate process-based models.
- Limited number of datasets with more than two fertilizer rates to estimate the scalars for emission factors associated with the base emission rates, particularly the possibility for non-linear scalars.
- The mineral soils method assumes a one percent emission factor for indirect N_2O emissions from volatilized nitrogen and 0.75 percent emission factor for leached NO_3^-. However, there is evidence that the EF for NO_3^- leaching varies from 0.75%, depending on the type of waterway (Beaulieu et al., 2011) and it is also likely that the soil N_2O emissions from atmospheric deposition of nitrogen will vary depending on the nitrogen status of the receiving ecosystem.
- The fraction of nitrogen that is volatilized (assumed to be 10 percent for inorganic nitrogen sources and 20 percent for organic nitrogen sources in Equation 3-15) is very uncertain. Likewise, the fraction of nitrogen that is leached from a profile or runs off is highly uncertain (assumed to be 30 percent of all nitrogen sources except where precipitation plus irrigation is less than 80 percent of potential evapotranspiration; U.S. Environmental Protection Agency, 2011). Experiments suggest that gross generalizations are not valid and that many practices can reduce both volatilized nitrogen and the nitrogen that is lost by leaching and runoff.[9]
- Climate change will affect model output insofar as baseline N_2O estimates are simulated for any given location using temperature and precipitation distributions for the past 30 years. Expected changes in temperature, precipitation, and extreme events such as droughts, floods, and heat waves will add further uncertainty to estimates of all N_2O emissions and potentially interact with scaling factors. Crop nitrogen management may further change with climate change (Robertson, 2013).

Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity Table 3-11 provides the probability distribution functions to estimate uncertainty in the direct and indirect soil N_2O emissions. Data are not shown for DNDC and DAYCENT output that are delineated by LRR, soil type, and climate.

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[9] The IPCC factors assume that the maximum aboveground nitrogen recovery by crops is 50 to 60 percent. However, rates of nitrogen recovery can be significantly higher with best practices.

Table 3-11: Available Uncertainty Data for Direct and Indirect N$_2$O Emissions

Parameter	Estimated Value	Units	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
Typical direct N$_2$O emission rate and 0-level input rate from process-based model	NS	Various	NS	NS	Multiple distributions	DAYCENT, DNDC
Scaling factor for slow-release fertilizers	-0.21	Proportional Change in Emissions	-0.30	-0.12	Normal	Appendix 3-A
Scaling factor for PRP manure N	+0.5	Proportional Change in Emissions	0.33	0.67	Normal	Appendix 3-A
Scaling factor nitrification inhibitors – semi-arid/arid climate	-0.38	Proportional Change in Emissions	-0.51	-0.21	Normal	Appendix 3-A
Scaling factor nitrification inhibitors – mesic climate	-0.40	Proportional Change in Emissions	-0.52	-0.24	Normal	Appendix 3-A
Scaling factor for no-till, semi-arid/arid climate, <10 years	0.38	Proportional Change in Emissions	0.04	0.72	Normal	van Kessel et al. (2012), Six et al. (2004)
Scaling factor for no-till, semi-arid/arid climate, ≥10 years	-0.33	Proportional Change in Emissions	-0.5	-0.16	Normal	van Kessel et al. (2012), Six et al. (2004)
Scaling factor for no-till, mesic/wet climate, <10 years	-0.015	Proportional Change in Emissions	-0.16	0.16	Normal	van Kessel et al. (2012), Six et al. (2004)
Scaling factor for no-till, mesic/wet climate, ≥10 years	-0.09	Proportional Change in Emissions	-0.19	0.01	Normal	van Kessel et al. (2012), Six et al. (2004)
Base EF scalar – cropland for non-grassland crops	0.0274	(metric tons N$_2$O-N (metric tons N)$^{-2}$) ha year			Normal	Appendix 3-A
Base EF scalar – for grasslands	0.117	(metric tons N$_2$O-N (metric tons N)$^{-2}$) ha year			Normal	Appendix 3-A
Emission rate for cropped Histosols	0.008	metric tons N$_2$O-N ha^{-1} year^{-1}	0.002	0.024	Uniform	IPCC (2006)
Fraction of synthetic nitrogen (N$_{SN}$) that volatilizes as NH$_3$ and NO$_x$	0.1	metric tons N (metric ton N$_{sfert}$)$^{-1}$	0.03	0.3	Uniform	IPCC (2006)

Parameter	Estimated Value	Units	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
Fraction of nitrogen in organic amendments (F_{ON}) that volatilizes as NH_3 and NO_x	0.2	metric tons N (metric ton N_{ON})$^{-1}$	0.05	0.5	Uniform	IPCC (2006)
Emission factor for volatilized nitrogen as NH_3 and NO_x that is transformed to N_2O.	0.01	metric tons N_2O-N (metric ton N)$^{-1}$	0.002	0.05	Uniform	IPCC (2006)
Fraction of N_t that leaches or runs off except in systems with cover crops	0.3	metric tons N (metric ton N)$^{-1}$	0.1	0.8	Uniform	IPCC (2006)
Fraction of N_t that leaches or runs off with a leguminous cover crop	0.18	metric tons N (metric ton N)$^{-1}$	0.14	0.26	Log-Normal	Tonitto et al. (2006)
Fraction of N_t that leaches or runs off with non-leguminous cover crop	0.09	metric tons N (metric ton N)$^{-1}$	0.06	0.15	Log-Normal	Tonitto et al. (2006)
Emission factor for leached and runoff nitrogen that is transformed to N_2O	0.0075	metric tons N (metric ton N)$^{-1}$	0.0005	0.025	Uniform	IPCC (2006)

NS = Not Shown. Data are not shown for parameters that have 100's to 1000's of values (denoted as NS). Data are provided in supplementary material available online.

3.5.5 Methane Uptake by Soils

> **Method for Estimating Methane Uptake by Soil**
>
> - Methane uptake by soil uses an equation based on average values for methane oxidation in natural vegetation—whether grassland, coniferous forest, or deciduous forest—attenuated by current land use practices.
> - Annual average CH_4 oxidation fluxes are from the data set used by Del Grosso et al. (2000a) who reviewed average fluxes from grassland and agricultural soils, coniferous forest soils, and deciduous forest soils. Management reduces potential (historic) oxidation to 30 percent of original rates based on available data (Del Grosso et al., 2000a; Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000). Kuchler potential vegetation maps can be used to determine the natural vegetation across the United States if the entity does not have information for land parcels in operation.
> - This newly developed methodology makes use of recent U.S.-based research that is not addressed by IPCC or the U.S. Inventory. The method incorporates entity specific annual data such as current management of the land parcel, cultivation for crop production, grazing activity, recently harvested forests, or fertilized grasslands or forests.

3.5.5.1 Rationale for Selected Method

There are no agronomic practices known to enhance CH_4 uptake (oxidation) in croplands, other than in wetlands converted to flooded rice (discussed in Section 3.2.2). Agronomic activity universally reduces methanotrophy in arable soils by 70 percent or more (Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000). Recovery of CH_4 oxidation upon abandonment from agriculture is slow, probably taking 50 to 100 years for the development of even 50 percent of former (original) rates (Levine et al., 2011). No recovery has been documented for CRP grasslands or perennial biofuel crops to date. There are currently no models for quantifying CH_4 oxidation recovery other than rate of reversion to natural vegetation, so this is a Tier 3 method as defined by the IPCC.

3.5.5.2 Description of Method

The model is based on average values for methane oxidation in natural vegetation—whether grassland, coniferous forest, or deciduous forest—attenuated by current land use practices. Average values are from the data set used by Del Grosso et al. (2000a), who reported average fluxes (± standard deviation) for temperate and tropical grassland soils of 3.2±1.9 kg CH_4 ha^{-1} year^{-1}; for coniferous forest soils, 2.8±1.4 kg CH_4 ha^{-1} year^{-1}; and for deciduous forest soils, 11.8±5 kg CH_4 ha^{-1} year^{-1}. Management reduces potential (historic) oxidation to 30 percent of original rates based on available data (Del Grosso et al., 2000a; Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000) as noted in Sections 3.2.3.3 and 3.3.2.3. Recovery of oxidation is assumed to occur over the period required for ecological succession to restore original vegetation (Del Grosso et al., 2000a; Mosier et al., 1991; Robertson et al., 2000; Smith et al., 2000), which is approximated at 100 years after abandonment from agriculture or forest harvest. Recovery is assumed to occur at a linear rate (Smith et al., 2000) such that successional forests and grasslands will consume CH_4 at a rate that is between 30 and 100 percent of the original oxidation capacity between the initial year of abandonment until year 100. The following equation is used to estimate methane oxidation for a land parcel:

Equation 3-17: Methane (CH$_4$) Oxidation

$$CH_{4SoilOxidation} = (PCH_4 \times AF) \times SF \times A \times CH_{4GWP}$$

Where:

CH$_{4SoilOxidation}$	= CH$_4$ oxidation in soils (metric tons CO$_2$-eq year^{-1})
PCH$_4$	= Potential CH$_4$ oxidation based on historic natural vegetation; grasslands = 3.2; coniferous forests = 2.8, deciduous forests = 11.8 (kg CH$_4$ ha^{-1} year^{-1})
AF	= CH$_4$ oxidation attenuation factor; cropland including set-aside (CRP) grassland, grazing land, and fertilized or recently harvested forests = 0.30; natural vegetation, 0-100 years after abandonment of agricultural production or timber harvest = 0.3 + (0.007 × years since abandonment); >100 years post-management or never used for agricultural management or timber harvest = 1.0
SF	= Scaling factor, 1/1000 (metric tons kg^{-1})
A	= Area (ha)
CH$_{4GWP}$	= Global warming potential of CH$_4$ (metric tons CO$_2$-eq (metric tons CH$_4$)$^{-1}$)

3.5.5.3   Activity Data

This method requires land use and type of vegetation for the past 80 years. Kuchler potential vegetation maps can be used to determine the natural vegetation across the United States (grassland, coniferous forest, or deciduous forest) if the entity does not have this information for land parcels in the operation. The entity will need to identify if the current management of the land parcel includes cultivation for crop production, grazing in grasslands, recently harvested forests, or fertilized grasslands or forests. Assuming the parcel of land is not under cultivation, fertilized, grazed grasslands, or recently harvested forest, the entity will need to provide the time since the land has been managed with one of these practices.

3.5.5.4   Ancillary Data

No ancillary data are required for this method.

3.5.5.5   Model Output

The model provides a value for diminished CH$_4$ oxidation capacity. The change in CH$_4$ oxidation capacity will be negative, and so there is no potential for increased CH$_4$ oxidation with this method. Unlike other methods in this section, the emissions intensity is not relevant for this method.

3.5.5.6   Limitations and Uncertainty

- Lack of precision in knowledge of prior land use.
- Uncertainties associated with estimating CH$_4$ oxidation rates prior to conversion (PCH$_4$ in Equation 3-17). In a review of available data, Del Grosso et al. (2000a) noted annual CH$_4$ oxidation rates of <1.8 kg CH$_4$ ha^{-1} year^{-1} for grassland and agricultural soils, 1.4 to 4.1 kg CH$_4$ ha^{-1} year^{-1} for coniferous and tropical forest soils, and 5.3 to 12 kg CH$_4$ ha^{-1} year^{-1} for deciduous forest soils.

- ▪ Uncertainty associated with the attenuation factor. In a review of temperate region comparisons of paired sites in natural vegetation vs. agricultural management, Smith et al. (2000) found that agricultural conversion to cropland or pasture reduced oxidation by 71 percent on average.

Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity, although this may not be the case if there is limited knowledge about land-use change. Table 3-12 provides the probability distribution functions associated with estimating uncertainty in methane oxidation.

Table 3-12 Available Uncertainty Data for Methane Oxidation

Parameter	Estimated Value	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
CH_4 oxidation rates prior to conversion (PCH_4) grasslands (kg CH_4 ha^{-1} year^{-1})	3.2	0	6.9	Normal	Del Grosso et al. (2000a)
CH_4 oxidation rates prior to conversion (PCH_4) coniferous forests (kg CH_4 ha^{-1} year^{-1})	2.8	0.1	5.5	Normal	Del Grosso et al. (2000a)
CH_4 oxidation rates prior to conversion (PCH_4) deciduous Forests (kg CH_4 ha^{-1} year^{-1})	11.8	1.9	21.6	Normal	Del Grosso et al. (2000a)
CH_4 oxidation attenuation factor: cropland including set-aside (CRP) grassland, grazing land, and fertilized or recently harvested forests	0.30	0.07	1	Log-Normal	Smith et al. (2000)
CH_4 oxidation attenuation factor: natural vegetation, 0-100 years after abandonment of agricultural production or timber harvest	0.3 + (0.007 × years since abandonment)	0.07 + (0.007 × years since abandonment)	1	Log-Normal	Smith et al. (2000)
CH_4 oxidation attenuation factor: >100 years post-management or never used for agricultural management or timber harvest	1	0.07	1	Log-Normal	Smith et al. (2000)

3.5.6   Methane and Nitrous Oxide from Flooded Rice Cultivation

Method for Estimating Methane and N$_2$O Emissions from Rice Cultivation

- IPCC equations developed by Lasco et al. (2006) for CH$_4$ and de Klein et al. (2006) for N$_2$O.
    - The baseline emission factor or typical daily rate at which CH$_4$ is produced per unit of land area represents fields that are continuously flooded during the cultivation period, not flooded at all during the 180 days prior to cultivation, and receive no organic amendments. Differences between the baseline continuously flooded fields without organic amendments are accounted for by scaling factors (e.g., water regime adjustments (pre-and during the cultivation period), or organic amendments). CH$_4$ scaling factors to account for water regimes and organic amendments come from Lasco et al. (2006).
    - N$_2$O emission factors rely on Lasco et al. (2006), and the scaling factor to account for drainage effects comes from Akiyama et al. (2005; USDA, 2011).
- This method uses the IPCC (2006) equations with the addition of a scaling factor for estimating N$_2$O emissions from drainage (Akiyama et al., 2005; U.S. EPA, 2011). The method for methane emissions uses entity specific seasonal parcel data as input into the IPCC equation.
- This method was chosen to minimize uncertainty. Process models were considered, but not chosen for this method due to a need for further research on U.S. rice cultivation conditions and practices.

3.5.6.1   Rationale for Selected Method

There are a number of possibilities for estimating GHG emissions from flooded rice systems. Process based models are being developed to quantify GHG emissions, such as the DNDC (e.g., Zhang et al., 2011) and DAYCENT models (Cheng et al., 2013). While, these models have been evaluated for various regions and countries in Asia, they have not been sufficiently evaluated for U.S. rice systems, which are significantly different from those found in Asia (establishment practices, residue management, water management, and varieties). Therefore, the selected method is based on the IPCC Tier 1 methodology. While the IPCC methodology has also been largely developed from Asian rice studies, it is more transparent and uncertainties can be derived in the emissions estimates. It is anticipated that the process-based models may be further tested and calibrated in the near future for U.S. conditions and possibly used in a future version of these methods.

Several management practices have the potential to influence CH$_4$ and N$_2$O emissions from flooded rice systems. However, there are currently not enough data available to quantitatively account for (or establish scaling factors for) the effects of all of these management practices. There is sufficient information to account for the influence of water management, residue management, and organic amendments on CH$_4$ emissions from flooded rice (Lasco et al., 2006; Yan et al., 2005).

3.5.6.2   Description of Method

Methane: The methodology assumes a baseline emission factor or "typical" daily rate at which CH$_4$ is produced per unit of land area. This baseline factor represents fields that are continuously flooded during the cultivation period, not flooded at all during the 180 days prior to cultivation, and receive no organic amendments. Differences between the baseline scenario and other scenarios are accounted for by the use of scaling factors that are used to adjust the baseline emission factor for

the effects of water management (occurring both before and during the cultivation period) and the amount of organic amendments. The rate at which CH_4 is emitted depends on water flooding/drainage regimes and on rates and types of organic amendments applied to the soil. As such, scaling factors for a broad range of scenarios are provided with this methodology. The factors are differentiated by hydrological context (e.g., irrigated, rainfed, upland—all rice fields in the United States are irrigated), cultivation period flooding regime (e.g., continuous, multiple aeration), time since last flooding (prior to cultivation; e.g., over 180 days, under 30 days) and type of organic amendment (e.g., compost, farm yard manure).

The following equation has been adopted from the methodology developed by the IPCC to estimate CH_4 emissions from a land parcel (Lasco et al., 2006):

Equation 3-18: Flooded Rice Methane Emissions

$$CH_{4Rice} = CH_{4GWP} \times \Sigma_{ijk} (EF_{ijk} \times t_{ijk} \times A_{ijk} \times 10^{-3})$$

Where:

CH_{4Rice}	= Annual methane emissions from rice cultivation (metric tons CO_2-eq year^{-1})
EF_{ijk}	= A daily emission factor for i, j, and k conditions (kg CH_4 ha^{-1} day^{-1})
t_{ijk}	= Cultivation period of rice for i, j, and k conditions (days)
A_{ijk}	= Annual harvested area of rice for i, j, and k conditions (ha year^{-1})
CH_{4GWP}	= Global warming potential for CH_4 (metric tons CO_2-eq (metric tons CH_4)$^{-1}$)
i, j, and k	= Represent different ecosystems, water regimes, type and amount of organic amendments, soil type, rice cultivar, sulfate containing amendments, and other conditions under which CH_4 emissions from rice may vary.

The daily emission factor is estimated based on the conditions (i, j, k, etc.) that influence CH_4 emissions for flooded rice production, including the ecosystem type, water regime, and organic amendment rate. As more data become available, additional conditions that influence CH_4 emissions may be added. The "i" in the equations below represents the specific scenario or "other conditions" that can significantly influence CH_4 emissions on a parcel. In the future, additional scenarios with factors that affect CH_4 emissions may be included as the relationship between these conditions becomes clear. The following equation is used to estimate the daily emission factor for a land parcel:

Equation 3-19: Flooded Rice Methane Emission Factor

$$EF_i = EF_c \times SF_w \times SF_p \times SF_o \times SF_{s,r}$$

Where:

EF_i = adjusted daily emission factor for a particular harvested area (kg CH_4 ha^{-1} day^{-1})

EF_c = baseline emission factor for continuously flooded fields without organic amendments (kg CH_4 ha^{-1} day^{-1})

SF_w = scaling factor to account for the differences in water regime during the cultivation period (from Lasco et al. 2006, Table 5.12) (unitless)

SF_p = scaling factor to account for the differences in water regime in the pre-season before the cultivation period (from Lasco et al. 2006, Equation 5.3 and Table 5.14) (unitless)

SF_o = scaling factor should vary for both type and amount of organic amendment applied (Equation 3-20) (unitless)

$SF_{s,r}$ = scaling factor for soil type, rice cultivar, etc., if available

The scaling factor for organic amendments to a land parcel is estimated using the following equation:

Equation 3-20: Organic Amendments Scaling Factor

$$SF_o = (1 + \Sigma(ROA_i \times CFOA_i))^{0.59}$$

Where:

SF_o = scaling factor for both type and amount of organic amendment

ROA_i = rate of application of organic amendment(s) (metric tons ha^{-1})

$CFOA_i$ = conversion factor for organic amendments (from Lasco et al. 2006, Table 5.14) (unitless)

The scaling factors for Equation 3-19 and Equation 3-20 are from Lasco et al. (2006) and shown below.

Table 3-13: Rice Water Regime Emission Scaling Factors (During Cultivation Period)

Water Regime During the Cultivation Period (assumes irrigated)	SF_w
Continuously flooded	1
Intermittently flooded – single aeration	0.6
Intermittently flooded – multiple aeration	0.52

Source: Lasco et al. (2006), Table 5.12.

Table 3-14: Rice Water Regime Emission Scaling Factors (Before Cultivation Period)

Water Regime Before the Cultivation Period	SF_p
Non flooded pre-season < 180 days	1
Non flooded pre-season > 180 days	0.68
Flooded pre-season > 30 days	1.9

Source: Lasco et al. (2006), Table 5.13.

Table 3-15: Rice Organic Amendment Emission Scaling Factors; adapted from Lasco et al. (2006)

Organic Amendments	CFOA
Straw incorporated shortly (<30 days) before cultivation	1
Straw incorporated long (>30 days) before cultivation	0.29
Compost	0.05
Farm yard manure	0.14
Green manure	0.50

Source: Lasco et al. (2006), Table 5.14.

Soil N_2O: The IPCC methodology (de Klein et al., 2006) has been adapted to estimate direct N_2O emissions from rice fields. The emission factor for rice soils accounts for nitrogen additions from mineral fertilizers, organic amendments, and crop residues. Note that an effect of nitrogen mineralized from mineral soil as a result of loss of soil carbon is not included in this equation. Flooded rice cultivation leads to minimal losses of soil carbon due to periodic flooding, which is the default assumption with the IPCC method (Lasco et al., 2006), and therefore it is not necessary to include the effect of enhanced nitrogen mineralization from loss of soil C.

The following equation is used to estimate the soil N_2O emissions from a parcel of land:

Equation 3-21: Direct Soil N_2O Emissions from flooded Rice

$$N_2O_{Rice} = N_t \times EF \times (1 + SF_D) \times N_2O_{MW} \times N_2O_{GWP}$$

Where:

N_2O_{Rice} = Direct emissions of N_2O from soils in flooded rice production systems (metric tons CO_2-eq year^{-1})

N_t = Total nitrogen inputs from all agronomic sources: mineral fertilizer, organic amendments, residues, and additional mineralization from land-use change or tillage change (metric tons N year^{-1})

EF = Emission factor or proportion of N_t transformed to N_2O (kg N_2O-N (kg N)$^{-1}$)

SF_D = Scaling factor to account for drainage effects; 0 for continuously flooded (dimensionless)

N_2O_{MW} = Ratio of molecular weights of N_2O to N_2O-N
= 44/28 (metric tons N_2O (metric tons N_2O-N)$^{-1}$)

N_2O_{GWP} = Global warming potential for N_2O (metric tons CO_2-eq (metric tons N_2O)$^{-1}$)

The emission factor and SF_D factors are based on research conducted by Akiyama et al. (2005). The IPCC (2006) does not account for differences in water management, and uses an emission factor of 0.3, but Akiyama et al. (2005) provide further disaggregation of the emission factors based on water management. Therefore, the selected emission factor value is 0.0022 based on Akiyama et al. (2005), and the scaling factors are 0 for continuously flooded rice and 0.59 for aerated systems (i.e., drainage events during the growing season).

Indirect N_2O Emissions: For indirect N_2O emissions from flooded rice, the same method is used as described in Section 3.5.4.2, by applying Equation 3-13, Total Indirect Soil N_2O Emissions from Mineral Soils; Equation 3-14, Indirect Soil N_2O Emissions from Mineral Soils —Volatilization; and Equation 3-15, Indirect Soil N_2O Emissions from Mineral Soils —Leaching and Runoff. In the latter

two equations, use the IPCC default fractions for FR_{SN}, FR_{ON}, and FR_{leach}, which are provided in the equation boxes.

3.5.6.3   Activity Data

The activity and related data requirements for this method include:

- Harvested area (ha);
- Cultivation period in days;
- Water management practices throughout the year (e.g., aeration or not);
- Organic matter amendment (including residue) rate;
- Organic fertilizer N;
- Fertilizer nitrogen management (rate);
- Type of fertilizer(s) applied (qualitative);
- Crop residue N; and
- Crop yield, metric tons dry matter crop yield year^{-1}.

3.5.6.4   Ancillary Data

No ancillary data are needed for this method.

3.5.6.5   Model Output

Model output is the combined emissions of CH_4 and N_2O in CO_2 equivalents, expressed on an area basis. The intensity of CH_4 emissions and nitrous oxide (i.e., emissions per unit of land area cultivated) is related to the quantity of crops grown and can be estimated with the following equation:

Equation 3-22: Flooded Rice Combined Methane and Nitrous Oxide Emissions Intensity

$$EI = (CH_{4Rice} + N_2O_{Rice})/Y$$

Where:

EI   = Emissions intensity (metric tons CO_2-eq per metric tons dry matter crop yield)

CH_{4Rice} = Annual methane emissions from rice cultivation (metric tons CO_2-eq year^{-1})

N_2O_{Rice} = Direct emissions of N_2O from soils in flooded rice production systems (metric tons CO_2-eq-year^{-1})

Y   = Total yield of crop (metric tons dry matter crop yield year^{-1})

3.5.6.6   Limitations and Uncertainty

This method has several limitations that will potentially create bias or imprecision in the results. Currently, scaling factors account only for water and organic matter management and do not account for other mitigation options. As indicated earlier there are other management opportunities that may reduce emissions, but further research is required in these areas. Baseline emissions are highly variable, but this methodology provides only one factor value representing the baseline emissions. In addition, the methodology assumes a period of drainage; however, drain events (even those of similar duration) can vary markedly based on soil and climatic conditions, from dry and cracking on the surface to saturated at the end of a drainage event. The influence of drainage on the soil saturation is not addressed with the current method. In addition, there is currently insufficient information to develop a method for the use of sulfur products as amendments; future guidance may be updated with a method for this practice.

CH_4 emissions are the result of a number of interacting biological processes, which by nature vary spatially and temporally. The greatest amount of uncertainty is the baseline emission factor. When using this methodology, the emission factor is an average emission factor for continuously flooded rice systems that have not been flooded the 180 days prior to cultivation and have not received organic amendments. In the case of CH_4 emissions from rice cultivation, the uncertainty ranges of Tier 1 values (emission and scaling factors) are adopted directly from Lasco et al. (2006). Ranges are defined as the standard deviation about the mean, indicating the uncertainty associated with a given default value for this source category.

Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity. Table 3-16 provides the probability distribution functions associated with estimating uncertainty in methane and N_2O emissions from rice cultivation.

Table 3-16: Available Uncertainty Data for Methane, Direct and Indirect N_2O Emissions

Methane from Flooded Rice Cultivation						
Parameter	Abbreviation/ Symbol	Estimated Value	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
Baseline emission factor for continuously flooded fields without organic amendments	EFc	1.3	0.8	2.2	Uniform	IPCC (2006)
Water regime during the cultivation period – Scaling factor	SFw for continuously flooded	1	0.79	1.26	Uniform	IPCC (2006)
Water regime during the cultivation period – Scaling factor	SFw for single aeration	0.6	0.46	0.8	Uniform	IPCC (2006)
Water regime during the cultivation period – Scaling factor	SFw for multiple aerations	0.52	0.41	0.66	Uniform	IPCC (2006)
Water regime before the cultivation period – Scaling factor	SFp for non-flooded pre-season <180 days	1	0.88	1.14	Uniform	IPCC (2006)
Water regime before the cultivation period – Scaling factor	SFp for non-flooded pre-season > 180 days	0.68	0.58	0.8	Uniform	IPCC (2006)
Water regime before the cultivation period – Scaling factor	SFp for flooded pre-season > 30 days	1.9	1.65	2.18	Uniform	IPCC (2006)
Organic amendment conversion factor	CFOAi for straw incorporation less than 30 days before cultivation	1	0.97	1.04	Uniform	IPCC (2006)

Methane from Flooded Rice Cultivation (*continued*)						
Parameter	Abbreviation/ Symbol	Estimated Value	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
Organic amendment conversion factor	CFOAi for straw incorporation more than 30 days before cultivation	0.29	0.2	0.4	Uniform	IPCC (2006)
Organic amendment conversion factor	CFOAi for compost	0.05	0.01	0.08	Uniform	IPCC (2006)
Organic amendment conversion factor	CFOAi for farm yard manure	0.14	0.07	0.2	Uniform	IPCC (2006)
Organic amendment conversion factor	CFOAi for green manure	0.5	0.3	0.6	Uniform	IPCC (2006)
N_2O from Flooded Rice						
Parameter	Abbreviation/ Symbol	Mean	Relative Uncertainty Low (%)	Relative Uncertainty High (%)	Distribution	Data Source
Emission factor or proportion of N_t transformed to N_2O	EF	0.0022	0.24%	0.24%	Normal	Akiyama et al. (2005)
Scaling factor to account for drainage effects	SF_D for aerated systems	0.59	0.35%	0.35%	Normal	Akiyama et al. (2005)

3.5.7   CO$_2$ from Liming

Method for Estimating CO$_2$ Emissions from Liming

- This method uses the IPCC equation (de Klein et al., 2006) with U.S. specific emissions factors.
- Entity specific annual parcel data as input into the IPCC equation (e.g., the amount of lime, crushed limestone, or dolomite applied to soils).
- This method was selected as it was the only readily available model for estimating CO$_2$ emissions from liming.

3.5.7.1   Rationale for Selected Method

Addition of lime to soils is typically thought to generate CO$_2$ emissions to the atmosphere (de Klein et al., 2006). However, prevailing conditions in U.S. agricultural lands lead to CO$_2$ uptake because the majority of lime is dissolved in the presence of carbonic acid (H$_2$CO$_3$). Therefore, the addition of lime leads to a carbon sink in the majority of U.S. cropland and grazing land systems. Whether liming contributes to a sink or source depends on the pathways of dissolution and rates of bicarbonate leaching. The emissions factor provided in this guidance has been estimated from a

review of existing models and mass balance analyses conducted for the application of lime in the United States and is a Tier 2 method as defined by the IPCC.

Since crushed limestone ($CaCO_3$) contains 12 percent C, an application of 1,000 kg $CaCO_3$ places 120 kg C on the soil surface. It is assumed that two-thirds of this (80 kg) is acidified to HCO_3- and leached to the ocean where it will be sequestered for decades to centuries (Oh and Raymond, 2006). Because this transfer represents a movement from one long-term pool (geologic formations) to another (ocean), this carbon transfer does not represent a net uptake of CO_2 from the atmosphere. However, with this transfer, there is 80 kg C of atmospheric CO_2 uptake into soils. The uptake of CO_2 from the atmosphere, after subtracting the one-third of carbon in the lime that is acidified directly to CO_2 (40 kg C), yields a total net CO_2 uptake of 40 kg C per 1,000 kg $CaCO_3$ applied. This results in a carbon coefficient or emission factor of 40/1000 = -0.04 kg C per kg $CaCO_3$. This equates to a carbon sink (40 kg C sequestered/120 kg C × 100). Dolomite contains only slightly more carbon than does $CaCO_3$ (13 percent vs. 12 percent) so the factors are essentially the same.

The emission factor is country-specific based on a revision of the estimates proposed in West and McBride (2005), which are currently used in the U.S. National GHG Inventory (U.S. EPA, 2011). The underlying difference with the earlier emission factor from West and McBride (2005) is that the revised value assumes that the amount of bicarbonate carried into rivers has a long turnover time and is essentially not returned to the atmosphere over decadal to century time scales.

3.5.7.2 Description of Method

The model to estimate CO_2 emissions from liming has been adapted from methods developed by the IPCC (de Klein et al., 2006), with refinement in the emission factors based on conditions in U.S. agricultural lands. The following equation is used to estimate emissions from carbonate lime additions to a land parcel:

Equation 3-23: Change in Soil Carbon Stocks from Lime Application

$$\Delta C_{Lime} = M \times EF \times CO_2MW$$

Where:

ΔC_{Lime} = Annual change in soil carbon stocks from lime application (metric tons CO_2-eq)

M = Annual application of lime as crushed limestone or dolomite

(metric tons of crushed limestone or dolomite year^{-1})

EF = Metric ton CO_2 emissions per metric ton of lime -0.04

(metric ton carbon (metric ton lime)$^{-1}$)

CO_2MW = Ratio of molecular weight of CO_2 to carbon (44/12) (metric tons CO_2 (metric tons C)$^{-1}$)

3.5.7.3 Activity Data

The method requires data on the amount of lime (crushed limestone or dolomite) applied to soils.

3.5.7.4 Ancillary Data

No ancillary data are needed in order to apply the method.

3.5.7.5 Model Output

Model output is generated on both an absolute quantity of emissions and emissions intensity. The latter is based on the amount of emissions per unit of yield for crops in cropland systems or grazing systems. The emissions intensity is estimated with the following equation:

Equation 3-24: Emissions Intensity from Lime Application

$$EI = \Delta C_{Lime}/Y$$

Where:

EI = Emissions intensity (metric tons CO_2 per metric ton dry matter crop yield)

ΔC_{Lime} = Annual change in soil carbon stocks from lime application (metric tons CO_2)

Y = Total yield of crop (metric tons dry matter crop yield year^{-1}), meat (kg carcass yield year^{-1}), or milk production (kg fluid milk yield year^{-1})

Yields are based on the total amount of product from the land managed with lime application.

3.5.7.6 Limitations and Uncertainty

Limitations include variation in soil carbon emissions due to soil pH and rate of nitrogen fertilizer application, which influence the chemical pathway of lime dissolution (Hamilton et al., 2007; West and McBride, 2005). More specifically, the EF will not accurately capture the result of lime dissolution in the presence of stronger nitric acid (HNO_3), which is produced when nitrifying bacteria convert ammonium (NH_4^+) based fertilizer and other sources of NH_4^+ to nitrate (NO_3^-).

Uncertainties in the lime emissions methods include imprecision at the farm scale, because the method of estimation is based on stream-gauge data that are collected at the watershed scale. Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the management activity data provided by the entity. Table 3-17 provides the probability distribution functions associated with CO_2 emissions per metric ton of lime applied.

Table 3-17: Available Uncertainty Data for CO_2 from Liming

Parameter	Mean	Relative Uncertainty Low (%)	Relative Uncertainty High (%)	Distribution	Data Source
Emissions factor (metric ton CO_2 emissions per metric ton of lime)	-0.04	46%	46%	Normal	Adapted from West and McBride (2005)

3.5.8   Non-CO_2 Emissions from Biomass Burning

Method for Estimating Non-CO_2 Emissions from Biomass Burning

- The method uses the IPCC equation and emission factors developed by Aalde et al. (2006).
- Entity specific annual parcel data (e.g., area burned for croplands and grazing land; crop type and harvest yield data; residue-yield ratios (West et al., 2010); type of forage, grazing area, and amount of biomass before the fire in grazing lands that are burned; and combustion efficiency) are inputs to the IPCC equation.
- This method was selected as it was the only readily available model for estimating non-CO_2 emissions from biomass burning.

3.5.8.1   Rationale for Selected Method

Non-CO_2 GHG emissions from biomass burning include CH_4 and N_2O. CO and NO_x are also emitted and are precursors that are later converted into GHGs following additional reactions (i.e., release of these gases leads to GHG formation). CO_2 is also emitted but not addressed for crop residues or grassland burning because the carbon is reabsorbed from the atmosphere in new growth of crops or grasses within an annual cycle.

There has been limited development and testing of process-based approaches for estimating non-CO_2 GHG emission from biomass burning. Moreover, country-specific data are limited on the amount of non-CO_2 GHG emissions. Therefore, this guidance has adopted the IPCC Tier 1 method as described by Aalde et al. (2006).

3.5.8.2   Description of Method

The model to estimate non-CO_2 GHG emissions and precursors has been adapted from methods developed by IPCC (Aalde et al., 2006). The following equation is used to estimate emissions due to burning biomass on a parcel of land:

Equation 3-25: GHG Emissions from Biomass Burning

$$GHG_{BiomassBurning} = A \times M \times C \times EF \times 10^{-3} \times GHG_{GWP}$$

Where:

$GHG_{BiomassBurning}$ = Annual emissions of GHG or precursor due to biomass burning (metric tons of CO_2-eq year^{-1})

A = Area burned (ha)

M = Mass of fuel available for combustion (metric tons dry matter ha^{-1} year^{-1})

C = Combustion efficiency, dimensionless

EF = Emission factor (g GHG (kg of burned biomass)$^{-1}$)

GHG_{GWP} = Global warming potential for each GHG (metric tons CO_2-eq (metric tons GHG)$^{-1}$)

Combustion efficiency, as defined in IPCC (2006) combines the proportion of biomass that is actually burned in a fire with the amount of carbon released as a proportion of the total carbon in the burned biomass. The mass of the fuel combusted includes live and dead biomass (i.e., dead

biomass includes plant residues in grazing and cropland systems) and is approximated for a land parcel with the following equation:

Equation 3-26: Mass of Fuel

$$M = (H_{peak}/C) \times (D/100)$$

Where:

M = Mass of fuel available for combustion (metric tons dry matter ha^{-1} year^{-1})

H_{peak} = Annual peak aboveground herbaceous biomass carbon stock (metric tons C ha^{-1} year^{-1})

C = Carbon fraction of aboveground biomass (dimensionless)

D = Percentage of biomass present at the stage of burning relative to peak (%)

Peak aboveground biomass is estimated with Equation 3-3 for crops and grass vegetation. For croplands that are burned following harvest, the residue mass is estimated by subtracting the harvest index (HI) from one and converting to a percentage, which is the residual biomass left in the field. Default harvest indices are given in Table 3-5. The estimated mass of fuel for grazing systems based on Equation 3-3 does not include the dead biomass. If there is significant residual litter in grazing systems, then multiply the mass of fuel by two as a conservative estimate of the total live and dead biomass on the land parcel. Alternatively, entities may enter an estimate for the proportion of residual litter mass relative to the live biomass, instead of using two, which doubles the mass of fuel. A summary of emission factors by land use category is provided in Table 3-18.

Table 3-18: Emission Factors for Biomass Burning

Land-Use Category	CO	CH$_4$	N$_2$O	NO$_x$
	(g kg^{-1})			
Grassland burning	65	2.3	0.21	3.9
Cropland residue	92	2.7	0.07	2.5
Forest biomass (with conversion to cropland or grazing lands)	107	4.7	0.26	3.0

Source: Aalde et al. (2006).

3.5.8.1 Activity Data

The following activity and related data are needed to apply the method:

- Area burned for croplands and grazing land;
- Crop type and harvest yield data for crops grown in fields with residue burning management;
- Residue: yield ratios (optional);
- Type of forage, grazing area, and amount of biomass before the fire in grazing lands that are burned; and
- Combustion efficiency (optional).

A list of default combustion efficiencies is provided for residues and forages (Table 3-19 and Table 3-20), but the entity can provide value specific to their operation. Default dry matter contents and residue-yield ratios are provided in Table 3-5, but can also be entered by the entity if the information is available.

Table 3-19: Default Combustion Efficiencies for Selected Crops

Crop	Combustion Efficiency (C)
Corn	0.88 x 0.93 = 0.82
Cotton	0.88 x 0.93 = 0.82
Lentils	0.88 x 0.93 = 0.82
Rice	0.88 x 0.93 = 0.82
Soybeans	0.88 x 0.93 = 0.82
Sugarcane	0.68 x 0.81 = 0.55
Wheat	0.88 x 0.93 = 0.82

Source: EPA (2013), Table 6-25.

In some years, the entity may not harvest the crop due to drought, pest outbreaks, or other reasons for crop failure. In those cases, the entity should provide the average yield that it has harvested in the past, and an approximate percentage of average crop growth that occurred prior to burning. The yield is estimated based on multiplying the average crop yield by the percentage of crop growth obtained prior to burning.

3.5.8.2   Ancillary Data

No ancillary data are needed in order to apply the method.

3.5.8.3   Model Output

Model output is generated on both an absolute quantity of emissions and emissions intensity. The latter is based on the amount of emissions per unit of yield for crops in cropland systems or animal products in grazing systems. The emissions intensity is estimated with the following equation:

Yields are based on the total amount of product from the land managed with burning.

Table 3-20: Default Combustion Efficiencies for Select Vegetation Types

Vegetation Type	Combustion Efficiency (C)
Boreal Forest (all)	0.34
Wildfire	0.40
Crown fire	0.43
Surface fire	0.15
Post logging slash burn	0.33
Land clearing fire	0.59
Temperate Forest (all)	0.45
Post logging slash burn	0.62
Felled and burned (land-clearing fire)	0.51
Shrublands (all)	0.72
Shrubland (general)	0.95
Calluna health	0.71
Fynbos	0.61
Savanna woodlands (early dry season burns) (all)	0.40
Savanna woodland (early)	0.22
Savanna parkland (early)	0.73
Savanna woodlands (mid/late dry season burns) (all)	0.74
Savanna woodland (mid/late)	0.72
Savanna parkland (mid/late)	0.82
Tropical savanna	0.73
Other savanna woodlands	0.68
Savanna grasslands (early dry season burns) (all)	0.74
Tropical/sub-tropical grassland	0.74
Savanna Grasslands/Pastures (mid/late dry season burns) (all)	0.77
Tropical/sub-tropical grassland	0.92
Tropical pasture	0.35
Savanna	0.86

Source: Aalde et al. (2006), Table 2.4 (C × M) and Table 2.6 (C)

Equation 3-27: Biomass Burning Emissions Intensity

$$EI = GHG_{BiomassBurning}/Y$$

Where:

EI = Emissions intensity (metric tons CO_2 per metric ton dry matter crop yield, metric tons CO_2 per kg carcass yield, metric tons CO_2 per kg fluid milk yield)

$GHG_{BiomassBurning}$ = Annual CO_2 equivalent emissions from burning (metric tons CO_2-eq year^{-1})

Y = Total yield of crop (metric tons dry matter crop yield year^{-1}), meat (kg carcass yield year^{-1}), or milk production (kg fluid milk yield year^{-1})

3.5.8.4 Limitations and Uncertainty

Uncertainty in the emission estimates is attributed to imprecision in carbon fractions, dry matter contents, harvest indices, combustion efficiencies, and the emission factors. Uncertainties in model parameters are combined using a Monte Carlo simulation approach. Uncertainty is assumed to be minor for the crop yields, peak forage, and relative amount of crop or forage growth compared to the peak production. However, these values are likely to have some level of uncertainty, and methods will need to be refined in the future to better address these uncertainties, particularly the mass of fuel in grazing lands. Table 3-21 provides the probability distribution functions for estimating uncertainty in non-CO_2 emissions from biomass burning.

Table 3-21: Available Uncertainty Data for Non-CO_2 Emissions from Biomass Burning

Parameter	Mean	Relative Uncertainty Low (%)	Relative Uncertainty High (%)	Distribution	Data Source
CH_4 EF for grassland (g CH_4 kg^{-1})	2.3	8%	8%	Normal	IPCC (2006)
CH_4 EF for crop residue (g CH_4 kg^{-1})	2.7	50%	50%	Normal	IPCC (2006)
N_2O EF for grassland (g N_2O kg^{-1})	0.21	93%	93%	Normal	IPCC (2006)
N_2O EF for crop residue (g N_2O kg^{-1})	0.07	50%	50%	Normal	IPCC (2006)
Combustion efficiency for shrublands	0.72	68%	68%	Normal	IPCC (2006)
Combustion efficiency for grasslands with early season burns	0.74	50%	50%	Normal	IPCC (2006)
Combustion efficiency for grasslands with mid to late season burns	0.77	66%	66%	Normal	IPCC (2006)
Combustion efficiency for small grains	0.9	50%	50%	Normal	Expert Assessment by authors
Combustion efficiency for large grain and other crop residues	0.8	50%	50%	Normal	Expert Assessment by authors
Combustion efficiency Boreal forest (all)	0.34	102%	102%	Normal	IPCC (2006)
Wildfire	0.40	340%	340%	Normal	IPCC (2006)
Crown fire	0.43	104%	104%	Normal	IPCC (2006)
Surface fire	0.15	96%	96%	Normal	IPCC (2006)
Post logging slash burn	0.33	130%	130%	Normal	IPCC (2006)
Combustion efficiency Temperate forest (all)	0.45	51%	51%	Normal	IPCC (2006)
Post logging slash burn	0.62	264%	264%	Normal	IPCC (2006)
Combustion efficiency Shrublands (all)	0.72	147%	147%	Normal	IPCC (2006)
Calluna health	0.71	121%	121%	Normal	IPCC (2006)
Fynbos	0.61	195%	195%	Normal	IPCC (2006)
Combustion efficiency Savanna woodlands (early dry season burns) (all)	0.40	93%	93%	Normal	IPCC (2006)

Parameter	Mean	Relative Uncertainty Low (%)	Relative Uncertainty High (%)	Distribution	Data Source
Combustion efficiency Savanna woodlands (mid/late dry season burns) (all)	0.74	99%	99%	Normal	IPCC (2006)
Savanna woodland (mid/late)	0.72	270%	270%	Normal	IPCC (2006)
Tropical savanna	0.73	598%	598%	Normal	IPCC (2006)
Other savanna woodlands	0.68	931%	931%	Normal	IPCC (2006)
Combustion efficiency Savanna grasslands (early dry season burns) (all)	0.74	183%	183%	Normal	IPCC (2006)
Tropical/sub-tropical grassland	0.74	270%	270%	Normal	IPCC (2006)
Tropical/sub-tropical grassland	0.92	151%	151%	Normal	IPCC (2006)
Tropical pasture	0.35	427%	427%	Normal	IPCC (2006)
Savanna	0.86	85%	85%	Normal	IPCC (2006)

3.5.9 CO_2 from Urea Fertilizer Applications

Method for Estimating CO_2 Emissions from Urea Fertilizer Application

- This method uses IPCC equation and emission factors developed by de Klein et al. (2006).
- This method uses entity specific annual parcel data as input into the IPCC equation (e.g., the amount of urea fertilizer applied to soils).
- This method assumes that the source of CO_2 used to manufacture urea is fossil fuel CO_2 captured during NH_3 manufacture.

3.5.9.1 Rationale for Selected Method

Urea fertilizer application to soils contributes CO_2 emissions to the atmosphere. The source of the CO_2 that is incorporated into the urea during the fertilizer production process is from fossil fuel sources in the U.S. fertilizer plants. The CO_2 captured during the production process is considered an emissions removal in the manufacturer's reporting so its release following urea fertilization on soils is included in the farm-scale entity reporting. If manufacturers do not estimate CO_2 capture during urea production and include the recaptured CO_2 as an emission, there is no need for a farm-scale entity to report release.

The Tier 1 method has been adopted from the IPCC (de Klein et al., 2006). No other methods have been developed or tested sufficiently for an operational system.

3.5.9.2 Description of Method

The model to estimate CO_2 emissions from urea application has been adopted from the methodology developed by the IPCC and uses the IPCC default emission factor (de Klein et al., 2006). The following equation is used to estimate the CO_2 emission from a land parcel where urea-based fertilizers have been applied:

Equation 3-28: CO$_2$ Emissions from Urea Fertilization

$$C_{Urea} = M \times EF \times CO_2MW$$

Where:

C_{Urea} = Annual release of carbon from urea added to soil (metric tons CO$_2$-eq year^{-1})

M   = Annual amount of urea fertilization (metric tons urea year^{-1})

EF   = Emission factor or proportion of carbon in urea, 0.20
                 (metric ton C (metric ton urea)$^{-1}$)

CO$_2$MW = Ratio of molecular weight of CO$_2$ to carbon (44/12)
                 (metric tons CO$_2$ (metric tons C)$^{-1}$)

3.5.9.3 Activity Data

This method requires data on the amount of urea fertilizer applied to soils.

3.5.9.4 Ancillary Data

No ancillary data are needed in order to apply the method.

3.5.9.5 Model Output

Model output is generated on both an absolute quantity of emissions and emissions intensity. The latter is based on the amount of emissions per unit of yield for crops in cropland systems or animal products in grazing systems. The emissions intensity is estimated with the following equation:

Equation 3-29: Emissions Intensity from Urea Fertilization

$$EI_{Urea} = C_{Urea}/Y$$

Where:

EI_{Urea} = Emissions intensity (metric tons CO$_2$ per metric ton dry matter crop yield, metric tons CO$_2$ per kg carcass yield, metric tons CO$_2$ per kg fluid milk yield)

C_{Urea} = Annual change in soil carbon stocks due to urea application (metric tons CO$_2$ year^{-1})

Y   = Total yield of crop (metric tons dry matter crop yield year^{-1}), meat (kg carcass yield year^{-1}), or milk production (kg fluid milk yield year^{-1})

Yields are based on the total amount of product from the land managed with urea application.

3.5.9.6 Limitations and Uncertainty

Urea (CO(NH$_2$)$_2$) is converted into ammonium and CO$_2$ in the presence of water and the enzyme urease. The CO$_2$ will dissolve in water to form carbonate, bicarbonate, and carbonic acid as a function of soil pH and temperature. Some of the bicarbonate may be transferred to groundwater, waterways, and eventually the ocean, and therefore reduce the CO$_2$ emissions to the atmosphere (de Klein et al., 2006; Hamilton et al., 2007)). However, there is insufficient information available to include this possibility in the urea method, so it is assumed that any increase in bicarbonate will lead to production of CO$_2$.

Uncertainty is assumed to be minor for the management activity data provided by the entity, although this may not be the case if there is limited knowledge about land use history for individual parcels. Uncertainty may also exist in the emission factor, assuming that some of the bicarbonate is not converted to CO_2. However, the method assumes all CO_2 is emitted because uncertainty estimates are not available for this emission factor. Therefore, no uncertainty is estimated for this source of GHG emissions based on this conservative assumption that all CO_2 is emitted.

3.6   Summary of Research Gaps for Crop and Grazing Land Management

This section discusses research gaps associated with cropland and grazing land management impacts on soil carbon stock changes and GHG emissions. The list is not necessarily exhaustive, but highlights some key gaps that will need further research before there is sufficient evidence for additional criteria to be included in the methodology. In general, the majority of prior experimental efforts have focused on components of GHGs, but few studies have been conducted on total GHG budgets to include CO_2, N_2O, and CH_4 in combination, which is needed to quantify interacting effects on the net emissions of these gases (Liebig et al., 2010). In addition, limited research has been conducted to address the influence of catastrophic weather events on GHG emissions, such as major floods, tornadoes, and hurricanes.

Carbon Stocks:[10] The following processes and practices require further study to improve the fundamental understanding or fill data gaps in the carbon inventory methods. In particular, deficiencies in understanding continue to undermine the development of robust estimates of net GHG emissions in rangelands and pastures. Such deficiencies stem from a lack of measurements across the major grassland ecoregions, as well as limitations associated with basic understanding of mechanistic processes related to GHG fluxes. There are also major gaps with respect to agroforestry, woody plant encroachment, and perennial woody crop systems.

- More data on allometric relationships for agroforestry, woody plant encroachment, and perennial woody crop systems, such as orchards.
- Improved ability to quantify the influence of agroforestry, woody plant encroachment, and perennial woody crops on soil organic carbon stocks, including optimal density of trees, the type of trees, and the landscape position of silvopasture systems.
- Improved mechanistic understanding and ability to quantify the fate of carbon with transport and sedimentation following erosion events.
- Field estimates of the amount of carbon added to soils through dynamic replacement on erodible lands.
- Improved mechanistic understanding of carbon dynamics in the subsoil horizons.
- Further study on the effect of irrigation on plant production and decomposition to quantify the net effect on soil organic carbon stocks.
- Further research on the variation in types and residence times of biochar amendments, in addition to biochar impact on other GHG emissions, priming of soil organic matter decomposition, and the overall physical breakdown and disintegration of biochar over time (Jaffé et al., 2013).
- Data on long-term responses of soil organic carbon to variation in stocking rate, grazing method (i.e., continuous, rotational, short-duration rotational, and ultra-high stocking density), and vegetation composition (i.e., forb and grass mixtures, cool- and warm-season grass mixtures, grass and legume mixtures, grass and woody mixtures, and plant architecture types), and whether these responses are mediated by different soils types, climatic conditions, botanical composition, grazing method used, fertilizer regime, etc.

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[10] Except agroforestry carbon stock changes, which are covered later in this section.

- Further study to address mitigation of GHGs in arid rangelands, particularly in shrublands, including interactions between management and environmental conditions (Ingram et al., 2008). Additional data collection and model improvement are also needed in arid rangelands, as uncertainty is extremely large for the soil carbon sequestration estimates associated with reduced stocking rates and seeding of legumes (Brown et al., 2010; Brown, 2010). Our basic knowledge of carbon sequestration and GHG mitigation in arid and semiarid environments is limited, and the effect of management is relatively understudied.

- Need for life-cycle assessment of grazing systems with particular attention to balance of soil organic carbon, N_2O emissions from soil, and CH_4 emissions from ruminants and soil, depending on stocking rate, stocking method, forage type associated with quality of intake, and environmental conditions of grazing system.

- Data from adaptive management approaches to inform understanding of soil organic carbon sequestration and GHG emissions under different grazing management strategies. This approach could help strengthen conservation-oriented programs to obtain greater impact for reducing GHG emissions and sequestering soil organic C.

- Additional field experiments and data on soil carbon emissions resulting from the combined application of lime and nitrogen fertilizers.

Soil Nitrous Oxide Emissions: The following practices have, in some studies, significantly affected N_2O emissions, but require additional research in side-by-side comparison studies across different soil types and climate, especially for extensively grown row crops that receive high levels of nitrogen fertilizers (corn and wheat in particular):

- Effects of split or delayed nitrogen applications on lowering N_2O fluxes and on increasing NUE to provide equivalent yields at lower total nitrogen input.

- Capacity of spatially precise fertilizer application technology (variable rate applicators) to lower N_2O fluxes (both direct and indirect) and increase NUE.

- Effects of banded nitrogen fertilizer applications, shown in some studies to increase NUE and in others to increase N_2O emissions.

- The generalizability of higher N_2O EFs and nitrate loss at nitrogen fertilizer rates greater than crop needs (i.e., at rates greater than those recommended by Maximum Return to Nitrogen approaches).

- The generalizability of different fertilizer formulations on N_2O emissions, in particular for urea vs. anhydrous ammonia vs. injected solutions.

- The generalizability of coated fertilizers such as polymer coated urea, urease inhibitors, biochar additions, and nitrification inhibitors for lowering N_2O emissions and nitrate loss.

- More research on the responses of soil N_2O emissions to variations in stocking rates, grazing methods (continuous, rotational, short-duration rotational, and ultra-high stocking density), and vegetation composition (forb and grass mixtures, cool- and warm-season grass mixtures, grass and legume mixtures, grass and woody mixtures, and plant architecture types), both individually and in combinations.

- The potential for mobile water and shelter sources in pastures to reduce N_2O emissions by allowing for a more even distribution of manure.

- Influence of crop residue harvesting on N_2O emissions, as well as soil organic carbon stocks, given the interest in using crop residues as a feedstock for bioenergy production.

- Influence of cover crops on N_2O emissions, including effects of plant type (e.g., legume vs. nonlegume) and residue management (e.g., harvested vs. incorporated).

- Influence of manure and compost on N_2O emissions insofar as effects may differ from synthetic nitrogen inputs with respect to rate, timing, placement, and form of organic nitrogen added (e.g., liquid vs. dry manure vs. compost with different C:N ratios).

- Improved quantification of spatial and temporal variation of N_2O emissions in different cropping systems and landscapes to provide a more accurate assessment of seasonal and annual emissions across whole fields.
- Improved estimates of indirect emissions, and in particular the percentage of nitrogen that is lost from a field through volatilization or leaching/runoff, and later converted to N_2O in downstream and downwind ecosystems. Additional study on practices that can reduce nitrate losses as well as practices that can reduce NH_3 and NO_x losses.

Research is also needed to improve modeling and empirical quantification of soil N_2O emissions in order to provide estimates of N_2O fluxes that integrate across multiple management practices simultaneously:

- Further development and validation of quantitative simulation models capable of accurately predicting N_2O fluxes in response to differing management practices, with particular respect to rate, timing, placement, and formulation of added fertilizers, both synthetic and organic; tillage type and intensity; and residue management.
- More data regarding seasonal and annual N_2O emissions, including emissions during the non-growing season and in particular winter and freeze-thaw periods.
- Better knowledge of fluxes across all Land Resource Regions (LRRs) concentrated especially in those areas and cropping and grazed systems expected to contribute most to local and regional N_2O fluxes, with side-by-side comparisons of different management practices.
- Development of standardized methodologies and creation of new technologies for rapid assessment of N_2O fluxes in the field.
- An improved understanding of the sources of N_2O in cropped soils (e.g., nitrification vs. denitrification) and consequences for feedbacks among adaptive management, soil physical and biological attributes, and SOC dynamics.
- Development of a set of geographically stratified test sites at which factors known to affect agronomic N_2O emissions could be tested in the context of different management systems. This would provide a robust empirical dataset for establishing Tier 2 and 3 models.

Flooded Rice Production Emissions: The primary research gap is the limited amount of research conducted in the United States on GHG from rice systems. Therefore, most of the current conclusions about management influences on rice CH_4 emissions are based on Asian studies where rice is transplanted as opposed to direct seeded. This may be problematic because water is managed differently in Asian transplanted flooded rice systems during the establishment period than in U.S. systems. Until recently, no studies evaluated seasonal or annual N_2O emissions from rice systems in the United States (Adviento-Borbe et al., 2007; Pittelkow et al., 2013). In the United States, much of the research on GHG emissions comes from Louisiana, Texas, and California. Lindau's lab conducted onstation research in Louisiana to evaluate CH_4 emissions (e.g., Lindau et al., 1995; Lindau et al., 1998). Sass's group also evaluated CH_4 emissions on experimental stations in Texas (e.g., Huang et al., 1997; Sass et al., 1994). In California, various researcher groups (e.g., Bossio et al., 1999; Fitzgerald et al., 2000) have been conducting research both onstation and offstation and have recently also included N_2O measurements (Adviento-Borbe et al., 2007; Pittelkow et al., 2013).

The following practices have in some studies significantly affected CH_4 or N_2O emissions but require further side-by-side comparisons with experimental designs across different soil types and climates within the United States.

- Water management practices (in particular midseason drains or intermittent irrigation) are often suggested as viable options to mitigate CH_4 emissions. While data support this conclusion, these management practices have not been widely tested in the United States. In

studies where the soil has been drained during the season, investigators have reported delayed crop maturation (a problem in temperate climates with relatively short growing seasons), reduced yields and grain quality, and increased weed and disease pressure. Therefore, although midseason drainage is mentioned as a mitigation option, more research is required before it is recommended for use in U.S. rice systems.

- Returning rice straw to soil often results in increased CH_4 emissions, but the removal of straw requires energy and time. Further compounding the problem is that there are relatively few uses for rice straw. The removal of rice straw also removes nutrients which would need to be replaced. Of particular concern is potassium, as rice straw contains an average of 1.4 percent of potassium. Therefore, it is possible to remove more than 100 kg/ha of potassium through removal of rice straw, which will need to be replaced in order to maintain a sustainable cropping system.

- In California, farmers typically incorporate rice straw and flood to facilitate straw decomposition during the winter. This practice increases CH_4 emissions from rice fields during the winter and the following growing season. However, it has also significantly improved habitat for overwintering waterfowl in the Pacific Flyway. Fitzgerald et al. (2000) reported that up to half of the annual CH_4 emissions occurred during the winter fallow period when straw was incorporated and flooded. Recent studies suggest that 50 percent may be a high estimate and that further research is needed (Adviento-Borbe et al., 2007; Pittelkow et al., 2013).

- While many studies have shown varietal differences in how much CH_4 is emitted, these studies are all relatively old and many of the varieties are no longer widely used. Further research on current varieties needs to be conducted.

- Limited data on nitrogen placement suggests that deep placement of fertilizer reduces CH_4 emissions, but more research is needed to confirm the findings.

- Side-by-side comparisons with experimental designs are needed of wet- and dry-seeded rice to evaluate their influence on CH_4 and N_2O emissions. These are the two most common rice establishment practices in the United States.

- Some studies from China suggest that more carbon is sequestered in rice systems than in upland (aerobic) systems, but this has not been evaluated in the United States.

Agroforestry: A sufficient database for developing the methods to readily measure and/or model the various GHG impacts of agroforestry is currently lacking. Full GHG monitoring and accounting in agroforestry will require a mix of methodologies from among the GHG accounting frameworks because of the diversity in uses associated with agroforestry systems. The following research gaps are highlighted.

- Assessment of approaches for estimating woody biomass in agroforestry plantings, which includes comparison of existing equations and lookup tables with agroforestry-generated volume and biomass equations to determine best approach for estimating carbon in the woody biomass of agroforestry plantings.

- Development of effective strategies for measuring/monitoring carbon sequestration and GHG emissions in soil and woody components.

- Effect of different species mixtures and combinations of management activities on soil carbon sequestration and minimizing total GHG emissions.

- Impact of management options and environment interactions on carbon sequestration and total GHG emissions within agroforestry systems.

- Development of tools relevant to the inventory/measurement/estimation of these "trees outside of forests." In addition, testing the validity of current carbon accounting tools (e.g., DAYCENT, HOLOS) in providing accurate estimates of carbon sequestered in the woody biomass of agroforestry plantings.

- Understanding soil carbon dynamics in agroforestry systems, along with the impact of soil erosion, transport and deposition on carbon stocks.
- Developing inventory methodologies (such as the use of Light Detection and Ranging) to establish a cost-effective national agroforestry inventory compatible for inclusion with current inventories contributing to regional/national GHG assessments.
- Developing standardized experimental procedures, measurement, and monitoring protocols, such as those being developed through the Greenhouse Gas Reduction through Agricultural Carbon Enhancement network (GRACEnet)[11] to agroforestry practices with the standardized measurement and monitoring for agricultural N_2O emissions.

Methane Oxidation in Soils: Soil CH_4 oxidation is known to decrease by ~70 percent upon conversion of longstanding natural vegetation to crop and pastureland (see Section 3.5.5). CH_4 oxidation rates for soils under natural vegetation are not well known for all climates and soils, so additional measurements would be useful. As with N_2O, the further development and validation of quantitative simulation models capable of accurately predicting CH_4 fluxes would also be helpful for better generalizing effects and for future inclusion of factors that may be discovered to restore oxidation in cropped soils. There is also limited research on the effect of grazing land management on CH_4 oxidation although variation in stocking rates, grazing methods, and associated practices may have an influence on this process.

Inorganic Soil Carbon: The effect of management on soil inorganic carbon dynamics and exchange of CO_2 with the atmosphere is also in need of further research. The following list is a brief summary of some of the key gaps identified for quantification of GHG emissions:

- When inorganic carbon is added to soil as agricultural lime or as a breakdown product of urea, part of the inorganic carbon becomes bicarbonate. Improved understanding of the fate of this bicarbonate in different soils and landscapes would help to better characterize the presence and strength of the resulting bicarbonate CO_2 sink.
- Improved quantification of emissions or uptake of atmospheric CO_2 with addition of carbonate limes to soils will require methods to determine the dominance of weathering due to carbonic acid (H_2CO_3) vs. the stronger nitric acid (HNO_3) in cropland and grazing land soils.
- Improved mechanistic understanding and quantification of inorganic carbon dynamics are needed in irrigated systems, as well as in nonirrigated systems—particularly in arid and semiarid regions.

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[11] GRACEnet is a research program initiated by USDA Agricultural Research Service to "identify and further develop agricultural practices that will enhance carbon sequestration in soils, promote sustainability, and provide a sound scientific basis for carbon credits and trading programs" (USDA ARS, 2013).

Appendix 3-A: Soil N$_2$O Modeling Framework Specifications

Soil N$_2$O emissions are estimated using a combination of process-based modeling, empirical scalars based on experimental data, and scaling factors for practices influencing the N$_2$O emissions as represented in the base emission rates (Section 3.5.4.1, Equations 3-8 and 3-9, and Text box 3-1). This appendix provides more information about the process-based models, in addition to the derivation of empirical scalars and the practice-based scaling factors.

DAYCENT and DNDC models were used to estimate N$_2$O emissions for the typical fertilizer rate and a 0-level nitrogen fertilization rate associated with major crops in each USDA LRR. Crops simulated are listed in Table 3-A. 1; base emission rates for other crops (e.g., sugar cane, millet, rye) were estimated using the Tier 1 emission factor (one percent of nitrogen inputs). To estimate emission factors from the model output, the N$_2$O emissions at the 0-level addition was subtracted from the N$_2$O emission for the typical fertilization rate. The difference was then divided by the synthetic agronomic nitrogen input to estimate the emission factor at the typical rate of fertilization. Scalars were used to scale the N$_2$O emissions for fertilization rates that were greater than the typical rate. The scalars were derived from empirical data based on the change in emission factors across a range of fertilization rates. See Text box 3-1 for more information about how the resulting emission factors were used to estimate base emission rates for the direct soil N$_2$O method.

Meta-analyses were used to derive practice-based scaling factors from experimental data. The scaling factors were used to adjust the base emission rates for specific practices that influence soil N$_2$O emissions. The scaling factors included the effect of nitrification inhibitors (S_{inh}), slow-release fertilizers (S_{SR}), pasture/range/paddock manure (S_{PRP}), and tillage (S_{till}). The resulting scaling factors are used in Equation 3-9 to scale the base emission rates for land parcels managed with these practices.

Figure 3-A.1 provides an overview of the decisions and steps involved in estimating N$_2$O emissions from mineral soils.

Figure 3-A.1: Decision Tree for Estimating N₂O Emissions from Mineral Soils
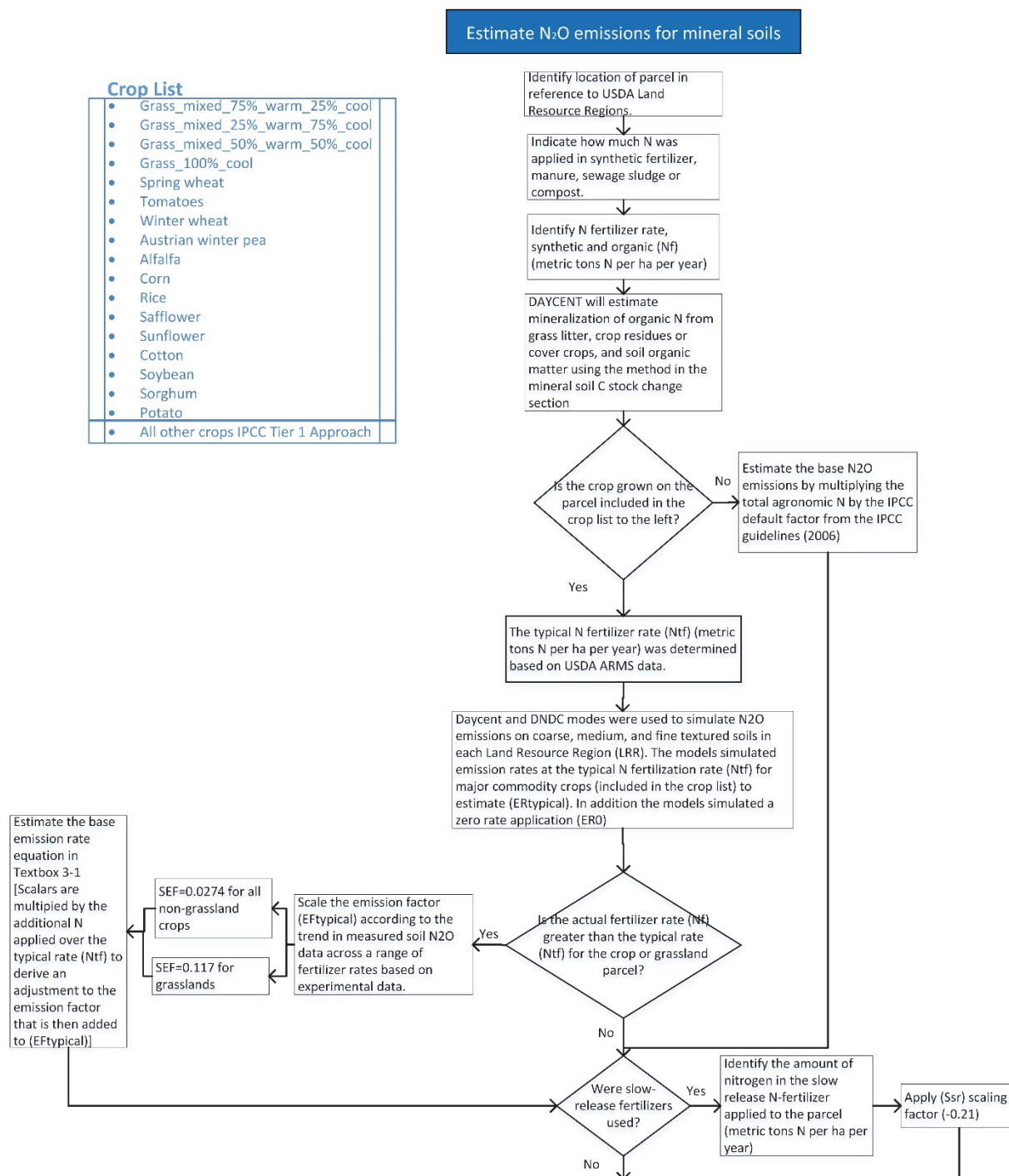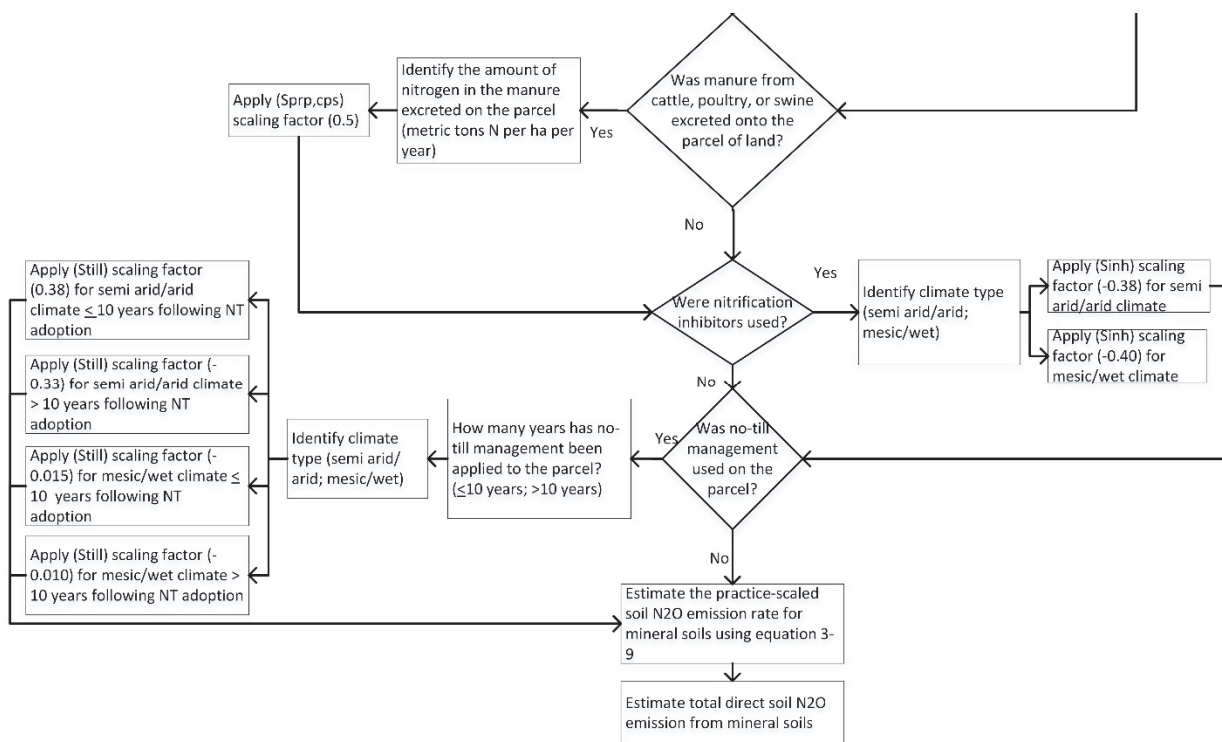
3-A.1 Description of Process-Based Models

DAYCENT[12] is a general terrestrial biogeochemical model that simulates carbon and nitrogen transformations involved in primary productivity, decomposition and nutrient dynamics (Del Grosso et al., 2000b; Parton et al., 2001). The model also simulates heat and water fluxes vertically through the soil profile (one-dimensional). Lateral flow of water is not simulated except that overland runoff occurs when rainfall events of sufficient magnitude occur given the permeability of the surface soil layer. Key submodels include plant growth with dynamic carbon allocation among plant components, soil organic matter decomposition and nutrient mineralization, and N_2O emissions from nitrification and denitrification. Plant growth is controlled by nutrient availability, soil water and temperature, and vegetation type specific parameters controlling maximum plant growth rates, maximum/minimum C:N ratios of biomass components, and phenology. Decomposition of senesced plant material and soil organic matter is controlled by the quality and quantity of litter inputs, soil texture, water, and temperature. N_2O emissions are controlled by soil NH_4 and NO_3, water content, temperature, gas diffusivity, and labile carbon availability. Land management/disturbance events such as cultivation, water and nutrient additions, fire, and grazing, can be readily implemented in the model. The model has been applied to simulate soil GHG fluxes at scales ranging from plots to regions to the globe (Del Grosso et al., 2010; Del Grosso et al., 2005; Del Grosso et al., 2009). The ability of DAYCENT to simulate crop yields, SOM, N_2O emissions, and NO_3 leaching has been tested against a variety of field experiments in cropland and grassland in the United States (Del Grosso et al., 2005; Del Grosso et al., 2008a; Del Grosso et al., 2008b).

DNDC[13] is a process-based biogeochemical model that is used to predict plant growth and production, carbon and nitrogen balance, and generation and emission of soil-borne trace gases by

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[12] The version of DAYCENT coded and parameterized for the U.S. National GHG inventory (U.S. EPA, 2013) was used to derive expected base emission rates.

[13] DNDC 9.5 compiled on Feb. 25, 2013, was used to derive expected base emission rates.

means of simulating carbon and nitrogen dynamics in natural and agricultural ecosystems (Li et al., 2000; Miehle et al., 2006; Stang et al., 2000) and forested wetlands (Zhang et al., 2002). The model integrates decomposition, nitrification-denitrification, photosynthesis and hydrothermal balance with the ecosystem. These components are mainly driven by environmental factors, including climate, soil, vegetation, and management practices. The model has been tested and used for estimating GHG emissions from forested ecosystems in a wide range of climatic regions, including boreal, temperate, subtropical, and tropical (Kesik et al., 2006; Kiese et al., 2005; Kurbatova et al., 2008; Li et al., 2004; Stang et al., 2000; Zhang et al., 2002), and similarly for grasslands and cultivated wetlands (Giltrap et al., 2010; Rafique et al., 2011).

Model inputs, for both models, include the weather data,[14] soil characteristics, and management data for these simulations. A total of 1,200 samples were drawn for cropland site simulations and another 1,200 samples for grassland site simulations. The sample number was originally determined from a plan to select three soil types from 20 counties dominated by agriculture in each of 20 LRRs (3 x 20 x 20 = 1,200). The emission rates that were produced by both models will be available online in supplementary material files. An example of the rates for corn, winter wheat, and grass are given in Figure 3-A. 2.

Figure 3-A. 2: Example of Median Base Emission Rates for Corn, Winter Wheat, and Grass Production in Land Resource Regions with Coarse, Medium, and Fine Textured Soils
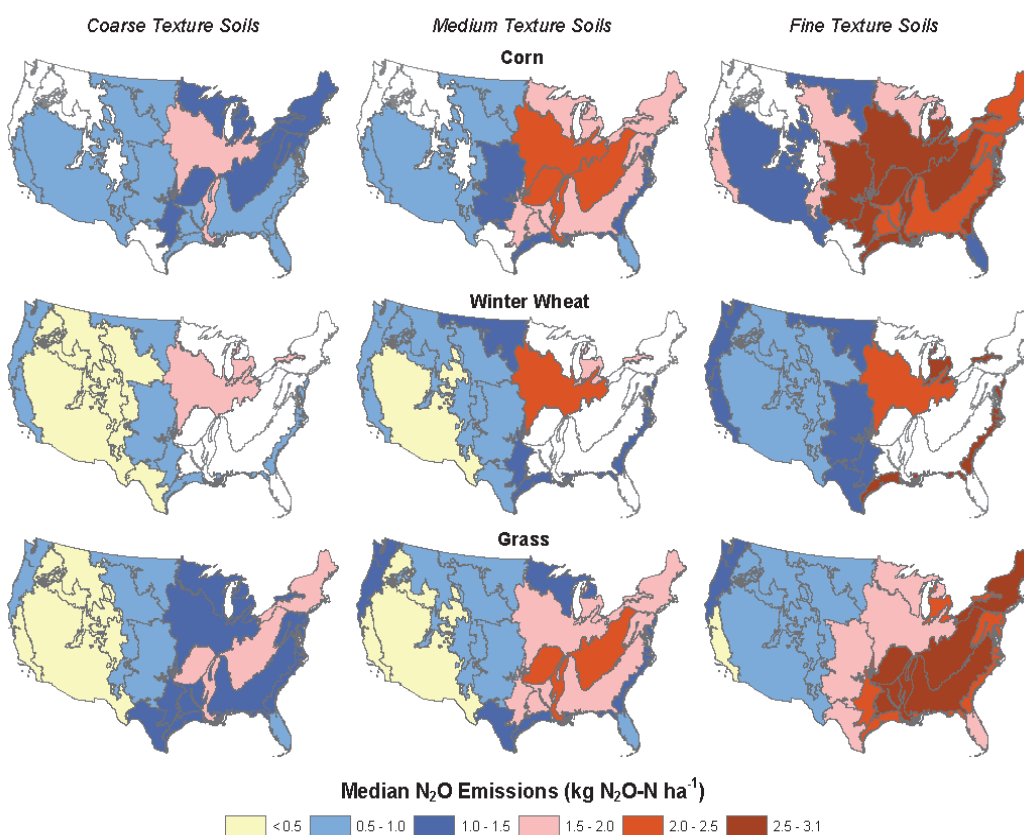
Table 3-A. 1 provides the 2.5, 50, and 97.5 percentile base emission rates for each crop, LRR, and soil texture combination. Emission rates are kgN_2O-N per ha when crops are fertilized at typical nitrogen rates.

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[14] The models used DAYMET weather for the centroid of grassland/cropland in each county.

Table 3-A. 1 Base Emission Rate (kg N_2O-N ha^{-1}) Percentiles by Land Resource Region (LRR), Crop, and Soil Texture at Typical Nitrogen Fertilizer Rates

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
A	Grass	Coarse	0.02	0.56	5.28
A	Grass	Medium	0.41	1.20	3.86
A	Grass	Fine	0.49	1.34	5.30
A	Tomato	Coarse	0.04	1.08	4.83
A	Tomato	Medium	0.28	1.69	8.31
A	Tomato	Fine	0.49	2.09	15.73
A	Wheat, Spring	Coarse	0.03	0.61	3.53
A	Wheat, Spring	Medium	0.16	1.00	2.87
A	Wheat, Spring	Fine	0.40	1.32	3.50
A	Wheat, Winter	Coarse	0.05	0.55	4.00
A	Wheat, Winter	Medium	0.19	0.91	2.99
A	Wheat, Winter	Fine	0.35	1.21	2.77
B	Grass	Coarse	0.01	0.40	5.25
B	Grass	Medium	0.02	0.45	5.41
B	Grass	Fine	0.05	0.74	8.20
B	Pea	Coarse	0.00	0.36	2.43
B	Pea	Medium	0.00	0.61	3.80
B	Pea	Fine	0.02	0.53	3.02
B	Wheat, Spring	Coarse	0.00	0.49	2.71
B	Wheat, Spring	Medium	0.01	0.80	4.43
B	Wheat, Spring	Fine	0.04	0.87	3.56
B	Wheat, Winter	Coarse	0.00	0.40	2.05
B	Wheat, Winter	Medium	0.01	0.54	3.58
B	Wheat, Winter	Fine	0.04	0.75	3.72
C	Alfalfa	Coarse	0.01	0.58	0.99
C	Alfalfa	Medium	0.01	0.66	1.60
C	Alfalfa	Fine	0.00	0.86	2.25
C	Corn	Coarse	0.21	0.78	3.00
C	Corn	Medium	0.27	0.93	8.23
C	Corn	Fine	0.60	1.60	12.96
C	Grass	Coarse	0.05	0.32	1.17
C	Grass	Medium	0.08	0.36	1.37
C	Grass	Fine	0.07	0.42	1.16
C	Rice	Coarse	0.04	0.63	1.34
C	Rice	Medium	0.03	0.70	2.19
C	Rice	Fine	0.02	0.95	7.50
C	Safflower	Coarse	0.17	0.89	2.86
C	Safflower	Medium	0.38	1.15	7.46
C	Safflower	Fine	0.56	2.09	12.92
C	Sunflower	Coarse	0.07	0.58	2.13
C	Sunflower	Medium	0.15	0.73	6.45
C	Sunflower	Fine	0.29	1.37	9.16
C	Tomato	Coarse	0.48	1.15	2.90
C	Tomato	Medium	0.57	1.21	8.01
C	Tomato	Fine	0.79	2.25	18.94
C	Wheat, Winter	Coarse	0.05	0.86	1.81
C	Wheat, Winter	Medium	0.06	0.96	3.30
C	Wheat, Winter	Fine	0.15	1.47	5.08

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
D	Alfalfa	Coarse	0.01	0.55	1.47
D	Alfalfa	Medium	0.01	0.49	2.91
D	Alfalfa	Fine	0.01	0.67	4.79
D	Corn	Coarse	0.20	0.85	2.03
D	Corn	Medium	0.26	0.87	3.28
D	Corn	Fine	0.30	1.32	5.99
D	Cotton	Coarse	0.01	1.04	2.53
D	Cotton	Medium	0.02	0.97	3.37
D	Cotton	Fine	0.09	1.63	5.68
D	Grass	Coarse	0.02	0.39	3.14
D	Grass	Medium	0.02	0.46	6.27
D	Grass	Fine	0.05	0.55	6.91
D	Wheat, Winter	Coarse	0.00	0.35	1.27
D	Wheat, Winter	Medium	0.00	0.36	2.21
D	Wheat, Winter	Fine	0.04	0.56	5.10
E	Grass	Coarse	0.01	0.46	7.35
E	Grass	Medium	0.02	0.63	8.00
E	Grass	Fine	0.12	0.66	5.52
E	Wheat, Spring	Coarse	0.02	0.59	2.46
E	Wheat, Spring	Medium	0.05	0.70	4.67
E	Wheat, Spring	Fine	0.07	0.87	2.92
E	Wheat, Winter	Coarse	0.02	0.39	1.97
E	Wheat, Winter	Medium	0.06	0.53	4.80
E	Wheat, Winter	Fine	0.10	0.63	2.89
F	Corn	Coarse	0.28	0.76	1.57
F	Corn	Medium	0.36	0.92	2.92
F	Corn	Fine	0.45	1.29	4.92
F	Grass	Coarse	0.12	0.57	2.80
F	Grass	Medium	0.15	0.66	2.69
F	Grass	Fine	0.16	0.80	3.52
F	Soybean	Coarse	0.20	0.95	3.26
F	Soybean	Medium	0.26	1.05	3.23
F	Soybean	Fine	0.29	1.48	4.40
F	Wheat, Spring	Coarse	0.10	0.69	1.85
F	Wheat, Spring	Medium	0.11	0.93	2.92
F	Wheat, Spring	Fine	0.12	1.19	4.90
F	Wheat, Winter	Coarse	0.14	0.85	3.17
F	Wheat, Winter	Medium	0.19	1.03	6.43
F	Wheat, Winter	Fine	0.18	1.41	11.05
G	Corn	Coarse	0.11	0.69	1.88
G	Corn	Medium	0.16	0.90	3.41
G	Corn	Fine	0.23	1.62	6.59
G	Grass	Coarse	0.09	0.55	1.85
G	Grass	Medium	0.09	0.54	1.92
G	Grass	Fine	0.18	0.91	3.67
G	Wheat, Winter	Coarse	0.08	0.49	1.64
G	Wheat, Winter	Medium	0.09	0.64	2.05
G	Wheat, Winter	Fine	0.10	0.91	4.43
H	Corn	Coarse	0.31	0.92	5.62
H	Corn	Medium	0.62	1.49	11.03
H	Corn	Fine	0.81	2.67	20.40

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
H	Cotton	Coarse	0.14	0.70	2.28
H	Cotton	Medium	0.18	1.17	4.38
H	Cotton	Fine	0.41	1.55	8.88
H	Grass	Coarse	0.30	0.88	2.53
H	Grass	Medium	0.29	0.95	3.53
H	Grass	Fine	0.57	1.64	4.34
H	Wheat, Winter	Coarse	0.15	0.65	2.29
H	Wheat, Winter	Medium	0.21	0.99	3.81
H	Wheat, Winter	Fine	0.32	1.30	9.16
I	Cotton	Coarse	0.25	0.63	4.38
I	Cotton	Medium	0.23	0.63	8.15
I	Cotton	Fine	0.34	1.27	8.70
I	Grass	Coarse	0.36	1.02	4.24
I	Grass	Medium	0.42	1.09	5.49
I	Grass	Fine	0.56	1.90	5.27
I	Sorghum	Coarse	0.34	0.78	5.69
I	Sorghum	Medium	0.31	0.79	8.75
I	Sorghum	Fine	0.43	1.60	9.35
I	Wheat, Spring	Coarse	0.38	0.78	6.87
I	Wheat, Spring	Medium	0.41	0.82	12.28
I	Wheat, Spring	Fine	0.60	1.60	15.24
I	Wheat, Winter	Coarse	0.19	0.43	4.66
I	Wheat, Winter	Medium	0.20	0.58	6.57
I	Wheat, Winter	Fine	0.22	1.06	7.75
J	Corn	Coarse	0.48	1.10	4.33
J	Corn	Medium	0.61	1.54	7.48
J	Corn	Fine	0.71	2.63	17.71
J	Grass	Coarse	0.48	1.41	3.95
J	Grass	Medium	0.61	1.86	5.13
J	Grass	Fine	0.69	2.41	5.77
J	Sorghum	Coarse	0.35	0.90	3.81
J	Sorghum	Medium	0.47	1.31	6.67
J	Sorghum	Fine	0.52	1.96	14.66
J	Wheat, Spring	Coarse	0.37	0.89	3.65
J	Wheat, Spring	Medium	0.48	1.30	5.93
J	Wheat, Spring	Fine	0.72	2.31	13.76
J	Wheat, Winter	Coarse	0.24	0.80	3.30
J	Wheat, Winter	Medium	0.33	1.02	5.63
J	Wheat, Winter	Fine	0.32	1.13	11.65
K	Alfalfa	Coarse	0.16	0.90	2.35
K	Alfalfa	Medium	0.28	1.39	2.95
K	Alfalfa	Fine	0.16	1.25	2.96
K	Corn	Coarse	0.40	1.14	2.41
K	Corn	Medium	0.72	1.75	4.57
K	Corn	Fine	0.45	1.81	5.27
K	Grass	Coarse	0.35	1.07	3.77
K	Grass	Medium	0.56	1.45	4.17
K	Grass	Fine	0.35	1.54	5.64
K	Soybean	Coarse	0.26	0.94	2.07
K	Soybean	Medium	0.57	1.37	2.80
K	Soybean	Fine	0.37	1.43	3.35

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
K	Wheat, Spring	Coarse	0.35	1.04	2.33
K	Wheat, Spring	Medium	0.77	1.65	4.58
K	Wheat, Spring	Fine	0.46	1.79	5.19
L	Corn	Coarse	0.41	1.42	3.31
L	Corn	Medium	0.63	1.97	5.92
L	Corn	Fine	1.36	3.09	15.09
L	Grass	Coarse	0.47	1.39	6.01
L	Grass	Medium	0.56	1.82	7.02
L	Grass	Fine	0.63	2.08	6.61
L	Soybean	Coarse	0.31	1.29	2.45
L	Soybean	Medium	0.45	1.66	3.10
L	Soybean	Fine	0.95	2.31	6.22
L	Wheat, Winter	Coarse	0.44	1.65	3.14
L	Wheat, Winter	Medium	0.54	1.97	3.34
L	Wheat, Winter	Fine	1.06	2.75	8.73
M	Corn	Coarse	0.55	1.51	4.33
M	Corn	Medium	0.87	2.28	11.87
M	Corn	Fine	0.99	2.76	15.46
M	Grass	Coarse	0.49	1.31	4.06
M	Grass	Medium	0.68	1.91	4.97
M	Grass	Fine	0.65	1.94	5.19
M	Soybean	Coarse	0.41	1.29	2.66
M	Soybean	Medium	0.71	1.86	5.03
M	Soybean	Fine	0.78	2.08	7.52
M	Wheat, Winter	Coarse	0.55	1.62	2.91
M	Wheat, Winter	Medium	0.85	2.16	5.17
M	Wheat, Winter	Fine	0.84	2.45	7.72
N	Corn	Coarse	0.60	1.48	12.11
N	Corn	Medium	0.76	2.11	19.17
N	Corn	Fine	1.14	2.80	32.82
N	Grass	Coarse	0.42	1.64	3.94
N	Grass	Medium	0.57	2.08	5.03
N	Grass	Fine	0.91	2.61	5.95
N	Soybean	Coarse	0.58	1.31	4.04
N	Soybean	Medium	0.73	1.80	5.24
N	Soybean	Fine	1.00	2.07	11.18
O	Corn	Coarse	0.60	1.55	4.52
O	Corn	Medium	0.67	2.14	9.63
O	Corn	Fine	1.07	3.08	24.03
O	Cotton	Coarse	0.51	1.19	4.95
O	Cotton	Medium	0.61	1.84	14.76
O	Cotton	Fine	0.99	3.24	25.42
O	Grass	Coarse	0.39	1.70	3.92
O	Grass	Medium	0.44	2.24	7.03
O	Grass	Fine	0.76	2.81	7.97
O	Rice	Coarse	0.52	1.11	5.15
O	Rice	Medium	0.73	1.29	9.18
O	Rice	Fine	1.00	2.45	11.14
O	Soybean	Coarse	0.53	1.22	3.73
O	Soybean	Medium	0.55	1.66	6.67
O	Soybean	Fine	0.86	2.18	14.83

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
P	Corn	Coarse	0.43	0.93	4.56
P	Corn	Medium	0.60	1.85	12.27
P	Corn	Fine	0.76	2.23	27.80
P	Cotton	Coarse	0.37	0.81	4.04
P	Cotton	Medium	0.63	1.68	10.68
P	Cotton	Fine	0.73	2.18	20.32
P	Grass	Coarse	0.29	1.26	4.30
P	Grass	Medium	0.41	1.95	5.44
P	Grass	Fine	0.50	2.79	7.47
P	Soybean	Coarse	0.36	0.80	2.98
P	Soybean	Medium	0.56	1.65	5.62
P	Soybean	Fine	0.67	1.72	12.55
R	Alfalfa	Coarse	0.09	1.35	3.01
R	Alfalfa	Medium	0.26	1.63	3.10
R	Alfalfa	Fine	0.25	1.85	3.61
R	Corn	Coarse	0.25	1.35	2.84
R	Corn	Medium	0.51	1.81	4.92
R	Corn	Fine	0.53	2.25	4.97
R	Grass	Coarse	0.30	1.77	7.53
R	Grass	Medium	0.49	1.96	7.25
R	Grass	Fine	0.56	2.82	9.59
R	Soybean	Coarse	0.20	1.24	2.69
R	Soybean	Medium	0.45	1.62	3.06
R	Soybean	Fine	0.41	1.95	3.80
S	Alfalfa	Coarse	0.16	1.03	2.23
S	Alfalfa	Medium	0.36	1.54	2.99
S	Alfalfa	Fine	0.44	1.53	3.44
S	Corn	Coarse	0.44	1.14	2.84
S	Corn	Medium	0.86	1.81	6.89
S	Corn	Fine	0.97	2.20	12.36
S	Grass	Coarse	0.60	1.37	3.02
S	Grass	Medium	0.77	1.85	4.99
S	Grass	Fine	0.93	2.35	6.43
S	Soybean	Coarse	0.39	1.04	1.66
S	Soybean	Medium	0.77	1.59	3.48
S	Soybean	Fine	0.89	1.78	4.72
T	Corn	Coarse	0.45	0.92	5.78
T	Corn	Medium	0.48	1.15	11.08
T	Corn	Fine	0.63	2.76	24.52
T	Grass	Coarse	0.33	1.05	4.89
T	Grass	Medium	0.41	1.23	8.49
T	Grass	Fine	0.50	2.32	9.65
T	Soybean	Coarse	0.40	0.81	4.06
T	Soybean	Medium	0.48	0.98	8.03
T	Soybean	Fine	0.50	1.79	17.49
T	Wheat, Winter	Coarse	0.33	0.81	4.89
T	Wheat, Winter	Medium	0.36	1.10	8.05
T	Wheat, Winter	Fine	0.46	2.72	17.87
U	Corn	Coarse	0.36	0.64	2.64
U	Corn	Medium	0.34	0.66	4.67
U	Corn	Fine	0.47	1.18	14.76

LRR	Crop	Soil Group	Emission Rate (25th Percentile)	Emission Rate (50th Percentile)	Emission Rate (97.5th Percentile)
U	Grass	Coarse	0.33	0.99	4.74
U	Grass	Medium	0.35	0.79	4.09
U	Grass	Fine	0.39	1.72	5.90
U	Potato	Coarse	0.57	0.82	2.53
U	Potato	Medium	0.63	1.05	13.93
U	Potato	Fine	0.79	1.53	13.88
U	Wheat, Spring	Coarse	0.23	0.55	2.08
U	Wheat, Spring	Medium	0.30	0.54	5.11
U	Wheat, Spring	Fine	0.32	0.84	10.58

3-A.2 Empirical Scalars for Base Emission Rates

As described in Text box 3-1, the base emission rate modeled by DAYCENT and DNDC is used to calculate an emission factor for the typical fertilizer case that is then scaled to reflect the increase in emission factor with increasing nitrogen inputs (S_{EF} in Text box 3-1). To calculate S_{EF} a meta-analysis was performed using data from all field studies in the literature where at least three different levels of nitrogen input, including a zero nitrogen rate, were applied to the same crop at the same site during the same growing season. Emission factors were calculated as the difference between the N_2O fluxes at 0N and at xN divided by the N_2O flux at 0N. The null hypothesis was that emission factors will be constant across different nitrogen rates.

A total of 44 data sets that meet the base criteria were identified. From each data set, slopes for each fertilizer addition interval were calculated and compared to the slope of the first interval (0N to the first nitrogen addition level). The value of the slope is a measure of how much the emission factor changes per additional unit of nitrogen fertilizer input (kg N ha^{-1}) for a given study site year. Thus, the slope measures the degree of nonlinearity of the emission factor. The slope is zero if the emission factor is constant, as assumed by the IPCC Tier 1 method. A positive slope indicates that the total emission function is convex with respect to total nitrogen input, i.e., that the unit of flux increase (the emission factor) is greater with each successive unit of nitrogen input. Uncertainty was quantified with a confidence interval obtained by performing a bootstrap analysis (n=100,000) on the original slopes.

There were sufficient data to analyze five different sub-categories: corn, grassland, other crops, clay-textured soils, and other-textured soils. The mean slope was significantly greater than zero for all analyzed categories but only the grassland category was significantly different from the others. Thus in the ER_b equation in Text box 3-1 there are two values for S_{EF}, one for grasslands and another for all other crops.

The studies used in the meta-analysis are provided below.

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Letica, S.A., C.A.M. de Klein, C.J. Hoogendoorn, R.W. Tillman, et al. 2010. Short-term measurement of N_2O emissions from sheep-grazed pasture receiving increasing rates of fertiliser nitrogen in Otago, New Zealand. *Animal Production Science*, 50:17-24.

Lin, S., J. Iqbal, R. Hu, J. Wu, et al. 2011. Nitrous oxide emissions from rape field as affected by nitrogen fertilizer management: a case study in central China. *Atmospheric Environment*, 45:1775-1779.

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McSwiney, C.P., and G.P. Robertson. 2005. Nonlinear response of N_2O flux to incremental fertilizer addition in a continuous maize (Zea mays L.) cropping system. *Global Change Biology*, 11(10):1712-1719.

Mosier, A.R., A.D. Halvorson, C.A. Reule, and X.J. Liu. 2006. Net global warming potential and greenhouse gas intensity in irrigated cropping systems in northeastern Colorado. *Journal of Environmental Quality*, 35(4):1584-1598.

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van Groenigen, J.W., G.J. Kasper, G.L. Velthof, A. van den Pol-van Dasselar, et al. 2004. Nitrous oxide emissions from silage maize fields under different mineral nitrogen fertilizer and slurry applications. *Plant and Soil*, 263.

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3-A.3 Practice-Based Scaling Factors

Data were analyzed to derive scaling factors for the following practices: nitrogen fertilizer placement, nitrification inhibitors, no-till management, and slow-release fertilizers. Practices were included if there was sufficient evidence from field experiments to suggest that the practice influenced N_2O emissions, or for which a previous meta-analysis had been conducted and shown that the practice had an effect on N_2O emissions (i.e., no-till management; van Kessel et al., 2012). All practices were found to have a significant effect on N_2O emission with the exception of nitrogen placement. The scaling factors are provided in Table 3-9.

Documentation for the no-till scaling factor can be found in van Kessel et al. Scaling factors for nitrification inhibitors were derived using a linear mixed-effect modeling approach (Pinheiro and Bates, 2000), similar to the method used by Ogle et al. (2005) to derive factors that were used in the 2006 IPCC Guidelines (IPCC, 2006). Variances associated with individual experimental results were not taken into consideration in the meta-analyses because many studies did not provide this information. A goal for future analyses supporting the USDA methods will be to include variances, under the assumption that studies will report this information in future publications. Covariates were included in the analysis to determine if the practice had a different effect depending on the land use, climate, soil type, water management, tillage practice, or crop type. Covariates were retained in the model if the variable was significant at an alpha level of 0.05. For other scaling factors, there were insufficient data to use the linear mixed-effect modeling approach, and so average differences between the control and treatments were estimated from the studies to estimate a scaling factor. The resulting estimates were evaluated for statistical significant from a value of 0 (or no effect) using an alpha level of 0.05. A 95 percent confidence interval was derived for each scaling factor and provided in Table 3-6 as an upper and lower bound on the estimated factor.

The studies used in each meta-analysis are provided below.

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Appendix 3-B: Guidance for Crops Not Included in the DAYCENT Model

The DAYCENT model is recommended for use in estimating Soil Carbon Stock Changes (Section 3.5.3), and was used (along with the DNDC model) to generate base emission rates for Equation 3-9 (See Appendix 3-A for a discussion of how models were used to estimate N_2O emissions from mineral soils). In addition, nitrogen mineralized from soil organic matter (N_{min}); additional nitrogen inputs from a change in soil organic matter mineralization due to a land-use change or tillage change (N_{dmin}); nitrogen mineralization from organic amendments (e.g., manure, sewage sludge, compost); and nitrogen mineralization from crop, grass, and cover crop residues (N_{resid}) are generated by the DAYCENT model.

The DAYCENT model is not used to generate estimates for all crops grown in the United States. The DAYCENT model is currently used to estimate SOC stocks for the following crops and sectors: agroforestry, almond, alfalfa, windbreak, woodlot, sorghum, spring wheat, winter wheat, woodlot—softwoods, woodlot—hardwoods, clover, cotton, dryland beans, corn, oats, millet, grass-clover pasture, grass, peas, potato, sugar beets, sunflower, soybean, sugar cane, peanut, tobacco, upland rice, windbreak three-row, and walnut. These crops represent 90 percent of the crops grown in the United States, and more crops are tested and added to the DAYCENT model-based assessment on a regular basis.

However, if an entity is managing a crop that is not included in the DAYCENT list of crops, the 2006 IPCC Guidelines may be used to estimate emissions or sinks for the sources listed above. This approach is consistent with the U.S. Environmental Protection Agency National Inventory Report (U.S. Environmental Protection Agency, 2013), and a complete discussion of this alternative methodology in provided in Annex 3 (Section 3.12) of the National Inventory Report. [15] Specifically, the National Inventory Report uses a combination of Tier 1, 2, and 3 approaches to estimate direct and indirect N_2O emissions and soil changes in agricultural soils. This report follows the same approach for the crops not included in the DAYCENT model when estimating soil carbon stock changes and direct N_2O emissions (See Table 3-B- 1).

Table 3-B- 1 Alternative Methodologies for Crops Not Included in the DAYCENT Model

Source	Tier 1	Tier 2
Soil carbon stock changes	IPCC 2006 Guidelines (See Chapter 5, Section 5.2.3.3)	
Direct N_2O emissions from mineral soils for the crops NOT estimated by the DAYCENT model		IPCC 2006 Guidelines with management based scaling factors (See Section 3.5.4)
$N_{smin,}$	Not estimated	
Nitrogen inputs from organic amendments (N_{man} and N_{comp})	IPCC 2006 Guidelines (See Chapter 11 Section 11.2.1.1)	
N_{resid}		Equation 3-B-1 Residue nitrogen (See below)

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[15] See U.S. Environmental Protection Agency, National GHG Inventory Annex 3: http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2013-Annex-3-Additional-Source-or-Sink-Categories.pdf

Equation 3-B-1: Residue N

For Crops:

$$N_{resid} = [((Y_{dm} / HI) - Y_{dm}) \times (1 - R_r) \times N_a] + [(Y_{dm} / HI) \times R{:}S \times N_b]$$

For Grazing Forage:

$$N_{resid} = [Y_{dm} \times (1 - F_r - R_r) \times N_a] + [Y_{dm} \times R{:}S \times N_b]$$

Where:

N_{resid} = Nitrogen in residues above and belowground on the parcel of land (metric tons N year^{-1} ha^{-1})

Y_{dm} = Crop harvest or forage yield, corrected for moisture content (metric tons biomass ha^{-1})
= Y x DM

Y = Crop harvest or total forage yield (metric tons biomass ha^{-1})

DM = Dry matter content of harvested biomass (dimensionless)

HI = Harvest Index (dimensionless)

F_r = Proportion of live forage removed by grazing animals (dimensionless)

R_r = Proportion of crop/forage residue removed due to harvest, burning or grazing (dimensionless)

N_a = Nitrogen fraction of aboveground residue biomass for the crop or forage (dimensionless)

N_b = Nitrogen fraction of belowground residue biomass for the crop or forage (dimensionless)

R:S = Root-shoot ratio (unitless)

Default values for dry matter content, root:shoot ratio and harvest index are provided in Table 3-5 in Section 3.5.1.2. Default values from the IPCC guidelines values are provided in Table 3-B-2 for the nitrogen content of aboveground and belowground residues in major crop types and individual crops.

Table 3-B-2: Nitrogen Content of Aboveground and Belowground Residues of Major and Individual Crops

Crop	Nitrogen Content of Aboveground Residues (kg N (kg dm)$^{-1}$)	Nitrogen Content of Belowground Residues (kg N (kg dm)$^{-1}$)
Major crop types		
Grains	0.006	0.009
Beans and pulses	0.008	0.008
Grass-clover mixtures	0.025	0.016
Nitrogen-fixing forages	0.027	0.022
Non-nitrogen-fixing forages	0.015	0.012
Perennial grasses	0.015	0.012
Root crops, other	0.016	0.014
Tubers	0.019	0.014
Individual crops		
Alfalfa	0.027	0.019
Barley	0.007	0.014
Dry bean	0.01	0.01
Maize	0.006	0.007
Millet	0.007	NA
Non-legume hay	0.015	0.012
Oats	0.007	0.008
Peanut (w/pod)	0.016	NA
Potato	0.019	0.014
Rice	0.007	NA
Rye	0.005	0.011
Sorghum	0.007	0.006
Soybean	0.008	0.008
Spring wheat	0.006	0.009
Wheat	0.006	0.009
Winter wheat	0.006	0.009

Source: de Klein (2006).

Chapter 3 References

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