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

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

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Executive Summary

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QUANTIFYING GREENHOUSE GAS FLUXES IN AGRICULTURE AND FORESTRY: METHODS FOR ENTITY-SCALE INVENTORY

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Executive Summary

Background

Provisions of Section 2709 of the Food, Conservation, and Energy Act of 2008 direct the U.S. Department of Agriculture (USDA) to prepare technical guidelines and science-based methods to measure environmental service benefits from conservation and land management activities, initially focused on carbon. The methods contained in this document address greenhouse gas (GHG) emissions and removals from agricultural and forestry activities.

Through the development of this report, USDA has prepared two primary products:

1. A comprehensive review of techniques currently in use for estimating GHG emissions and removals from agricultural and forestry activities; and
2. A technical report outlining the preferred science-based approach and specific methods for estimating GHG emissions at the farm or forest scale (i.e., this document).

Uses of the Report and Methods:

- Estimating increases and decreases in GHG emissions and carbon sequestration resulting from current and future conservation programs and practices;
- Providing methods suitable for GHG inventory efforts at the entity, farm, or forest scale, with possible implications for regional and national scale assessments as well; and
- Estimating increases and decreases in GHG emissions and carbon sequestration associated with changes in land management.

Purpose of the Report

The objective for this report is to create a standard set of GHG estimation methods for use by USDA, landowners, and other stakeholders to assist them in evaluating the GHG impacts of their management decisions. The methods presented in the report address GHG emissions and carbon sequestration for the entire entity or operation and also provide the opportunity to assess individual practices or management decisions. Therefore, ease of use is critical.

A co-objective is to demonstrate capacity within USDA, establishing a standardized, consensus set of methods that become the scientific basis for entity-scale estimation of the GHG impacts of landowner management decisions. Therefore, scientific rigor and transparency are also critical.

Because the report is intended as a means of evaluating management practices across the full scope of the farm, ranch, and forest management system, the methods in the report need to be as comprehensive as possible. Research and data gaps exist that result in some management practices not being accounted for or are reflected in higher levels of estimate uncertainty. Completeness is important, though, and the report attempts to identify the most significant research gaps and data needs.

This report will be used within USDA and by farmers, ranchers, and forest landowners, and will be made publicly available. These methods are designed to:

1. Provide a scientific basis for methods that can be used by landowners and managers, USDA, and other stakeholders to estimate changes in GHG emissions and removals at the local entity scale;
2. Create a standard set of GHG quantification guidelines and methods for use by stakeholders;
3. Quantify all significant emissions and removals associated with specific source categories;
4. Quantify emissions from land-use change and carbon sequestration from land management practices and technologies; and
5. Support the development of entity-, farm-, or forest-scale GHG inventories that will facilitate the participation of landowners in public and private environmental market registries and reporting systems.

The report also serves as input into the development of a USDA GHG Estimation Tool. The report and the methods are not intended as an addition to or replacement of any current Federal GHG reporting systems or requirements.

Process for the Development of the Report

This report was developed by three author teams (i.e., working groups) under the direction of one lead author for each team (plus one co-lead author for the forestry chapter). The lead authors were chosen based on their experience with GHG inventories and accounting methodologies and their professional research experience. With input from each lead author, USDA chose 8 to12 working group members per team to write the report. These working group members each had different backgrounds that fit with the anticipated content of the document and also had experience with GHG accounting and/or field research that was unique and addressed one or more of the niche methods that were essential for ensuring the comprehensiveness of the methods for each sector. The author teams were provided with a preliminary outline of their chapters and with two background reports developed as part of the project. One background report was an analysis of the scientific literature related to rates of carbon sequestration or emissions reduction resulting from various management practices and technologies (Denef et al., 2011). The other report was a compilation of all of the available tools, protocols, and models, with basic information on each one (Denef et al., 2012).

The methods were developed according to several criteria in order to maximize their usefulness. In particular, the methods must:

1. Stand on their own, independent of any other accounting system, yet maintain consistency with other accounting systems to the maximum extent possible;
2. Be scalable for use at entity-scale sites across the United States, with applicability at county and/or State levels as well;
3. Facilitate use by USDA in assessing the performance of conservation programs;

4. Provide a broad framework to assess management practices to evaluate the GHG aspect of production sustainability;
5. Maintain maximum applicability for use in environmental markets, including possible future Federal, State, or local GHG offsets initiatives;
6. Be scientifically vetted through USDA, U.S. Government and academic expert review, and public comment;
7. Provide reliable, real, and verifiable estimates of onsite GHG emissions, carbon storage, and carbon sequestration (the methods will be designed so that over time they can be applied to quantify onsite GHG reductions and increases in carbon storage due to conservation and land management activities); and
8. Provide a basis for consistency in estimation and transparency in reporting.

Development of the report has been iterative as various drafts of the document have been put through several review stages, including a USDA intra-agency technical review, a Federal interagency technical review, a scientific expert review, and a public comment period.

Overview of Recommended GHG Estimation Methods in the Report

This section provides an overview of the current estimation methods or approaches an entity could use to estimate GHG emissions and sinks on his or her property. This overview is followed by a summary of each sector's proposed methodologies for entity GHG estimations.

There are several approaches that a farmer, rancher, or forest landowner can use to estimate GHG emissions at an entity scale, and each approach gives varying accuracy and precision. The most accurate way of estimating emissions is through direct measurement, which often requires expensive equipment or techniques that are not feasible for a single landowner or manager. On the other hand, lookup tables and estimation equations alone often do not adequately represent local variability or local conditions. This report attempts to delineate methods that balance user-friendliness, data requirements, and scientific rigor in a way that is transparent and justified.

The following approaches were considered for these guidelines:

- Basic estimation equations (cf., IPCC [Intergovernmental Panel on Climate Change] Tier 1)—involve combinations of activity data[1] with parameters and default emission factors.[2] Any default parameters or default emission factors (e.g., lookup tables) are provided in the text, or if substantial in length, in an accompanying compendium of data.
- Models (cf., IPCC Tier 3)—use combinations of activity data with parameters and default emission factors. The inputs for these models can be ancillary data[3] (e.g., temperature, precipitation, elevation, and soil nutrient levels that may be pulled from an underlying source), biological variables (e.g., plant diversity) or site-specific data (e.g., number of acres,

[1] Activity data is defined as data on the magnitude of human activity resulting in emissions or removals taking place during a given period of time (IPCC, 1997).
[2] Emission factor is defined as a coefficient that quantifies the emissions or removals of a gas per unit of activity. Emission factors are often based on a sample of measurement data, averaged to develop a representative rate of emission for a given activity level under a given set of operating conditions (IPCC, 2006).
[3] Ancillary data is defined as additional data necessary to support the selection of *activity data* and *emission factors* for the estimation and characterization of emissions. Data on soil, crop or animal types, tree species, operating conditions, and geographical location are examples of ancillary data.

number of animals). The accuracy of the models is dependent on the robustness of the model and the accuracy of the inputs.

- Field measurements—actual measurements that a farmer or landowner would need to take to more accurately estimate the properties of the soil, forest, or farm or to estimate actual emissions. Measuring actual emissions on the land requires special equipment that monitors the flow of gases from the source into the atmosphere. This equipment is not readily available to most entities, so field measurements are more often incorporated into other methods described in this section to create a hybrid approach. A field measurement such as a sample mean tree diameter could be incorporated into other models or equations to give a more accurate input.
- Inference (cf., IPCC Tier 2)—uses State, regional, or national emissions/sequestration factors that approximate emissions/sequestration per unit of the input. The input data is then multiplied by this factor to determine the total onsite emissions. This factor can have varying degrees of accuracy and often does not capture the mitigation practices on the farm or the unique soil conditions, climate, livestock diet, livestock genetics, or any farm-specific characteristics, although they can be developed with specific soil types, livestock categories, or climactic regions.
- Hybrid estimation approach (cf., IPCC Tier 2 or IPPC Tier 3)—an approach that uses a combination of the approaches described above. The approach often uses field measurements or models to generate inputs used for an inference-based approach to improve the accuracy of the estimate.

The types of approaches that the authors recommended in this report include basic estimation equations with default emission factors (cf., IPCC Tier 1); geography-, crop-, livestock-, technology-, or practice-specific emission factors (cf., IPCC Tier 2); and modified IPCC/empirical and/or process-based modeling (cf., IPCC Tier 2 or IPCC Tier 3).[4] Table ES-1 categorizes the sources of emissions with the types of approaches that are recommended in this report.

Table ES-2 summarizes the sources of agricultural and forestry GHG emissions and removals discussed in this report, the recommended method for estimating emissions and removals for each source category, and the reference(s) used for the development of the method.

[4] A tier represents a level of methodological complexity. Usually three tiers are provided. Tier 1 is the basic method, Tier 2 intermediate, and Tier 3 most demanding in terms of complexity and data requirements. Tiers 2 and 3 are sometimes referred to as *higher tier* methods and are generally considered to be more accurate (IPCC, 2006).

Table ES-1: Summary of the Sources of Emissions and Types of Approaches in this Report

Source	Basic Estimation Equation (cf., IPCC Tier 1)	Inference (cf., IPCC Tier 2)	Modified IPCC or Empirical Model (cf., IPCC Tier 2 or IPPC Tier 3)	Processed-Based Model (cf., IPPC Tier 3)
Croplands/Grazing Lands	▪ Direct N_2O Emissions from Drainage of Organic Soils ▪ CH_4 Emissions from Rice Cultivation ▪ CO_2 from Urea Fertilizer Application	▪ Soil Organic Carbon Stocks for Organic Soils ▪ CO_2 from Liming ▪ N_2O Emissions from Rice Cultivation ▪ Non-CO_2 Emissions from Biomass Burning ▪ Indirect N_2O Emissions	▪ Biomass Carbon Stock Changes ▪ CH_4 Uptake by Soils ▪ Direct N_2O Emissions from Mineral Soils	▪ Soil Organic Carbon Stocks for Mineral Soils
Wet-lands	—	—	—	▪ Biomass Carbon ▪ Soil C, N_2O, and CH_4
Animal Production[5]	▪ Enteric CH_4 from Swine ▪ Enteric CH_4 from Other Animals (Goats, American Bison) ▪ CH_4 from Poultry Housing	▪ CH_4 from Dairy Cattle, Beef Cattle, and Swine Housing ▪ CH_4 and N_2O from Aerobic Lagoons ▪ CH_4 and N_2O from Temporary Stack and Long-Term Stockpile ▪ CH_4 and N_2O from Composting	▪ Enteric CH_4 from Dairy Cattle, Sheep, Beef Cow-Calf, Bulls, Stockers, Feedlot Cattle ▪ CH_4 from Manure from Barn Floors – Dairy Cattle ▪ N_2O from Dairy Cattle, Beef Cattle, Swine, and Poultry Housing ▪ CH_4 and N_2O from Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks ▪ CH_4 and N_2O from Combined Aerobic Treatment Systems ▪ CH_4 from Anaerobic Digester	—
Forestry	—	—	▪ Establishing, Re-establishing, and Clearing Forest ▪ Harvested Wood	▪ Forest Carbon Accounting ▪ Forest Management ▪ Urban Forests

[5] Ammonia (NH_3), as an important precursor to GHGs, is included in the animal production systems discussion where necessary, but is not of primary focus. If readers are interested in more technical information, methods for estimating NH_3 emissions can be found in Appendix 5-C.

Source	Basic Estimation Equation (cf., IPCC Tier 1)	Inference (cf., IPCC Tier 2)	Modified IPCC or Empirical Model (cf., IPCC Tier 2 or IPPC Tier 3)	Processed-Based Model (cf., IPPC Tier 3)
			Products	▪ Natural Disturbance—Wildfire and Prescribed Fire
Land-use Change	▪ Annual Change in Carbon Stocks in Dead Wood and Litter Due to Land Conversion ▪ Change in Soil Organic Carbon Stocks for Mineral Soils	—	—	—

Organization of the Report

The report is largely organized by sector, with each chapter providing an overview of management practices and resulting GHG emissions and removals. For each sector, background and information on management practices are presented first, followed by the detailed methods proposed for estimating emissions and removals for those practices.

- Chapter 1 provides an overview of the report, report objectives, contents of the report, and uses and limitations of the report.
- Chapter 2 describes the linkages and cross-cutting issues relating to sector-specific and entity-scale estimation of GHG emissions and removals.
- Chapter 3 describes the GHG emissions from crop and grazing land systems. The chapter presents methods for estimating the influence of land use and management practices on GHG emissions (and removals) in crop and grazing land systems. Methods are described for estimating biomass and soil carbon stocks changes, direct and indirect soil nitrous oxide (N_2O) emissions, methane (CH_4) and N_2O emissions from wetland rice, CH_4 uptake in soils, carbon dioxide (CO_2) emissions or removals from liming, non-CO_2 GHG emissions from biomass burning, and CO_2 emissions from urea fertilizer application.
- Chapter 4 provides guidance for estimation of carbon stock changes and CH_4 and N_2O emissions from actively managed wetlands.
- Chapter 5 describes on-farm GHG emissions from the production of livestock and manure management. The chapter presents GHG estimation methods appropriate to the production of each common livestock sector (beef, dairy, sheep, swine, and poultry), with methods related to manure management combined for all livestock types.
- Chapter 6 provides guidance on estimating carbon sequestration and GHG emissions from managed forest systems. The chapter is organized to provide an overview of the elements of forest carbon accounting, including definitions of the key carbon pools and basic methods for their estimation.

- Chapter 7 provides guidance on estimating the net GHG emissions and removals resulting from changes between land types—i.e., conversions into and out of cropland, wetland, grazing land, or forest land—at the entity scale.
- Chapter 8 presents the approach for accounting for the uncertainty in the estimated net emissions based on the methods presented in this report. A Monte Carlo approach was selected as the method for estimating the uncertainty around the outputs from the methodologies in this report as it is currently the most comprehensive, sound method available to assess the uncertainty at the entity scale.

Summary

In developing this report, the authors have sought to outline the most state-of-the art and suitable science-based approaches and specific methods for estimating farm- or forest-scale GHG emissions (see Table ES-2). In some cases, the proposed methods have not previously been applied in specifically the way that is proposed. For example, the forestry systems chapter describes the integration of the Forest Vegetation Simulator (FVS) within other estimation tools for forest carbon accounting. This application of FVS, while technically sound, will require additional effort to implement. In other cases, the authors have proposed new methods that build on or enhance previously used methods. For example, a new hybrid approach is proposed for estimating direct soil N_2O emissions from mineral soils on croplands and grazing lands. The hybrid approach uses models to derive expected emission rates at the typical fertilization rate for the major soil textures, weather patterns, and crop rotation systems in each USDA Land Resource Region and uses a meta-analysis of empirical studies to develop emission scaling factors for cropland and grazing land systems. The method also applies practice-based scaling factors derived from a meta-analysis of the most recent data. This hybrid approach is the result of a workshop held in February 2012 that convened experts on N_2O emissions from croplands in order to develop estimation methods that were inclusive and best met the objectives of USDA.

In addition to proposing science-based methods, the authors also acknowledge that for certain practices and technologies, adequate data do not currently exist to accurately estimate GHG emissions and/or carbon sequestration. In each sector chapter, the authors have included a discussion of research gaps or priority areas for future data collection that are important in order to improve the completeness and accuracy of the estimation methods put forth in this report. Estimation of GHG emissions from managed wetland systems is a good example. While a method is put forward that reflects the best currently available science, the authors state in Section 4.3 that the methods for these lands are not as well developed as for other sectors. Later in that same section there is text discussing the considerable limitations to estimating GHG fluxes from these systems and the large levels of uncertainty around flux estimates. In Section 4.4, the authors outline a significant list of research and data priorities that would help to refine and strengthen the estimation methods.

In the continual effort to advance the science and improve the understanding of these complex and dynamic systems, this report provides the foundation for entity-level tools to quantify the GHG benefits from conservation and land management activities. The report also identifies priorities for future effort in order to broaden the scope of entity-scale GHG flux estimation and reduce estimation uncertainties.

Table ES-2: Summary of Source Categories, Recommended Methods, and Emission Factors in this Report

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Croplands/Grazing Lands				
Biomass Carbon Stock Changes	Herbaceous biomass is estimated with an empirical method using entity specific data as input into the IPCC[6] 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 a simulation model (DAYCENT) using entity input.	Changes in the estimated biomass carbon stock for cropland and grazing land if there is a land-use change or a change in the crop or forage species.	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 proposed for estimating the carbon fraction value. Yield in units of dry matter can be estimated by the entity, or average values from USDA-Natural Agricultural Statistics Service statistics can be used.	This method was chosen because it captures the influence of land-use change and changes in crop or forage species on biomass carbon stocks by using U.S.-specific default values where entity specific data are not available.

[6] IPCC = Intergovernmental Panel on Climate Change

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Soil Organic Carbon stocks for Mineral Soils	The DAYCENT model is used to estimate the soil organic carbon at the beginning and end of the year for mineral soils. The stocks are entered into the IPCC equations developed by Lasco et al. (2006) and Verchot et al. (2006) to estimate carbon stock changes.	Addition of carbon in manure and other organic amendments; tillage intensity; residue management (retention in field without incorporation; retention in the field with incorporation; and removal with harvest, burning, or grazing); influence of bare and vegetated fallows; irrigation effects on decomposition in cropland and grazing land systems; setting-aside cropland from production; influence of fire on oxidation of soil organic matter; and woody plant encroachment, agroforestry, and silvopasture effects on carbon inputs and outputs.	The DAYCENT model (Parton et al., 1987).	DAYCENT model has been demonstrated to represent the dynamics of soil organic carbon and estimate soil organic carbon stock change in cropland and grasslands (Parton et al., 1993). There have been uncertainties noted in the model in 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.
Soil Organic Carbon Stocks for Organic Soils	CO_2 emissions from drainage of organic soils (i.e., histosols) are estimated with an inference method (cf., IPCC Tier 2) using the IPCC equation developed by Aalde et al. (2006) and region-specific emission factors from Ogle et al. (2003).	Cropland drainage	Emission factors are from Ogle et al. (2003) and are region-specific based on typical drainage patterns and climatic controls (e.g., temperature/precipitation) on decomposition rates.	Uses entity-specific annual data as input into the equation used in the U.S. Inventory.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Direct N_2O Emissions from Mineral Soils	Direct N_2O methods are estimated with a hybrid estimation method. For major commodity crops, (e.g., corn, cotton, alfalfa) a combination of experimental data and process-based modeling using DAYCENT[7] and DE nitrification-decomposition (DNDC)[8] are used to derive expected base emission rates for different soil texture classes in each USDA Land Resource Region. For minor commodity crops (e.g., barley, oats, peanuts) 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 (i.e., 0.01) multiplied by the agronomic 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.	Nitrogen application to crops. In addition, specific management practices are included as scaling factors that influence a portion or the entire pool of mineral nitrogen.[9] Management practices that influence a portion of the emission rate include: ▪ Use of slow release formulation ▪ Nitrification inhibitor application ▪ Manure nitrogen directly deposited on pasture/ range/paddock Management practices that influence the entire pool of mineral nitrogen include: ▪ Tillage	The base emission factors are adjusted by scaling factors related to specific crop management practices that are derived from experimental data.	The method is based on using results from process-based models and measured N_2O emissions in combination with scaling factors based on U.S.-specific empirical data on a seasonal timescale.[10]

[7] 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.
[8] DNDC9.5 compiled on Feb 25, 2013 was used to derive expected base emission rates.
[9] Practice-based emission scaling factors (0 to 1) are used to adjust the portion of the emission rate associated with slow release fertilizers, nitrification inhibitors, and pasture/range/paddock (PRP) manure nitrogen additions. 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 are used to scale the entire emission rate under the assumption that this practice influences the entire pool of mineral nitrogen.
[10] A full description of the method is included in Chapter 3 and its appendix. Supplemental data outputs from the model runs will be available online to download.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Direct N_2O Emissions from Drainage of Organic Soils	Direct N_2O emissions from drainage of organic soils, i.e., histosols, are estimated with a basic estimation equation (cf., IPCC Tier 1) method (de Klein et al., 2006).	Drainage of organic soils.	Emission rate for cropped histosols based on an IPCC Tier 1 emission factor of 0.008 tonnes N_2O-nitrogen ha^{-1} $year^{-1}$.	Uses entity specific annual data as input into the equation used in the USDA Inventory (USDA, 2011).
Indirect N_2O Emissions	Indirect soil N_2O emissions are estimated with an inference (cf., IPCC Tier 2) based on IPCC methodology (de Klein et al., 2006).	Irrigation.	IPCC defaults are used for estimating the proportion of nitrogen that is subject to leaching, runoff, and volatilization. Where cropping systems with leguminous and non-leguminous winter cover crops are grown, a U.S.-specific emission factor is provided.	This method uses entity-specific seasonal data on nitrogen management practices.
Methane Uptake by Soils[11]	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 empirical model (cf., IPCC Tier 2 or IPPC Tier 3).	Land management including cultivation for crop production, grazing in grasslands, forest harvest, grassland, or forest fertilization.	Annual average CH_4 oxidation emissions and removals are from the data set used by Del Grosso et al. (2000).	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.

[11] Methane uptake by soils is a natural process in undisturbed soils. Processes for restoring methanotrophic activity are not well understood, and require decades to develop. A method is outlined in this report, but additional data and understanding are required prior to use or implementation in quantification tools.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Methane and Nitrous Oxide Emissions from Rice Cultivation	A basic estimation equation (cf., IPCC Tier 1) is used to estimate CH_4, and an inference (cf., IPCC Tier 2) method is used for N_2O emissions from flooded rice production (Akiyama et al., 2005; de Klein et al., 2006; Lasco et al., 2006; USDA, 2011).	CH_4: scaling factors are differentiated by hydrological context (e.g., irrigated, rain fed, upland (i.e., dry soil)—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., more than 180 days, less than 30 days) and type of organic amendment (e.g., compost, farmyard manure). N_2O: additions from mineral fertilizers, organic amendments, and crop residues.	CH_4: 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. CH_4 scaling factors to account for water regimes come from Lasco et al. (2006). N_2O: 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).	The N_2O method uses the IPCC (2006) equation with the addition of a scaling factor for drainage from Akiyama et al. (2005). The method for methane emissions uses entity-specific annual data as input into the equation and is consistent with U.S. Inventory method.
CO_2 from Liming	An inference (cf., 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).	The amount of lime, crushed limestone, or dolomite applied to soils.	U.S.-specific emissions factors (West and McBride, 2005).	Uses U.S.-specific emission factors as annual input into the IPCC equation, which is consistent with the U.S. Inventory.
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 an inference (cf., IPCC Tier 2) method (Aalde et al., 2006).	Area burned.	Emission factors are from values in the IPCC guidelines (Aalde et al., 2006) and West et al. (2010) for the residue:yield ratios.	Uses entity-specific annual data as input into the IPCC equation.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
CO_2 from Urea Fertilizer Application	CO_2 emissions from application of urea or urea-based fertilizers to soils are estimated with a basic estimation equation (cf., IPCC Tier 1) method (de Klein et al., 2006).	The amount of urea fertilizer applied to soils.	Emission factors are from values in the IPCC guidelines (de Klein et al., 2006). This method assumes that the source of CO_2 used to manufacture urea is fossil fuel CO_2 captured during NH_3 manufacture.	Uses entity-specific annual data as input into the IPCC equation, which is used for the U.S. Inventory.
Wetlands				
Biomass Carbon in Wetlands	Methods for estimating forest vegetation and shrub and grassland vegetation biomass carbon stocks use a combination of the Forest Vegetation Simulator (FVS) model and lookup tables for dominant shrub and grassland vegetation types found in the Cropland and Grazing Land Chapter. If there is a land-use change, methods for cropland herbaceous biomass are suggested.	*Forested Wetlands*: Same as those described for upland forests in Section 6.2.3. *Shrub and Grassland Vegetation*: Same as those described for total biomass carbon stock changes presented in the Cropland and Grazing Land Chapter, Section 3.5.1.	*Forest Wetlands*: Regional variants are available for FVS that allow for region-specific focus on species and forest vegetation communities. The driver for productivity is the availability of site index curves, and the regional variants include many wetland tree species. However, if a species-specific curve is not available, then a default function is used to estimate carbon stock changes. *Shrub and Grassland Vegetation:* Same as the Croplands and Grazing Lands Chapter, Section 3.5.1.	Uses entity-specific seasonal data. No IPCC methodologies currently exist for this source; hence, this is a newly developed method.
Soil Carbon, N_2O, and CH_4 in Wetlands	The DeNitrification-DeComposition (DNDC) process-based biogeochemical model is the method used for estimating soil carbon, N_2O, and CH_4 emissions from wetlands.	Vegetation management, water management regime, soil management, fertilization practices, and land-use history.	Process based model is used; hence, no emissions factors are used in this method.	This method leverages the DNDC model to simulate soil carbon, N_2O, and CH_4 emissions from wetlands on a seasonal timescale.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Animal Production Systems				
Enteric Fermentation				
Mature Dairy Cows	Mits3 equation developed by Mills et al. (2003) and further utilized by DairyGEM (Rotz et al., 2011). Mits3 equation is based primarily on metabolizable energy intake. Dry matter intake (DMI), starch, acid detergent fiber, crude protein, and total digestible nutrients provide the inputs for the equation.	*Dietary changes:* increasing DMI, using fibrous concentrate rather than starch concentrate, feeding rapidly degraded starch (such as barley), and addition of dietary fat. *Activity changes:* confining currently grazing animals, fewer work hours per day, fewer days on feed prior to slaughter.	Emission factors calculated with approach developed by Mills et al. (2003) and Rotz et al. (2011).	Use of the DairyGEM/Mits3 equation is recommended over the IPCC Tier 2 equation (2006) because it has proven to be more accurate, in general, for dairy cows.
Beef Cow-Calf and Bulls	IPCC Tier 2 approach (2006). The calculation considers weight, weight gain, mature weight, pregnancy, lactation, other activity (grazing, confined, daily work), and the energy content of the animals' diets.	*Dietary changes:* increasing DMI, using fibrous concentrate rather than starch concentrate, feeding rapidly degraded starch (such as barley), and addition of dietary fat. *Activity changes:* confining currently grazing animals, fewer work hours per day.	Emission factors are determined with the IPCC Tier 2 equation (2006). Methane conversion factor (Ym) based on animal-specific guidance in U.S. EPA (2013).	The equations utilized are the same as existing inventory methods; however, the methods utilize farm-specific feed types and utilize monthly, rather than annual, level data (i.e., account for seasonal variation in forage quality).
Stockers	IPCC Tier 2 approach (2006). The calculation considers weight, weight gain, mature weight, pregnancy, lactation, other activity (grazing, confined, daily work), and the energy content of the animals' diets.	*Dietary changes:* increasing DMI, using fibrous concentrate rather than starch concentrate, feeding rapidly degraded starch (such as barley), and addition of dietary fat. *Activity changes:* confining currently grazing animals, fewer work hours per day, fewer days on feed prior to slaughter.	Emission factors are determined with the IPCC Tier 2 equation (2006) on an entity-by-entity basis. Ym based on animal-specific guidance in U.S. EPA (2013).	The equations utilized are the same as existing inventory methods; however, the methods utilize farm-specific feed types and utilize monthly, rather than annual, level data (i.e., account for seasonal variation in forage quality).

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Feedlot Cattle	IPCC Tier 2 approach (2006). The calculation considers weight, weight gain, mature weight, pregnancy, lactation, other activity (grazing, confined, daily work), and the energy content of the animals' diets.	*Dietary changes:* increasing DMI, using fibrous concentrate rather than starch concentrate, feeding rapidly degraded starch (such as barley), and addition of dietary fat. *Activity changes:* confining currently grazing animals, fewer work hours per day, fewer days on feed prior to slaughter.	Emission factors are determined with the IPCC Tier 2 equation (2006). Ym based on guidance developed by Hales (2012).	The calculation considers weight, weight gain, mature weight, pregnancy, lactation, other activity (grazing, confined, daily work), and the energy content of the animals' diets.
Sheep	Howden equation (Howden et al., 1994), based on dietary DMI.	Dietary changes, but no well-developed research due to difficulty of obtaining accurate feed-intake estimates for grazing sheep.	The equation from Howden et al. (1994) estimates emissions based solely on DMI; hence, emission factors not utilized.	This method uses actual monthly estimates of DMI, rather than head count, as utilized by the IPCC Tier 1 equation (2006).
Swine	IPCC Tier 1 approach (2006).	None.	Utilizes IPCC Tier 1 emission factor (IPCC, 2006).	None.
Other Animals (Goats, American Bison)	IPCC Tier 1 approach for American bison (based on buffalo, modified by average animal weight) and goats (IPCC, 2006).	None.	Utilizes IPCC Tier 1 emission factors(IPCC, 2006).	None.
Housing				
Methane Emissions from Manure on Barn Floors for Dairy Cattle	DairyGEM (a subset of the Integrated Farm Systems Model) is used to estimate CH_4 emissions.	None.	Empirical relationship as provided in Chianese et al. (Chianese et al., 2009).	Utilizes climate and entity characteristics.
Methane Emissions from Dairy Cattle, Beef Cattle, and Swine Housing	IPCC Tier 2 approach.	Type and duration of manure storage.	Utilizes a combination of IPCC and U.S. EPA Inventory emission factors.	None.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Nitrous Oxide Emissions from Dairy Cattle, Beef Cattle, Swine, and Poultry Housing	IPCC Tier 2 approach, using American Society of Agricultural Engineers (ASAE) equations to estimate nitrogen excretion and default values for ammonia losses to account for nitrogen balance.	Animal diets and type of manure storage.	Utilizes IPCC emission factors (IPCC, 2006) and ammonia losses from Koelsh and Stowell (2005).	Uses nitrogen balance approach to adjust nitrogen in housing to account for ammonia losses.
Methane Emissions from Poultry Housing	IPCC Tier 1 approach.	None.	Utilizes IPCC emission factors that vary by temperature and whether manure is managed as dry manure or as a liquid (IPCC, 2006).	Of the models evaluated for poultry, an estimate of confidence for output was only available for the IPCC Tier 1 approach. Specific to estimates of poultry, on manure CH_4 emissions, the uncertainty was less than 20% (Little et al., 2008).
Manure Storage and Treatment				
Solid Manure Storage and Treatment - Temporary Stack and Long-Term Stockpile				
Methane Emissions	IPCC Tier 2 approach using IPCC and U.S. EPA Inventory emission factors, utilizing monthly data on volatile solids and dry manure.	Animal diets.	Utilizes a combination of IPCC and U.S. EPA Inventory emission factors.	Uses U.S.-specific emission factors and takes into account diet characterization.
Nitrous Oxide Emissions	IPCC Tier 2 approach using U.S. EPA Inventory emission factors and monthly data on total nitrogen, and dry manure.	Duration of manure storage and animal diets.	Utilizes emission factors from U.S. EPA Inventory.	Uses U.S.-specific emission factors and takes into account diet characterization.
Manure Storage and Treatment-Composting				
Methane Emissions	IPCC Tier 2 approach utilizing monthly data on volatile solids and dry manure.	Configuration of storage unit (e.g., composting in-vessel, static pile, intensive windrow, passive windrow) and animal diets.	Utilizes emission factors from IPCC.	Takes into account diet and climate characteristics.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Nitrous Oxide Emissions	IPCC Tier 2 approach utilizing data on total initial nitrogen and dry manure.	Manure handling (i.e., no mix or active mix) and animal diets.	Utilizes emission factors from IPCC.	Takes into account diet and climate characteristics.
Liquid Manure Storage and Treatment – Aerobic Lagoon				
Methane Emissions	The methane correction factor for aerobic treatment is negligible and was designated as 0% in accordance with the IPCC.	Not applicable.	Utilizes emission factors from IPCC.	Not estimated.
Nitrous Oxide Emissions	IPCC Tier 2 method.	Configuration of storage (e.g., volume of lagoon), natural or forced aeration, and animal diets.	Utilizes emission factors from IPCC.	None.
Liquid Manure Storage and Treatment – Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks				
Methane Emissions	Sommer Model (Sommer et al., 2004) is used with degradable and non-degradable fractions of volatile solids from Møller et al. (2004).	Configuration of storage unit (e.g., covered or uncovered storage, presence or absence of crust) and animal diets.	Parameters for estimation from Sommer et al. (2004).	Takes into account diet and storage temperature characteristics.
Nitrous Oxide Emissions	Emissions are a function of the exposed surface area and U.S.-based emission factors.	Configuration of storage unit (e.g., surface area of manure).	Utilizes emission factors from Rotz et al. (2011a).	Utilizes U.S.-specific emission factors.
Liquid Manure Storage and Treatment – Anaerobic Digestion with Biogas Utilization				
Methane Emissions	Leakage from anaerobic digestion system is estimated using IPCC Tier 2 approach and system-specific emission factors.	Configuration of digester (e.g., steel or lined concrete or fiberglass digesters) and animal diets.	Utilizes emission factors from CDM (CDM, 2012).	Takes into account system design and diets.
Combined Aerobic Treatment Systems				

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Methane Emissions	Assumed to be 10 percent of the emissions resulting from method to estimate emissions from Liquid Manure Storage and Treatment – Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks.	Configuration of storage unit (e.g., covered or uncovered storage, presence or absence of crust) and animal diets.	Parameters for estimation from Sommer et al. (2004).	Takes into account diet and storage temperature characteristics.
Nitrous Oxide Emissions	Assumed to be 10 percent of the emissions resulting from method to estimate emissions from Liquid Manure Storage and Treatment – Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks.	Configuration of storage unit (e.g., surface area of manure).	Utilizes emission factors from Rotz et al. (2011a).	Uses U.S.-specific emission factors.
Liquid Manure Storage and Treatment – Sand/Manure Separation	No method provided as GHG emissions are negligible. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.	Not applicable.	Not applicable.	Not applicable.
Liquid Manure Storage and Treatment – Nutrient Removal	Not estimated due to limited quantitative information on GHGs from nitrogen removal processes.	Not applicable.	Not applicable.	Not applicable.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Liquid Manure Storage and Treatment – Solid/Liquid Separation	No method provided as GHG emissions are negligible. Efficiency factors for different mechanical solid-liquid separation systems provided. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.	Not applicable.	Not applicable.	Not applicable.
Liquid Manure Storage and Treatment – Constructed Wetlands				
GHG Removals	Currently no method is provided, although GHG removals are noted to likely be greater than CH_4 and N_2O emissions, which are considered negligible.	Not applicable.	Not applicable.	Not applicable.
Solid Manure Storage and Treatment – Thermo-Chemical Conversion	Not estimated as CH_4 and N_2O emissions considered negligible.	Not applicable.	Not applicable.	Not applicable.
Manure Application				

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Solid Manure Application Systems (manure handling prior to land application)	Not estimated due to limited quantitative information on GHGs from manure mixing and removal from storage systems or during transport to fields where manure is land applied.	Not applicable.	Not applicable.	Not applicable.
Liquid Manure Application Systems (manure handling prior to land application)	No method is provided as CH_4 and N_2O GHG emissions are negligible; however, CO_2 emissions would result from the operation of equipment.	Not applicable.	Not applicable.	Not applicable.
Forestry				
Forest Carbon	Methods include: (1) FVS model with Fire and Fuels Extension module (FVS-FFE) with Jenkins et al. (2003) allometric equations; and (2) default look up tables.	FVS-FFE models hundreds of management practices (thinning from below/above/evenly through a stand, thinning with species preference, conditional thinning/planting/ regeneration, piling of surface fuels and prescribed fires, salvage operations, mastication treatments, insect/disease management, etc.)	Allometric equations are from Jenkins et al. (2003); default look up tables from Smith et al. (2006).	The method allows large landowners to estimate base year carbon stocks from field surveys and repeat the field survey at recommended intervals (e.g., 5-year, 10-year) depending on the region/forest type group. Small landowners estimate carbon stocks from lookup tables based on USDA Forest Inventory and Analysis program data, which is comparable to other GHG methodologies (e.g., Section 1605(b) Guidance).

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Establishing, Re-establishing, and Clearing Forest	IPCC equations developed by Aalde et al. (2006); with Jenkins et al. (2003) allometric equations.	Planting trees on previously unforested lands; replanting trees on previously forested lands; and permanently clearing trees from forested lands.	Allometric equations are from Jenkins et al. (2003)	This method allows large landowners to estimate base year carbon stocks from field surveys and repeat the field survey at recommended intervals (e.g., 5-year, 10-year) depending on the region/forest type group. The National Inventory Report (NIR) uses a carbon stock change method, which explicitly includes the establishment, re-establishment, and clearing of forests.
Forest Management	Methods include: (1) FVS-FFE with Jenkins et al. (2003) allometric equations and (2) default lookup tables of management practice scenarios.	Stand density management; site preparation techniques; vegetation control; planting; natural regeneration; fertilization; selection of rotation length; harvesting and utilization techniques; fire and fuel load management; reducing risk of emissions from pests and disease; short-rotation woody crops.	Default lookup tables of carbon stocks over time by region, forest type categories, including species group (e.g., hardwood, softwood, mixed); regeneration (e.g., planted, naturally regenerated); management intensity (e.g., low, moderate, high, very high) and site productivity (e.g., low, high), to be developed as a supporting product using FVS.	This method provides a consistent and comparable set of carbon stocks for each region, forest type group, management intensity, and site productivity over time, under management scenarios common to the forest types and management intensities.

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Harvested Wood Products	U.S.-specific harvested wood products tables developed by Skog (2008), taking the estimated average amount of harvested wood product carbon from the current year's harvest that remains stored in end uses and landfills over the next 100 years.	The approach models various management practices including the disposition of each primary product (e.g., lumber, structural panels) to major end uses (e.g., percentage of product going to residential housing, non-residential housing, manufacturing (furniture)), and percentage going to exports; with decay functions indicating how quickly products go out of use for each end use; fraction of material going out of use that goes to landfills; fraction of material in landfills that does not decay, and the decay rate for material in landfills that does decay.	WOODCARB II model used to estimate annual change in carbon stored in products and landfills (Skog, 2008).	Provides a method that is suitable to count the average amount of carbon stored in products in use and in landfills, and the underlying model is the same used for the National Inventory Report (NIR) (i.e., The NIR also uses WOODCARB II model to estimate annual change in carbon stored in products and landfills). The harvested wood product tables (Skog, 2008) provide annual values for zero to 10 years after production and 5-year intervals for 10 to 100 years after production.
Urban Forests	Methods include: (1) Field Data Method using i-Tree Eco (formerly UFORE) model; and (2) Aerial Method using i-Tree Canopy model with aerial tree cover estimates and look up tables.	Maintenance (use of vehicles, chain saws, etc.) and altering building energy use (use of trees for shading and wind breaks); quantitative methods for estimating emissions from these management practices are included for information purposes only.	i-Tree Eco model; i-Tree Canopy model.	This method provides a range of options dependent on the data availability of the entities' urban forest land. The NIR uses equations based on look up tables and average tree canopy values.
Natural Disturbance—Wildfire and Prescribed Fire	Methods include: (1) First Order Fire Effects (FOFEM) model entering measured biomass; and (2) FOFEM model using default values generated by vegetation type.	Fire and fuel load management.	FOFEM (Reinhardt et al., 1997).	This method provides a range of options dependent on the data availability of the entities' disturbed forest land. The use of a U.S.-specific fire and fuel load management model is an improvement compared to the NIR, which uses equations based on IPCC (2006).

Source	Methodology Approach	Potential Management Practices	Source of Emission Factors	Improvements Compared to Other Greenhouse Gas Methodologies
Land-use Change				
Annual Change in Carbon Stocks in Dead Wood and Litter Due to Land Conversion	A basic estimation equation (cf., IPCC Tier 1) is used to estimate change in carbon stocks in dead wood and litter (Aalde et al., 2006).	Land conversion.	IPCC 2006 Guidelines (Aalde et al., 2006).	Uses entity-specific annual data as input into the equation and is consistent with IPCC 2006 guidance.
Change in Soil Organic Carbon Stocks for Mineral Soils	The methodologies to estimate soil carbon stock changes for organic soils and mineral soils are adopted from IPCC (Aalde et al., 2006) and are a basic estimation equation.	Land conversion.	IPCC 2006 Guidelines (Aalde et al., 2006).	Uses entity-specific annual data as input into the equation and is consistent with IPCC 2006 guidance.

IPCC= Intergovernmental Panel on Climate Change

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Chapter 1 Introduction

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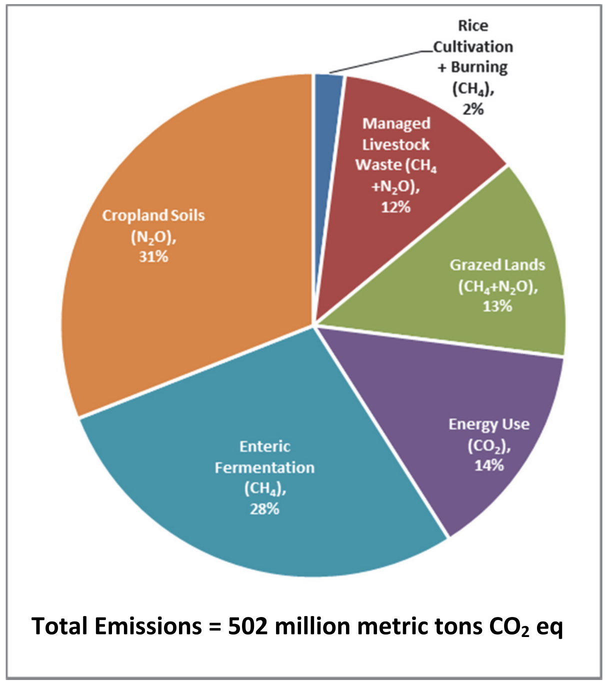
Acronyms, Chemical Formulae, and Units

C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
EO	Executive Order
EPA	U.S. Environmental Protection Agency
GHG	Greenhouse gas
ISO	International Organization for Standardization
LCA	Life cycle assessment
LCI	Life cycle inventory
N_2O	Nitrous Oxide
USDA	U.S. Department of Agriculture

1 Introduction

In 2008, agriculture contributed 6.1 percent of the total greenhouse gas (GHG) emissions in the United States (USDA, 2011).[1] The distribution of emissions across the agriculture sector is illustrated in Figure 1-1. In addition, forestry sequestered enough carbon to offset about 13 percent of total U.S. GHG emissions (USDA, 2011). Since the late 1990s, the U.S. Department of Agriculture (USDA) has analyzed and reported GHG emissions and removals via national-scale inventories, and field-scale measurement of these fluxes has been done for decades by USDA researchers. USDA also has done significant work in the development of GHG estimation models and tools within the agriculture and forestry sectors.

Figure 1-1: Agriculture Sources of Greenhouse Gas Emissions in 2008[a]



[a] Cropland soils emissions include emissions from major crops; non-major crops; histosol cultivation; and managed manure that accounts for the loss of manure nitrogen during transport, treatment, and storage, including volatilization and leaching/runoff.
Source: USDA (2011).

This report provides methods and a scientific basis for estimating GHG emissions and sequestration at the landowner, land-manager scale—entity scale. The report was authored by recognized experts from across USDA, other U.S. Government agencies, and academia and reflects estimation methods that balance scientific rigor, scale, practicality, and availability of data.

This chapter provides an overview of the report as well as the objectives set out for the project and the process used in developing the report. The remainder of the chapter is organized as follows:

- Overview of the Report
- Report Objectives
- Process for the Development of the Methods
- Contents of the Report
- Uses and Limitations of the Report
- Chapter 1 References

[1] Here the agriculture sector includes GHG emissions and removals from livestock, grasslands, croplands, and energy use on farms; it does not include GHG emissions and removals from industrial processes (e.g., fertilizer production) or from off-farm energy use (e.g., transportation fuels used in exporting commodity crops).

1.1 Overview of the Report

Under provision of Section 2709 of the Food, Conservation, and Energy Act of 2008, USDA has been directed to "establish technical guidelines that outline science-based methods to measure the environmental service benefits from conservation and land management activities in order to facilitate the participation of farmers, ranchers, and forest landowners in emerging environmental services markets." The legislation further states that the initial emphasis of the methods development should focus on GHG emissions. Agreement on that set of methods is the primary scope and purpose for this report. The findings in this report provide the foundation for entity-level tools to measure the GHG benefits from conservation and land management activities.

This report and the estimation methods are not intended as an addition to or replacement of any current Federal or State GHG reporting systems or requirements. This report has been prepared to outline methods to calculate direct GHG emissions and carbon sequestration from agriculture and forestry processes and builds upon existing inventory efforts such as U.S. Environmental Protection Agency (EPA) and USDA's national inventories and the U.S. Department of Energy's Voluntary Greenhouse Gas Reporting Program Section 1605(b) Guidelines, with an aim of providing simple, transparent, and robust inventory and reporting methods.

The report provides technical methods for estimating and reporting GHGs from significant agriculture and forestry sources and sinks. These methods are designed to quantify significant emissions and sinks associated with specific source categories as well as annual reductions in those emissions or fluxes in carbon storage resulting from land-use change and land management practices and technologies. Therefore, the report will support the development of entity-, farm-, or forest-scale GHG estimates and inventories.

Because the report is intended as a means of evaluating management practices across the full scope of the farm, ranch, and forest management system, the methods in the report need to be as comprehensive as possible. Research and data gaps exist that result in some management practices not being accounted for or are reflected in higher levels of estimate uncertainty. Completeness is important, though, and the report attempts to identify the most significant research gaps and data needs.

The methods were developed according to several criteria in order to maximize their usefulness. In particular, the methods must:

1. Stand on their own, independent of any other accounting system, yet maintain consistency with other accounting systems to the maximum extent possible;
2. Be scalable for use at entity-scale sites across the United States, with applicability at county and/or State levels as well;
3. Facilitate use by USDA in assessing the performance of conservation programs;
4. Provide a broad framework to assess management practices to evaluate the GHG aspect of production sustainability;
5. Maintain maximum applicability for use in environmental markets, including possible future Federal, State, or local GHG offsets initiatives;
6. Be scientifically vetted through USDA, U.S. government, and academic expert review and public comment;
7. Provide reliable, real, and verifiable estimates of on-site GHG emissions, carbon storage, and carbon sequestration (methods will be designed so that over time they can be applied to

quantify on-site GHG reductions and increases in carbon storage due to conservation and land management activities); and

8. Provide a basis for consistency in estimation and transparency in reporting.

1.2 Report Objectives

The objectives for this report are to create a standard set of GHG estimation methods for use by USDA, landowners, and other stakeholders and to serve as input into the development of USDA estimation tools. The methods presented in the report address GHG emissions and carbon removal for the entire entity or operation and provide the opportunity to assess individual practices or management decisions.

Greenhouse Gas Mitigation Options and Costs for Agricultural Land and Animal Production within the United States covers mitigation practices in crop production, animal production, and land retirement systems in the United States. This report reviews available scientific methods for estimating GHG sources and sinks at an entity level and recommends particular estimation methods for each livestock type and agriculture/forestry practice. To estimate the costs, USDA has developed another report that estimates the implementation costs, GHG mitigation potential at the farm level, and break-even prices (i.e., GHG incentive) for different mitigation practices on a farm level.

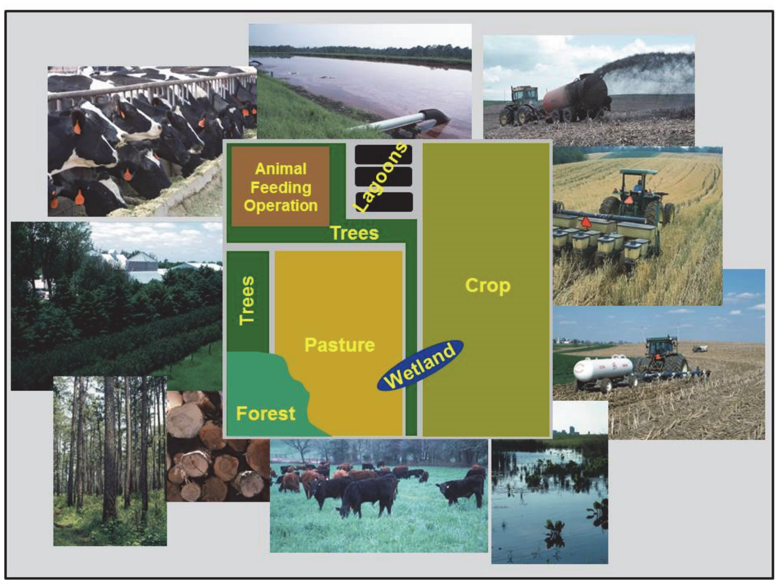
The report is available for download on the project website at: http://www.usda.gov/oce/climate_change/mitigation_technologies/GHGMitigationProduction Cost.htm.

A co-objective is to establish consensus on a standardized set of methods for the Department, which become the scientific basis for entity-scale estimation of the GHG impacts of landowner management decisions. Therefore, scientific rigor and transparency are also critical.

While USDA has long been involved in development of GHG inventories and estimation tools, this report brings together estimation approaches from all agriculture and land management sectors into one place. These methods are combined in such a way that an integrated estimate can be derived for all activities within the boundary of the farm and forest management operation. Figure 1-2 shows the diversity of activities and the complexity of estimating GHG emissions and carbon sequestration across the entire management entity.

Figure 1-2: Conceptual Diagram of Activities Covered in This Report

Combining a landowners' crop, livestock and forest management activities into a seamless greenhouse gas estimate for the entity.



Source: Eve (2012).

1.3 Process for the Development of the Methods

This report was developed by three author teams (i.e., working groups) under the direction of one lead author for each team (plus one co-lead author for the forestry chapter). The lead authors were chosen based on their experiences with GHG inventories and accounting methodologies and their professional research experiences. With input from each lead author, USDA chose 10 to 13 working group members per team to write the report. These working group members each had different backgrounds that fit with the anticipated content of the document. Members also had experience with GHG accounting and/or field research that was unique and addressed one or more of the niche methods that were essential for ensuring the comprehensiveness of the methods for each sector. The author teams were provided with a preliminary outline of a chapter and two background reports developed as part of the project. One background report was an analysis of the scientific literature related to rates of carbon sequestration or emissions reduction resulting from various management practices and technologies (Denef et al., 2011); the other was a compilation of all of the available tools, protocols, and models and basic information on each one (Denef et al., 2012). Both reports are available for download on the project website at: http://usda.gov/oce/climate_change/estimation.htm.

There are several general ways to estimate GHG emissions and sequestration at an entity scale, and each approach gives varying accuracy and precision. Typically, the most accurate way to estimate GHG fluxes is through direct measurement, which often requires expensive equipment or techniques that are not feasible for a single landowner or manager.[2]

Lookup tables and estimation equations can be much simpler to implement and use, but when used alone may not adequately represent local variability or local conditions. This report attempts to delineate methods that balance user friendliness, data requirements, and scientific rigor in a way that is transparent and justified.

Figure 1-3 illustrates the scope of the GHG emission sources and removals and processes in managed ecosystems that these methods estimate.

The author teams considered the following general approaches in deriving the methods for this report:

- Basic estimation equations – Involve combinations of activity data[3] with parameters and default emission factors.[4] Any default parameters or default emission factors (e.g., lookup tables) are provided in the text, or if substantial in length, in an accompanying (or referenced) compendium of data.

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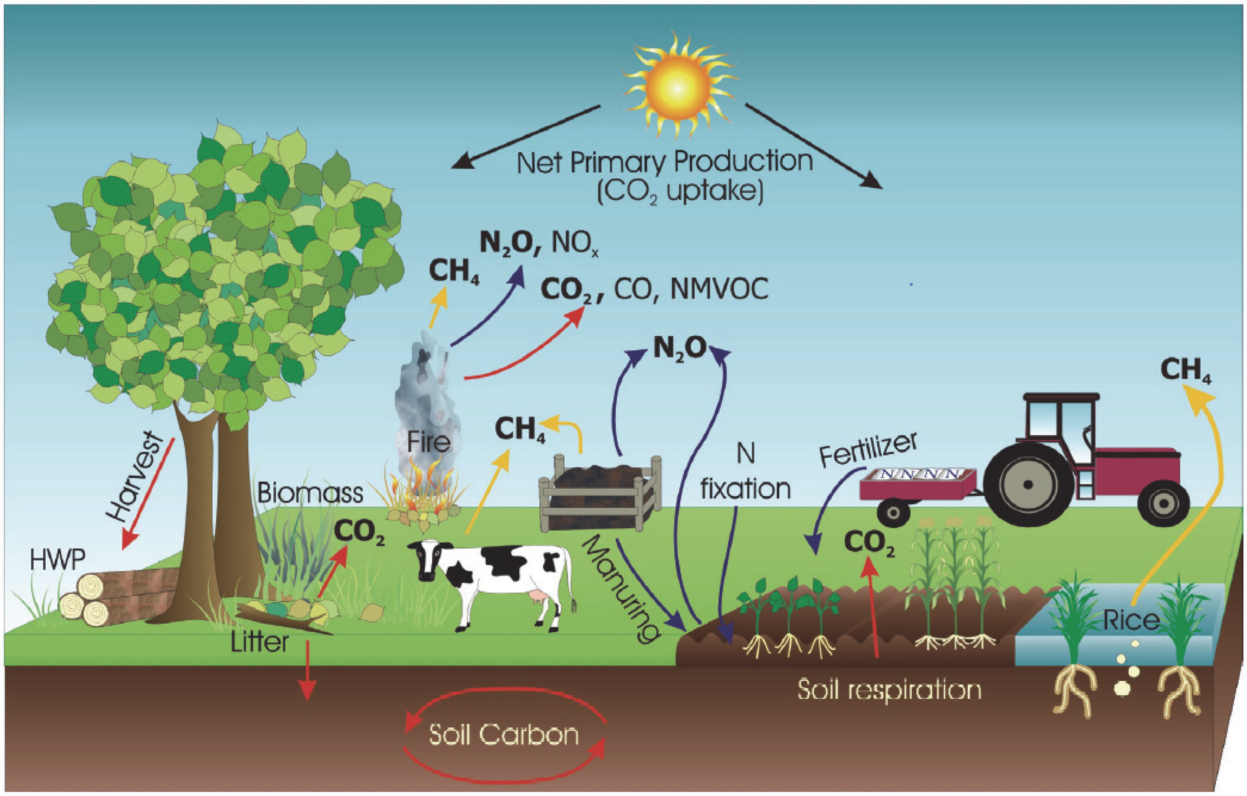
[2] Examples include intermittent measurement of soil organic carbon and biomass reserves. Estimates of flux for dynamic process measures like N_2O emissions need to be based on multiple measures taken at reasonable frequency. Direct measurement may work for comparative analysis but must be extended to estimate total emissions using assumptions or modeling method.
[3] Activity data is defined as data on the magnitude of human activity resulting in emissions or removals taking place during a given period of time (IPCC, 1997).
[4] Emission factor is defined as a coefficient that quantifies the emissions or removals of a gas per unit of activity. Emission factors are often based on a sample of measurement data, averaged to develop a representative rate of emission for a given activity level under a given set of operating conditions (IPCC, 2006).

Figure 1-3: Greenhouse Gases Emission Sources/Removals and Processes in Managed Ecosystems



Source: Paustian et al. (2006).
NMVOC= non-methane volatile organic compounds

- Models – Use combinations of activity data with parameters and default emission factors. The inputs for these models can be ancillary data[5] (e.g., temperature, precipitation, elevation, and soil nutrient levels that may be pulled from an underlying source), biological variables (e.g., plant diversity), or site-specific data (e.g., number of acres, number of animals). The accuracy of the process model is dependent on the robustness of the model and the accuracy of the inputs.
- Field measurements – Actual measurements that a farmer or landowner would need to take to more accurately estimate the properties of the soil, forest, or farm or to estimate actual emissions. Measuring actual emissions on the land requires special equipment that monitors the flow of gases from the source into the atmosphere. This equipment is not readily available to most entities, so more often, field measurements are incorporated into other methods described in this section to create a hybrid approach. A field measurement such as a sample mean tree diameter could be incorporated into other models or equations to give a more accurate input.
- Inference – Uses State, regional, or national emissions/sequestration factors that approximate emissions/sequestration per unit of the input. The input data is then

[5] Ancillary data is defined as additional data necessary to support the selection of *activity data* and *emission factors* for the estimation and characterization of emissions. Data on soil, crop or animal types, tree species, operating conditions, and geographical location are examples of ancillary data.

multiplied by this factor to determine the total onsite emissions. This factor can have varying degrees of accuracy and may not capture the mitigation practices on the farm or the unique soil conditions, climate, livestock diet, livestock genetics, or any farm-specific characteristics, unless they are developed for specific soil types, livestock categories, or climactic regions, etc.

- Hybrid estimation approach – An approach that uses a combination of the approaches described above. The approach often uses field measurements or models to generate inputs used for an inference-based approach to improve the accuracy of the estimate.

With this background, and evaluating these and other data and resources, each author team developed the text for its chapter. Development of the text has been iterative as various drafts of the document have been put through several review stages. The review process for the report of methods consists of:

- *USDA Technical Review*. USDA performed an intra-agency review. The result of this review was a series of comments and questions for the lead authors and their working groups. These comments were received by, discussed within, and responded to by the working groups and lead authors. For example, one specific outcome of this review process was a nitrous oxide (N_2O) Cropping Practices Workshop consisting of 20 experts in the field of N_2O emissions from croplands and grazing lands. The workshop was convened to review the methods that were originally proposed by the working group and to determine if there was a more scientifically rigorous method to quantifying N_2O emissions from agricultural soils.

- *Inter-agency Technical Review*. The May 2012 version of the report was circulated for review by an inter-agency group of GHG emissions and inventory experts. The reviewers included over 50 members from nine agencies including USDA, U.S. Department of Energy, U.S. Department of the Interior, EPA, U.S. Department of State, and several of the White House Offices. The result of this review was a series of comments and questions for the lead authors and their working groups. These comments were received by, discussed within, and responded to by the working groups and lead authors.

- *Scientific Expert Review*. Following the inter-agency review, the next version of the report was reviewed by a team of scientific experts. The reviewers were chosen based on recognized expertise, experience in expert reviews, availability, and willingness to participate. Each reviewer was asked to review those chapters and/or sections of the report relating to his or her expertise. A subset of the group of expert reviewers was asked to review the report in its entirety and provide comments specifically regarding issues of consistency, completeness, and accuracy. Again, the lead authors and author teams responded to each of the comments posed by the expert panel and edited the document as appropriate.

- *Public Comment Period*. Once all of the expert comments were addressed and appropriate edits were made, the report was made available for public comment. This coincided with a final review by USDA and other Federal agency GHG experts. Comments from this review were assessed, and the report was edited as necessary prior to final publication of the report.

How to Use the Report

In order to accomplish the objectives noted above, the report is laid out by broad land-use sector (i.e., croplands and grazing lands, wetlands, animal production, and forestry). Each sector chapter is further delineated into two main parts: first the current scientific understanding and available data for estimating GHG fluxes within the sector; second, the methods that demonstrate the current best approach to estimating GHG fluxes, balancing the available science and data with the criteria and considerations mentioned previously. The report is intended to be considered in its entirety with contextual information provided in the first and second chapters as background to the content presented in the following chapters. The authors realize that many users may find specific chapters or sections especially valuable or useful; therefore, summarized contextual information is also included at the beginning of each chapter. The beginning to the croplands and grazing lands, wetlands, animal production, and forestry chapters include tables that summarize the methods for each source or removal of GHG emissions. The subsequent sections in the report are organized according to the sources mentioned in the summary table.

1.4 Contents of the Report

The remainder of the report is organized by sector. For each sector, background and information on management practices are presented first, followed by the detailed methods proposed for estimating emissions and sequestration for those practices. Each of the chapters is summarized as follows:

- **Chapter 2: Considerations When Estimating Agriculture and Forestry GHG Emissions and Removals.** Chapter 2 sets the context for the methods, including linkages and cross-cutting issues that span the sectors. This includes, for example, definition of entity, definition of system boundaries, etc.
- **Chapter 3: Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems.** Chapter 3 describes the GHG emissions from crop and grazing land systems. The chapter presents 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 stocks changes, direct and indirect soil N_2O emissions, methane (CH_4) and N_2O emissions from wetland rice, CH_4 uptake in soils, carbon dioxide (CO_2) emissions or sinks from liming, non-CO_2 GHG emissions from biomass burning, and CO_2 emissions from urea fertilizer application.
- **Chapter 4: Quantifying Greenhouse Gas Sources and Sinks in Managed Wetland Systems.** Chapter 4 provides guidance for estimation of carbon stock changes, CH_4, and N_2O emissions from actively managed wetlands.
- **Chapter 5: Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems.** Chapter 5 describes on-farm GHG emissions from the production of livestock and manure management. The chapter presents GHG estimation methods appropriate to the production of each common livestock sector (i.e., beef, dairy, sheep, swine, and poultry), with methods related to manure management combined for all livestock types.
- **Chapter 6: Quantifying Greenhouse Gas Sources and Sinks in Managed Forest Systems.** Chapter 6 provides guidance on estimating carbon sequestration and GHG emissions for the forestry sector. The chapter is organized to provide an overview of the elements of forest carbon accounting, including definitions of the key carbon pools and basic methods for their estimation.

- **Chapter 7: Quantifying Greenhouse Gas Sources and Sinks from Land-Use Change.** Chapter 7 provides guidance on estimating the net GHG flux resulting from changes between land types—i.e., conversions into and out of cropland, wetland, grazing land, or forestland—at the entity scale.
- **Chapter 8: Uncertainty Assessment for Quantifying Greenhouse Gas Sources and Sinks**. Chapter 8 provides a framework for a Monte Carlo assessment of estimation uncertainty.

The report also describes methods for uncertainty assessment for each source as well as for the estimate in total. The authors recognize that for some sources, current data are not complete enough to allow for a reliable statistical estimate of uncertainty. In some cases, expert judgment was used to delineate estimated uncertainty bounds. In other cases, the report simply notes that more data are required to reliably estimate uncertainty. Each sector chapter of the report contains a section on uncertainty and limitations.

The authors acknowledge that for many practices and technologies, adequate data do not currently exist to accurately estimate GHG emissions and/or carbon sequestration. For each sector, the authors have included a discussion of research gaps or priority areas for future data collection that are important in order to improve the completeness or accuracy of the estimation methods put forth in this report.

1.5 Uses and Limitations of the Report

Specific potential uses of the methods include aiding:

1. Landowners and other stakeholders in quantifying increases and decreases in GHG emissions and carbon sequestration associated with changes in land management;
2. USDA in understanding GHG and carbon sequestration increases and decreases resulting from current and future conservation programs and practices; and
3. USDA and others in evaluating and improving national and regional GHG inventory efforts.

The report will provide additional cobenefits. For example, the report may provide improved methods for voluntary GHG registries, help to facilitate regional GHG markets, or inform existing and/or future GHG reporting programs (e.g., sequestration/emissions from land use and agriculture under Executive Order [EO] 13514).[6]

These methods are designed to provide the most appropriate, single, accounting method for quantifying GHG emissions/sequestration for each particular source category (e.g., CH_4 from rice cultivation) determined from the activity data, published emission factors, and accounting methods and tools typically available for the entity scale. These methods are **not** designed to provide a range of emission/sequestration accounting options, or a range of similar options, at varying levels of complexity (i.e., tiers) for each particular source category. That said, there may be specific instances (e.g., forest carbon stocks) where different individual options might be specified for entities within

[6] It should be noted that under EO13514, agency-level reporting of emissions and sequestration as a result of land management practices is not required at this time. In addition, reporting of emissions from wildfire management, prescribed burning, land use, and land-use changes is not required. Agencies choosing to report activities undertaken to date in calculating such emissions would address them in the qualitative portion of their GHG inventory. Emissions resulting from manure management and enteric fermentation when the animals are owned by the Federal agency would be reported voluntarily in scope 1 at this time. If the activities take place on Federal land, but are operated by others, these emissions may be voluntarily reported as scope 3.

source categories where there are significantly different operational scales (e.g., commercial forest plantations versus small woodlots).

This report is designed to provide GHG accounting methods to determine actual GHG emissions at the entity scale (i.e., an emissions inventory) and/or to quantify the emission (or emission reductions) associated with an existing or future mitigation practice/technology. At the time of this writing, the United States does not have a national policy guiding GHG emissions reduction, monitoring, or crediting in the agriculture and forestry sectors. Presented are the recommended methods for quantifying GHG emissions and emission reductions. The report is **not** intended as an accounting framework for emission reduction crediting or trading—i.e., the methods do not constitute an offset protocol. As a result, this report does **not** provide specific guidance on critical policy features of such offset protocols including additionality, permanence, and leakage. Any national policy would provide precise definitions of these terms, and then the methods described in this report would be adapted to conform to policy standards and requirements.

As stated above, this report does not address policy issues related to crediting reductions such as permanence, additionality, or leakage. The intended purpose is simply to provide a quantitative estimate of what is occurring under a given set of practices and activities, or what could be expected to occur given a change in management. While the report is not addressing policy issues, it may address practical concerns around GHG estimation, such as the risk of reversal if management practices revert back in the foreseeable future. For example, a land manager must understand that a change in management that results in soil carbon sequestration, if reversed, will lead to the extra stored carbon likely being rereleased to the atmosphere. For the context of this report, we are most concerned with "what the atmosphere sees" or what the long-term net effect is to GHG levels in the atmosphere.

The source categories covered in the report are specific to the agriculture and forestry sectors (e.g., croplands, grazing lands, managed wetlands, animal agriculture, and forestry). The report does not approach emissions from these sources from a life-cycle perspective. In other words, the report does **not** include source categories that are associated with management activities related to certain agriculture and forestry activities (e.g., transportation, fuel use, heating fuel use), upstream production (e.g., animal feed production, fertilizer manufacture), or downstream (e.g., wastewater treatment, pulp and paper manufacture, or landfills). As a result, the report does **not** provide GHG accounting methods for sectors including: energy and industrial processes (e.g., fertilizer production).

The report also does **not** include emissions from stationary source combustion (e.g., burning heating oil or natural gas to heat animal housing) or mobile source combustion (e.g., fuel use in vehicles) at this time. However, where there are obvious changes in the level of combustion due to a change in practices, that change is qualitatively discussed. For example, a shift from conventional tillage to no till can result in a large reduction in fuel consumption because of fewer trips across the field. These relationships are noted qualitatively in the report, but quantitative methods are not proposed. Methods for quantifying emissions from stationary or mobile combustions are available from other Federal agencies.

The scope of this report is assessing the impact of specific decisions made by the farm or forest manager within the confines of the farm or forest gate. A life-cycle perspective, while valuable, is outside the scope of this report. A life-cycle assessment (LCA) is a useful tool for quantification of environmental impacts and benefits on a basis that allows for analysis of environmental burden shifting and trade-offs between different options. LCAs include the environmental impact of management decisions during product manufacturing and processing of raw inputs to, as well as products output from, the farm or forest system, continuing through its use by the end consumer.

The methodologies presented in this report do not constitute an LCA, but support several components of LCAs. For example, this report covers emissions (e.g., from croplands) that could be used as part of an attribution LCA for a commodity crop product, or used as part of a consequential LCA studying the impacts of agricultural policy decisions on GHG mitigation potential.

The text box below provides further information on LCAs as they relate to quantifying GHG sources and sinks in agriculture and forestry systems, including background information on the purpose of LCAs, the LCA process, the interpretation of LCA results, and current LCA efforts by USDA and other organizations related to agriculture and forestry.

Life Cycle Assessment

An LCA is a tool for addressing the environmental aspects (e.g., use of resources) and potential environmental impacts (e.g., global warming potential) throughout the life-cycle of a product or material. When applied to agriculture and forestry products, the scope of an LCA would likely include upstream impacts from extraction and production of material inputs (e.g., fuels, fertilizers); the environmental impacts of management decisions during crop, livestock, or tree growth on site; and the outputs from the farm or forest system, including the downstream impacts from use and disposal by the end consumer. The accounting boundary of GHG emission sources and sinks quantified in an LCA for an agricultural or forest consumer product would extend beyond the accounting boundary of the methodologies presented in this report. For example, an LCA for a grain product would not only include N_2O emissions from fertilizer application, but also other upstream inputs such as emissions from synthetic fertilizer production, and downstream impacts such as emissions from grain transportation and storage, processing, use, and disposal.

The International Organization for Standardization (ISO) has established several international standards addressing LCA, including ISO 14040 (ISO, 2006a) describing the principles and framework for LCAs, ISO 14044 (ISO, 2006b) addressing LCA requirements and guidelines, and ISO 14048 (ISO, 2002) presenting a standardized LCA data documentation format.[a] As defined in ISO 14040 (ISO, 2006a), the LCA development process includes the following primary steps: defining the goal and scope; conducting a life-cycle inventory (LCI) analysis by gathering data and quantifying all relevant inputs and outputs of the product system, as defined in the scope of the study; conducting a life-cycle impact assessment through evaluation of the significance of the environmental impacts defined in the scope of the study and determined during the LCI process; and interpreting the results (ISO, 2002; 2006a; 2006b). USDA has several initiatives applying LCAs to agriculture and forestry.

- USDA's National Agricultural Library has developed the LCA Digital Commons Project, a database and tool intended to provide LCI data for use in LCAs of food, biofuels, and other bio-products. The database currently includes data on inputs (e.g., fertilizers) and outputs (e.g., air emissions, residues) per unit of field crop production from 1996–2009 for corn, cotton, oats, peanuts, rice, soybeans, and wheat (durum, spring, and winter) in States covered by the USDA Economic Research Service annual Agricultural Resource Management Survey. Future phases of this work will include the addition of data representing irrigation, manure management, farm equipment operation, crop storage, transport, and production of mineral and organic fertilizers, herbicides, insecticides, and fungicides.

(continued)

Life Cycle Assessment (continued)

- USDA also recently worked with the National Cattlemen's Beef Association and the chemical company BASF in the development of an eco-efficiency assessment for the U.S. beef industry by quantifying life-cycle inputs and outputs for beef production over time. The process involved measuring the life-cycle environmental impacts and life-cycle costs for different beef production processes at a defined level of output. The USDA Agriculture Research Service's Integrated Farm System Model was used to estimate environmental impacts (e.g., air emissions, water use, abiotic depletion potential, toxicity, etc.) based on data from the USDA's Roma L. Hruska Meat Animal Research Center (Battagliese et al., 2013).

Beyond USDA, other LCAs and studies related to quantifying environmental impacts from agriculture and forestry products have been published. Below is a list of recent studies, projects, or resources that use LCAs or could be used in the development of LCAs to evaluate climate impacts from agriculture and forestry.

- The Innovation Center for U.S. Dairy analyzed fluid milk, cheese, and dairy processing and packaging. These data have recently been made publicly available through the USDA's LCA Digital Commons database.[b]
- The Innovation Center for U.S. Dairy developed the FarmSmart tool that compares energy use, GHG emissions, and water use against regional and national averages. The tool takes approximately 20 minutes to complete and will have enhanced decision support features added in 2014.[c]
- The National Pork Board funded a study of pork products conducted by researchers at the University of Arkansas.[d]
- The United Kingdom's Carbon Trust developed a "carbon footprinting" methodology that has been used by the grocery chain Tesco to determine the life-cycle GHG impacts of many of their products.[e]
- The United Kingdom Food Climate Research Network maintains a compendium of food LCAs.[f]
- Kumar Venkat of CleanMetrics Corp. compared 12 organic and conventional farming systems from a life-cycle GHG emissions perspective using agricultural production data from the University of California-Davis.[g]
- Field to Market prepared a report presenting environmental and socioeconomic indicators for measuring outcomes from on-farm agricultural production in the United States.[h]
- A coalition of food industry companies, academic organizations, and non-governmental organizations created The Cool Farm Tool, a GHG calculator designed to help farmers reduce emissions.[i]

(continued)

[a] See http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_tc_browse.htm?commid=54854.
[b] See http://www.usdairy.com/sustainability/Greenhouse%20Gas%20Projects/Pages/ProcessingandPackagingLCA.aspx and http://www.lcacommons.gov/?q=node/16.
[c] See http://www.usdairy.com/FarmSmart/Pages/Home.aspx.
[d] See http://www.pork.org/filelibrary/NPB%20Scan%20Final%20-%20May%202011.pdf.
[e] See http://www.carbontrust.com/our-clients/t/tesco/.
[f] See http://www.fcrn.org.uk/research-library/lca.
[g] Venkat, K. 2012. Comparison of Twelve Organic and Conventional Farming Systems: A Life Cycle Greenhouse Gas Emissions Perspective. *Journal of Sustainable Agriculture* 36 (6): 620-649.
[h] See http://www.fieldtomarket.org/report/national-2/PNT_SummaryReport_A11.pdf.
[i] See http://www.coolfarmtool.org/Home.

Life Cycle Assessment (continued)

- The National Pork Board developed a predictive model that provides estimates on the GHG emissions, water consumption, and associated costs involved in sow and grow-finish production. The Pig Production Environmental Footprint Calculator requires fundamental inputs only (herd size, feed composition, manure handling system, farm location, barn size, characteristics of heating, ventilation, and air conditioning system) and generates an annual "cradle to gate" estimate.[j]
- The Technical Working Group on Agricultural Greenhouse Gases has published three editions of a synthesis of literature related to the GHG mitigation potential of agricultural land management in the United States.[k]
- The EPA developed and maintains the Waste Reduction Model, an interactive tool that calculates and totals GHG emissions of baseline and alternative waste management practices for 46 common material types, including food waste, yard waste, dimensional lumber, and other organic materials. EPA is currently in the process of developing detailed food waste energy and emission factors to quantify the life-cycle impacts of production and disposal of five common food types—grains, fruits and vegetables, beef, chicken, and dairy.[l]

There are many potential applications for LCA results. When conducted for several comparable agricultural or forest products, LCAs can allow for analysis of the tradeoffs between yield and environmental impacts between different production processes or inputs. For example, comparing LCA results for grain products using different production inputs could show fewer life-cycle GHG emissions and similar yields by switching to a different fertilizer. However, there are limitations to how LCA results can be applied, including use of GHG emissions results in annual reporting or emission inventories. Since LCAs are intended to quantify the environmental impacts across the entire product life cycle, the GHG emissions and sinks frequently occur across several years (and several source categories) and are therefore not appropriate for use in applications that require annual emissions data.

[j] See http://www.pork.org/Resources/1220/CarbonFootprintCalculatorHomepage.aspx#.Us7mGbSwWSo.
[k] See http://nicholasinstitute.duke.edu/ecosystem/land/TAGGDLitRev#.Usbx9tJDuSp.
[l] See http://www.epa.gov/warm.

Finally, the methods in this report are not intended as a sustainability assessment. Other environmental services and cobenefits are not addressed by these methods. Nor are potential tradeoffs or detriments to other environmental concerns addressed here. The methods are specific to GHG emissions only, and sustainable farm, ranch, or forest management should consider the GHG implications of management in tandem with other environmental concerns such as water quality, soil health, and ecosystem health.

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Chapter 2
Considerations When Estimating Agriculture and Forestry GHG Emissions and Removals

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Acronyms, Chemical Formulae, and Units

CH_4	Methane
CO	Carbon monoxide
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
GHG	Greenhouse gas
GWP	Global warming potential
ha	Hectares
HWP	Harvested wood products
IPCC	Intergovernmental Panel on Climate Change
N_2O	Nitrous oxide
NH_3	Ammonia
NO	Nitric oxide
NO_x	Mono-nitrogen oxide
NO_2	Nitrite
NO_3	Nitrate
PDF	Probability density function
USDA	U.S. Department of Agriculture

2 Considerations When Estimating Agriculture and Forestry GHG Emissions and Removals

This chapter describes the linkages and cross-cutting issues relating to sector-specific and entity-scale estimation of greenhouse gas (GHG) sources and sinks. In particular, this chapter describes the common elements that must be considered both within an emissions sector or source category as well as across sectors or source categories in order for an entity to report accurate GHG inventory estimates.

Chapter 2 is organized as follows:

- Scope
- Review of Relevant Current Tools and Methods
- Selection of Most Appropriate Method and Mitigation Practices to Include
- Overview of Sectors
  - Croplands and Grazing Lands
  - Wetlands
  - Animal Production
  - Forestry
  - Uncertainty

2.1 Scope

In order for an entity to accurately inventory its direct GHG emissions to (and removals from) the atmosphere and compare emissions and removals between years, practices, or entities, it is important that estimation elements—e.g., definitions of entity and system boundaries—are common to all emission sectors and source categories. These common elements are described in more detail in the sections that follow and include:

- Definition of Entity
- Definition of System Boundaries:
  - Physical Boundaries
  - Temporal Boundaries
  - Activity Boundaries
  - Material Boundaries

2.1.1 Definition of Entity

The definition of an entity will, to a large degree, determine the (spatial) bounds of the estimation methodologies. This will primarily be driven by what data a landowner chooses to input—i.e., the definition will be user-specific and primarily depend on the user's definition.[1] However, it is anticipated that the science-based methods will be suitable to quantify GHG sources and sinks at a process or practice scale. The methods in this report provide an integrated assessment of the net

[1] It should be noted that the definition of an entity used in this report is not a policy or regulatory definition, and is only provided to help the land manager determine what practices should be included in the estimation.

GHG emissions for an entity, all lands for which the landowner has management responsibility. They also provide the basis for an integrated tool to be used by the U.S. Department of Agriculture (USDA) as well as by individual farmers, ranchers, forest owners, and other stakeholders to evaluate the net GHG emissions on parcels of land under their management. So while the entity would be defined as all of the activities occurring on all tracts of land under the management control of the landowner, the report describes practice-level methodologies that can be summed collectively to arrive at an estimate for the entity. The definition of entity applied here is intentionally broad, understanding that any policy, registry, or market will provide its own narrower definition.

2.1.2 Definition of System Boundaries

The system boundaries should include the GHG emissions and carbon sequestration occurring (or established) onsite for the source category and management practice in question. For example, this report does not address indirect land-use changes occurring offsite or biogenic GHG flux related to subsequent use of agricultural or forestry outputs (e.g., food processing, pulp and paper manufacture, biomass combustion). However, certain offsite carbon storage considerations (e.g., flow of harvested wood into harvested wood products [HWPs]) have been considered in the report to maintain consistency with national inventory efforts.

Four types of system boundaries are important for consideration:

- Physical Boundaries
- Temporal Boundaries
- Activity Boundaries
- Material Boundaries

2.1.2.1 Physical Boundaries

Physical boundaries (e.g., spatial, sectoral) address the area and the management to be considered in the reporting. Setting the boundaries for which emissions and sequestration will be estimated is more difficult than it first seems. Although there may be multiple alternatives, clarity and consistency are important. There are many facets to consider. One factor is what constitutes an entity or a farm/ranch/forest operation; another is what operations are associated with that entity. For example, does the use of fertilizer on a farm include the processes of manufacturing and delivering that fertilizer? Another consideration is how to subdivide that larger entity into the relevant sectors as presented in the individual chapters in this report. For example, is the entity entirely grazing land or is some of it in forest management? Finally, there may be questions of how to associate management practices to the most relevant categories for use of the accounting guidelines provided, including any guidance on size limits, what constitutes management, and how to address changing land uses. Definitions are an important part of setting boundaries and will be provided here as well. Examples of management practices (e.g., irrigation, tillage, or residue management for croplands) are included within the various sector descriptions below (i.e., croplands and grazing lands, wetlands, animal production, and forestry); when considering what constitutes a management practice, an entity should note that in the context of these guidelines, a management practice refers to changes in the management of agriculture, animal, or forest production that impact GHG emissions and removals.

The objective of these methods is to provide a complete estimation of GHG emissions and carbon sequestration within the boundaries of an entity. This is not intended as a life cycle analysis, as will be further explained below in the discussion of material boundaries. The methods are designed to be applied at the local scale, but need to be flexible enough to be valid for very large entities. The methods are designed to estimate fluxes for the entirety of an entity, but must also be capable of evaluating a single practice (e.g., project) implemented within a single entity or aggregated across multiple entities.

As noted in Chapter 1, the definition of an entity can be complicated. For the purposes of this report, users should simply delineate the spatial extent of its entity as the land area that is under their ownership and/or management control for the foreseeable future. This is a generalized application of the term entity, and the user should recognize that any policy, program, or contractual agreement may define the user's entity differently and result in a different boundary of the entity. Within the entity boundary, there will be a variety of land uses that will rely on methods from various chapters in this report. An entity should be subdivided if it includes different categories of land use, such as grazing land and cropland, but the entire entity should fall into some land-use category. No rigid lower bound is specified here for the areal extent of a land-use categorization, but, in general, areas of an acre or more merit identification.

Within the boundaries of the overall entity, areas of *cropland* will need to be identified. Beyond just areas producing row or close-grown crops or hay, cropland also includes land that is fallow and areas of hay and pasture that are managed in a rotation with other crops. Wetlands (including drained wetlands and hydric soils) and land under agroforestry practices where the predominant production activity is cropping should also be considered as cropland for the purposes of this report. Finally, areas of cropland that are set aside, such as lands in the Conservative Reserve Program, are included in this management type. The methods for these lands are included in Chapter 3 of this report. The cropland areas should be delineated as fields or groups of fields for which the basic rotations and management practices are all similar.

Cropland:
A land-use category that includes areas used for the production of adapted crops for harvest, including both cultivated and non-cultivated lands. Cultivated crops include row crops or close-grown crops and also hay or pasture in rotation with cultivated crops. Non-cultivated cropland includes continuous hay, perennial crops (e.g., orchards), and horticultural cropland. Cropland also includes land with alley cropping and windbreaks, as well as lands in temporary fallow or enrolled in conservation reserve programs (i.e., set-asides). Roads through cropland, including interstate highways, State highways, other paved roads, gravel roads, dirt roads, and railroads are excluded from cropland area estimates and are, instead, classified as settlements.

The next land management type to be identified is *grazing land*. This is land that is used primarily for grazing animals and not as part of a rotation with other crops. This portion of the entity will primarily be comprised of *pastureland* (which is more intensively managed), and *rangeland* (which is typically less intensively managed and usually has a higher proportion of native species). Wetlands (including drained wetlands and hydric soils) and land managed as agroforestry should be included in this category if the primary use of the tract of land is for grazing livestock. There will be obvious overlap between grazing land and forestland methods where the land matches the definition of both uses. For example, if any active management is focused on enhancing tree growth and timber production, the user should identify these areas as forestland and the methods will need to be integrated to account for the impact of grazing management on the forestland. Grazing lands should be delineated as contiguous areas that are under a similar stocking rate and set of management practices, and the methods for grazing lands as presented in Chapter 3 should be followed. In addition, the GHG estimation methods associated with the grazing animals as presented in Chapter 5 should be followed. Development of an integrated tool that follows these methods will need to account for these management interactions.

Grazing Land:
A land-use category on which the plant cover is composed principally of grasses, grass-like plants, forbs, or shrubs suitable for grazing and browsing, and includes both pastures and native rangelands. This includes areas where practices such as clearing, burning, chaining, and/or chemicals are applied to maintain the grass vegetation. Savannas, some wetlands and deserts, and tundra are considered grazing land. Woody plant communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also classified as grazing land if they do not meet the criteria for forest land. Grazing land includes land managed with agroforestry practices such as silvopasture and windbreaks, assuming the stand or woodlot does not meet the criteria for forest land. Roads through grazing land, including interstate highways, State highways, other paved roads, gravel roads, dirt roads, and railroads are excluded from grazing land area estimates and are, instead, classified as settlements.

Forestland should be delineated as land that is used primarily for woody biomass production, whether for saw wood, pulp, biofuels, or other forest or woodland related industry, or land that is tree covered and managed for recreational or conservation purposes. This will include areas of agroforestry and silvopasture where the primary management objective on the landscape is forest-related production. An integrated tool would need to be flexible enough to also capture the impact of the additional cropping or grazing activities occurring on the parcel. Similarly, wetland areas that are wooded or forested and managed primarily as forests and woodlands will be considered in this category. Also, because harvesting is one of the major management practices in forestland and because harvested wood moves to several long-term carbon pools that undergo differing rates of decay, it is important that the methods account for emissions from HWPs, even though they may be moved outside of the boundary of the farm/ranch/forest operation.

The forestland methods are presented in Chapter 6 of this report. Tracts of forest should be delineated such that any given tract is made up of trees of a similar stand age and species mix, and that the entire tract is under one uniform set of management practices. On a given entity, there may be trees that exist outside of clearly defined forests, such as orchards and vineyards, farmstead shelterbelts and field windbreaks, and agroforestry practices. Even though these lands may not meet the definition of a forest, the carbon storage in the trees is likely significant. In some cases it may be useful to evaluate individual trees or small stands of trees (using methods presented in Chapter 6). In other cases, the estimation may require a blending of methods such as cropland methods from Chapter 3 with forest methods from Chapter 6.

Forestland:
A land-use category that includes areas at least 120 ft (36.6 m) wide and 1 acre (0.4 ha) in size with at least 10 percent cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated. Forest land includes transition zones, such as areas between forest and non-forest lands that have at least 10 percent cover (or equivalent stocking) with live trees and forest areas adjacent to urban and built-up lands. Roadside, streamside, and shelterbelt strips of trees must have a crown width of at least 120 ft (36.6 m) and continuous length of at least 363 ft (110.6 m) to qualify as forest land. Unimproved roads and trails, streams, and clearings in forest areas are classified as forest if they are less than 120 ft (36.6 m) wide or 1 acre (0.4 ha) in size; otherwise they are excluded from forest land and classified as settlements. Tree-covered areas in agricultural production settings, such as fruit orchards, or tree-covered areas in urban settings, such as city parks, are not considered forest land (Smith et al., 2009).

Wetland areas will fall into one of two categories: managed wetlands or natural, unmanaged wetlands. Many wetland areas may have already been delineated in one of the above categories, and their management will be captured through estimation for that category. If, however, there are wetland areas that have not already been included in the cropland, grazing land, or forestland delineations above, those should be identified here. A naturally occurring wetland that does not have active management being applied in order to increase productivity or provide other environmental services will not be included in the estimation of GHG fluxes. These natural, unmanaged wetlands should simply be included in the category of "other land" as defined below. Any wetland areas that are outside the boundaries of the defined areas mentioned above and where the land manager is actively applying management decisions in order to enhance productivity or provide environmental services should be delineated as a managed wetland and included. This report provides estimation methods in Chapter 4 for emissions from palustrine wetlands,[2] influenced by a variety of management options such as water table management, timber or other plant biomass harvest, and wetlands that are managed with fertilizer applications. Currently, there are insufficient data and therefore, the GHG fluxes will likely not be included in an entity's GHG estimation until adequate data exist to provide that estimation with a reasonable and measurable level of uncertainty.

Wetland:
A land-use category that includes land with hydric soils, native or adapted hydrophytic vegetation, and a hydrologic regime were the soil is saturated during the growing season in most years. Wetland vegetation types may include marshes, grasslands or forests. Wetlands may have water levels that are artificially changed, or where the vegetation composition or productivity is manipulated. These lands include undrained forested wetlands, grazed woodlands and grasslands, impoundments managed for wildlife, and lands that are being restored following conversion to a non-wetland condition (typically as a result of agricultural drainage). Provisions for engineered wetlands including storm water detention ponds, constructed wetlands for water treatment, and farm ponds or reservoirs are not included. Natural lakes and streams are also not included.

Settlements will fall into two broad categories: (1) land where the entity manager imposes management decisions; and (2) land where the manager does not regularly impose management decisions that impact carbon balances. Examples of settlement land that may be significant from a carbon management perspective would be developed livestock feed yards, dairy barns, poultry houses, manure piles, and manure or runoff lagoons. Examples of developed land where management is not of concern to carbon balances is homes, yards, driveways, workshops, roads, and parking areas. For purposes of the GHG flux estimation, only the areas with carbon management implications (e.g., animal housing, manure waste treatment areas) need to be identified within the spatial boundary delineation. These livestock and manure management methods are presented in Chapter 5 of this report. The remaining settlement lands without carbon management implications (e.g., roads and railroads) can simply be excluded from the spatial boundaries an entity chooses to account for within the settlement land-use category.

[2] Palustrine wetlands are nontidal wetlands that are primarily composed of trees, shrubs, persistent emergent, emergent mosses or lichens, and all wetlands that occur in tidal areas where salinity due to ocean-derived salts is below 0.5 percent. Palustrine wetlands must have an area less than 20 acres, not have active wave-formed or bedrock shoreline, have a maximum water depth of less than 2 m [6.6 ft], and have a salinity less than 0.5 percent (USGS, 2006).

Settlements:
A land-use category representing developed areas consisting of units of 0.25 acres (0.1 ha) or more that includes residential, industrial, commercial, and institutional land; construction sites; public administrative sites; railroad yards; cemeteries; airports; golf courses; sanitary landfills; sewage treatment plants; water control structures and spillways; parks within urban and built-up areas; and highways, railroads, and other transportation facilities. Also included are tracts of less than 10 acres (4.05 ha) that may meet the definitions for forest land, cropland, grassland, or other land but are completely surrounded by urban or built-up land, and so are included in the settlement category. Rural transportation corridors located within other land uses (e.g., forest land, cropland, and grassland) are also included in settlements.

Any land that is actively managed in such a way as to impact biomass growth or otherwise impact production-related GHG emissions should have been captured within the spatial boundaries defined for the land-use categories listed above. Any remaining land should be categorized as *other lands* or *unmanaged land* and will not be considered in the estimation of GHG fluxes. This includes the wetland and developed areas that were previously noted as not having active management—i.e., unmanaged wetlands and unmanaged settlements. It also includes any other areas within the entity boundary that represent barren, mined, abandoned, or otherwise unmanaged land—i.e., other land.

Land-cover change is simply a variation from year to year in what is growing on a parcel of land, such as rotating corn and soybean crops, and is not considered land-use change. In contrast, *land-use change* is a fundamental shift in purpose or production of a parcel, such as a shift from cropping to forest production or vice versa. Land-use change needs to be accounted for in the annual GHG flux, as the impact (either positive or negative) on biomass and soil carbon can be significant. These land-use change methods are presented in Chapter 7 of this report.

Other Land:
A land-use category that includes bare soil, rock, ice, and all land areas that do not fall into any of the other five land-use categories, which allows the total of identified land areas to match the identified land base.

Animal production is not necessarily a spatially defined activity within the entity, but has to be considered as part of the physical boundary of the manager's operation. There are three main areas that need to be considered as important to estimating GHG emissions from an animal production system: methane emissions from the animals, methane and nitrous oxide emissions from management of manure, and any emissions impacts related to animal housing. Animal production in the chapter is discussed by animal system type, including beef, dairy, sheep, swine, and poultry. The collective noun for a group of animals typically varies by species, but for the purposes of this report, we will refer to any group of animals of the same animal type that are kept together under a common set of production management practices as a *herd*. Following this definition, the entity's manager may have several distinctly different herds that make up the entity. GHG emissions from animal production will vary greatly depending upon species (digestive processes), growth stage, diet, and manure storage and management. Timing is also a challenge in estimating emissions from the animal production sector, as emissions per animal change dramatically as a young animal grows and matures, as feedlot cattle are finished, or as dairy cows cycle between gestating and lactating. In some cases, it will likely be necessary for the user to estimate emissions for a herd using average weight, average age, and other representative characteristics to represent the herd population. In other cases, it will be necessary to generalize by seasons—manure management may be different in winter than summer, animal feed mixture may vary by season or by animal growth stage. Averaging and generalizing in this way should be adequate in capturing the information needed to provide a reasonable estimate of GHG emissions as long as the manager applies assumptions consistently across the herds and throughout the time under consideration. For example, assuming an average finish weight for feeder animals in the herd should provide a reasonable GHG estimate as long as the assumed weight does not change from year to year, unless a specific management decision (such as a change in animal diet) results in an actual change in finishing weight, in which case the change in averages would be appropriate. Specific methods for animal production systems are presented in Chapter 5 of the report. In some cases, such as manure applied to cropland, methods from Chapter 3 will be utilized as well.

Occasionally, physical boundaries will change over time. Whether a portion of a cropland field is converted to an animal feedlot, shelterbelt or riparian trees are planted onto former cropland, or abandoned land reverts to grazing land or forestland, these changes could result in the need for a new delineation of parcel boundaries or a dissection of one parcel into several parcels with more than one management strategy. For the portion of the parcel where this change has occurred, the land-use change methods (Chapter 7) will be used to estimate GHG fluxes.

Figure 2-1 can be used to help landowners determine the land use category for their land area, according to the definitions above.

2.1.2.2 Temporal Boundaries

Temporal issues include such considerations as the frequency of the reported estimates, the treatment of activities that occur within an accounting period but have long-term implications for carbon balances (e.g., changes in soil carbon following a change in tillage practices), and how to account for short-term management or short-term adjustments to long-term management decisions. Also significant is how to address movement of spatial boundaries over time and with land-use change. This section will attempt to resolve some of these temporal issues around GHG emission estimation and reporting.

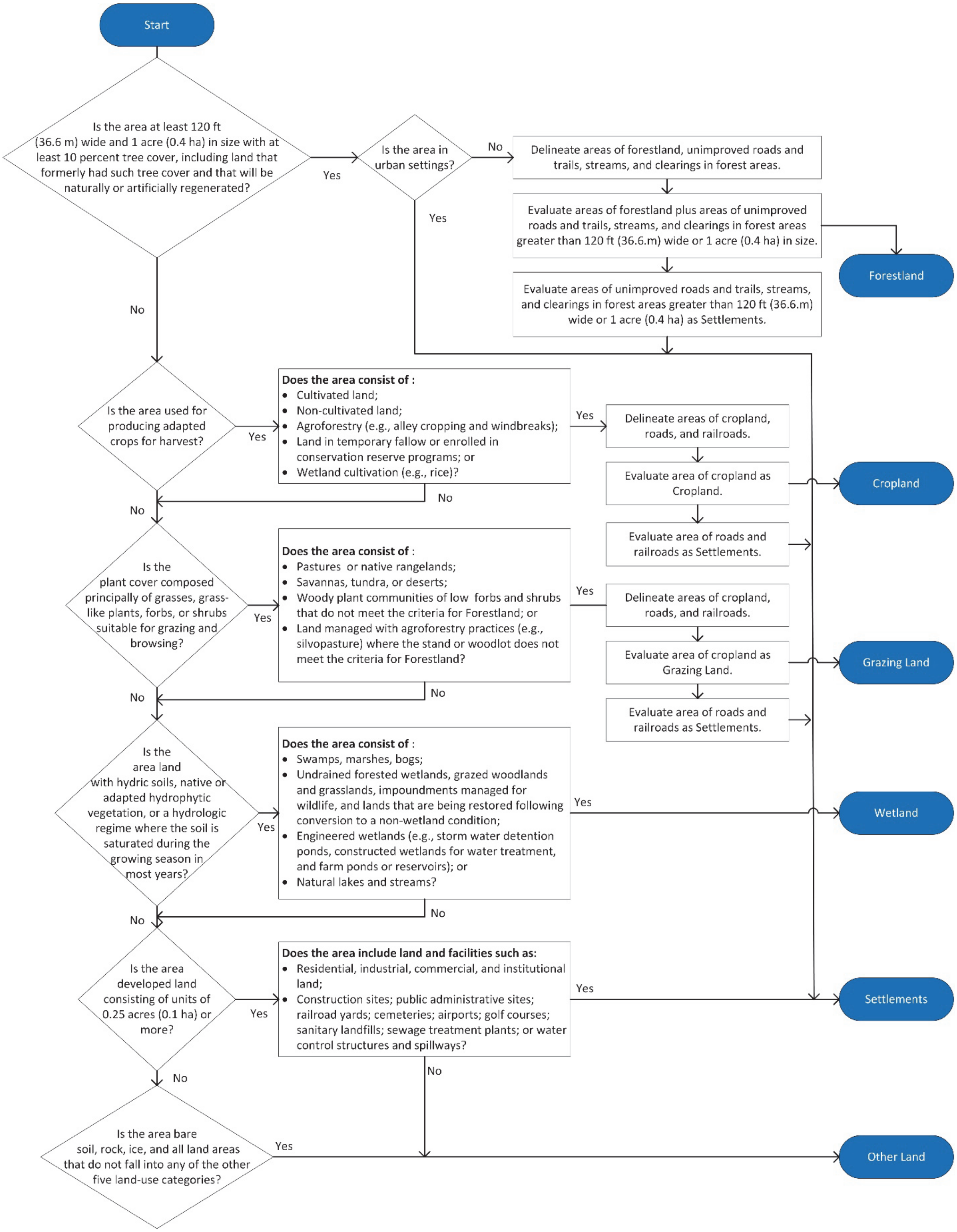
The methods reported here are intended to provide a means of annual accounting and reporting of GHG fluxes. Annual changes in some emissions are easily quantified, but for others it is much more difficult. Carbon stored in trees, for example, may need to be estimated over a longer period, with the change then converted to an annualized estimate.

The report methodologies assume an accounting period of one calendar year (e.g., 365 days) when estimating annualized emissions in a particular sector or source category.

Management decisions also are significant to the accounting time horizon. For example, a forest management plan might call for timber harvest or thinning. In the year of harvest, the annual accounting will reflect a loss of standing live and/or standing dead carbon stocks, yet in the longer term management strategy, the net result could be an increase in total carbon stocks. If a land manager has a management plan that prescribes forest thinning, but then harvests more aggressively than the plan, consideration should be given as to whether this constitutes a change in forest management, which would be discussed in the forest management methods (see Chapter 6).

There are also times when management has to take corrective action or temporarily deviate from a long-term management plan. This could be the case where a cropland manager has adopted a no-till management strategy, but after several years has to use tillage one year because of weather, pests, or other extenuating circumstances. In this case, the methods will ideally be sensitive enough to capture the GHG impact of the deviation from the management plan.

Figure 2-1: Decision Tree for Determining Land-Use Category for Land Areas



2.1.2.3 Activity Boundaries

It is important to distinguish which activities within an entity are subject to accounting. This accounting system is focused on land-based activities such as tillage and harvesting, and not on emissions of GHGs that are related to fossil fuel use. Thus, emissions from tractor fuel or fuel used for crop drying are not counted, nor are the energy inputs required to manufacture fertilizer or farm tools, or to heat farm buildings—i.e., indirect GHG emissions (see Chapter 1). However, as mentioned in Chapter 1, where there are obvious changes in the level of combustion due to a change in practices, that change is qualitatively discussed. For example, a shift from conventional tillage to no till can result in a large reduction in fuel consumption because of fewer trips across the field. These relationships are noted qualitatively in the report, but quantitative methods are not proposed. Methods for quantifying emissions from stationary or mobile combustions are available from other Federal agencies.

As previously mentioned, the methods in this report do not constitute a life-cycle assessment for two primary reasons. First the activity boundaries do not include emissions from fossil-fuel use. Second, the temporal boundaries are focused on annual reporting and do not encompass the range of activities such as capital investment, material supplies, and disposal.

2.1.2.4 Material Boundaries

Material boundaries include the GHGs that are to be considered in the estimation and should also delineate what sources of those gases are included and what are excluded. Also included in this section is a discussion of the global warming potentials (GWPs) used throughout the report. It is important to determine up front which gases are included and which are not. It is also important to determine how much freedom the user has in what is estimated and where these boundaries lie in order to ensure that a change in management that reduces emissions in one sector does not inadvertently cause emissions to rise outside of the boundaries being reported.

The report includes estimation methodologies covering the GHG emissions from the croplands and grazing lands, wetlands, animal production, forestry, and land-use change sectors. Within these sectors and source categories, emissions and removals of the main GHGs—carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)—are accounted for. It should be noted that carbon sequestration (i.e., increases in carbon stocks) is estimated in terms of carbon dioxide equivalents (CO_2-eq). It should also be noted that the animal production chapter includes discussion of ammonia (NH_3), as this is an important precursor to N_2O emissions from manure management. Estimating NH_3 emissions is beyond the scope of this report—NH_3 is not considered a GHG—but since NH_3 is significant as a precursor to N_2O, understanding changes in NH_3 emissions resulting from changes in management is important.

Emissions and sequestration values are presented in this report in terms of the mass (not volume) of each gas, using metric units (e.g., metric tons of methane). In the integrated tool, the masses of each gas will be converted into CO_2 equivalent units using the GWPs for each gas in the International Panel on Climate Change (IPCC) Second Assessment Report.

A GWP is an index used to compare the relative radiative forcing of different gases without directly calculating the changes in atmospheric conditions. GWPs are calculated as the ratio of the radiative forcing that would result from the emissions of one kilogram of a GHG to that from the emissions of one kilogram of CO_2 over a defined period of time, such as 100 years. Emissions in terms of CO_2 equivalents (CO_2-eq) are estimated by multiplying the mass of a particular GHG (e.g., CH_4, N_2O) by the respective GWP for that particular GHG. The GWPs used in this report are shown in Table 2-1 below.

The methods in this report focus primarily on the direct emissions resulting from management decisions made within the boundaries of the entity—e.g., within the farm and forest gate. The indirect emissions related to inputs into the entity are not considered. The reason for this is that those emissions would likely be reported by the manufacturer producing the inputs. If one were conducting a full life-cycle assessment, these emissions would need to be included, but for purposes of the emissions being estimated here we focus primarily on the emissions resulting within the spatial boundary of the entity. The one notable exception that is accounted for is when management decisions on the operation have a specific related influence on emissions leaving the entity's boundary. An example of this is indirect emissions such as nitrogen that is applied within the operation but then carried offsite via erosion or leaching and contributes to N_2O emissions offsite. Another example to consider is harvested commodities. In the case of grains or other agricultural commodities, the product is assumed to be consumed within a relatively short amount of time, resulting in no net gain or loss related to GHG accounting. HWPs are somewhat different, as much of that harvest will end up in long-term carbon pools either as structures, furniture, or other wood products, or in landfills. This report does provide a discussion of N_2O losses that result from erosion and leaching of fertilizer nitrogen and the carbon pools related to the fate of HWPs.

Table 2-1: Global Warming Potentials Used in the Report

Species	Chemical Formula	Lifetime (years)	GWP[a]
Carbon dioxide	CO_2	Variable	1
Methane	CH_4	12±3	21
Nitrous oxide	N_2O	120	310

[a] GWPs used are 100-year time horizon, in accordance with the IPCC Second Assessment Report (IPCC, 2007).

2.2 Review of Relevant Current Tools and Methods

This section provides an overview of the current estimation methods or approaches an entity could use to estimate GHG emissions and sinks on their property. This overview is followed by a summary of each sector's proposed methodologies for entity GHG estimations.

There are several approaches that a farmer or landowner can use to estimate GHG emissions at an entity scale, and each approach gives varying accuracy and precision. The most accurate way of estimating emissions is through direct measurement, which often requires expensive equipment or techniques that are not feasible for a single landowner or manager. On the other hand, lookup tables and estimation equations alone often do not adequately represent local variability or local conditions. This report attempts to delineate methods that balance user-friendliness, data requirements, and scientific rigor in a way that is transparent and justified.

The following approaches were considered for these guidelines:

- Basic estimation equations – Involve combinations of activity data[3] with parameters and default emission factors.[4] Any default parameters or default emission factors (e.g., lookup tables) are provided in the text, or if substantial in length, in an accompanying compendium of data.

[3] Activity data are data on the magnitude of human activity resulting in emissions or removals taking place during a given period of time (IPCC, 1997).

[4] Emission factor is defined as a coefficient that quantifies the emissions or removals of a gas per unit activity. Emission factors are often based on a sample of measurement data, averaged to develop a representative rate of emission for a given activity level under a given set of operating conditions (IPCC, 2006).

- Models – Use combinations of activity data with parameters and default emission factors. The inputs for these models can be ancillary data[5] (e.g., temperature, precipitation, elevation, and soil nutrient levels that may be pulled from an underlying source), biological variables (e.g., plant diversity), or site-specific data (e.g., number of acres, number of animals). The accuracy of the models is dependent on the robustness of the model and the accuracy of the inputs.
- Field measurements – Actual measurements that a farmer or landowner would need to take to more accurately estimate the properties of the soil, forest, or farm to estimate actual emissions. Measuring actual emissions on the land requires special equipment that monitors the flow of gases from the source into the atmosphere. This equipment is not readily available to most entities, so more often field measurements are incorporated into other methods described in this section to create a hybrid approach. A field measurement such as a sample mean tree diameter could be incorporated into other models or equations to give a more accurate input.
- Inference – Uses State, regional, or national emissions/sequestration factors that approximate emissions/sequestration per unit of the input. The input data is then multiplied by this factor to determine the total onsite emissions. This factor can have varying degrees of accuracy and often does not capture the mitigation practices on the farm or the unique soil conditions, climate, livestock diet, livestock genetics, or any farm-specific characteristics, although they can be developed with specific soil types, livestock categories, or climactic regions.
- Hybrid estimation approaches – An approach that uses a combination of the approaches described above. The approach often uses field measurements or models to generate inputs used for an inference-based approach to improve the accuracy of the estimate.

2.3 Selection of Most Appropriate Method and Mitigation Practices to Include

In drafting the report, a number of selection criteria were considered (e.g., transparency, consistency, comparability, completeness, accuracy, cost effectiveness, ease of use). A description of each appears below:

- Transparency – The assumptions and methodologies used for an inventory should be clearly explained to facilitate replication and assessment of the inventory by users of the reported information. The transparency of inventories is fundamental to the success of the process for the communication and consideration of information.
- Consistency – The methods used to generate inventory estimates should be internally consistent in all its elements and the estimates should be consistent with other years. An inventory is consistent if the same methodologies are used for the base and all subsequent years and if consistent data sets are used to estimate emissions or removals from sources or sinks. Consistency is an important consideration in merging differing estimation techniques from diverse technologies and management practices.
- Comparability – For the guidelines to be comparable, the estimates of emissions and sequestration being reported by one entity are comparable to the estimates being reported by others. For this purpose, entities should use common methodologies and formats for

[5] Ancillary data are additional data necessary to support the selection of *activity data* and *emission factors* for the estimation and characterization of emissions. Data on soil, crop or animal types, tree species, operating conditions, and geographical location are examples of ancillary data.

estimating and reporting inventories. Consequently, in general, the methods specify one method for any technology or management practice (i.e., methods suggested in this report do not allow users to select from a menu of methods).

- Completeness – The methods must account for all sources and sinks, as well as all GHGs to the greatest extent possible. Completeness also means full coverage of sources and sinks under the control of the entity. Completeness is an important consideration to be balanced with ease of use in reporting appropriately for an entity that may have a minor activity or an activity with severely limited data availability.
- Accuracy – A relative measure of the exactness of an emission or removal estimate. Estimates should be accurate in the sense that they are systematically neither over nor under true emissions or removals, as far as can be judged, and that uncertainties are reduced as far as practicable.
- Cost effectiveness – A measure of the relative costs and benefits of additional efforts to improve inventory estimates or reduce uncertainty. For example there is a balance between the relative costs and benefits of additional efforts to reduce uncertainty.
- Ease of use – A measure of the complexity of the user interface and underlying data requirements.

The working groups developed the following selection criteria for the mitigation practices that could be included in the methods:

1. The science reflects a mechanistic understanding of the practice's influence on an emission source.
2. Published research supports a reasonable level of repeatability/consistency (can use international studies if similar management, climate, and soils as U.S. conditions).
3. There is general agreement that at least the sign and range of responses are reasonably well understood.
4. There is consensus of the authors that the practice can be adequately included. To reach consensus, the authors discussed issues such as: Would leaving a mitigation practice out make the report incomplete? Is there strong enough evidence that the method will hold up for this practice for at least the next five years?

There were mitigation practices that did not fulfill these criteria, and those practices were cited as areas that require more research in order to fully understand the effect of changes in the practice to GHG emissions. These research gaps are intended to become areas that USDA, non-governmental organizations, universities, and other research institutions will consider as important areas to focus agriculture and forestry climate-change research priorities. Other topics, such as albedo effects, were not considered. Currently, with the exception of urban areas, albedo effects are highly variable and are difficult to reliably quantify.

2.4 Overview of Sectors

This report covers emissions sources and sinks from croplands/grazing lands, managed wetlands, animal production systems, and forestry, along with changes in land use. Figure 2-2 can be used to help landowners determine which chapter can be used to estimate their GHG sources and sinks from their land.

Figure 2-2: Decision Tree for Determining Which Methods to Follow in This Report

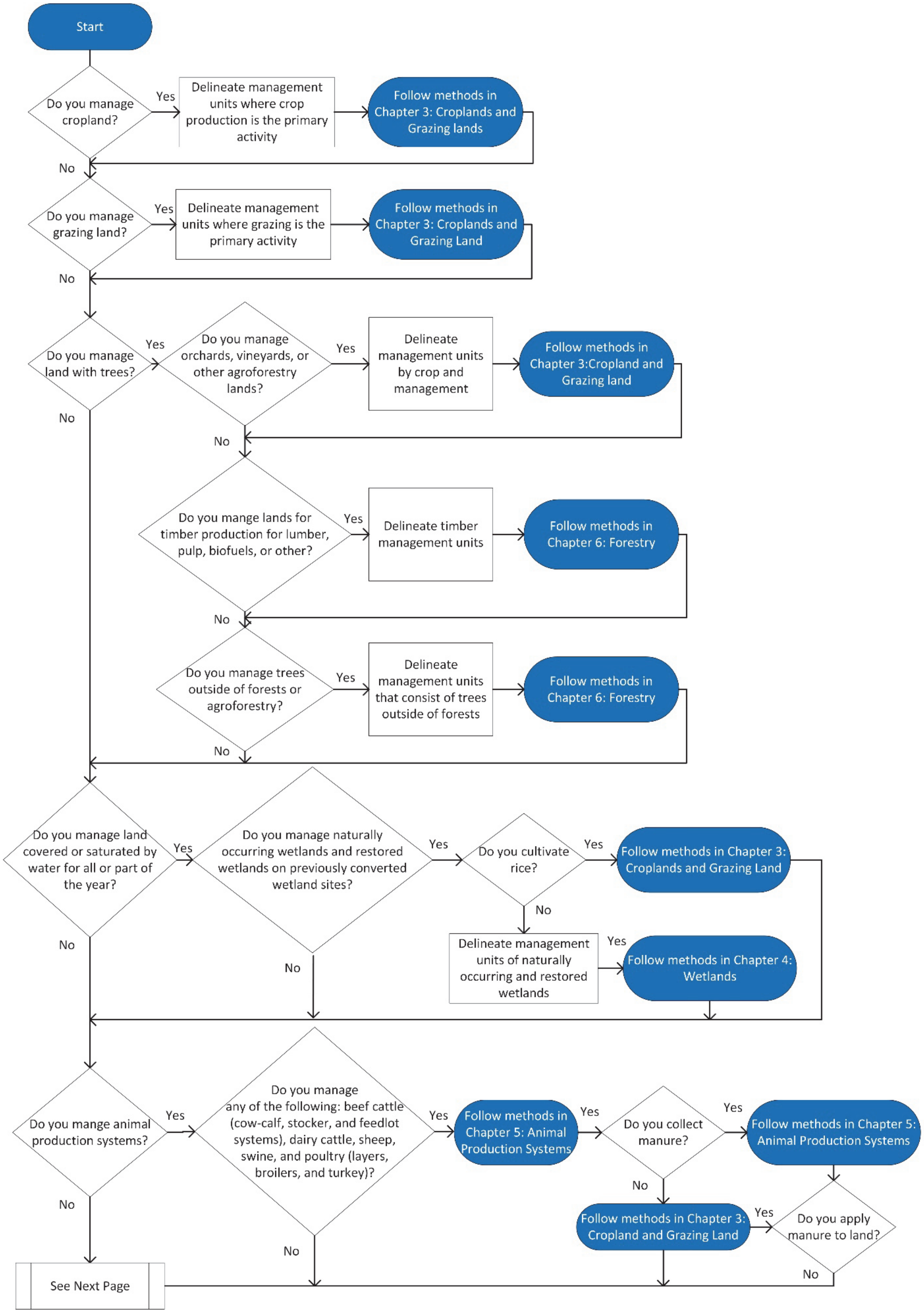
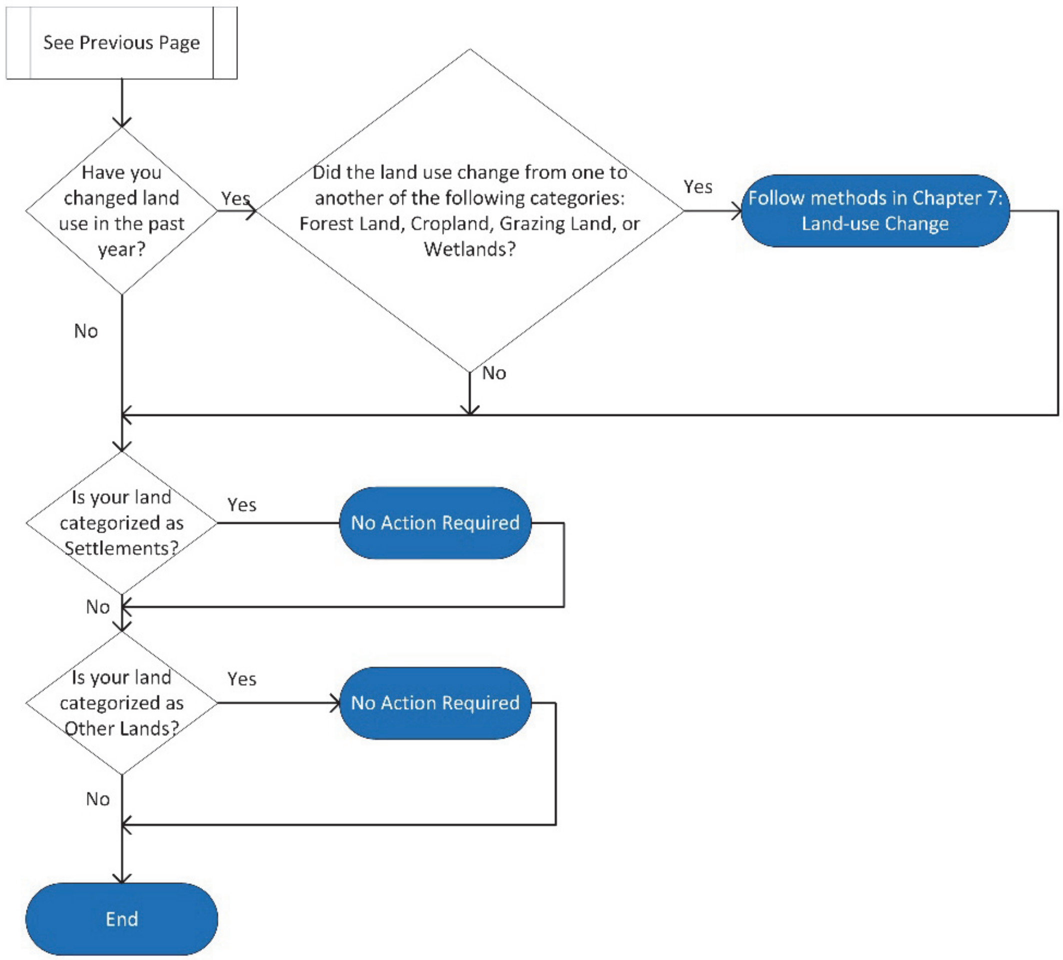


Figure 2-2: Decision Tree for Determining Which Methods to Follow in This Report (continued)



*Methods are not provided for land areas categorized as Settlements or Other Lands

The following sections provide an overview of the sectors covered in this report. For each sector, the emission sources and sinks are introduced as well as the management practices impacting GHG emissions.

2.4.1 Croplands and Grazing Lands

Croplands include all systems used to produce food, feed, and fiber commodities, in addition to feedstocks for bioenergy production. Most U.S. croplands are drylands (irrigated or unirrigated); rice and a few other crops are grown in wetlands. Some croplands are set aside in the Conservation Reserve Program. Croplands also include agroforestry systems that are a mixture of crops and trees, such as alley cropping, shelterbelts, and riparian woodlots. 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; they include both pastures and native rangelands (EPA, 2011). Savannas, some wetlands and deserts, and tundra can be considered grazing lands if used for livestock production. Grazing land systems include managed pastures that may require periodic management to maintain the grass vegetation and native rangelands that typically require limited management to maintain.

Cropland and grazing lands are significant sources of CO_2, N_2O, and CH_4 emissions and can also be a sink for CO_2 and CH_4 (U.S. EPA, 2011). N_2O emissions from soils are influenced by land use and management activity, particularly nitrogen application. Land use and management also influence carbon stocks in biomass, dead biomass, and soil pools. Crop and grazing land systems can be either a source or sink for CO_2, depending on the net changes in these carbon pools. The main influences on nitrogen use efficiency and N_2O emissions are fertilizer rate, timing, placement, and nitrogen source. Tillage intensity, cropping intensity, and the use of crop rotation can have significant effects on soil carbon stocks.

Other management activities also affect GHG emissions from soils. Irrigation can impact CH_4 and N_2O emissions as well as carbon stocks. Burning decreases biomass carbon stocks and also soil organic carbon stocks due to decreased carbon input to the soil system. Burning will also lead to emissions of CH_4 and N_2O and other gases (CO, NO_x) that are GHG precursors. CH_4 can be removed from the atmosphere through the process of methanotrophy in soils, which occurs under aerobic conditions and generally in undisturbed soils. CH_4 is produced in soils through the process of methanogenesis, which occurs under anaerobic conditions (e.g., wetland soils used for production of rice). Both processes are driven by the activity of micro-organisms in soils, but the rate of activity is influenced by land use and management.

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 fertilizer placement and timing. Because of these interconnections, estimating GHG emissions from crop and grazing land systems will depend on a complete description of the practices used in the operation, as well as ancillary variables such as soil characteristics and weather or climate conditions. It is also important to note that trends in GHG emissions associated with a change in crop and grazing land management can be reversed if the landowner reverts to the original practice. For example, a farmer might switch from conventional tillage to no-till for 10 years and see an increase in soil carbon sequestration; if, however, the farmer then reverts to conventional tillage, the gains in soil carbon will be quickly lost as the stored soil carbon is released back into the atmosphere as CO_2, negating the GHG mitigation of the previous 10 years. However, 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.

The text box, Management Practices Impacting GHG Emissions from Croplands and Grazing Lands, lists the most significant mitigation practices discussed in Chapter 3. Additional mitigation practices are discussed in the chapter, but these often have sparse or conflicting evidence in support of their mitigation effects. Therefore, the text box lists the more robustly supported practices.

Management Practices Impacting GHG Emissions from Croplands and Grazing Lands

- Nutrient Management (Synthetic and Organic)
- Tillage Practices
- Crop Rotations and Cropping Intensity
- Irrigation
- Residue Management
- Set-Aside/Reserve Cropland
- Wetland Rice Cultivation
- Livestock Grazing Practices
- Forage Options
- Silvopasture

2.4.2 Wetlands

Wetlands occur across the United States on many landforms, particularly in floodplains and riparian zones, inland lacustrine, glaciated outwash, and coastal plains. The National Wetlands Inventory broadly classifies wetlands into five major systems, including (1) marine, (2) estuarine, (3) riverine, (4) lacustrine, and (5) palustrine (Cowardin et al., 1979). These systems are further classified by major vegetative life form. For example, forested wetlands are often classified as palustrine-forested. Similarly, most grassland wetlands are classified as palustrine wetlands with emergent vegetation (e.g., grasses and sedges). Wetlands also vary greatly with respect to groundwater and surface water interactions that directly influence hydroperiod, water chemistry, and soils (Cowardin et al., 1979; Winter et al., 1998). All these factors along with climate and land-use drivers influence overall carbon balance and GHG flux.

Grassland and forested wetlands are subject to a wide range of land use and management practices that influence the carbon balance and GHG flux (Faulkner et al., 2011; Gleason et al., 2011). For example, forested wetlands may be subject to silvicultural prescriptions and intensity of management, and hence, the carbon balance and GHG emissions should be evaluated on a rotation basis. In contrast, grassland wetlands may be grazed, hayed, or directly cultivated to produce a harvestable commodity. All these manipulations influence the overall GHG flux. This report will focus primarily on restoration and management practices associated with riverine and palustrine systems in forested, grassland, and riparian ecosystems; although other major wetlands systems are significant in the global carbon cycle (e.g., estuarine), these wetlands systems have received the most attention in terms of implementation of restoration and management practices to conserve wetlands habitats and sustain ecosystems services (Brinson and Eckles, 2011). Wetlands that have been drained for a commodity production, such as annual crops, are not considered wetlands in this guidance. Therefore, management of drained wetlands is addressed in other sections of the guidance, such as in Chapter 3.

Wetland emissions are largely controlled by the degree of water saturation as well as climate and nutrient availability. In aerobic conditions, common in most upland wetland ecosystems, decomposition releases of CO_2, and CH_4 emissions are more prevalent in anaerobic conditions. Typically, wetlands are a source of CH_4, with estimated global emissions of 55 to 150 million metric tons CH_4 per year (Blain et al., 2006). N_2O emissions from wetlands are typically low, unless an outside source of nitrogen is entering the wetland. If wetlands are drained, N_2O emissions are largely controlled by the fertility of the soil. Wetland drainage results in lower CH_4 emissions and an increase in CO_2 emissions due to oxidation of soil organic matter and an increase in N_2O emissions in nutrient rich soil. On the other hand, the creation of wetlands generates higher levels of CH_4 and lower levels of CO_2 (Blain et al., 2006).

Management Practices Impacting GHG Emissions from Wetlands

- Silvicultural Water Table Management
- Forest Harvesting Systems
- Forest Regeneration Systems
- Fertilization
- Conversion to Open Wetland
- Forest Type Change
- Water Quality Management
- Wetland Management for Waterfowl
- Constructed Wetlands for Wastewater Treatment
- Land-Use Change to Wetlands
- Actively Restoring Wetlands
- Actively Restoring Scrub-Grass Wetlands
- Constructing Wetlands
- Passive Restoration of Wetlands

Biomass carbon can change significantly with management of wetlands, particularly in peatlands, forested wetlands, or changes from forest to wetlands dominated by grasses and shrubs or open water. Peatlands cover approximately 400 million hectares or three percent of the global land surface, accounting for 450 billion metric tons of stored carbon (Couwenbert, 2009). Emissions from peatland degradation and fires are estimated at 2 billion metric tons of CO_2-eq per year (IPCC, 2011). In forested wetlands, there can also be significant carbon in dead wood, coarse woody debris, and fine litter. Harvesting practices will also influence the carbon stocks in wetlands to the extent that the wood is collected for products, fuel, or other purposes. Wetlands are also a source of N_2O emissions, primarily because of nitrogen runoff and leaching into groundwater from agricultural fields and/or livestock facilities. N_2O emissions from wetlands due to nitrogen inputs from surrounding fields or livestock facilities are considered an indirect emission of N_2O (de Klein et al., 2006). Direct N_2O emissions can also occur if management practices include nitrogen fertilization of the wetlands.

The text box, Management Practices Impacting GHG Emissions from Wetlands, lists the management practices in wetlands that have an influence on GHG emissions (CH_4 or N_2O) or carbon stock changes, and will be covered in more detail later in the report. Individual sections will deal with different types of wetlands including forested, grassland, and constructed wetlands that could occur in agricultural and forestry operations. The methods are restricted to estimation of emissions on palustrine wetlands that are influenced by a variety of management options such as water table management, timber or other plant biomass harvest, and wetlands that are managed with fertilizer applications.

2.4.3 Animal Production

GHG emissions from animal production systems consist of three main categories: enteric fermentation, housing, and manure management. The three categories are described in the sections that follow. Discussion about enteric fermentation and housing are addressed together in this report.

2.4.3.1 Enteric Fermentation and Housing

Enteric fermentation refers to the methane emissions resulting from animal digestive processes, while housing emissions refer to GHG emissions from manure that is stored within the housing structure (i.e., manure stored under a barn floor). GHG emissions arising from manure stored in housing have similar emissions to manure that is managed in stockpiles. More discussion on housing manure emissions can be found in Section 2.4.3.2 and Chapter 5.

For enteric fermentation, CH_4-producing micro-organisms, called methanogens, exist in the gastrointestinal tract of many animals. Ruminant animals (hoofed mammals) that have three or four chambered stomachs (and chew cud as a part of the digestive process), produce much more CH_4 than do other animals because of the presence and fermentative capacity of the rumen (the first stomach in a ruminant animal).

Management Practices Impacting GHG Emissions from Enteric Fermentation and Housing

- Dietary Fat
- Grain Source, Grain Processing, Starch Availability
- Feeding Co-Product Ingredients
- Roughage Concentration and Form
- Level of Intake
- Feed Additives and Growth Promoters
- Novel Microorganisms and Their Products
- Genetics

In the rumen, CH_4 formation is a disposal mechanism by which excess hydrogen from the anaerobic fermentation of dietary carbohydrate can be released. Control of hydrogen ions through methanogenesis assists in maintenance of an efficient microbial fermentation by reducing the partial pressure of hydrogen to levels that allow normal functioning of microbial energy transfer enzymes (Martin et al., 2010). CH_4 can also arise from hindgut fermentation, but the levels associated with hindgut fermentation are much lower than those of foregut fermentation. Although animals produce CO_2 through respiration, the only gas of concern in enteric fermentation processes is CH_4. In field studies, respiration chambers equipped with N_2O and NH_3 analyzers have confirmed that enteric fermentation does not result in the production of N_2O or NH_3 (Reynolds et al., 2010).

The text box, Management Practices Impacting GHG Emissions from Enteric Fermentation and Housing, lists several of the practices that can modify enteric fermentation emissions. Most of the practices relate to diet composition. These practices are covered in greater detail in Chapter 5.

2.4.3.2 Manure Management

Storage of animal manure (dung and urine) is a popular management practice because it reduces the need to buy commercial fertilizer, allows for more control over manure application, and has lower demands on farm labor. The treatment and storage of manure in management systems contributes to the GHG emissions of the agricultural sector. Anaerobic conditions, as found in many long-term storage systems, produce CH_4 through anaerobic decomposition. N_2O is produced either directly, as part of the nitrogen cycle through nitrification and denitrification, or indirectly, as a result of volatilization of nitrogen as NH_3 and nitrogen oxides (NO, NO_2, or NO_3) and runoff during handling.

Animal manure can be classified as:

- Slurry, where the dry matter is greater than 10 percent;
- Solid, where the dry matter is greater than 15 percent; or
- Liquid, where the dry matter is lower than 10 percent.

The four solid manure storage/treatment practices are: (1) temporary stack; (2) long-term stockpile; (3) composting; and (4) thermo-chemical conversion. The eight main liquid manure storage/treatment practices are: (1) anaerobic digestion; (2) nutrient removal; (3) anaerobic lagoon/runoff holding pond/storage tanks; (4) aerobic lagoon; (5) constructed wetland; (6) sand-manure separation; (7) combined aerobic treatment system; and (8) solid-liquid separation. Greater analysis of each of these systems is provided in Chapter 5.

Management Practices Impacting GHG Emissions from Manure Management

- Thermo-Chemical Conversion
- Anaerobic Digestion
- Liquid Manure Storage and Treatment-Sand-Manure Separation
- Liquid Manure Storage and Treatment–Solid-Liquid Separation

The magnitude of CH_4 and N_2O emissions that result from animal manure is dependent largely on the environmental conditions that the manure is subjected to. CH_4 is emitted when oxygen is not available for bacteria to decompose manure. Storage of manure in ponds, tanks, or pits, as is typical with liquid/slurry flushing systems, promote anaerobic conditions and the formation of CH_4. Storage of solid manure in stacks or dry lots or deposition of manure on pasture, range, or paddock lands tend to result in more oxygen-available conditions, and little or no CH_4 will be formed. Other factors that influence CH_4 generation include the ambient temperature, moisture content, residency time, and manure composition (which is dependent on the diet of the livestock, growth rate, and type of digestive system) (U.S. EPA, 2011).

The production of N_2O from managed livestock manure depends on the composition of the manure and urine, the type of bacteria involved, the oxygen and liquid content of the system, and the environment for the manure after excretion (U.S. EPA, 2011). N_2O occurs when the manure is first subjected to aerobic conditions where NH_3 and organic nitrogen are converted to nitrates and nitrites (nitrification), and if conditions become sufficiently anaerobic, the nitrates and nitrites can be denitrified (reduced to nitrogen oxides and nitrogen gas) (Groffman et al., 2000). N_2O is an intermediate product of both nitrification and denitrification and can be directly emitted from soil as a result of either of these processes. Dry waste handling systems are generally oxygenated but have pockets of anaerobic conditions from decomposition; these systems have conditions that are most conducive to the production of N_2O (USDA, 2011).

Some manure management systems can effectively mitigate the release of GHG emissions from livestock manure. The text box, Management Practices Impacting GHG Emissions from Manure Management, lists several of the practices that can modify manure management emissions.

2.4.4 Forestry

Forest systems represent a significant opportunity to mitigate GHGs through the sequestration and temporary storage of forest carbon stocks. Forests remove CO_2 from the atmosphere through photosynthesis and store carbon in forest biomass (e.g., stems, root, bark, leaves). Respiration releases CO_2 to the atmosphere. Net forest carbon stocks increase over time when carbon sequestration during photosynthesis exceeds carbon released during respiration. Other GHGs are also exchanged by forest ecosystems—e.g., CH_4 from microbial communities in forest soil and N_2O from fertilizer use.

Harvesting forests releases some sequestered carbon to the atmosphere, while the remaining carbon passes in HWPs, the fate of which (e.g., combustion for energy, manufacture of durable wood products, disposal in landfills) determines the rate at which the carbon is returned to the atmosphere.

Management Practices Impacting Net GHG Emissions from Forestry

- Establishing and Reestablishing Forest
- Avoiding Clearing Forest
- Stand Density Management
- Site Preparation Techniques
- Vegetation Control
- Planting
- Natural Regeneration
- Fertilization
- Selection of Rotation Length
- Harvesting and Utilization Techniques
- Fire and Fuel Load Management
- Reducing the Risk of Emissions from Natural Disturbances
- Short Rotation Woody Crops

There are many forestry activities (i.e., management practices) relevant to reducing GHG emissions and/or increasing carbon stocks in the forestry sector including establishing and/or re-establishing forest, avoided forest clearing, and forest management. More information on each is included below.

The Chapter 6 describes methods for the various source categories contributing to the GHG flux from forests. These source categories include forest carbon accounting—e.g., live trees, understory, standing dead, down dead wood, forest floor or litter, forest soil organic carbon—establishing, re-establishing, and clearing forest, forest management, HWPs, urban forestry, and natural disturbances (e.g., forest fires). This subsection briefly describes these source categories. Descriptions of the current tools and methods used to estimate GHG flux from these source categories is discussed later in Chapter 6.

Forest Carbon. Accounting for forest carbon (i.e., forest biomass) typically divides the forest into forest carbon pools—e.g., live trees, understory, standing dead, down dead wood, forest floor or litter, forest soil organic carbon—the definitions for which are developed around a common set in use by a number of publications, which are further outlined in Chapter 6. The methods for estimating the key forest carbon pools are well developed and fairly standard.

Establishing, Re-Establishing, and Clearing Forest. In addition to forestland remaining forestland, there are three distinct processes that can significantly alter forest carbon stocks, and are termed:

forest establishment (i.e., afforestation), forest re-establishment (i.e., reforestation), and forest clearing (i.e., deforestation). Each of these processes alters stocks of carbon in aboveground and belowground carbon pools. Establishment involves the intentional planting (or allowing the natural process of secondary succession) on land that was not previously forest. Reestablishment is returning land that was recently forest back into forest. In either case, establishing forest will generally increase the carbon stocks in aboveground and belowground carbon pools over time. Forest clearing is the removal and/or conversion of a forest system into another land cover (cropland, grazing land, etc.) and is the most significant source of GHG emissions from forests.

Forest Management. Forest management describes the range of practices employed by landowners to meet their objectives (e.g., timber production) while satisfying biological, economic, and social constraints. A number of the practices used by forest managers to achieve their objectives impact the carbon dynamics in forests either by enhancing forest growth or accelerating the loss of forest carbon. The management practices include: stand density management (e.g., under planting, pre-commercial and commercial thinning); site preparation techniques (e.g., mechanical methods, chemical application, prescribed burning); vegetation control; planting (e.g., planting density, species selection, genetic improvement); natural regeneration; fertilization (e.g., nitrogen and phosphorous fertilizer application); selection of rotation lengths; harvesting and utilization techniques; fire and fuel load management; reducing the risk of emissions from pests and disease; and establishing biomass plantations (i.e., short rotation woody crops).

Harvested Wood Products. A proportion of the wood carbon harvested from forests ends up in solid wood, paper, or other products, which are collectively known as HWPs. The carbon contained in these products can remain stored for years or decades depending on the end use, and may eventually be combusted, decay, or be diverted to landfills.

Urban Forestry. Urban (or urban community) forest describes the population of trees within an urban area. Urban trees directly store atmospheric carbon as woody biomass and also affect local climate (e.g., secondary effects). The maintenance of urban trees also affects GHG emissions in urban areas (i.e., indirect effects).

Natural Disturbances. Natural disturbances in forest systems (e.g., forest fires, pests and disease, storms) can significantly impact forest carbon stocks either directly in the case of combustion from forest fires or indirectly by converting live biomass to dead or converting standing trees to downed dead wood and accelerating decomposition.

The text box, Management Practices Impacting Net GHG Emissions from Forestry, lists the management practices relevant to reducing GHG emissions and/or increasing carbon stocks in the forestry sector including establishing and/or reestablishing forest, avoiding forest clearing, and improving forest management.

2.5 Land-Use Change

Converting land parcels from one land-use category to another can have a significant effect on a parcel's carbon stocks. For example, carbon stock gains can be realized by converting cropland soils to wetlands or forestland, while carbon stock losses often result from a conversion from forestlands to grazing lands. A land-use categorization system that is consistent and complete (both temporally and spatially) is needed in order to assess land use and land-use change status within an entity's boundaries. All of the land within an entity's boundary should be classified according to the following land-use types: cropland, grazing land, forestland, wetland, settlements (e.g., residential and commercial buildings), and other land (e.g., bare soil, rock); see definitions provided above. Individual parcel areas should sum to the total land area before and after land-use change.

In many cases, the methods proposed to estimate contributions to the GHG flux resulting from land-use change are the same as those used to estimate carbon stock changes in the individual cropland and grazing land, wetland, and forestry chapters; although, in specific cases, guidance is also provided on reconciling carbon-stock estimates between discrete data sets and estimation methods (e.g., reconciling forest soil carbon estimates and cropland soil carbon estimates for land-use change from forestland to cropland). The methods for quantifying GHG flux from land-use change are intended for use at the entity scale on lands managed to enhance the production of food, feed, fiber, and renewable energy. Methods are currently not provided for estimating emissions from energy used when converting land use from one category to another. Neither are methods provided for land-use change from settlements or the "other land" category to cropland, grazing land, wetland or forestland. The methods have been developed for U.S. conditions and are considered applicable to agricultural and forestry production systems in the United States. This subsection briefly describes the source categories covered. Further descriptions of the current tools and methods used to estimate GHG flux from these source categories are discussed later in Chapter 7.

Annual Change in Carbon Stocks in Dead Wood and Litter Due to Land Conversion. Live and dead biomass carbon stocks and soil organic carbon constitute a significant carbon sink in many forest and agricultural lands. Following land-use conversion, the estimation of dead biomass carbon stock changes during transition periods requires that the area subject to land-use change on the entity's operation be tracked for the duration of a 20-year transition period.

Change in Soil Organic Carbon Stocks for Mineral Soils. Soil organic carbon stocks are influenced by land-use change (Aalde et al., 2006) due to changes in productivity that influence carbon inputs and to changes in soil management that influence carbon outputs (Davidson and Ackerman, 1993; Ogle et al., 2005; Post and Kwon, 2000). The most significant changes in soil organic carbon occur with land-use change, particularly conversions to croplands, due to changes in the disturbance regimes and associated effects on soil aggregate dynamics (Six et al., 2000).

Specific mitigation practices are not explicitly described in Chapter 7; however, avoiding land-use conversions that result in significant carbon losses could mitigate net GHG emissions (e.g., avoiding the conversion of forestlands to grazing lands).

2.6 Uncertainty

Quantifying the uncertainty of GHG emissions and reductions from agriculture and forestry practices is an important aspect of decisionmaking for farmers and landowners as the uncertainty range for each GHG estimate communicates our level of confidence that the estimate reflects the actual balance of GHG exchange between the biosphere and the atmosphere. In particular, a farm, ranch, or forest landowner may be more inclined to invest in management practices that reduce net GHG emissions if the uncertainty range for an estimate is low, meaning that higher confidence in the estimates exists. This report presents the approach for accounting for the uncertainty in the estimated net emissions based on the methods presented in this report.[6] A Monte Carlo approach

[6] The IPCC Good Practice Guidance (IPCC, 2000) recommends two approaches—Tier 1 and Tier 2—for developing quantitative estimates of uncertainty for emissions estimates for source categories. The Tier 1 method uses error propagation equations. These equations combine the uncertainty associated with the activity data and the uncertainty associated with the emission (or other) factors. This approach is appropriate where emissions (or removals) are estimated as the product of activity data and an emission factor or as the sum of individual sub-source category values. The Tier 2 method utilizes the Monte Carlo Stochastic Simulation technique. Using this technique, an estimate of emission (or removal) for a particular source category is generated many times via an uncertainty model, resulting in an approximate PDF for the estimate.

was selected as the method for estimating the uncertainty around the outputs from the methodologies in this report, as it is currently the most comprehensive, sound method available to assess the uncertainty at the entity scale. Limitations and data gaps exist; however, as new data become available, the method can be improved over time. Implementation of a Monte Carlo analysis is complicated and requires the use of a statistical tool to produce a probability density function (PDF)[7] around the GHG emissions estimate.[8] From the probability density function, the uncertainty estimate can be derived and reported.

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Where sufficient and reliable uncertainty data for the input variables are available, the Tier 2 method is the preferred option.

[7] The integral of a PDF over a given interval of values is the probability for a random variable to take on some value in the interval. That is, the PDF is a function giving probability "densities" and its integral gives probabilities. A narrower PDF for an estimate indicates smaller variance around the central/most likely value, i.e., a higher probability of the value to be closer to the central/most likely value. The uncertainty for such an estimate is lower.

[8] Given the complexity of Monte Carlo analysis and the necessity for a tool, the approach presented here is not intended for development by a landowner, rather it is intended for use in developing a tool that a landowner would use to assess uncertainty estimates.

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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.
	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).
	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).

[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 (≥ 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).

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

$$CaCO_3 + 2HNO_3 = Ca^{2+} + 2NO_3^- + H_2O + CO_2 \qquad \text{Eq. 2}$$

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_2O 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_2O 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_2O 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 andN_2O rates, while CH_4 production from grassland and woody detritus was likely caused by termite activity. The

[3] Woody encroachment will eventually lead to a transition from grazing land to a forest. See Chapter7: 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

[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

[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 = Mean annual herbaceous biomass (metric tons CO_2-eq year^{-1})

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

t = Current year stocks

t-1 = 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 = Mean annual herbaceous biomass carbon stock (metric tons CO_2-eq year^{-1})

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

R:S = Root-shoot ratio (unitless)

A = Area of land parcel (ha)

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

Y_f = 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		Distribution	Data Source
			Low (%)	High (%)		
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_2O 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_2O 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_2O-N ha^{-1} $year^{-1}$).
- This method relies on entity specific activity data as input into the equations.

Method for Estimating Soil Indirect N_2O Emissions

- This method uses the IPCC equation for indirect soil N_2O (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_2O 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_2O 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_2O 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_2O emissions from cropland. Two are based on emission factors, the proportion of nitrogen added to a crop that becomes N_2O. 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

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

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

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

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).

[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:

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

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 $\Delta N_f >$ 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 $\Delta N_f \leqq$ 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_2O 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_2O emitted by ecosystem receiving volatilized nitrogen (metric tons N_2O-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_2O in receiving ecosystem; IPCC Tier 1 EF = 0.01 (metric tons N_2O-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_2O 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_2O 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_2O emitted by ecosystem receiving leached and runoff nitrogen (metric tons N_2O-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_2O in receiving ecosystem; IPCC Tier 1 EF = 0.0075 (metric tons N_2O-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_2O 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.

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).

3.5.4.4 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.

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.

[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_2O Emissions

Parameter	Estimated Value	Units	Effective Lower Limit	Effective Upper Limit	Distribution	Data Source
Typical direct N_2O 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_2O-N (metric tons N)$^{-2}$) ha year			Normal	Appendix 3-A
Base EF scalar – for grasslands	0.117	(metric tons N_2O-N (metric tons N)$^{-2}$) ha year			Normal	Appendix 3-A
Emission rate for cropped Histosols	0.008	metric tons N_2O-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 NOx 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 ($\pm$ 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 abandon ment)	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_2O Emissions from Rice Cultivation

- IPCC equations developed by Lasco et al. (2006) for CH_4 and de Klein et al. (2006) for N_2O.
  - 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_2O 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_2O 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_2O 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

$$CH_{4Rice} = CH_{4GWP} \times \Sigma_{i\,j\,k} \left(EF_{i\,j\,k} \text{ x } t_{i\,j\,k} \text{ x} A_{i\,j\,k} \text{x } 10^{-3}\right)$$

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:

SFo = 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

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

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_2CO_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_2O	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

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

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 ($g\ N_20\ kg^{-1}$)	0.21	93%	93%	Normal	IPCC (2006)
N_2O EF for crop residue ($g\ N_20\ kg^{-1}$)	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_2MW = 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.

[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_2O Modeling Framework Specifications

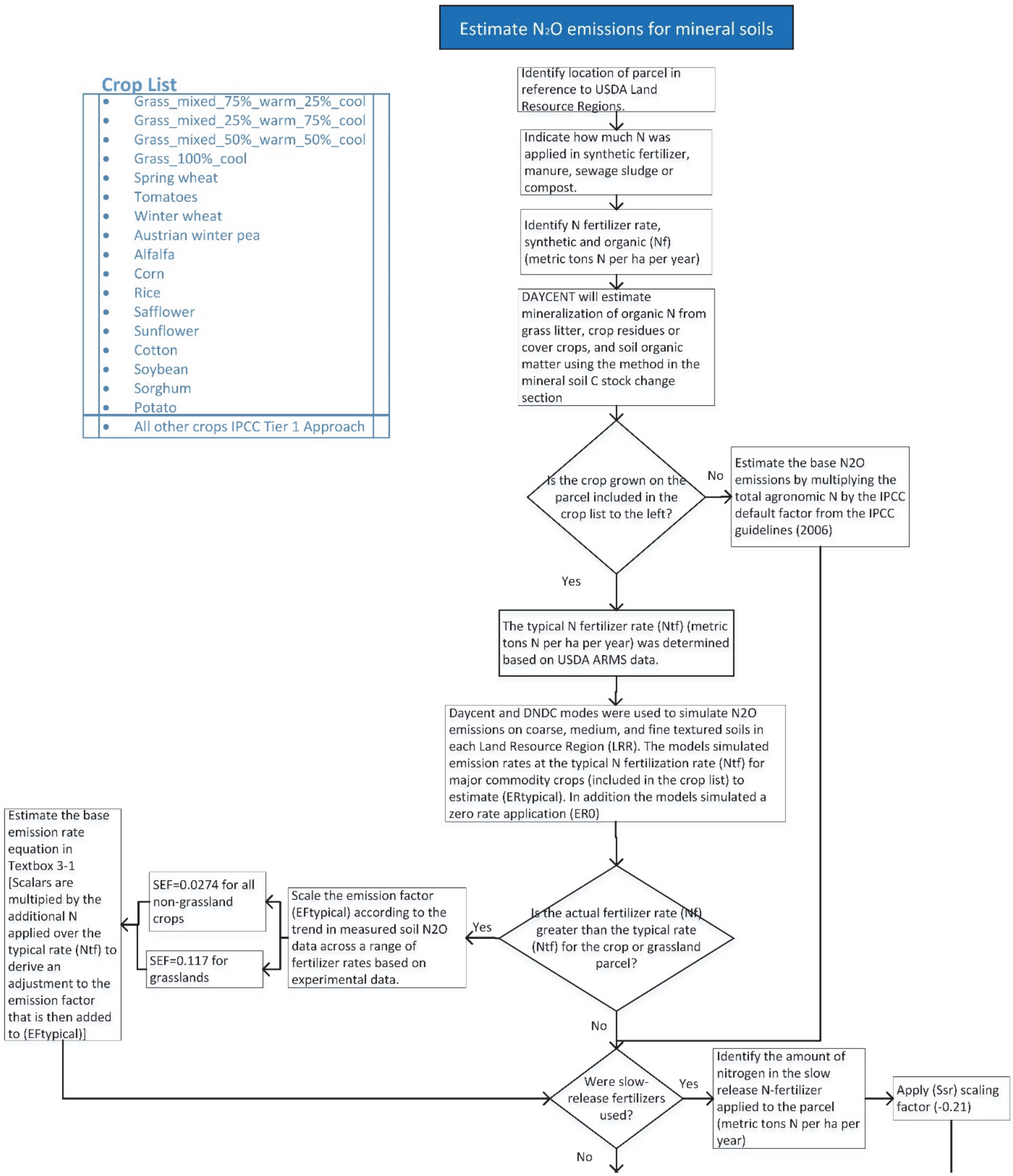
Soil N_2O emissions are estimated using a combination of process-based modeling, empirical scalars based on experimental data, and scaling factors for practices influencing the N_2O emissions as represented in the base emission rates (Section 3.5.4.1, Equations 3-8 and 3-9, and Text box 3-1). This appendix provides more information about the process-based models, in addition to the derivation of empirical scalars and the practice-based scaling factors.

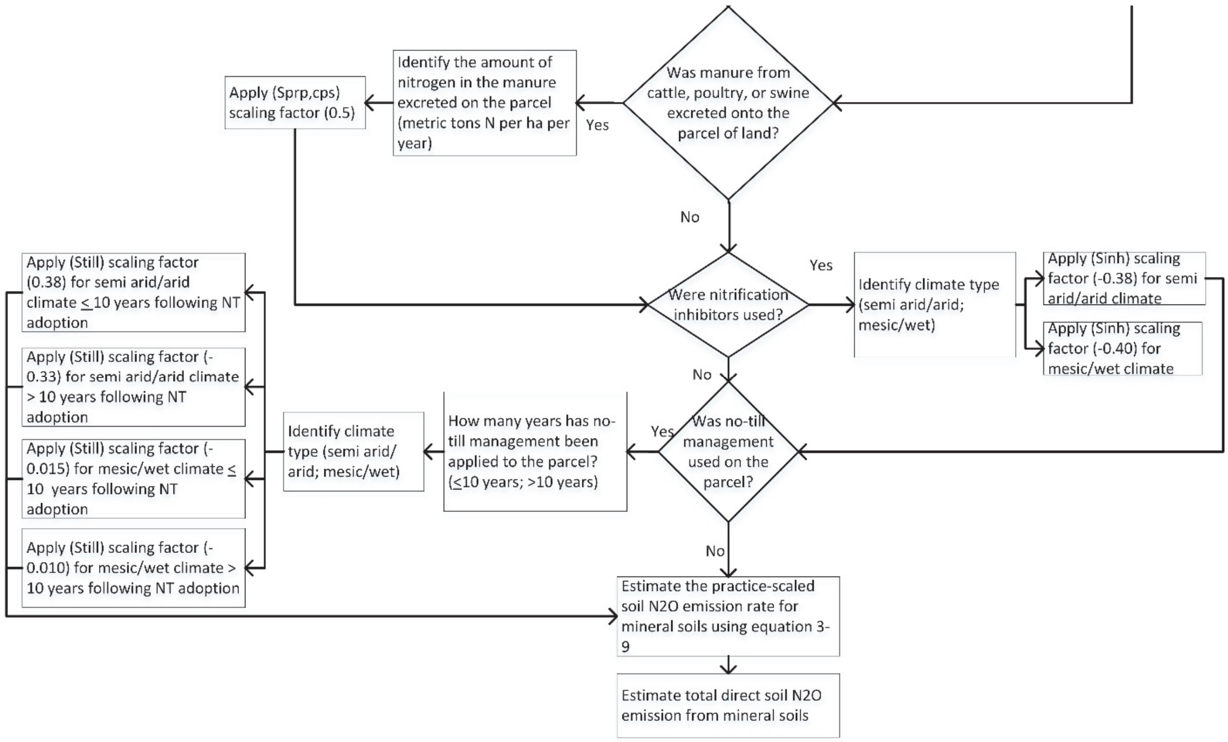
DAYCENT and DNDC models were used to estimate N_2O emissions for the typical fertilizer rate and a 0-level nitrogen fertilization rate associated with major crops in each USDA LRR. Crops simulated are listed in Table 3-A. 1; base emission rates for other crops (e.g., sugar cane, millet, rye) were estimated using the Tier 1 emission factor (one percent of nitrogen inputs). To estimate emission factors from the model output, the N_2O emissions at the 0-level addition was subtracted from the N_2O 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_2O 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_2O 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_2O 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_2O emissions from mineral soils.

Figure 3-A.1: Decision Tree for Estimating N_2O 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

[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

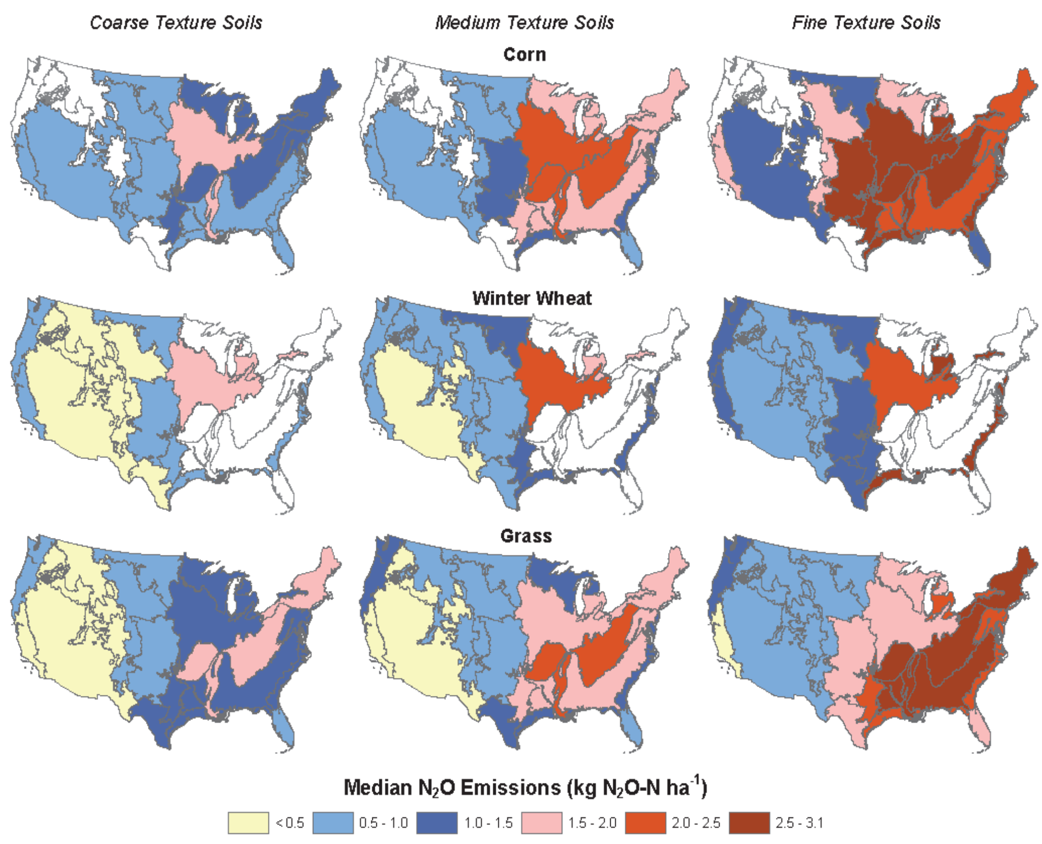


Table 3-A. 1 provides the 2.5, 50, and 97.5 percentile base emission rates for each crop, LRR, and soil texture combination. Emission rates are kgN_2O-N per ha when crops are fertilized at typical nitrogen rates.

[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 *x*N 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.

Breitenbeck, G.A., and J.M. Bremner. 1986. Effects of rate and depth of fertilizer application on emission of nitrous oxide from soil fertilized with anhydrous ammonia. *Biology and fertility of soils*, 2(4):201-204.

Cardenas, L.M., R. Thorman, N. Ashlee, M. Butler, et al. 2010. Quantifying annual N_2O emission fluxes from grazed grassland under a range of inorganic fertiliser nitrogen inputs. *Agriculture, Ecosystems and Environment*, 136:218-226.

Chang, C., C.M. Cho, and D.H. Janzen. 1998. Nitrous oxide emission from long-term manured soils. *Soil Science Society America Journal*, 62:677-682.

Ding, W., Y. Cai, X. Cai, K. Yagi, et al. 2007. Nitrous oxide emissions from an intensively cultivated maize-wheat rotation in soil in the North China Plain. *Science and the Total Environment*, 373.

Halvorson, A.D., S.J. Del Grosso, and C.A. Reule. 2008. Nitrogen, Tillage, and Crop Rotation Effects on Nitrous Oxide Emissions from Irrigated Cropping Systems. *Journal of Environmental Quality*, 37(4):1337-1344.

Hoben, J.P., R.J. Gehl, N. Millar, P.R. Grace, et al. 2011. Nonlinear nitrous oxide (N_2O) response to nitrogen fertilizer in on-farm corn crops of the US Midwest. *Global Change Biology*, 17(2):1140-1152.

Kaiser, E.A., K. Kohrs, M. Kücke, E. Schnug, et al. 1998. Nitrous oxide release from arable soil: importance of N-fertilization, crops and temporal variation. *Soil Biology and Biochemistry*, 30:1553-1563.

Kammann, C., L. Grünhage, C. Müller, S. Jacobi, and H.-J. Jäger. 1998. Seasonal variability and mitigation options for N_2O emissions from differently managed grasslands. *Environmental Pollution*, 102(S1):179-186.

Kim, D.-G., G. Hernandez-Ramirez, and D. Giltrap. 2013. Linear and nonlinear dependency of direct nitrous oxide emissions on fertilizer nitrogen input: a meta-analysis. *Agriculture, Ecosystems and Environment*, 168:53-65.

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.

Ma, B.L., T.Y. Wu, N. Tremblay, W. Deen, et al. 2010. Nitrous oxide fluxes from corn fields: on-farm assessment of the amount and timing of nitrogen fertilizer. *Global Change Biology*, 16(1):156-170.

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.

Signor, D., C.E.P. Cerri, and R. Conant. 2013. N_2O emissions due to nitrogen fertilizer applications in two regions of sugarcane cultivation in Brazil. *Environmental Research Letters*, 8(1):015013.

Song, C., and J. Zhang. 2009. Effects of soil moisture, temperature, and nitrogen fertilization on soil respiration and nitrous oxide emission during maize growth period in northeast China. *Acta Agriculturae Scandinavia*, 59:97-106.

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.

Velthof, G.L., O. Oenema, R. Postma, and M.L. Van Beusichem. 1997. Effects of type and amount of applied nitrogen fertilizer on nitrous oxide fluxes from intensively managed grassland. *Nutrient Cycling in Agroecosystems*, 46:257-267.

Zebarth, B.J., P. Rochette, and D.L. Burton. 2008. N_2O emissions from spring barley production as influenced by fertilizer nitrogen rate. *Canadian Journal of Soil Science*, 88:197-205.

Zhang, J., and X. Han. 2008. N_2O emission from the semi-arid ecosystem under mineral fertilizer (urea and superphosphate) and increased precipitation in northern China. *Atmospheric Environment*, 42:291-302.

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.

Nitrogen Fertilizer Placement:

Burton, D.L., X. Li, and C.A. Grant. 2008. Influence of fertilizer nitrogen source and management practice on N_2O emissions from two Black Chernozemic soils. *Canadian Journal of Soil Science*, 88:219-227.

Drury, C.F., W.D. Reynolds, C.S. Tan, T.W. Welacky, et al. 2006. Emissions of Nitrous Oxide and Carbon Dioxide. *Soil Science Society of America Journal*, 70(2):570-581.

Engel, R., D.L. Liang, R. Wallander, and A. Bembenek. 2010. Influence of Urea Fertilizer Placement on Nitrous Oxide Production from a Silt Loam Soil. *Journal of Environmental Quality*, 39(1):115-125.

Halvorson, A.D., and S.J. Del Grosso. 2013. Nitrogen placement and source effects on nitrous oxide emissions and yields of irrigated corn. *Journal of Environmental Quality*, 42(In press).

Hou, A.X., and H. Tsuruta. 2003. Nitrous oxide and nitric oxide fluxes from an upland field in Japan: effect of urea type, placement, and crop residues. *Nutrient Cycling in Agroecosystems*, 65:191-200.

Hultgreen, G., and P. Leduc. 2003. *The effect of nitrogen fertilizer placement, formulation, timing, and rate on greenhouse gas emissions and agronomic performance*: Agriculture Agri-Food Canada, Prairie Agricultural Machinery Institute.

Liu, X.J., A.R. Mosier, A.D. Halvorson, and F.S. Zhang. 2005. Tillage and nitrogen application effects on nitrous and nitric oxide emissions from irrigated corn fields. *Plant and Soil*, 276:235-249.

Maharjan, B., and R.T. Venterea. In review. Nitrite dynamics explain fertilizer management effects on nitrous oxide emissions in maize. *Submitted to Soil Biology and Biochemistry*.

Zebarth, B.J., P. Rochette, D.L. Burton, and M. Price. 2008. Effect of fertilizer nitrogen management on N_2O emissions in commercial corn fields. *Canadian Journal of Soil Science*, 88:189-195.

Nitrification Inhibitors:

Akiyama, H., H. Tsuruta, and T. Watanabe. 2000. N_2O and No Emissions from Soils after the Application of Different Chemical Fertilizers. *Chemosphere-Global Change Science*, 2(3):313-320.

Ball, B.C., K.C. Cameron, H.J. Di, and S. Moore. 2012. Effects of Trampling of a Wet Dairy Pasture Soil on Soil Porosity and on Mitigation of Nitrous Oxide Emissions by a Nitrification Inhibitor, Dicyandiamide. *Soil Use and Management*, 28(2):194-201.

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Cui, M., X.C. Sun, C.X. Hu, H.J. Di, et al. 2011. Effective Mitigation of Nitrate Leaching and Nitrous Oxide Emissions in Intensive Vegetable Production Systems Using a Nitrification Inhibitor, Dicyandiamide. *Journal of Soils and Sediments*, 11(5):722-730.

de Klein, C.A.M., K.C. Cameron, H.J. Di, G. Rys, et al. 2011. Repeated Annual Use of the Nitrification Inhibitor Dicyandiamide (Dcd) Does Not Alter Its Effectiveness in Reducing N_2O Emissions from Cow Urine. *Animal Feed Science and Technology*, 166-167:480-491.

Delgado, J.A., and A.R. Mosier. 1996. Mitigation Alternatives to Decrease Nitrous Oxides Emissions and Urea-Nitrogen Loss and Their Effect on Methane Flux. *Journal of Environmental Quality*, 25(5):1105-1111.

Dittert, K., R. Bol, R. King, D. Chadwick, et al. 2001. Use of a novel nitrification inhibitor to reduce nitrous oxide emission from N15-labelled dairy slurry injected into soil. *Rapid Communications in Mass Spectrometry*, 15:1291-1296.

Dobbie, K.E., and K.A. Smith. 2003. Impact of Different Forms of N Fertilizer on N_2O Emissions from Intensive Grassland. *Nutrient Cycling in Agroecosystems*, 67(1):37-46.

Ghosh, S., D. Majumdar, and M.C. Jain. 2003. Methane and Nitrous Oxide Emissions from an Irrigated Rice of North India. *Chemosphere*, 51(3):181-195.

Hadi, A., O. Jumadi, K. Inubushi, and K. Yagi. 2008. Mitigation Options for N_2O Emission from a Corn Field in Kalimantan, Indonesia. *Soil science and plant nutrition*, 54(4):644-649.

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Halvorson, A.D., S.J. Del Grosso, and F. Alluvione. 2010. Tillage and Inorganic Nitrogen Source Effects on Nitrous Oxide Emissions from Irrigated Cropping Systems. *Soil Science Society of America Journal*, 74(2):436-445.

Halvorson, A.D., S.J. Del Grosso, and C.P. Jantalia. 2011. Nitrogen Source Effects on Soil Nitrous Oxide Emissions from Strip-Till Corn. *Journal of Environmental Quality*, 40(6):1775-1786.

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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 ($N_{\mathbf{man}}$ and $N_{\mathbf{comp}}$)	IPCC 2006 Guidelines (See Chapter 11 Section 11.2.1.1)	
N_{resid}		Equation 3-B-1 Residue nitrogen (See below)

[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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Chapter 4 Quantifying Greenhouse Gas Sources and Sinks in Managed Wetland 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
DNDC	Denitrification-Decomposition
EPA	Environmental Protection Agency
FVS	Forest Vegetation Simulator
GHG	Greenhouse gas
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
N	Nitrogen
N_2O	Nitrous oxide
NO_x	Mono-nitrogen oxides
NRCS	USDA Natural Resources Conservation Service
P	Phosphorous
SOC	Soil organic carbon
Tg	Teragrams
USDA	U.S. Department of Agriculture
USDA-ARS	U.S. Department of Agriculture, Agricultural Research Service

4 Quantifying Greenhouse Gas Sources and Sinks in Managed Wetland Systems

This chapter provides methodologies and guidance for reporting greenhouse gas (GHG) emissions and sinks at the entity scale for managed wetland systems. More specifically, it focuses on methods for managed palustrine wetlands.[1] Section 4.1 provides an overview of wetland systems 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 4.2 presents the various management practices that influence GHG emissions in wetland systems and land-use change to wetlands. Section 4.3 provides the estimation methods for biomass carbon in wetlands and for soil carbon, N_2O, and CH_4 emissions and sinks. Finally, Section 4.4 includes a discussion of research gaps in wetland management.

4.1 Overview

Wetlands occur across most landforms, existing as natural unmanaged and managed lands, restored lands following conversion from another use (typically agriculture), and as constructed systems for water treatment, such as anaerobic lagoons. All wetlands sequester carbon and are a source of GHGs. Table 4-1 provides a description of the sources of emissions or sinks and the gases estimated in the methodology.

Table 4-1: Overview of Wetland Systems Sources and Associated Greenhouse Gases

Source	Method for GHG Estimation			Description
	CO_2	N_2O	CH_4	
Biomass carbon	✓			Provisions for estimating aboveground biomass for wetland forests and above and belowground biomass and carbon are included for shrub and grass wetlands in this chapter. Aboveground biomass for forested wetlands and shrub and grass wetlands includes live vegetation, trees, shrubs, and grasses, standing dead wood (dead biomass), and down dead organic matter—litter layer (dead biomass).
Soil C, N_2O, and CH_4 in wetlands	✓	✓	✓	The production and consumption of carbon in wetland-dominated landscapes are important for estimating the contribution of GHGs, including CO_2, CH_4, and N_2O emitted from those areas to the atmosphere. The generation and emission of GHGs from wetland-dominated landscapes are closely related to inherent biogeochemical processes, which also regulate the carbon balance (Rose and Crumpton, 2006). However, those processes are highly influenced by the land use, vegetation, soil organisms, chemical and physical soil properties, geomorphology, and climate (Smemo and Yavitt, 2006).

[1] Palustrine wetlands include non-tidal and tidal wetlands that are primarily composed of trees, shrubs, persistent emergent, emergent mosses, or lichens, where salinity due to ocean-derived salts is below 0.5 ‰ (parts per thousand). Palustrine wetlands also include those wetlands lacking vegetation that have the following four characteristics: (1) are less than 20 acres; (2) do not have active wave-formed or bedrock shorelines; (3) have a maximum water depth of less than 6.5 ft. at low water; and (4) have a salinity due to ocean-derived salts less than 0.5% (Stedman and Dahl, 2008).

4.1.1 Overview of Management Practices and Resulting GHG Emissions

This chapter provides methods for estimating carbon stock changes and CH_4 and N_2O emissions from naturally occurring wetlands[2] and restored wetlands on previously converted wetland sites. Constructed wetlands for water treatment, including detention ponds, are engineered systems that are beyond the scope considered here because they have specific design criteria for influent and effluent loads. In addition, the methods are restricted to estimation of emissions on palustrine wetlands that are influenced by a variety of management options such as water table management, timber, or other plant biomass harvest, and wetlands that are managed with fertilizer applications. The methods are based on established principles and represent the best available science for estimating changes in carbon stocks and GHG fluxes associated with wetland management activities. However, given the wide diversity of wetlands types and the variety of management regimes, the basis for the methods provided in this section are not as well-developed as other sections in this guidance (i.e., Cropland and Grazing Lands, Animal Production, and Forestry Methods). Table 4-2 provides a summary of the methods and their corresponding section for the sources of emissions estimated in this report.

Table 4-2: Overview of Wetland Systems Sources, Method, and Section

Section	Source	Method
4.3.1	Biomass carbon	Methods for estimating forest vegetation and shrub and grassland vegetation biomass carbon stocks use a combination of the Forest Vegetation Simulator (FVS) model and lookup tables for dominant shrub and grassland vegetation types found in Chapter 3, Cropland, and Grazing Land. If there is a land-use change to agricultural use, methods for cropland herbaceous biomass are provided in Chapter 3.
4.3.2	Soil C, N_2O, and CH_4 in wetlands	The Denitrification-Decomposition (DNDC) process-based biogeochemical model is the method used for estimating soil C, N_2O, and CH_4 emissions from wetlands. DNDC simulates the soil carbon and nitrogen balance and generates emissions of soil-borne trace gases by 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 (Dai et al., 2011; Zhang et al., 2002), using plant growth estimated as described in Section 4.3.1.

4.1.1.1 Description of Sector

The National Wetlands Inventory broadly classifies wetlands into five major systems: (1) marine, (2) estuarine, (3) riverine, (4) lacustrine, and (5) palustrine (Cowardin et al., 1979). Four of those systems (marine, estuarine, riverine, and lacustrine) are open-water bodies and not considered within the methods described in this guidance. Palustrine wetlands encompass the wetland types occurring on the land and are further classified by major vegetative life form and wetness or flooding regime. Common palustrine wetlands are illustrated in Figure 4-1. For example, forested wetlands are often classified as palustrine—forested. Similarly, most grass wetlands are classified as palustrine—emergent, reflecting emergent vegetation (e.g., grasses and sedges). Wetlands also vary greatly with respect to groundwater and surface water interactions that directly influence

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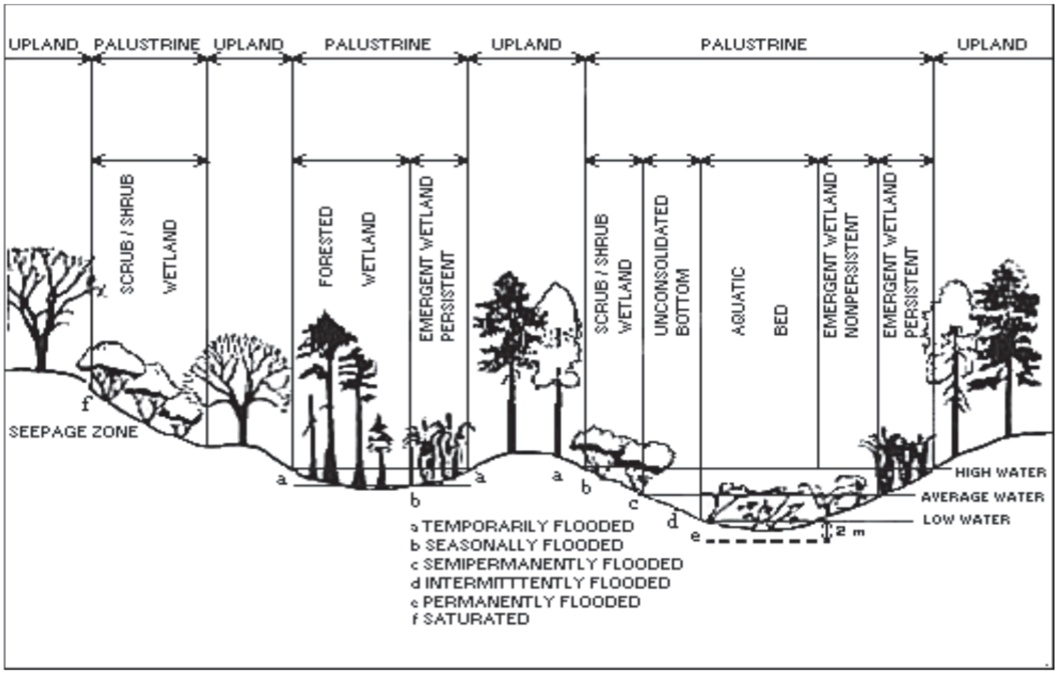
[2] Wetlands are defined in Chapter 7, Land Use Change. Wetlands that are converted to a non-wetland status should be considered in the appropriate chapter (e.g., Cropland and Grazing Lands, Animal Production, and Forestry Methods).

hydroperiod (i.e., the length of time and portion of the year the wetland holds water), water chemistry, and soils (Cowardin et al., 1979; Winter et al., 1998). All these factors along with climate and land use drivers influence the overall carbon balance and GHG fluxes.

Figure 4-1: Palustrine Wetland Classes Based on Vegetation and Flooding Regime



Source: Cowardin et al. (1979).

Grassland and forested wetlands are subject to a wide range of land use and management practices that influence the carbon balance and GHG flux (Faulkner et al., 2011; Gleason et al., 2011). For example, forested wetlands may be subject to silvicultural prescriptions with varying intensities of management through the stand rotation; hence, the carbon balance and GHG emissions should be evaluated on a rotation basis, which could range from 20 to more than 50 years. In contrast, grass wetlands may be grazed, hayed, or directly cultivated to produce a harvestable commodity annually. While each management practice may influence carbon sequestration and GHG fluxes, the effect is dependent on vegetation, soil, hydrology, climatological conditions, and the management prescriptions. This section focuses on restoration and management practices associated with palustrine wetlands that are typically forested or grassland.

4.1.1.2 Resulting GHG Emissions

GHG emissions from wetlands are largely controlled by water table depth and duration as well as climate and nutrient availability. Under aerobic soil conditions, which are common in most upland ecosystems, organic matter decomposition releases CO_2, and atmospheric CH_4 can be oxidized in the surface soil layer (Trettin et al., 2006). In contrast, the anaerobic soils that characterize wetlands can produce CH_4 (depending on the water table position) in addition to emitting CO_2. Accordingly, wetlands are an inherent source of CH_4, with globally estimated emissions of 55 to 150 teragrams (Tg) of CH_4 per year (Blain et al., 2006).

To accommodate entity-scale reporting in the United States for agricultural and forestry operations, Tier 2 and 3 methods address palustrine wetlands containing both organic and mineral hydric soils. These wetlands may be influenced by agricultural and forestry management, and methods are currently available for both types of management. This chapter provides methodologies for the following wetland source categories:

1. Biomass carbon in forested, shrub, and grass wetlands;
2. Soil carbon sinks in wetlands; and
3. N_2O and CH_4 emissions in wetlands.

Biomass carbon can change significantly with management of wetlands, particularly in forested wetlands, changes from forest to wetlands dominated by grasses and shrubs, or open water. In forested wetlands, there can also be significant carbon in dead wood, coarse woody debris, and fine litter. Harvesting practices will also influence the carbon stocks in wetlands to the extent the wood is collected for products, fuel, or other purposes.

Wetlands are also a source of soil N_2O emissions, primarily because of nitrogen runoff from adjoining uplands and leaching into groundwater from agricultural fields and/or animal production facilities. N_2O emissions from wetlands due to nitrogen inputs from surrounding fields or animal production are considered indirect emissions of N_2O (de Klein et al., 2006). Methodologies for estimating indirect N_2O are provided in the respective source chapter (i.e., Chapter 3, Cropland and Grazing Lands, or Chapter 5, Animal Production). However, direct N_2O emissions occur in wetlands if management practices include nitrogen fertilization, hence, guidance is provided for this source of emissions.

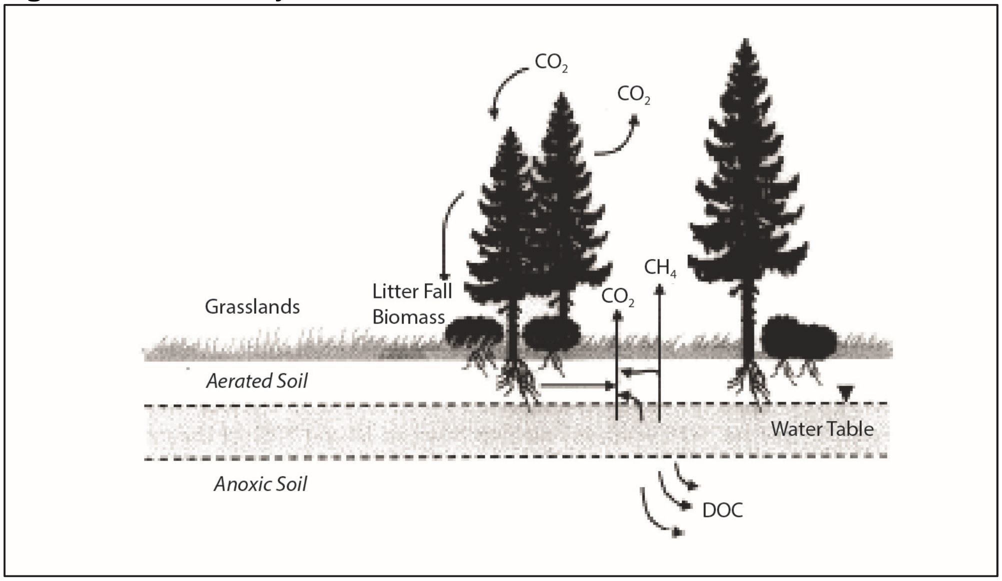
4.1.1.3 Risk of Reversals

Wetlands inherently accumulate carbon in the soils due to anaerobic conditions, and they are natural sources of CO_2 and CH_4 to the atmosphere. Management may alter conditions that affect both the pools and fluxes. For example, accumulated soil carbon can be returned to the atmosphere if the wetland is drained (Armentano and Menges, 1986). In contrast, silvicultural water management in wetlands can lead to higher biomass production, which may partially offset increased soil organic matter oxidation. Conversely, the soil carbon pool in converted wetlands is typically lower than the unmanaged soil, and restoring wetland conditions may increase carbon storage over time if inherent hydric soil conditions are maintained with consistent organic matter inputs.

Reversals of emission trends can occur if a manager reverts to a prior condition or an earlier practice. For example, an entity may decide to return a wetland that had been drained and cropped back to a forested wetland condition. Another common example would be if a restored forested wetland is reverted back to agriculture. These reversals do not negate the mitigation of CH_4 or N_2O emissions to the atmosphere that had occurred previously, to the extent that wetland restoration or change in management can reduce or change these emissions. Correspondingly, the starting point from the reversion will determine the effect on carbon sequestration and GHG flux. For example, in a restored forested wetland, reversion of the site to crop production would return carbon sequestered during the restoration period to the atmosphere over time.

There is a trade-off in CH_4 and N_2O emissions with management of the water table position. Wetlands with anaerobic soil conditions that are persistent near the surface for a longer period during the year will tend to have higher CH_4 emissions and lower emissions of N_2O. N_2O emissions are greatly reduced if soils are saturated because there is little inherent nitrification, and denitrification will lead to N_2 production (Davidson et al., 2000). For example, restoration of wetlands will normally lead to a higher water table for a longer period of the year, and thus contribute to higher emissions of CH_4 but lower emissions of N_2O. These trends can be reversed if the water table is lowered through management or drought, which will tend to enhance N_2O emissions if there is a source of nitrate, while reducing emissions of CH_4. Figure 4-2 provides an illustration of the carbon cycle typically found in wetland forest and grassland wetlands and represents the scope of the methods presented in this guidance.

Figure 4-2: Carbon Cycle for Forest and Grassland Wetlands



Source: Trettin and Jurgensen (2003).

4.1.2 System Boundaries and Temporal Scale

System boundaries are defined by the coverage, extent, and resolution of the estimation methods. The location of the wetlands may be approximated by use of the National Wetlands Inventory,[3] the location of hydric soils as conveyed by the NRCS soils map, or through direct delineation of wetlands. The coverage of the methods can be used to estimate a variety of emission sources, including emissions associated with biomass C, litter C, and soils carbon stock changes and CO_2, CH_4, and N_2O fluxes from soils. System boundaries are also defined by the extent and resolution of the estimation method. The methods provided for wetlands have a spatial extent that would include all wetlands in the entity's operation, with estimation occurring at the resolution of an individual wetland. Emissions are estimated on an annual basis for as many years as needed for GHG emissions reporting.

4.1.3 Summary of Selected Methods/Models and Sources of Data

The IPCC (2006) has developed a system of methodological tiers for estimating GHG emissions. Tier 1 represents the simplest methods using default equations and factors provided in the IPCC guidance. Tier 2 uses default methods but emission factors that are specific to different regions. Tier 3 utilizes a region-specific estimation method, such as a process-based model. Higher tier methods are expected to reduce uncertainties in the emission estimates if there is sufficient information and testing to develop these methods. In this guidance, biomass, litter, and soil carbon stock changes, in addition to soil N_2O and CH_4 emissions, are estimated using Tier 2 and 3 methods.

The data required to apply these methods range from basic information on soils, vegetation, weather, land use, and management history to data on fertilization rates or drainage conditions. While some of these data are operation-specific and must be provided by the entity, other data can be obtained from national databases, such as weather data and soil characteristics.

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[3] See National Wetlands Inventory http://www.fws.gov/wetlands/.

4.1.4 Organization of Chapter/Roadmap

The wetlands section of this report is organized into three primary sections. Section 4.2 provides a description of wetland management effects on GHG emissions, elaborating on the scientific basis for how various practices influence GHG emissions. Section 4.3 provides a rationale for the selected method, a description of the method, including a general description (with equations and factors), activity data requirements, ancillary data requirements, limitations of the method, and uncertainties associated with the estimation. A single method is provided for each source presented in this chapter (i.e., biomass carbon in forested, shrub, and grass wetlands; soil carbon and CH_4 in wetlands; and direct N_2O emissions in wetlands). A single method was selected to ensure consistency in emission estimation by all reporting entities, and the selected method is considered the best option among possibilities for entity-scale reporting. Methods may be refined in the future as they are further developed. The last section provides a summary of selected research gaps.

4.2 Management and Restoration of Wetlands

How wetlands are managed can have a significant effect on GHG emissions and sinks, which are primarily influenced by the degree of water saturation, climate, and nutrient availability. In a majority of wetlands, 90 percent of carbon in gross primary production is returned to the atmosphere through decay, and the remaining 10 percent accumulates in the bottom of the water body accumulating on previously deposited materials (Blain et al., 2006). Management of the water table within a wetland will result in both lower CH_4 emissions due to decreased production and oxidation of CH_4 produced in the subsoil and an increase in CO_2 emissions due to increased oxidation of soil organic matter. N_2O emissions from wetlands are typically low, unless an anthropogenic source of nitrogen enters the wetland. In drained wetlands, N_2O emissions are largely controlled by the fertility of the soil and water management regime. In contrast, restored and constructed wetlands generate higher levels of CH_4 and lower levels of CO_2 because of the change in a water table depth (Blain et al., 2006).

4.2.1 Description of Wetland Management Practices

This section provides a description of management practices in wetlands that influence GHG emissions (CH_4 or N_2O) or carbon stocks. Individual sections deal with forested and grass wetlands that could occur in agricultural and forestry operations. It is important to note that drainage of wetlands for commodity production, such as annual crops, or for other purposes are not considered wetlands in these guidelines. Methods for drained wetlands can be found in Chapter 3, Croplands and Grazing Lands, or Chapter 6, Forest Lands, depending on the land use after drainage of the wetland.

4.2.1.1 Silvicultural Water Table Management

Silvicultural water management systems are principally used to regulate the water table depth in order to reduce soil disturbance associated with harvesting operations and alleviate stress from saturated soil conditions on artificially regenerated plantations. The silvicultural water management system should not eliminate the wetland conditions of the site.

Silvicultural water management systems affect the carbon balance and GHG emissions from the site (Bridgham et al., 2006). Typically organic matter decomposition is enhanced with the imposition of a drainage system, CH_4 emissions are reduced, and N_2O emissions may increase (Li et al., 2004). Carbon sequestration in biomass may be enhanced on sites with silvicultural drainage systems due to increased tree productivity (Minkkinen and Laine, 1998).

4.2.1.2 Forest Harvesting Systems

There are two general types of systems used to harvest trees from forested wetlands: partial cutting and clear cutting. A partial cut involves the removal of selected trees from the stand. The number of trees removed or the residual density of the stand will depend on the stand type, species, intended product(s), and stand age. The amount of tree biomass removed during the partial cut may also vary; tops may be left onsite if only logs are removed, or they may be concentrated in a landing if whole-tree harvesting is used. With the latter system, the tops may also be utilized and removed from the site. Partial cutting is typically used in riparian zones and sites that are managed for solid wood products. Clear cutting results in the removal of all overstory trees from the site. Clear cutting is typically used on natural stands occurring in floodplains of the southeastern coastal plain and lacustrine and outwash plains of the upper Midwest. Clear cutting is also the typical system employed to harvest conifer and hardwood plantations.

Partial cutting affects the carbon balance of the site by direct removal of biomass; increased biomass on the forest floor, which is then subject to decay processes; and increased growth of the remaining trees for several years. Decomposition of dead biomass within the stand may be accelerated temporarily due to the changes in ambient conditions and the added residue from the harvest.

Clear cutting affects carbon stocks of the site by directly removing the biomass; increasing amounts of biomass added to the forest floor; altering the carbon sequestration for several years, depending on the type of regeneration; and altering the rate of organic matter decomposition in the forest floor and soil (Lockaby et al., 1999). Clear cutting affects the ambient conditions of the site because of the removal of the overstory vegetation. It also alters the water balance of the wetland due to the reduction in evapotranspiration following harvesting. Typically, as a result of lower evapotranspiration, the water table rises, and the site will exhibit longer periods of saturation. This change in the water table position has direct effects on the production of CH_4 and N_2O and subsequent fluxes to the atmosphere (Li et al., 2004).

4.2.1.3 Forest Regeneration Systems

There are two basic forest regeneration systems, characterized as (a) natural regeneration, and (b) artificial regeneration. Natural regeneration, as the name implies, relies upon regeneration of the trees from seed or sprouts that are left by harvested trees. Natural regeneration is used in both partial-cut and clear-cut harvest systems. Natural regeneration will lead to even-aged stands of shade-intolerant or early successional communities, typically in floodplains in the southeastern United States and the coniferous plains of the upper Midwest.

Artificial regeneration results from planting seedlings on a prepared site. The site preparation practices may involve removal of the harvest residue biomass, mechanical scarification and/or the application of herbicide to temporarily reduce weed competition with seedlings, and the creation of planting beds.

The effect of the forest regeneration system on carbon stocks and trace GHG emissions depends on the type of harvesting system that was used (Lockaby et al., 1999; Trettin et al., 1995). The combination of partial cutting and natural regeneration has little additive effect because the extent of regeneration is typically quite low following a partial cut that removes less than half of the basal area. Carbon stocks following clear-cut harvesting with natural regeneration is affected by the rate of growth of the regeneration, changes in ambient conditions, and changes in the soil water regime. Those factors also affect artificial regeneration systems; additionally, the type and extent of site preparation also affects the carbon stocks.

4.2.1.4 Fertilization

Fertilization is used primarily in forested wetlands, such as tree plantations, to enhance growth (Albaugh et al., 2004). Grass wetlands also receive fertilizer as a result of adjacent agricultural activities, and when dry conditions permit, are directly tilled, planted, and fertilized. Nitrogen is the most commonly applied fertilizer, and increased nitrogen inputs are known to increase emissions of N_2O (Bedard-Haughn et al., 2006; Davidson et al., 2000; Gleason et al., 2009; Merbach et al., 2002; Phillips and Beeri, 2008; Thornton and Valente, 1996). Nitrogen fertilizers will also enhance N_2O emissions both directly on the site and indirectly if nitrogen is lost from the site as nitrate in groundwater or runoff, as well as volatilization of nitrogen as ammonia or NO_x. The indirect losses will contribute to N_2O emissions at other sites.

The effect of fertilization on carbon stocks is principally realized through changes in tree growth rates. The effect would result from nitrogen fertilizers, but phosphorus may also be applied in the southeastern United States.

4.2.1.5 Conversion to Open-Water Wetland

The conversion of wetland to open water occurs primarily as a result of beaver impoundments and to a lesser degree improperly installed roads or other artificial embankments through a wetland that impedes natural drainage. The conversion to open water significantly reduces carbon sequestration through plant growth, because uptake is limited to submerged aquatic vegetation. The higher water table for a longer period of the year will also tend to increase CH_4 flux.

4.2.1.6 Forest Type Change

Changing a managed forest to a characteristic native condition is also considered a form of restoration. The effect of the restoration activities on the carbon stocks and CH_4 emissions depends on the extent of the hydrologic modifications that were employed in the previous silvicultural system. The two most common situations are a site that has been managed for a particular species or product without hydrologic modification; the other common situation is where the site has been managed for plantation forestry and the hydrology and vegetation have been extensively modified.

4.2.1.7 Water Quality Management

Riparian zones along streams, rivers, and lakes may be managed to protect water quality by mitigating nonpoint source pollution (Balestrini et al., 2011; Chaubey et al., 2010).[4] Pollutants are removed by physical filtration, chemical adsorption, plant uptake, and microbial transformations (Abu-Zreig et al., 2003; Borin et al., 2005).[5] However, riparian buffers are limited in their adsorption capacities for some constituents, which may then flow into waterways. The buffer zone size and configuration varies according to runoff patterns of the site, phosphorus/nitrogen inputs, hydrologic connectivity, organic carbon, mineral content, and oxidative/reductive state (Abu-Zreig et al., 2003; Hoffmann et al., 2009; Novak et al., 2002; Young and Briggs, 2008).

Riparian buffer zones are comprised of native and non-native vegetation or may also contain cultivated plants in some cases. Management activities of the native vegetation buffer zones are typically constrained or limited to small removals. In the case of forest riparian buffers, a selective-

[4] Additional references include (Cho et al., 2010; Flite et al., 2001; Hoffmann et al., 2009; Hunt et al., 2004; Lee et al., 2004; Lowrance et al., 2007; Montreuil et al., 2010; Peterjohn and Correll, 1984; Ranalli and Macalady, 2010; Schoonover et al., 2005; Tabacchi et al., 1998; Young and Briggs, 2008).
[5] Additional references include (Dillaha et al., 1989; Dillaha et al., 1988; Hoffmann et al., 2009; Jordan et al., 2003; Kelly et al., 2007; Novak et al., 2002; Vellidis et al., 2003; Young and Briggs, 2008).

harvest regime would be used that influences both carbon stocks and GHG emissions. In mixed buffers (i.e., grass strips followed by forest), the management of the cultivated buffer would largely determine the effect of the practice, which will be analogous to hay cultivation. Riparian zones may contain a mosaic of hydric (wetland) and non-hydric soils; accordingly, the distribution of soil types is important for assessing the effect of the management activity.

Whereas riparian buffers occupy low landscape positions and are typically wet, they are often very effective in removing nitrogen via denitrification (Ambus, 1991; Davis et al., 2008; Dodla et al., 2008; Hill et al., 2000; Hunt et al., 2007; Jordan et al., 1998; Roobroeck et al., 2010; Smith et al., 2006; Stone et al., 1998; Woodward et al., 2009), which leads to indirect N_2O emissions (Jetten, 2008). Denitrification in riparian buffers is often spatially uneven because riparian buffers vary considerably in their size and landscape positions as well as their soil, vegetative, and hydrological conditions (Bowden et al., 1992; Bruland and MacKenzie, 2010; Flite et al., 2001; Hill et al., 2000). Studies have suggested that N_2O emissions in riparian zones were not a significant "pollution-swapping phenomenon" (Dhondt et al., 2004; Kim et al., 2009a; Kim et al., 2009b). Significant emissions are likely to be limited to spatial and temporal hot spots (Groffman et al., 2000; Hunt et al., 2007; Kim et al., 2009b). Moreover, some riparian wetland systems can serve as sinks for nitrogen (Roobroeck et al., 2010). While many factors affect the microbial production of N_2O, one of the most dominating factors is the carbon to nitrogen ratio; larger ratios generally have low N_2O emissions because nitrogen is immobilized in the soil organic matter (Hunt et al., 2007; Klemedtsson et al., 2005). However, it is important to note that indirect N_2O emissions are attributed to the source of the nitrogen, which can be a neighboring field or livestock facility; so the methods to estimate indirect N_2O emissions are provided in other sections of this report (i.e., Chapter 3, Cropland and Grazing Lands, or Chapter 5, Animal Production).

Riparian buffers can serve as both sources and sinks of CH_4 (Hopfensperger et al., 2009; Soosaar et al., 2011). Their hydrology and biogeochemical characteristics exhibit significant influence on the net CH_4 emission. These characteristics include water table position, temperature, oxidative/reductive potential, and plant community compositions (Pennock et al., 2010; Whalen, 2005). Moreover, N_2O emissions from denitrification can be significantly influenced by methanotrophs (Costa et al., 2000; Knowles, 2005; Modin et al., 2007; Osaka et al., 2008).

Similar buffers exist for grass wetlands, either as part of a conservation program or as a naturally occurring area around a wetland where moist-soil conditions prevent tillage. Grass buffers reduce runoff and intercept sediments that would affect water quality by increasing turbidity and inputs of fertilizers and agrichemicals. Moreover, planting the entire catchment with grass can reduce CH_4 emissions by decreasing the artificially high water levels and extended hydroperiods that often are associated with cropland sites (Euliss Jr and Mushet, 1996; Gleason et al., 2009; van der Kamp et al., 2003).

4.2.1.8 Wetland Management for Waterfowl

Wetlands may be managed for waterfowl habitat. Activities that are specific to wetland waterfowl management have direct influences on carbon stocks and GHG emissions, including regulation of the water regime, specifically depth and duration of inundation, as well as planting and cultivation of crops for food and habitat. Water regimes imposed for waterfowl management may be different than the natural water table cycle of the site. Accordingly, changing the water table alters the periods of soil aeration and saturation influencing rates of CH_4 and N_2O, as well as carbon stock changes in timber stands and other wetland vegetation. Cultivating crops in wetlands managed for waterfowl will also influence carbon stocks and N_2O emissions based on selection of crops and/or rotation practice, tillage, liming, and nutrient management.

4.2.1.9 Constructed Wetlands for Wastewater Treatment, Sediment Capture, and Drainage Water Abatement

Constructed wetlands are engineered systems for wastewater treatment, capture of sediments, and drainage water abatement in agricultural and forestry operations (Chen et al., 2011; Elgood et al., 2010). Surface-flow and subsurface flow systems are the two principal types of constructed wetlands (Kadlec and Knight, 1996). The principal difference between these two types of constructed wetlands is the water flow path. In the case of the subsurface flow wetlands, all the water flows are beneath the soil surface; the surface-flow systems have flow both above and within the soil.

The subsurface wetlands typically consist of wetland plants growing in a bed of highly porous media such as gravel or wood chips that have a water table from one to two meters above the soil surface with a rectangular shape. There is lack of agreement about the relative impact of microbial and plant processes in the function of subsurface wetlands including GHG emissions. However, plants and microbes are typically interdependently involved in the processes that contribute to emissions (Faubert et al., 2010; Lu et al., 2010; Picek et al., 2007; Tanner and Headley, 2011; Wang et al., 2008; Zhu et al., 2007). While the microbial community drives the biogeochemical processes that specifically emit GHGs (Dodla et al., 2008; Faulwetter et al., 2009; Hunt et al., 2003; Tanner et al., 1997; Zhu et al., 2010), the plant community modifies the environmental conditions contributing to emission rates, including transporting oxygen into the depth of the wetlands, providing root surfaces for rhizosphere reactions, and venting gases to the atmosphere. The plant processes are significantly impacted by plant community composition and weather conditions (Stein et al., 2006; Stein and Hook, 2005; Taylor et al., 2010; Towler et al., 2004; Wang et al., 2008; Zhu et al., 2007).

Surface flow wetlands have a much more direct exchange of oxygen and GHGs with the atmosphere. They can be variable in shape and are generally less than 0.5 meters in depth. Surface wetlands minimize clogging problems, but they can have a significant loss of treatment as a result of channel flow. They are typically designed for either carbon or nitrogen removal (Stein et al., 2006; Stein et al., 2007; Stone et al., 2002; Stone et al., 2004), including the prevention of excessive ammonia emissions (Poach et al., 2004; Poach et al., 2002).

Constructed wetlands are typically created in upland settings (e.g., non-wetland); accordingly, the site assumes the same biogeochemical processes that are inherent to natural wetlands. Carbon stocks and GHG emissions are affected by the type and quantity of effluent being treated, the type of vegetation in the wetland cells, and management of the hydrologic regimes within the cells. The management of CH_4 and N_2O from constructed wetlands is somewhat similar to managing GHG emissions from wetland rice systems (Fey et al., 1999; Freeman et al., 1997; Johansson et al., 2003; Maltais-Landry et al., 2009; Mander et al., 2005a; Mander et al., 2005b; Picek et al., 2007; Tanner et al., 1997; Teiter and Mander, 2005; Wu et al., 2009). Of particular importance is the maintenance of wetland oxidative/reductive potentials that are sufficiently positive to avoid CH_4 production (Insam and Wett, 2008; Seo and DeLaune, 2010; Tanner et al., 1997). This requires higher levels of oxygen and lower levels of available carbon. The management of N_2O emissions is complicated by the fact that nitrates are often present in the wastewaters or drainage waters, and so GHG emissions can be reduced in the constructed wetlands if N_2 gas is emitted instead of N_2O. Complete denitrification to N_2 gas requires higher carbon/nitrogen ratios (Hunt et al., 2007; Hwang et al., 2006; Klemedtsson et al., 2005). Thus, there is an important balance between sufficient carbon for complete denitrification and copious carbon that drives wetlands into the low redox conditions associated with CH_4 production.

This section is included for completeness, but no method for constructed wetlands is provided in this section. Section 5.4.10 in Chapter 5, Animal Agriculture, provides a qualitative discussion of estimating emissions from liquid manure storage and treatment-constructed wetlands. However, Chapter 5 does not provide methods to estimate greenhouse gas emissions from constructed wetlands.

4.2.2 Land-Use Change to Wetlands

Conversion of land to wetlands may involve restoring agricultural land into a functioning wetland. However, wetlands can be restored from previously drained forest or grasslands, and the change tends to vary for different regions of the United States. Wetlands can also be constructed in any location for wastewater treatment. The original conversion of wetlands to another use typically involves an alteration of the natural wetland hydrology. Chapter 7, Land Use Change, addresses this type of conversion. Restoration of wetlands entails reestablishment of the requisite hydrology to support forest, scrub-shrub, sedge, or emergent wetland plant communities and occurs in floodplains, riparian zones, depressions, and slopes and valleys.

4.2.2.1 Actively Restoring Wetlands

The effect of restoring both forested and grass wetlands will lead to carbon sequestration and CH_4 emissions that would be characteristic for that wetland type. However, the extent to which carbon sequestration, organic matter turnover, and gas fluxes return to rates typical for the wetland type depends on many factors, particularly the degree of alteration, time since restoration, hydrology, and development of the vegetation. In general, restored sites will be carbon sinks due to sequestration in the developing biomass (e.g., forest stand) and soils (Euliss Jr et al., 2008). Soil carbon is expected to increase slowly in forested settings and somewhat more rapidly in grassland sites (Gleason et al., 2009); however, the extent and rates of change are uncertain. Reestablishment of the wetland hydrology will also alter the CH_4 flux from the restored site since hydrologic modifications for other land uses will typically involve drainage or diversions. Raising the water table and increasing the period of time that the soil surface is covered with water will increase CH_4 production. However, many restored grassland sites are not directly drained, and reestablishment of grasses in the catchment can shorten the hydroperiod (Van Der Kamp et al., 1999; Voldseth et al., 2007), thus reducing CH_4 production.

Conversion of scrub-shrub wetlands typically involves drainage to a non-wetland state, and the imposition of cultivation or other practices depending on the land use. Accordingly, the restoration of prior-converted scrub-shrub wetlands typically involves reestablishment of the natural wetland hydrology and selective planting to establish native vegetation. The development of the characteristic wetland hydrology is the principal factor affecting the carbon stocks and GHG emissions from the site following conversion, but the type of vegetation and time since establishment will also have some influence.

4.2.2.2 Created Wetlands

Created wetlands are engineered into non-wetland or upland sites. Typical examples include mitigation sites, anaerobic lagoons (See Section 5.4.10 in Chapter 5, Animal Agriculture) on livestock operations, and storm water detention basins. The principal activity affecting the carbon stocks and GHG emissions is the imposition of a hydrologic regime that induces hydric soil properties and supports hydrophytic plants, in addition to clearing of the previous vegetation that may lead to a change in biomass carbon stocks.

4.2.2.3 Passive Restoration of Wetlands

Allowing an area to regenerate through natural succession is also considered a form of restoration. The effect of the restoration activities on the carbon stocks and CH_4 emissions depends on whether there was hydrologic remediation and the degree of vegetation change over time.

4.3 Estimation Methods

Section 4.3.1 provides methods for estimating live and dead biomass in forested, shrub, and grassland wetlands. Section 4.3.2 provides methods for estimating soil C, N_2O, and CH_4 emissions from managed naturally occurring wetlands.

4.3.1 Biomass Carbon in Wetlands

Method for Estimating Live and Dead Biomass Carbon in Wetlands

- Methods for estimating forest vegetation and shrub and grassland vegetation biomass carbon stocks use a combination of the Forest Vegetation Simulator model and the biomass carbon stock changes method in Section 3.5.1 of Chapter 3, Cropland and Grazing Land. If there is a land-use change to agricultural use, methods for cropland herbaceous biomass are provided in Chapter 3.
- These methods were chosen because they offer the most consistent approach within the context of this report.

4.3.1.1 Rationale for Selected Method

Various approaches are used for estimating tree biomass carbon, but ultimately each relies on allometric relationships developed from a characteristic subset of trees. The Forest Vegetation Simulator (FVS) has been selected as the method to estimate tree biomass. FVS is model-based approach that is specific to U.S. conditions and a Tier 3 method as defined by the IPCC. The simulator is the most complete model in the United States to estimate tree biomass. Regional versions of FVS have been refined based on large databases developed from many years of data collection on forest stands throughout the United States, thereby providing improved estimates while requiring few input parameters from the user.

Both IPCC (2006) and EPA (2011) consider herbaceous biomass carbon stocks to be ephemeral, and recognize that there are no net emissions to the atmosphere following growth and senescence. However, with respect to changes in land use (e.g., forest to cropland), the IPCC (Lasco et al., 2006) recommends that grazing land biomass be counted in the year that land conversion occurs (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. The method is considered a Tier 2 method as defined by the IPCC because it incorporates factors that are based on U.S. specific data.

The methods presented in this section are based on the following definitions.

- *Live vegetation biomass:* Live vegetation includes trees, shrubs, and grasses. The tree carbon pool includes aboveground and belowground carbon mass of live trees, as defined in Section 6.2.3.1, and the aboveground biomass of the forest understory is defined in Section 6.2.3.2. The methods to estimate full-tree and aboveground biomass for trees greater than one inch in diameter at breast height are based on the models provided in the forest section.

The forest understory vegetation includes all biomass of undergrowth plants in a forest, including woody shrubs and trees less than one inch in diameter at breast height.

- *Standing dead wood (dead biomass)*: The carbon pool of standing deadwood in a forested wetland is defined and estimated according to the methods in Section 6.2.3.3 of Chapter 6, Forestry.
- *Down dead organic matter—litter layer (dead biomass)*: Down dead organic matter includes the litter layer composed of small pieces of dead wood, branches, leaves, and roots in various stages of decay. This layer is typically designated as the organic layer of the soil. This pool also includes logs in various stages of decay that lie on the soil surface (e.g., Section 6.2.3.4, down-dead wood, and Section 6.2.3.5, forest floor or litter).

4.3.1.2 Description of Method

Provisions for estimating aboveground biomass for wetland forests and above and belowground biomass and carbon are included for shrub and grass wetlands in this section. Since the vegetative cover on wetlands may vary from natural communities to agricultural crops, cross-references are made to ensure congruity with Section 3.5.1 of Chapter 3, Croplands, and Grazing Lands, and Section 6.2.3 of Chapter 6, Forestry.

Forest vegetation: Biomass carbon stocks are estimated for forests in wetlands using the methods described in Section 6.2.3 of Chapter 6, Forestry. The approach uses the FVS, which is a system of growth and yield models that estimate growth and yield for U.S. forests. FVS is an individual tree model and can estimate biomass carbon stock change for nearly any type of forest stand. The Fire and Fuels Extension to FVS can be used to generate reports of all live and dead biomass carbon pools in addition to harvested wood products. Regional variants are available for FVS that allow for region-specific focus on species and forest vegetation communities. The driver for productivity is the availability of site index curves,[6] and the regional variants include many wetland tree species. Regional variants of FVS may also provide provisions for refining the basis for estimating productivity by classifying the area of interest into ecological units, habitat type, or plant associations. However, if a species-specific curve is not available, then a default function is used to estimate carbon stock changes.

Grassland vegetation: The change in carbon stock for grass wetlands is generally small unless there are drought conditions or the area is actively managed. In cases where reporting is required, biomass carbon stock changes can be estimated following a land use change using the method in Section 3.5.1 of Chapter 3, Croplands and Grazing Lands. There are no methods currently available to estimate the shrub cover.

4.3.1.3 Activity Data

Forested wetlands: The data and requirements for estimating the changes in carbon stocks in wetland forests are the same as those described for upland forests in Section 6.2.3.

Grassland vegetation: The data and requirements for estimating the changes in carbon stocks in grassland vegetation are the same as those described for total biomass carbon stock changes presented in the Croplands/Grazing Lands Sections 3.5.1.

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[6] Site index is the measure of a forest's potential productivity. The height of the dominant or co-dominant trees at a specified age in a stand are calculated in an equation that uses the tree's height and age. Site index equations differ by tree species and region. Site index curves are constructed by using the tree heights at a base age and an equation is derived from the curves to estimate the site index when an individual tree's age is not the same as the base age (Hanson et al., 2002).

4.3.1.4 Model Output

Change in aboveground carbon pools associated with wetland forests are provided for live vegetation, standing dead biomass, and down dead biomass. Change in live biomass carbon is also provided for belowground biomass. The units of reporting are metric tonnes ha^{-1} CO_2-eq.

4.3.1.5 Limitations and Uncertainty

Estimates of the forest biomass carbon pools in wetlands are constrained by limited data on productivity response to management and are sensitive to the wide array of characteristic vegetative communities and soil types. Although FVS is the most inclusive model available, many results for wetlands will still be based on default model functions, because there is limited data on the growth of specific wetland species under particular management regimes. Accordingly, the results will provide a relative basis for tracking changes over time in biomass carbon. Table 4-3 summarizes additional limitations in the current approach.

Table 4-3: Key Limitations to Estimating Biomass Carbon Pools in Forest Wetland Vegetation

Consideration	Limitation
Ratio for belowground biomass	A ratio is used to estimate belowground biomass in upland and wetland forests based on aboveground biomass. While a common ratio will provide a basis for estimating relative change, it will likely over or underestimate actual stocks in many wetlands.
Response to management or climatic conditions	Wetland vegetation is known to respond to management practices, soil, and climatic conditions. Those relationships are not necessarily reflected in FVS because there is insufficient basis for generalized assessment purposes. For example, in response to dynamic water-level fluctuations during wet and dry cycles, wetlands often exhibit major intra and interannual shifts in vegetative structure, ranging from open water to emergent herbaceous vegetation. Correspondingly, the altered site conditions under the management regime and the genetic quality of the planted trees may exhibit responses that are not captured by the existing allometric relationships in FVS.

This shrub and grassland method is based on the assumptions found in Chapter 3, Cropland and Grazing Land. Essentially, the method assumes that half of the crop biomass at harvest or peak forage/shrub biomass provides an accurate estimate of the mean annual carbon stock. This assumption warrants further study, and the method may need to be refined in the future.

Major sources of uncertainty include belowground biomass, vegetation response to management, and hydrologic regime (e.g., seasonal hydroperiod). 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 and carbon fractions, as well as the uncertainties associated with estimating the biomass carbon stocks for the other land uses.

Measurement, sampling, and regression/modeling errors are all part of the estimation process in FVS. Some similar measure of the representativeness of selected forest inventory and analysis plots to the entities' forests is needed. Uncertainties about carbon conversion factors are also significant in some cases.

4.3.2 Soil C, N_2O, and CH_4 in Wetlands

Method for Estimating Soil C, N_2O and CH_4 in Wetlands

- The DNDC process-based biogeochemical model is the method used for estimating soil C, N_2O, and CH_4 emissions from wetlands.
- DNDC predicts soil carbon and nitrogen balance and generation and emission of soil-borne trace gases by 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 (Dai et al., 2011; Zhang et al., 2002), using plant growth estimated as described in Section 4.3.1.

4.3.2.1 Rationale of Method

The production and consumption of carbon in wetland-dominated landscapes are important for estimating the contribution of GHGs, including CO_2, CH_4, and N_2O emitted from those areas to the atmosphere. The generation and emission of GHGs from wetland-dominated landscapes are closely related to inherent biogeochemical processes that also regulate the carbon balance (Rose and Crumpton, 2006). However, those processes are highly influenced by the land use, vegetation, soil organisms, chemical and physical soil properties, geomorphology, and climate (Smemo and Yavitt, 2006).

Given this complexity, a process-based modeling approach is desirable because these approaches typically account for more of the variability than simpler emission factor methods (IPCC, 2006). However, few process-based models have been tested sufficiently to be used for operational reporting of GHG emissions. One of the more widely tested models for estimating GHG fluxes from wetlands is the DNDC model. DNDC 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 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 is designed to explicitly consider anaerobic biogeochemical processes, which are fundamental to addressing soil carbon dynamics and trace GHG dynamics in wetlands (Trettin et al., 2001). It integrates decomposition, nitrification–denitrification, photosynthesis, and hydro-thermal balance within the ecosystem. These components are mainly driven by environmental factors, including climate, soil, vegetation, and management practices.

DNDC 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).

4.3.2.2 Description of Method

The method consists of using the process-based model—DNDC—to estimate the changes in soil organic carbon (SOC) stocks, CH_4, and N_2O emissions, based on the standing biomass and plant growth that are provided by the vegetation method outlined above (Section 4.3.1), wetland characteristics, and the planned management activities. The model simulates SOC stocks, CH_4, and N_2O emissions at the beginning of the reporting period based on an assessment of initial conditions at the site; then the model simulates the reporting period based on the current/recent management activity and any changes in the wetland conditions. This information characterizes the physical and chemical soil properties that in turn interact with the climatic regime, management practices, and the vegetation response. The reported emissions for the land parcel must reflect the total for the entire land area. Accordingly, the per-unit area emission rates from DNDC are expanded based on the total wetland area for the land parcel to estimate total emissions.

Equation 4-1 is used to estimate SOC stock changes from a parcel of land in a wetland:

Equation 4-1: Change in Soil Organic Carbon Stocks for Wetlands

$$\Delta C_{Soil} = (SOC_t - SOC_{t-1}) \times A \times CO_2MW$$

Where:

ΔC_{Soil}	= 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})
A	= Area of parcel (ha)
CO_2MW	= Ratio of molecular weight of CO_2 to C = 44/12 (metric tons CO_2 (metric tons C)$^{-1}$)

Equation 4-2 is used to estimate CH_4 emissions from a parcel of land in a wetland:

Equation 4-2: Methane Emissions from Wetlands

$$CH_4 = ER \times A \times CH_4MW \times CH_4GWP$$

Where:

CH_4	= Total CH_4 emissions from the land parcel (metric tons CO_2-eq year^{-1})
ER	= Emission rate on a per unit wetland area (metric tons CH_4 ha^{-1} year^{-1})
A	= Area (ha)
CO_2MW	= Ratio of molecular weight of CH_4 to C = 16/12 (metric tons CH_4 (metric tons C)$^{-1}$)
CH_4GWP	= Global warming potential of CH_4

N_2O emissions are estimated for a land parcel in a wetland using Equation 4-3:

Equation 4-3: Nitrous Oxide Emissions from Wetlands

$$N_2O = ER \times A \times CO_2MW \times CH_4GWP$$

Where:

N_2O	= Total N_2O emissions from the land parcel (metric tons CO_2-eq year^{-1})
ER	= Emission rate on a per unit land area (metric tons N_2O ha^{-1} year^{-1})
A	= Area (ha)
CO_2MW	= Ratio of molecular weight of N_2O to N = 44/28 (metric tons N_2O (metric tons N_2O-N)$^{-1}$)
CH_4GWP	= Global warming potential of N_2O

To estimate the SOC stock changes, CH_4, and N_2O emissions, DNDC requires a considerable amount of information to characterize the plant production (Section 4.3.1), wetland characteristics, and management activities. The initial step in applying the method is to parameterize DNDC using the baseline soil conditions, along with the corresponding forest or grassland conditions. For example, if a forest plantation is to be harvested and regenerated during the reporting period, the initial conditions should reflect the pre-harvest conditions. Based on the initial conditions, the model simulates baseline fluxes and the SOC stock prior to the reporting period for the entity. Subsequently, the entity specifies the type of management activity(s) changes that occurred during the reporting period (if any occurred). Provisions are available to have multiple management activities on a single tract if there were mixed activities. Climatic factors, especially precipitation, can affect carbon turnover and wetland conditions. Consequently, weather data are a key input to DNDC, and will be provided from a climatological data set.

The simulation output at the end of each year is used to estimate change in SOC stocks and the total amount of CH_4 and N_2O emissions for the year. Annual changes in SOC can be estimated based on the difference between years, and the total change in emissions can be estimated by combining the changes in SOC pools with the annual CH_4 and N_2O flux.

4.3.2.3 Activity Data

Activity data for the application of DNDC are summarized in Table 4-4. Vegetation management information affects the amount of organic matter that is available for decomposition processes. Water management information conveys how the drainage system affects the soil water table dynamic as compared to an undrained condition. The soil tillage information is used to convey when the surface soil is disturbed or its elevation changed because of the associated effects on decomposition. The fertilization information is needed because the addition of nitrogen greatly affects decomposition and N_2O production. In addition, land use history influences the amount of soil organic carbon. If an entity is composed of different wetland types, it is recommended that separate estimates be prepared because the carbon turnover rate and GHG emissions can vary widely depending on hydric soil properties and the type of vegetation.

Table 4-4: Activity Data for Application of DNDC

Category	Management Practice	Data
Vegetation management	Grazing or management events should be included to capture the influence on carbon input to soils and subsequent effects on the soil carbon stocks.	▪ Harvesting: date, harvest, or cut fraction ▪ Understory thinning or chopping: date, chopped fraction ▪ Prescribed fire: date, proportion of forest floor, and understory consumed ▪ Tree planting: date, species, density
Water management regime	Water table response to the drainage system, daily data.	▪ Drainage system: date, controlled water table elevation
Soil management	Application of soil amendments or site preparation practices for tree planting.	▪ Type of site preparation
Fertilization practices	Applications of mineral or organic nitrogen fertilizers will be needed to simulate the effect on N_2O emissions.	▪ Fertilization frequency, date, application rate (N, P kg ha^{-1})
Land use history	Summary of land use practices over the past 5 years. For assessing if prior use affects parameterization. The time since a change in land management practice for assessing effects on decomposition.	▪ Fertilization regimes, drainage regimes, cropping, or forest management history

4.3.2.4 Ancillary Data

The DNDC model requires relatively detailed information about the site (Table 4-5). While default values are available for most parameters, some entity-specific data are needed to produce reasonable estimates. Most of the required soils input data are available from the national soils data base.[7] Similarly, climate data are available from the National Climate Data Center.[8]

Table 4-5: Input Information Needed for the Application of DNDC

Category	Data
Climate	Daily maximum and minimum temperature, daily rainfall; nitrogen deposition in rainfall, or use default value.
Vegetation	Standing biomass and biomass and detrital inputs provided in Section 4.3.1; belowground biomass estimated based on aboveground biomass.
Soil	Hydraulic parameters and physical and chemical components, including thickness; layers; hydraulic conductivity; porosity; field capacity; wilting point; carbon content; pH; organic matter fractions; content of stone, sand, silt, and clay; and bulk density for major soil layers.
Hydrology	Water table below surface as daily input or starting position and DNDC can estimate GHG emissions and sinks using empirical functions.

4.3.2.5 Model Output

Model output includes annual estimates of CH_4, N_2O emissions, and changes in soil organic carbon stocks. The units of reporting are metric tons CO_2-eq ha^{-1}.

4.3.2.6 Limitations and Uncertainty

The models to estimate carbon sequestration in vegetation are robust with respect to species and community composition. However, uncertainties may be higher than for uplands because of limited background information. The merit of the recommended approach is that it ensures consistency for estimating changes in the vegetative carbon pool among land types and uses by using common methods as described in Section 4.3.1. However, this approach complicates the application of DNDC for estimating changes in soil carbon pools and fluxes because it contains provisions for sequestering carbon in crops, grasslands, and forest vegetation. Accordingly, DNDC would have to undergo substantial revisions to accommodate the vegetative component as an input variable because the vegetation growth functions are integral with the consideration of hydrologic processes (especially evapotranspiration) and biogeochemical processes. The DNDC model could be used as a stand-alone tool for wetlands, but unfortunately, the production or carbon sequestration functions have not been validated for many of the wetland plant communities.

The availability of water table data is essential to modeling the carbon cycle in wetland soils. Since the lack of site-specific water table data for a sufficient period is likely a constraint for most entities, an approach incorporating a hydrologic module or look-up table is needed. Hydrologic models that provide information on water table dynamics are inherently complex, but they can be effective (Dai. et al., 2010). Accordingly, the development of characteristic water table conditions for a range of climatological and soil settings would be a viable approach that can also incorporate water management effects (e.g., Skaggs et al., 2011).

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[7] See National Cooperative Soil Survey Soil Characterization data http://soils.usda.gov/survey/nscd/.
[8] See NOAA National Climatic Data Center http://www.ncdc.noaa.gov/.

Tidal freshwater forested wetlands, which occur to a limited extent along the Atlantic, Gulf, and Pacific coasts, are a special case. The tidal influence on water table dynamics can make characterizing the water table regime of such sites more difficult. For DNDC to simulate the carbon dynamics would require detailed data on daily water table dynamics, and such detailed data are unavailable.

While the effects of the various management regimes on soil carbon pools and GHG fluxes have not been widely studied, this is more of a consideration with respect to uncertainties in the estimates as opposed to a limitation to its application. The DNDC framework is robust because it is a process-based model that has been validated in a wide variety of wetland types and soils. However, it has not been extensively tested on Histosols or peat soils, especially with respect to changes in soil carbon stocks. The model was validated successfully for estimating CH_4 from micotopographic positions in a peatland (Zhang et al., 2002), but additional work is needed to better address the wide array of managed Histosols that exist across the country.

Similarly, this method is not applicable to constructed wetlands, impoundments, or shallow reservoir systems that have extended periods of ponding; those sites would tend to have dynamics more similar to a lake or pond as opposed to a terrestrial ecosystem.

With respect to the forest model, accuracy of the estimates is dependent on applicability of the available site index curves. While the general curves are available for all species, they may not accurately represent the site or the entity's management regime. Provisions are included within FVS for customizing the tree site index curves, which could be important for an entity especially if genetically-improved planting stock and fertilization regimes are employed.

Detrital organic matter is the source for decomposition processes. The effect of vegetation on wetland carbon dynamics is promulgated through the amount of organic matter and the water regime (e.g., evapotranspiration). Accordingly, the accuracy of the vegetation productivity and turnover will affect the estimates of the soil carbon pools and GHG flux.

Water table position is the most critical factor affecting CH_4 and N_2O flux from the wetland soil (Trettin et al., 2006). Accordingly, considerations to improve that estimate as discussed in Section 4.3.2 will improve the estimates of GHG emissions from the soil. There are other uncertainties in the activity and ancillary data, as well as model structure that can create bias and imprecision in the resulting estimates. Wetlands typically exist in a mosaic with upland forests, grasslands, and cultivated lands. Accordingly, the accuracy of partitioning the entity into upland (agriculture, forest) and wetlands will affect the accuracy of the estimates.

4.4 Research Gaps for Wetland Management

Wetland management, and its influence on GHG emissions, is not as well studied as some of the other management practices in this report, such as tillage in croplands or forest harvesting practices in uplands. There is the potential for improving the estimation of GHG emissions associated with different management practices in the future if there are monitoring activities and studies to fill information gaps. A select number of information needs and research gaps are identified here.

- The 2013 Supplement to the 2006 Intergovernmental Panel on Climate Change (IPCC) Guidelines provide new guidance for estimating emissions from drained inland organic soils, rewetted organic soils, coastal wetlands, inland wetland mineral soils, and constructed wetlands for wastewater treatment (Blain et al., 2013). These newly developed guidelines will be compared to the technical methods provided in this report.

- Water table position is the principal factor affecting carbon dynamics in wetlands; unfortunately there is a lack of long-term data, which is needed to characterize the water table response to a management regime and to provide a basis for validating assessment tools. Establishment of a network of water table monitoring sites within selected USDA Forest Service experimental forests and ranges and USDA-Agricultural Research Service (ARS) experiment stations could provide the continuity in measurements and linkages with common management practices to represent the major soil and climatic condition in the United States.
- Improving modeling capabilities that integrate surrounding areas with the wetlands that receive surface and subsurface drainage waters will allow for modeling the flows of nutrients and organic matter into wetlands and subsequent losses to other wetlands beyond the entity's operation. This type of assessment framework is used in several established spatially-explicit hydrologic models; the need is to integrate the biogeochemistry. Linked models can be used at present; but development of a functionally-integrated system is needed to support broad-based applications.
- There is a need, generally, for improved information on biomass production and allocation in managed wetlands. These data could be obtained through a coordinated monitoring program employing USDA-Forest Service experimental forests and ranges, USDA-ARS experiment stations, and U.S. Department of the Interior wildlife refuges to monitor production of key species or vegetation types in association with common management prescriptions. There is also need for more detailed mechanistic research to provide information on energy, water, and GHG dynamics on selected managed sites; this information is critical for validating process-based models.
- Field-based studies are needed to develop more complete databases that provide ancillary data for GHG estimation, particularity CH_4 emissions for DNDC or similar process-based models, rather than relying on entity input, which will likely be challenging. A key attribute of this work should be the consideration of the inherent spatial and temporal variability within a site.
- Further quantification of the controlling and threshold parameters and associated uncertainty within DNDC or similar process-based models to estimate trace gas emissions is warranted. This work could also suggest a path towards development of an assessment tool that was not reliant on a wide array of parameters to effectively simulate the GHG dynamics of the site.
- A more robust and extensive database on GHG emissions from freshwater tidal (salinity < 0.5 ‰) palustrine wetlands is needed to more fully understand the drivers of emissions, in addition to providing a more complete dataset for parameterization and evaluation of process-based models.
- Studies on individual sites and meta-analyses of existing data are needed to fully evaluate the net GHG flux for CH_4, N_2O, and soil carbon. Most studies only consider one of the GHGs and may mask some of the differences in fluxes among the GHGs associated with a management activity.
- Constructed wetlands are discussed qualitatively in Section 5.4.10 of Chapter 5, Animal Production Systems for Liquid Manure Storage and Treatment in Constructed Wetlands. More research is needed in this area to accurately estimate emissions from constructed wetlands.

This list is not exhaustive but is intended to provide some direction for improving the estimation methods for GHG emission from wetlands.

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Chapter 5
Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems

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Acronyms, Chemical Formulae, and Units

AA	Amino acids
AD	Anaerobic digestion
ADF	Acid detergent fiber
AGP	Antibiotic growth promoters
ASABE	American Society of Agricultural and Biological Engineers
B_0	Maximum methane production capacities
bLS	backward Lagrangian stochastic
BNR	Biological nitrogen removal
BW	Body weight
CH_4	Methane
CNCPS	Cornell Net Carbohydrate and Protein System
CO_2-eq	Carbon dioxide equivalents
CP	Crude protein
CSTR	Continuous stirred tank reactor
DDGS	Dried distillers grains with solubles
DE	Digestible energy
DFM	Direct fed microbials
DGS	Distillers grains with solubles
DIP	Dietary crude protein
DMI	Dry matter intake
DRC	Dry-rolled corn
EF	Emission factor
g	Grams
Gg	Gigagrams
GEI	Gross energy intake
GHG	Greenhouse gas
HCW	Hot carcass weight
HMC	High-moisture corn
IFSM	Integrated Farm System Model
kcal	Kilocalorie
kg	Kilograms
lb(s)	Pound(s)
LCA	Life cycle analysis
LU	Livestock unit
m	Meters
MCF	Methane conversion factor
ME	Metabolizable energy
mg	Milligram
MGA	Melengestrol acetate
MJ	Millijoules
NE	Net energy
N_{ex}	Nitrogen excreted
N	Nitrogen
N_2O	Nitrous oxide
NDF	Neutral detergent fiber
NFC	Non-fiber carbohydrate
NH_3	Ammonia
NPN	Non-protein nitrogen

NSP	Non-starch polysaccharide
O_2	Oxygen
OM	Organic matter
ppb	parts per billion
ppm	parts per million
RDP	Ruminal degradable protein
RFI	Residual feed intake
RMSPE	Residual mean square prediction error
SF_6	Sulfur hexafluoride
SFC	Steam-flaked corn
TAN	Total ammoniacal nitrogen
TDN	Total digestible nutrients
TKN	Total Kjeldahl nitrogen
TMR	Total mixed ration
UASB	Upflow anaerobic sludge blanket
UP	Unprocessed
U.S. EPA	U.S. Environmental Protection Agency
VFA	Volatile fatty acids
VS	Volatile solids
WDGS	Wet distillers grains with solubles
Ym	Methane conversion factor, percent of gross energy in feed converted to methane

5 Quantifying Greenhouse Gas Sources and Sinks in Animal Production Systems

This chapter provides guidance for reporting greenhouse gas (GHG) emissions associated with entity-level fluxes from animal production systems. In particular, it focuses on methods for estimating emissions from beef cattle (cow-calf, stocker, and feedlot systems), dairy cattle, sheep, swine, and poultry (layers, broilers, and turkey). Information provided is based on available data at the time of writing. In many cases systems are oversimplified because of limited data availability. It is expected that more data will become available over time. This chapter provides insight into the current state of the science and serves as a starting point for future assessments.

- Section 5.1 summarizes animal management practices and the resulting GHG emissions.
- Section 5.2 presents an overview of each production system and a general discussion of common management systems and practices.
- Section 5.3 describes the methods for estimating GHG emissions from enteric fermentation and housing (enteric fermentation being a much more significant emissions source than housing).
- Section 5.4 describes methods for estimating GHGs from manure management systems.
- Section 5.5 identifies research gaps that exist for quantifying GHGs from animal production systems. The intent of identifying research gaps is to highlight where improvements in knowledge can best improve the usefulness of this document at farm-, regional-, and industry-scales.

Ammonia Emissions in Animal Production Systems

Ammonia (NH_3), although not a GHG, is emitted in large quantities from animal housing and manure management systems and is an indirect precursor to nitrous oxide (N_2O) emissions as well as an environmental concern. Inside barns and housing units, NH_3 is considered an indoor air quality concern because it can have a negative impact on animal health and production. Volatilized ammonia can react with other compounds in the air to form particulate matter with a diameter of 2.5 microns. This fine particulate matter can penetrate into the lungs, causing respiratory and cardiovascular problems, and contribute to the formation of haze.

Information about ammonia has been included in this chapter and proposed quantification methods are presented in Appendix 5-C.

5.1 Overview

This section summarizes the key practices in animal management and the resulting GHG emissions that are discussed in detail in this chapter. The agricultural practices discussed include those required to breed and house livestock, including the management of resultant livestock waste. Emissions considered here include those from enteric fermentation (resulting from livestock digestive processes), livestock waste in housing areas, and livestock waste managed in systems (such as stockpiles, lagoons, digesters, solid separation, and others). Options for management changes that may result in changes in GHG emissions are also discussed.

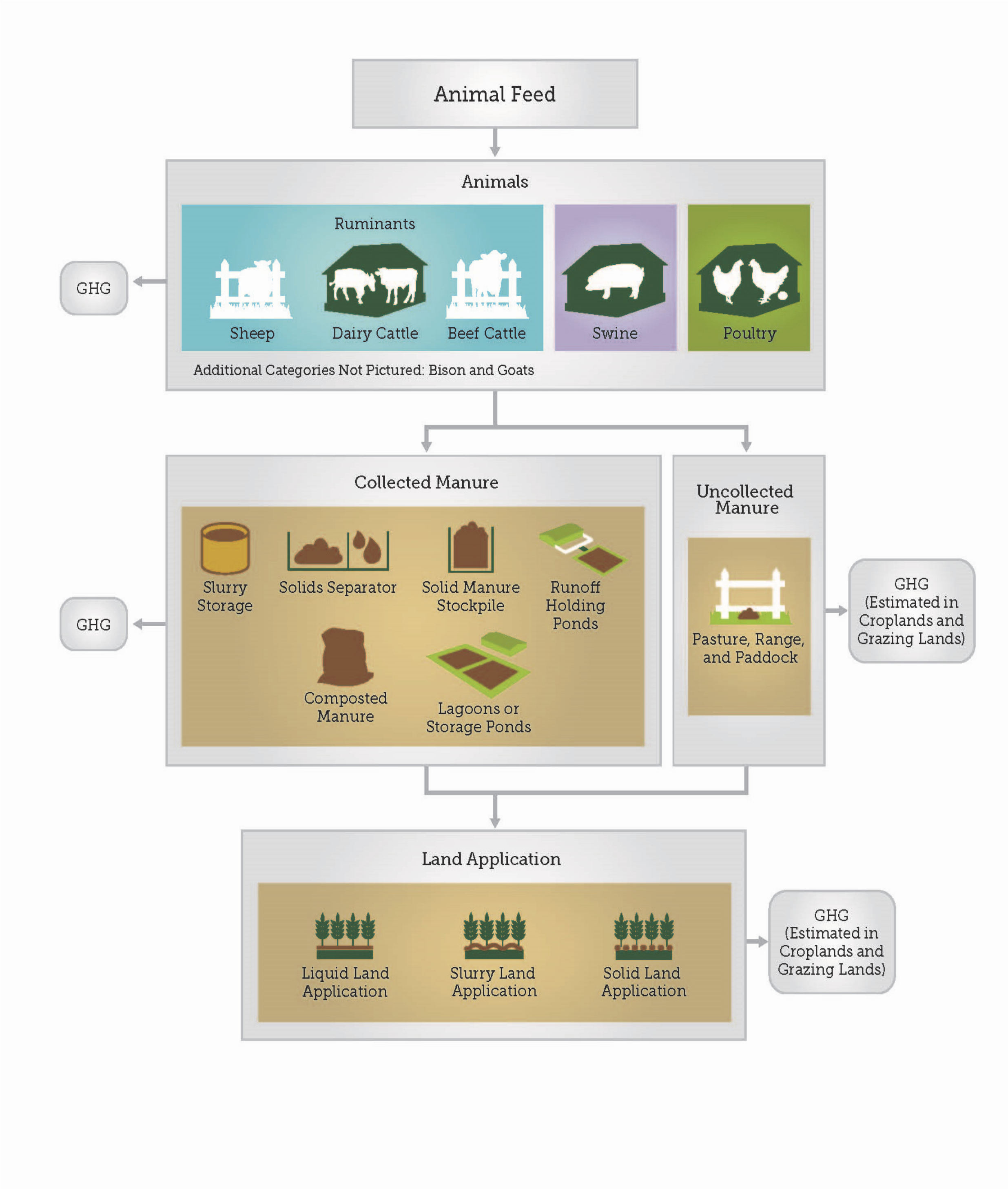
5.1.1 Overview of Management Practices and Resulting GHG Emissions

Animal production systems include agricultural practices that involve breeding and rearing livestock for meat, eggs, dairy, and other animal products such as leather, wool, fur, and industrial products like glue or oils. Farmers and other facility owners raise animals in either confined, semi-confinement, or unconfined spaces; the practices used to raise them are dependent on animal type, region, land availability, and individual preferences (e.g., conventional or "organic" standards). Regardless of the conditions in which animals are raised and housed, they produce GHG emissions. The magnitude of emissions depends primarily on the quality of the diet, the animals' requirements and intake (e.g., grazing, pregnant, lactating, performing work), and the types of systems in place to manage manure. The primary source of methane (CH_4) emissions from animal production systems is *enteric fermentation,* which is a result of bacterial fermentation during digestion of feed in ruminant animals. The second largest source of emissions from animal production systems is from the management of livestock manure. Methane emissions also occur from the digestive processes in monogastric animals; however, the quantity is significantly less than these other two sources. For simplicity, in the report, the term enteric fermentation refers to emissions from the digestive process of both ruminant and monogastric animals.

Manure management is the collection, storage, transfer, and treatment of animal urine and feces. Storage of animal manure has become increasingly popular as it allows synchronization of land application of manure nutrients with crop needs, reduces the need for purchased commercial fertilizer, and reduces potential for soil compaction due to poor timing of manure application. Depending on the storage and treatment practices, manure management has the added benefit of reducing air and water pollution. However, manure stored in anaerobic conditions results in the production and potential release of GHGs and odors. Greenhouse gas emissions from three solid manure storage/treatment practices (temporary stack and long-term stockpile, composting, and thermo-chemical conversion) and eight liquid manure storage/treatment practices (aerobic lagoon, anaerobic lagoon/runoff holding pond/storage tanks, anaerobic digestion, combined aerobic treatment system, sand-manure separation, nutrient removal, solid-liquid separation, and constructed wetland) are considered in the report.

Figure 5-1 provides an overview of the connections between feed, animals, manure, and GHG emissions in an animal production system. At the top of the conceptual model, livestock are fed a variety of diets. Ruminant animals eat feedstuffs and, through fermentation by the ruminal microbes, CH_4 is produced. Poultry and swine, although they do not release a significant amount of CH_4 through enteric fermentation, deposit manure into bedding, and upon manure decomposition, may release nitrous oxide (N_2O), CH_4 and ammonia (NH_3) into the atmosphere. Methodology to estimate emissions from bedding and dry manure in housing is similar to, and often parallel to, the method described for dry manure handling and storage systems. Manure from grazing livestock is left on fields or paddocks, and the manure may be collected to be treated and stored. Manure that has been collected and stored can be applied to croplands. GHG emissions from grazing lands and croplands are addressed in Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems.

Figure 5-1: Connections Between Feed, Animals, Manure, and GHG for Animal Agriculture



5.1.1.1 Resultant GHG Emissions

For this report, methods are categorized according to those from enteric fermentation, housing, and manure management systems. The housing discussion includes emissions from manure deposited in the housing unit and manure that is managed inside those areas (such as interior stockpiles). Manure management includes emissions from managed, treated, and stored manure.[1]

Enteric Fermentation and Housing Emissions

Methane-producing microorganisms, called methanogens, exist in the gastrointestinal tract of many animals. However, the volume of CH_4 emitted by ruminants is vastly different from that of other animals because of the presence and fermentative capacity of the rumen. In the rumen, CH_4 formation is a disposal mechanism by which excess hydrogen from the anaerobic fermentation of dietary carbohydrate can be released. Control of hydrogen ions through methanogenesis assists in maintenance of efficient microbial fermentation by reducing the partial pressure of hydrogen to levels that allow normal functioning of microbial energy transfer enzymes (Morgavi et al., 2010). The only GHG of concern resulting from enteric fermentation is CH_4. Respiration chambers equipped with N_2O analyzers indicate that enteric fermentation does not result in the production of N_2O (Reynolds et al., 2010). Methane can also arise from hindgut fermentation, but the levels associated with hindgut fermentation are much lower than those of foregut fermentation.

Background: Ruminants

Ruminants are animals that have four-chambered stomachs, which allow for easier digestion of high-fiber, hard-to-digest feedstuffs. They include:

- Cattle
- Goats
- Sheep
- Deer
- American Bison

Because the magnitude of enteric emissions is so great and, therefore, a significant contributor to many countries' GHG emissions, decades of research have gone into characterizing, understanding, and attempting to mitigate enteric CH_4 emissions. A fundamental challenge in this type of research has been the measurement of these emissions.

Methane, N_2O, carbon dioxide (CO_2), and NH_3 are produced from livestock feces and urine, and some gaseous forms are emitted soon after manure excretion. In dry-lot situations, feces and urine are deposited on the pen surface and are mixed via animal hoof action. Microorganisms in the feces or underlying soil metabolize nutrients in the manure to produce GHGs. In feedlots, where manure is normally cleaned from pens once or twice per year, distinctive, hard-packed layers of manure and soil may develop that produce microenvironments favorable to oxidative and reductive processes (Woodbury et al., 2001; Cole et al., 2009b). Periods of rainfall or dry conditions may alter the microbial and chemical nature of the pen surface. Production of CH_4 and N_2O occur in the underlying manure/soil layers and in water-saturated areas where oxygen is limited, such as wet areas of the pen around water troughs and depressions that collect rain water and snow melt. In contrast, most NH_3 produced in the pen probably comes from fresh urine spots on the pen surface. To date, few measurements of GHG emissions from feedlot or dry-lot pen surfaces have been made.

Runoff from dry-lot and feedlot pens is normally collected in retention ponds (more typical in feedlots), or lagoons (more common in dairies). In some cases, runoff may undergo partial removal of suspended solids in settling basins (feedlots and dairies) or in mechanical separators (dairies only) that parallels treatment of manure collected in these same systems. Losses of GHGs and NH_3

[1] Emissions from manure deposited on grazing lands are addressed in Chapter 3: Croplands and Grazing Lands.

from these facilities depend upon climatic factors and the oxidative-reductive potential, pH, and chemistry of the effluent in the pond or lagoon. A limited number of studies have measured GHG or NH_3 emissions from retention ponds or lagoons.

Manure Management

Manure is managed in a wide variety of systems. The resulting GHG emissions differ by GHG and magnitude of emissions per quantity of manure. Table 5-1 provides an overview of the liquid and solid manure systems considered in this report and the resulting GHGs.

Table 5-1: Overview of Manure Management Systems and Associated Greenhouse Gases

Storage and Treatment Practices		Estimation Method			Description
		CH_4	N_2O	NH_3 [a]	
Solid Manure	Temporary and long-term storage	✓	✓	✓	Manure may be stored temporarily for a few weeks to avoid land application during unfavorable weather or it can be stored for several months.
	Composting	✓	✓	✓	Composting involves the controlled aerobic decomposition of organic material and can occur in different forms. Estimation methods are provided for in vessel, static pile, intensive windrow, and passive windrow composting.
	Thermo-chemical conversion				Thermo-chemical conversion involves the combustion of animal waste, converting CH_4 to CO_2. Pyrolysis/gasification is one method that has received much interest. No method is provided as GHGs are considered negligible.
Liquid Manure	Aerobic lagoon	✓	✓	✓	Aerobic lagoons involve the biological oxidation of manure as a liquid with natural or forced aeration.
	Anaerobic lagoon/runoff holding ponds/storage tanks	✓	✓	✓	Anaerobic lagoons are earthen basins that provide an environment for anaerobic digestion and storage of animal waste. Lagoons may be covered or uncovered and have a crust or no crust formation. Runoff and holding ponds are constructed to capture and store runoff from feedlots and dry-lots. In some cases wash water from dairy parlors may be stored in holding ponds. Storage tanks typically store slurry or wastewater that was scraped or pumped from housing systems.
	Combined aerobic treatment system	✓	✓	✓	This process involves removing solids using flocculation and then composting the solid stream and aerating the liquid stream of manure.
	Anaerobic digester	✓			Anaerobic digesters are manure treatment systems designed to maximize conversion of organic wastes into biogas. These can range from covered anaerobic lagoons to highly engineered systems. Methane gas leakage is the main source of GHG emissions; NH_3 and N_2O leakage is negligible.
	Sand–manure separation				Manure is separated from sand and bedding by mechanical and sedimentation separation. No method is provided as emissions are negligible. Separated liquids and solids could be inputs into other storage systems.
	Nutrient removal				There are four main nitrogen removal approaches: biological nitrogen removal, Anammox (i.e., anaerobic ammonium oxidation), NH_3 stripping, ion exchange, and struvite crystallization. No method is provided due to limited quantitative information on GHG generation from nutrient removal systems.

Storage and Treatment Practices		Estimation Method CH_4	N_2O	NH_3 [a]	Description
	Solid–liquid separation				Mechanical separation of liquids and solids through screens, centrifuges, pressing, filtration, or microscreening. Separated liquids and solids could be inputs into other storage systems.
	Constructed wetland				Typically consist of wetland plants growing in a bed of highly porous media. No method is provided as emissions are negligible; GHG sinks are noted to likely be greater than emissions.

[a] Although NH_3 is considered in this chapter as an important precursor to particulate formulation (affecting radiation balance) and GHGs and is a key element of discussion, NH_3 itself is not a GHG. Therefore, methods for estimating NH_3 emissions are provided in Appendix 5-C.

An entity can reduce its GHG emissions from manure by utilizing alternative treatment options and/or management systems. Anaerobic digesters do not reduce the amount of CH_4 released but offer an option to capture and convert the CH_4 to CO_2 and energy through combustion. Digesters offer both CH_4 reductions as well as GHG avoidance by reducing an entity's electricity demand.

Combined Aerobic Treatment Compared to Anaerobic Lagoons

A combined aerobic treatment system involves the treatment of a manure stream with flocculants to remove the majority of solids from the stream. The solids portion is composted while the remaining liquid is transferred to a storage tank where it is aerated. Methane is avoided by aerobically treating the solids via composting while NH_3 in the wastewater is avoided via nitrification. The GHGs resulting from a combined aerobic treatment are only 10 percent of what would be emitted from an anaerobic lagoon, thus combined aerobic treatments represent a potential mitigation option for entities.

5.1.1.2 Management Interactions

Table 5-2 depicts the key types of information desired for estimating GHG emissions from an animal production facility. This table illustrates the attributes of a system that have the greatest influence over emissions within each component. A number of existing models can be used to estimate GHG emissions that utilize the key activity data indicated in Table 5-2.

Table 5-2: Desired Activity and Ancillary Data for Estimating GHG Emissions from Animal Production Systems

General Category	Specific Data	Cattle Cow–calf	Stockers	Feedlot	Dairy	Sheep	Swine	Poultry	Goats	Amer. Bison
Animal Characteristics	Body weight	•	•	•	•	•	•	•	•	•
	Body condition score	•	•		•	•				
	Stage of production (dry, lactating, pregnant)	•			•	•				
Dietary Facto	Diet intake (or factors that can be used to predict intake)	•	•	•	•	•	•	•	•	•

General Category	Specific Data	Cattle				Sheep	Swine	Poultry	Goats	Amer. Bison
		Cow-calf	Stockers	Feedlot	Dairy					
	Type of forage (conserved or grazed, pasture composition, stage of plant growth)	•	•		•	•			•	•
	Diet dry matter intake, crude protein, neutral detergent fiber, acid detergent fiber, non-structural carbohydrates, fiber, fat, energy content	•	•	•	•	•	•	•	•	•
	Diet digestibility and/or rate of passage	•	•	•	•	•	•			
	Degradability of carbohydrates and proteins	•		•	•					
	Supplementation practices – type (e.g., grains, protein, liquid, dry blocks, non-protein nitrogen) and quantity	•	•			•			•	•
	Supplemental or diet ionophore concentration	•	•	•	•					
	Dietary beta-agonists			•			•			
Nutrient Excretion: Quantity	Carbon, nitrogen, and volatile solids	•	•	•	•	•	•	•	•	•
Other Animal Factors	Growth promoting implants		•	•						
Manure Management Factors	Animal management regimen used to spread manure over pasture to reduce concentration near water or feed sources	•	•		•	•			•	•
	Soil type	•	•	•	•	•	•	•	•	•
	Practices to control runoff from pastures/lots/fields	•	•	•	•	•	•	•	•	•
	If housed, the length of time they are housed, animal concentration, manure handling procedures	•	•	•	•	•	•	•	•	•
	Type of manure collection/storage system			•	•		•	•	•	
	Frequency of manure collections and composition			•	•		•	•	•	
	Bedding/litter use and source			•	•		•	•	•	

5.1.2 System Boundaries and Temporal Scale

System boundaries are defined by the coverage, extent, and resolution of the estimation methods. The methods in this report can be used to estimate GHG emission sources that occur within the production area of an animal production system, including the animals, animal housing, and manure handling, treatment, and storage. Methane emissions from enteric fermentation, as well as the CH_4 and N_2O emissions from manure management systems or manure stored in housing, are considered in this report. Ammonia, while not a GHG, is a precursor to N_2O formation and is, therefore, included, primarily in Appendix 5-C. The act of transporting manure to the field for land application is included in the production area boundary, but emissions from vehicle transport are not included in the scope of this report as there are many variables that would determine emissions from vehicles (age of vehicle, type, fuel efficiency, idle time), and they are not direct agricultural emissions and could instead be considered part of the transport sector (off-road). Additionally, this report does not encompass a full life cycle analysis (LCA) of GHG emissions from animal production systems. The adjacent text box summarizes several studies on LCAs for animal production systems; however, they are not utilized in this report. Emissions that result following manure application are addressed separately in Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems.

Qualitative Discussion on Manure Sources

Estimation methods are not available for some sources. Qualitative discussion is provided for:

- Sand-Manure Separation
- Nutrient Removal
- Solid-Liquid Separation
- Constructed Wetlands
- Thermo-chemical Conversion

For emissions from animal production systems, the methods provided have a resolution of individual herds within an entity's operation. A herd is defined as a group of animals that are the same species, graze on the same parcel of land (same diet composition), and utilize the same manure management systems. Emissions are estimated for each individual herd within an operation and then added together to estimate the total animal production emissions for an entity. The animal production totals are then combined with emissions from croplands, grazing lands, and forestry to determine the overall emissions from the operation based on the methods provided in this document. Emissions are estimated on an annual basis.

5.1.3 Summary of Selected Methods/Models/Sources of Data

The Intergovernmental Panel on Climate Change (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.

Estimating CH_4 emissions from enteric fermentation in swine, goats, American bison, llamas, alpacas, and managed wildlife use Tier 1 methods. Enteric emissions from sheep are estimated using the Howden equation (Howden et al., 1994), and emissions from dairy production systems are estimated using the Mitscherlich 3 (Mits3) equation (Mills et al., 2003) as provided in the Dairy Gas Emissions Model (DairyGEM) (Rotz et al., 2011a). Emissions from beef cows are estimated using the IPCC Tier 2 approach. Emissions from feedlots are estimated using a modification of the IPCC Tier 2 approach.

Life Cycle Analysis of Cattle Production Systems

Peters et al. (2010) reported that the estimated carbon footprint of cattle production systems around the world ranged from 8.4 kg of CO_2-eq (kg HCW)$^{-1}$ (HCW=hot carcass weight) in an African pastoral system to 25.5 kg CO_2-eq (kg HCW)$^{-1}$ in an intensive Japanese grain feeding system. Five North American studies (Verge et al. (2008) and Beauchemin et al. (Sweeten, 2004; 2010) in Canada, Pelletier et al. (2010) and Lupo et al. (2013) in the U.S. Midwest, and Stackhouse et al. (2012) and Stackhouse-Lawson et al. (2012) in California) estimated the carbon footprint of various beef cattle production systems: The carbon footprint for the total beef production systems ranged from 10.4 to 19.2 kg CO_2-eq (kg final body weight) $^{-1}$ (or 16.7 to 32.5 kg CO_2-eq (kg HCW)$^{-1}$). Sixty four to 80 percent of the total CO_2-eq was produced in the cow-calf sector of production; whereas 8 to 20 percent of CO_2-eq was produced in the stocker phase, and only 12 to 16 percent was produced during the finishing phase. The majority (55 to 63 percent) of the total CO_2-eq was enteric CH_4, 18 to 23 percent was manure N_2O, and 14 to 24 percent was from fossil energy use and secondary emissions.

In general, the daily carbon footprint was greater during the grazing (stocker) phase than during the feedlot finishing phase. Both Pelletier et al. (2010) and Stackhouse et al. (2012) reported that the carbon footprint was slightly lower for calves that were weaned and went directly to the feedlot (21.1 and 23.0 kg CO_2-eq (kg HCW)$^{-1}$ or 2,382 and 3,493 kg head^{-1}, respectively) than for cattle that went through a stocker grazing phase before entering the feedlot (22.6 and 26.1 kg CO_2-eq (kg HCW)$^{-1}$ or 2,904 and 4,522 kg CO_2-eq head^{-1}, respectively). Pelletier et al. (2010) and Lupo et al. (2013) both reported that the carbon footprint of grass-finished cattle was greater than for calves that were weaned and went directly to the feedlot. These differences are due in part to slower weight gain and lighter final body weights and carcass weights of grass-fed cattle than cattle finished on grain- and byproduct-based diets in the feedlot.

Most LCAs assume that carbon sequestration is minimal in established, unfertilized pastures. Phetteplace et al. (2001) and Liebig et al. (2010) suggested there may be some small net carbon sequestration, in established native pastures. However, Liebig et al. (2010) noted that fertilized, improved pastures had net CO_2-eq emissions; primarily because of increased losses of N_2O from fertilizer nitrogen. Lupo et al. (2013) noted that the assumed carbon sequestration of pastures (equilibrium vs. net sequestration) affected the carbon footprint of grass-finished cattle; however, regardless of the carbon sequestration assumption, grass-finished cattle had a greater carbon footprint than grain-finished cattle.

For manure management, the IPCC Tier 2 methodology is used for CH_4 emissions from temporary stack and long-term stockpile, CH_4 and N_2O emissions from composting, and N_2O emissions from aerobic lagoons. The Sommer model is used to estimate CH_4 emissions from anaerobic lagoons.

All methods include a range of data sources from 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., dietary information, volatile solids content of manure). National datasets are recommended for ancillary data requirements, such as climate data and soil characteristics.

A summary of proposed methods and models for estimating GHG emissions from animal production systems is provided in Table 5-3.

Table 5-3: Summary of Sources and Proposed GHG Estimation Methods for Animal Production Systems

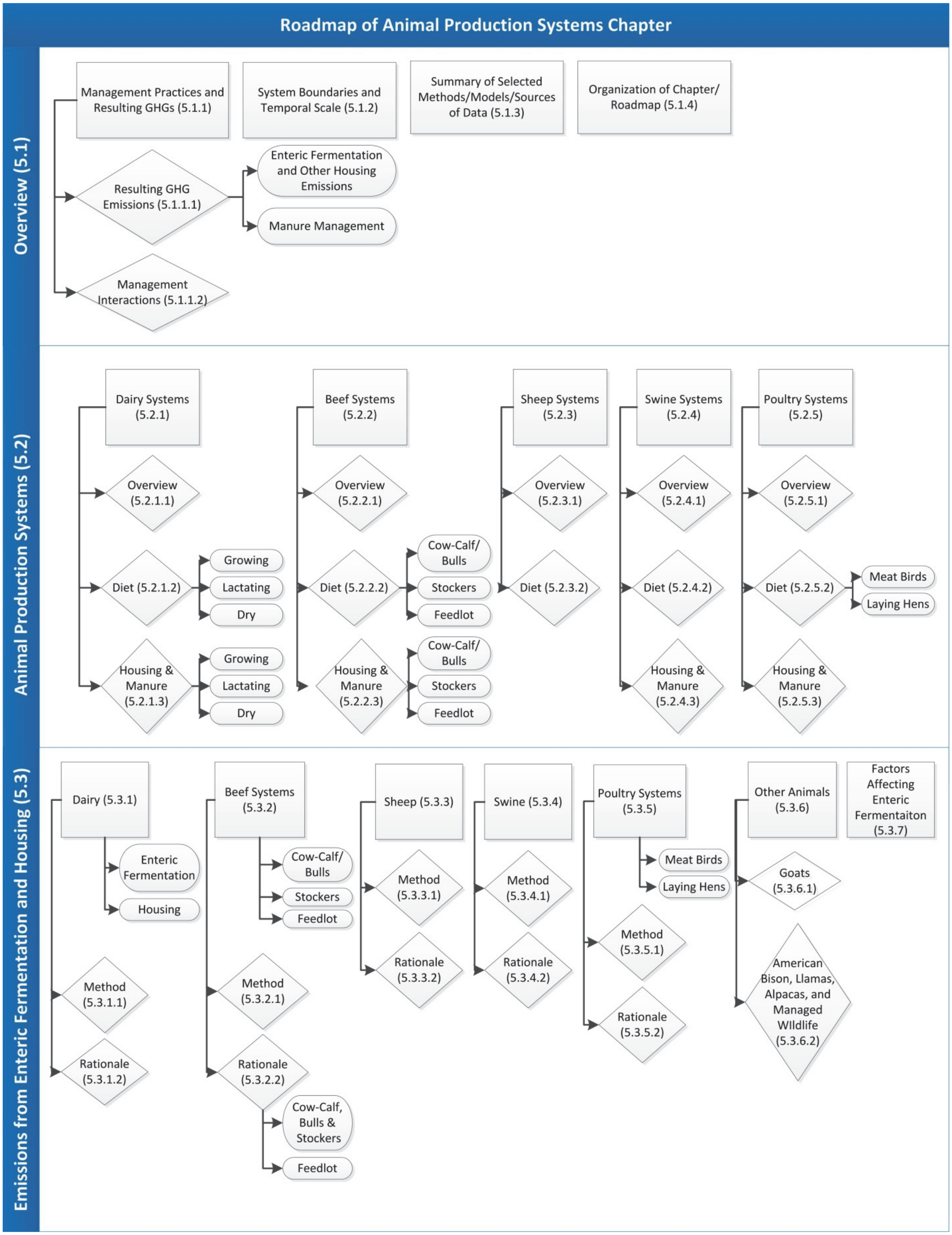
Section	Source	Method
Animal Production Systems, Including Enteric Fermentation and Housing Emissions		
5.3.1.2	Dairy Cattle	Mits3 equation; ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.2.2	Beef Cattle	Modified IPCC Tier 2 (enteric and housing); ASABE Standard D384.2 (housing)
5.3.3.2	Sheep	Howden equation for grazing sheep (Howden et al., 1994) and Blaxter and Clapperton (1965) for feedlot sheep
5.3.4.2	Swine	IPCC Tier 1 (enteric methane); ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.5.2	Poultry	IPCC Tier 1; ASABE Standard D384.2 and IPCC Tier 2 (housing)
5.3.6.1	Goats	IPCC Tier 1
5.3.6.2	American Bison, Llamas, Alpacas, and Managed Wildlife	IPCC Tier 1
Manure Storage and Treatment		
Temporary Stack & Long-Term Stockpile		
5.4.1.2	Methane	IPCC Tier 2 using U.S. EPA Inventory emission factors (EFs) and diet characterization
5.4.1.4	Nitrous Oxide	IPCC Tier 2 using U.S.-based EFs and monthly data
Composting		
5.4.2.2	Methane	IPCC Tier 2 with monthly data
5.4.2.4	Nitrous Oxide	IPCC Tier 2
Aerobic Lagoon		
5.4.3.2	Methane	Methane Conversion Factor for aerobic treatment is negligible and was designated as 0% in accordance with IPCC
5.4.3.4	Nitrous Oxide	IPCC Tier 2 using IPCC EFs
Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks		
5.4.4.2	Methane	Sommer model based on fractions of volatile solids (Møller et al., 2004)
5.4.4.4	Nitrous Oxide	Function of the exposed surface area and U.S.-based emission factors
Anaerobic Digestion		
5.4.5.2	Methane	IPCC Tier 2 using Clean Development Mechanism EFs for digester types to estimate CH_4 leakage from digesters
Combined Aerobic Treatment Systems		
5.4.6.2 5.4.6.2	Methane Nitrous Oxide	10% of emissions from estimation of liquid manure storage and treatment – anaerobic lagoon, runoff holding pond, storage tanks
Other Treatment Methods		
5.4.7	Sand–Manure Separation	No method provided because GHG emissions are negligible
5.4.8	Nutrient Removal	Not estimated due to limited quantitative information
5.4.9	Solid Liquid Separation	No method provided because GHG emissions are negligible
5.4.10	Constructed Wetland	No method provided because emissions are negligible; GHG sinks are noted to likely be greater than emissions
5.4.11	Thermo-chemical Conversion	No method provided as GHG emissions are negligible

5.1.4 Organization of Chapter/Roadmap

The remainder of this chapter is organized into four primary sections, as illustrated in Figure 5-2. Section 5.2 provides overviews of dairy cattle, beef cattle, sheep, swine, and poultry production systems and provides information on diet and housing. Section 5.3 provides the methods for estimating GHGs from housing, primarily focusing on GHGs from enteric fermentation. Methods are also provided for all the species described in Section 5.2, plus additional animal types (i.e., goats, American bison, llamas, alpacas, and managed wildlife). Section 5.4 provides the methodology for estimating emissions from different manure management systems. Methodology is provided to estimate CH_4 and N_2O from temporary stack and long-term stockpiles, composting, aerobic lagoons, anaerobic lagoons, and combined aerobic treatment systems. Section 5.4 also provides methods for estimating CH_4 from anaerobic digestion. A qualitative discussion is provided for sand-manure separation, nutrient removal, solid-liquid separation, constructed wetlands, or thermo-chemical conversion. Section 5.5 presents research gaps for both enteric fermentation and manure management.

There are five appendices to the animal production systems chapter of this report. Appendix 5-A provides Ym adjustment factors for calculating enteric CH_4 from feedlot cattle. Appendix 5-B provides nutritional information about animal feedstuffs (Ewan, 1989; Preston, 2013). Appendix 5-C discusses available methodologies for estimating NH_3 emissions from animal production systems. Appendix 5-D describes the shape factors and related equations that can be applied in Appendix 5-C to more accurately estimate emissions from manure stockpiles that are shaped differently (as surface area partially determines emissions). Appendix 5-E provides a detailed review of models evaluated for suitability for estimating emissions from animal production systems.

Figure 5-2: Animal Production Systems Road Map



Appendix 5-A: Enteric CH_4 from Feedlot Cattle – Methane Conversion Factor (Ym)

Appendix 5-B: Feed Stuffs Composition Table

Appendix 5-C: Estimation Methods for Ammonia Emissions from Manure Management Systems

Appendix 5-D: Manure Management Systems Shape Factors

Appendix 5-E: Review of Enteric Fermentation Models

5.2 Animal Production Systems

This section provides discussion on the production systems for beef and dairy cattle, sheep, swine, and poultry. This provides the background necessary for understanding Section 5.3, which covers GHG emissions from animal production systems.

5.2.1 Dairy Production Systems

5.2.1.1 Overview of Dairy Production Systems

The U.S. dairy production system is comprised of several key processes for dairy cattle, their manure, and their end products (meat, dairy) as depicted in Figure 5-3. This conceptual model provides an overview of the typical dairy system, following cattle from birth to slaughter and following manure from the animal through a management system. Manure is produced during each stage, and depending on the location, is managed differently. The management of the resultant manure has implications on the quantity of GHG emissions and sinks; the key practices are discussed in detail below. The estimation methods in this chapter include discussions for emissions from enteric fermentation, housing, and manure management and are not a full LCA.

The U.S. dairy industry is composed primarily of four major segments of production: 1) calf rearing; 2) replacement heifers; 3) lactating cows; and 4) nonlactating (dry) cows. The U.S. dairy cattle population in 2012 consisted of approximately 9.2 million milk cows and first calf heifers and approximately 4.6 million replacement heifers. The majority of dairy cattle in the United States are Holstein (Holstein-Friesian), followed by Jersey, with smaller numbers of Guernsey, Brown Swiss, and Ayrshire. Over the last 65 years there have been dramatic increases in milk production per animal, due to changes in herd management, nutrition, composition, and breeding programs. Present-day dairy herds are dominated by Holstein cows (90 percent) as opposed to a mix of the five most common breeds (Jersey, Guernsey, Ayrshire, Brown Swiss, and Holstein) as was common in the 1940s. With a change in breed dominance and enhanced genetics, the typical milk production per cow has increased from 2,074 to 9,193 kg of milk per year (Capper et al., 2009).

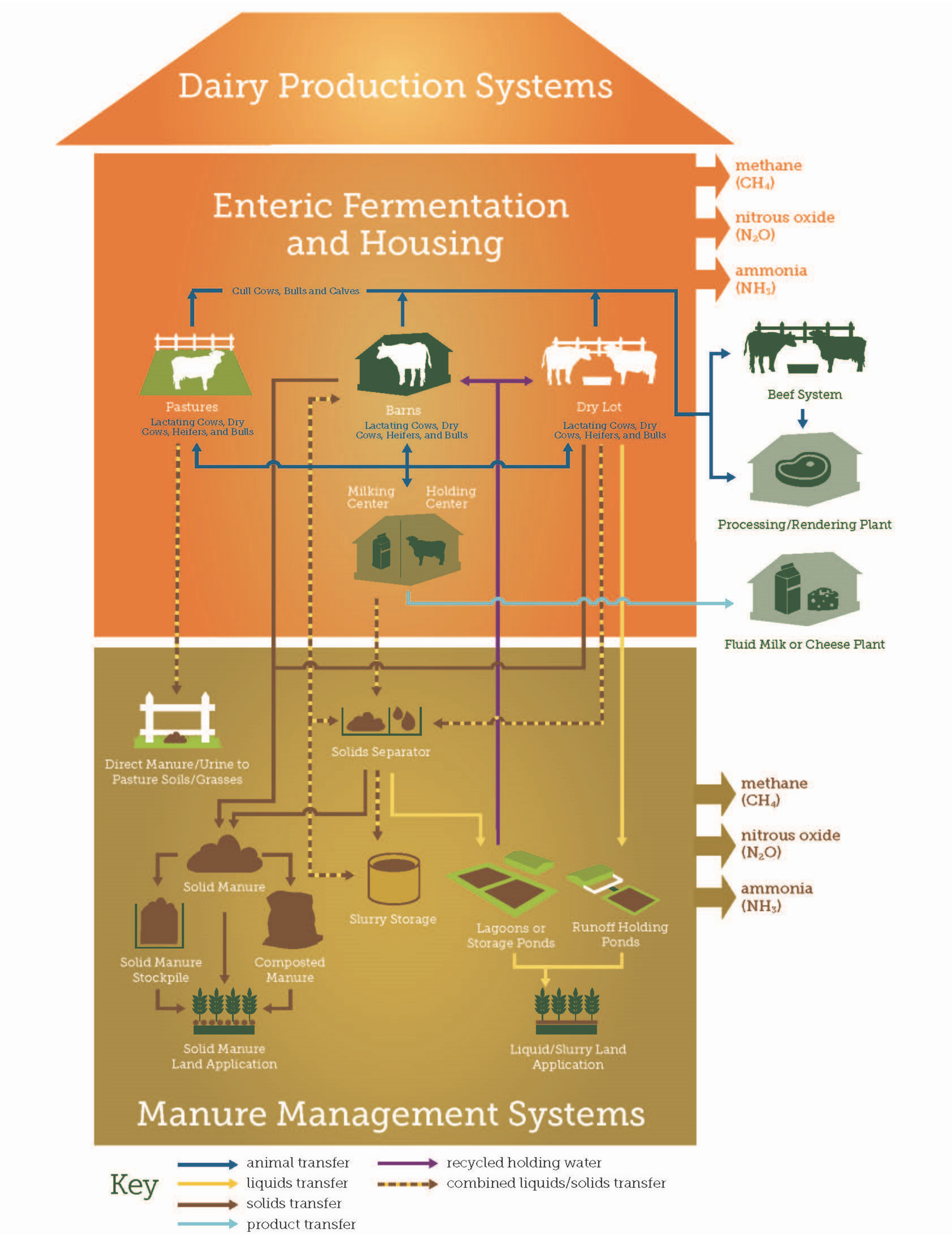
5.2.1.2 Diets for Dairy Cattle

Cows in intensive dairy production systems are fed diets that reflect regionally available feeds and typically contain between 40 and 60 percent concentrates, such as feed grains, protein supplements, and byproducts such as distiller's grains. Typical diets include corn silage, alfalfa or grass silage, alfalfa hay, ground or high-moisture shelled corn, soybean meal, fuzzy whole cottonseed, and often byproduct feeds (e.g., corn gluten, distiller's grains, soybean hulls, citrus pulp, beet pulp). Byproduct feeds may make up a large portion of the diet composition, providing key nutrients and a means of disposal for otherwise landfilled ingredients. Proximity to crop processing plants and industries may dictate the availability of byproduct feeds by region.

Growing Heifers

Diets for growing heifers are formulated based on growth rate and stage of rumen development. Diets range from liquid diets (e.g., milk or milk replacer) in newborn calves to pelleted complete feeds in the growing calf (e.g., calf starter) to diets that are similar to that offered to lactating cows as the cows grow and rumens develop. Roughage content of the diet increases as the rumen develops with hay or silage often offered in conjunction with a calf starter during a transition period. Following that transition, typical feeds include those listed above. Feeds are often mixed together in a mixer and fed as a Total Mixed Ration (TMR). In some cases, feed not consumed by the lactating herd is fed to growing heifers when the rumen is fully developed (> 9 months of age).

Figure 5-3 Conceptual Model of Dairy Systems in the United States



Lactating Cows

Diets for lactating cows are formulated by target milk production or stage of lactation, which reflects the differences in energy and protein required for different amounts of milk produced. Peak lactation occurs about 60 days after calving, and production slowly declines over the next several months. Feedstuffs are commonly blended together in a mixer and fed as a TMR.

Dry Cows

Dry cow diets are often formulated into two stages: far-off dry and close-up dry. During the far-off dry period, cows are fed diets with high forage content (>60%) using ingredients similar to that fed to the lactating herd. As dry cows approach calving, energy content of the diet increases by decreasing forage to include more concentrate feeds and mineral formulation changes in order to avoid pre- and post-partum metabolic disorders that often center around calcium mobilization as the cow begins to lactate. Feedstuffs are commonly blended together in a mixer and fed as a TMR

5.2.1.3 Dairy Housing and Manure Handling

Two general dairy farm types can be distinguished in the United States: confinement feeding systems (including barns and dry-lots) and pasture-based systems (USDA, 2004a). Typical housing systems for confinement feeding operations include tie stall barns, freestall barns, freestall barns with drylot access, and drylots. Drylot systems consist of housing animals in pens similar to beef cattle feedlots, but at a lower stocking density. In pasture-based systems, cattle graze pasture for periods of time, based on feed availability and environmental conditions, and are housed in barns and fed stored feed when pasture is not available. The dairy cattle lifecycle production phase is generally divided into three segments: growing animals (calves and replacement heifers), lactating mature cows, and dry mature cows. Nutrient needs, and therefore diets, and intake are very different between the different lifecycle phases: growing cattle (calves and heifers), lactating cows, and dry cows. Housing and manure management systems vary considerably throughout the country and can differ in a region and by the size of the herd. In cases where housing and manure management varies by animal group (e.g., heifers, dry, and lactating cows), estimates of GHG emissions from one group are not applicable to other groups. When housing and manure management are similar between groups (e.g., all cattle on dry-lots), diet and intake adjustment factors can be used to compare GHG emissions for the different groups.

With the exception of calves, replacement heifers and dry cows may be housed and managed in similar ways as lactating cows. When this is the case, much of the discussion is relevant to the three groups. In cases where the lactating herd is managed in confinement but replacement and dry animals are managed on pasture or in dry-lots, emissions from lactating cattle are not applicable not only due to differences in diet and intake but also due to housing differences. There are no readily available studies that have focused strictly on emissions from dairy calf management and housing. Summarized below are key characteristics of difference in housing by life cycle phase of a dairy cow.

- **Growing (calves and replacement heifers).** Following birth, calves are usually removed from the cow within a few hours and are typically reared on milk or milk replacer in calf hutches or barns for three to seven weeks until weaning. Female calves (replacement heifers) are typically moved to group housing (e.g., super hutches, transition barns, open housing, or pasture) until they reach appropriate breeding weight at about 14 to 15 months of age. Some replacements are contract-reared by heifer growers or sold. Following breeding, heifers are often raised in lots, pasture, or barns until they are ready to calve. Manure in group housing may be handled as a solid (bedded pack or compost barn) or as a slurry, similar to that described below for lactating cows in freestall barns.

- **Lactating Cows.** Heifers typically have their first calf at about 23 to 24 months of age, after which they join the production herd. A cow typically remains in the herd until about five years of age, although many cows are capable of remaining productive in the herd for 12 to 15 years. Each period of production or lactation lasts for 11 to 14 months or longer and spans the time period from calving to dry-off, which is when milking is terminated about 40 to 60 days before the next anticipated calving. Thus, cows are bred while they are producing milk, usually beginning at about 60 days after calving, to maintain a yearly calving schedule. Following the 35 to 60-day dry period, the cow calves again, and the lactation cycle begins anew. Cows average about 2.8 lactations, although many remain productive considerably longer (Hare et al., 2006).

  Lactating cows may be housed in tie stall (stanchion) barns, which limit the cows' mobility because the cows are tethered, fed, and milked in the stalls. A gutter is used to remove the manure by a barn cleaner, which typically places the manure directly into a manure spreader or in a temporary storage pile. Freestall barns allow the cows to move freely in and out of stalls, and the cows are moved to a separate area (milking center or parlor) for milking. Manure typically accumulates in alleyways and is removed via scraping, vacuuming, or flushing with either clean or recirculated water. Some freestall barns have slotted floors with long-term manure storage below the floors. Manure is generally worked naturally through the slots by the cows' feet and with assistance via mechanical scraping equipment. Dairy facilities may also use pastures and dry-lots to house lactating cows. Lots are scraped periodically, as are pastures occasionally, and the solid manure is collected. Although not prevalent, some dairy facilities may house lactating cows in bedded pack or compost barns, again handling manure as a solid material.

- **Dry Cows.** Much like growing cows, housing options for dry cows are the same as described above for lactating cows. The key determinant is management preference for the farm owner and/or facility availability.

Manure and soiled bedding from barns can be handled in a number of ways. Manure can be removed from the barns mechanically and directly loaded into manure spreaders, although this is not common on medium and large farms. Manure can also be processed in an anaerobic digester where bacteria can break down manure to produce biogas that can be flared or captured for energy purposes prior to storage of digester effluent. When manure has a lower solids content, it may be stored in a tank or pit as a slurry, or transported to a solid-liquid separation system with the liquid fraction conveyed (pumped or by gravity) to a long-term storage pond, while the solids can be dewatered naturally and reused as bedding, composted, land-applied, and/or sold. In dry-lot systems, the manure in the pens is typically stacked and following storage is either land-applied or composted. Lot runoff and milking parlor wash water is pumped to a storage pond. There are some dry-lot dairies that use a flush system to clean manure from alleyways behind the feed bunks; this washwater is eventually stored in a wastewater pond. Open freestall dairies have a combination of barns with exercise yards between the barns, and therefore manure is handled similarly as in a traditional freestall barn and dry-lot production system. Wastewater from milking centers (manure, clean-in-place water, and floor washdown water) is typically combined with barn manure destined for long-term storage, and may go through a solid-liquid separation process first. In pasture-based systems, manure is deposited directly onto the pasture and therefore not intensively managed, but may accumulate in areas where animals tend to congregate (e.g., watering areas, shade).

5.2.2 Beef Production Systems

5.2.2.1 Overview of Beef Production Systems

The U.S. beef production system is comprised of several key components for beef cattle, their waste, and their end products, as depicted in Figure 5-4. This conceptual model provides an overview of the typical beef processing systems, following the segments of the beef cattle industry (i.e., cow-calf, stocker, feeder/finisher, and packer) from birth to slaughter and following waste from the animal through a management system. Waste is produced during each stage of activity occurring in the system, and depending on the location, is managed differently.

Of the 90 million beef cattle in the United States, approximately 50 million are mature cows and their calves on cow-calf operations (USDA NASS, 2012), which range in size from a few cows to several thousand cows. These operations are normally based on forages, either improved pastures or native range, and vary in size from a few acres to hundreds of sections. Typically, when calves are 150 to 220 days of age they are weaned and moved to pasture for periods of 60 to 200 days (the stocker phase), although some may move directly to a feedlot. The pastures may be native range, improved perennial pastures, or annuals such as wheat pasture, forage-sorghums, and crop residues such as corn stalks. After the stocker phase, calves normally move to feedlots where they are fed grain- and byproduct-based diets for 110 to 160 days, until they are ready for harvest. In addition, steers and cull heifers from dairy operations are also fed. Approximately 23 million cattle are fed in feedlots annually in the United States. Feedlots range in size from a few hundred head to more than 100,000 head capacity.

5.2.2.2 Diet Information for Beef Cattle

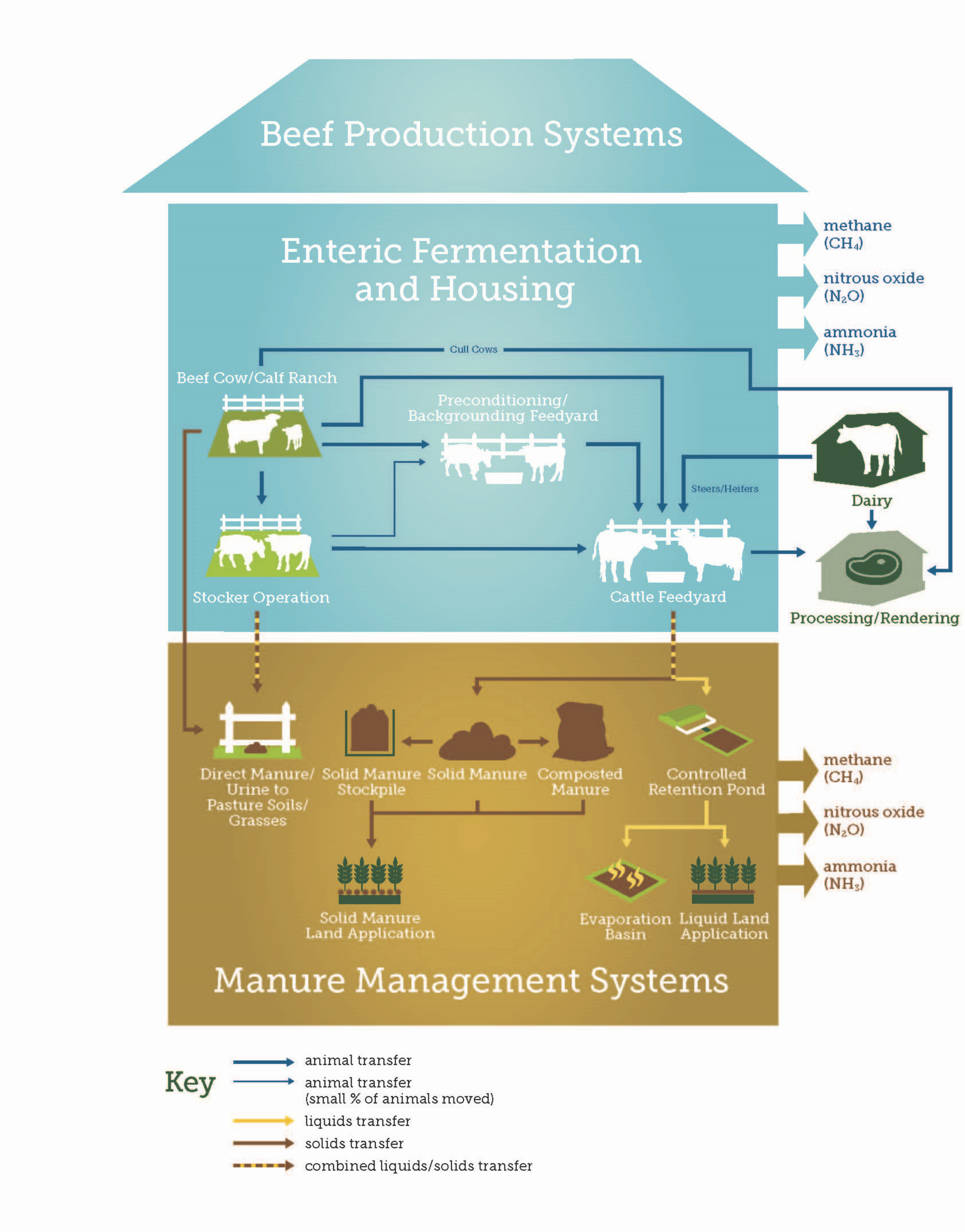
Cow-Calf and Bulls

Grazing pastures may be native range, improved perennial pastures, or annuals such as wheat pasture, forage-sorghums, and crop residues such as corn stalks. Beef cows and bulls are typically fed supplemental feeds during times when pasture or range forage does not meet their nutritional requirements, usually in winter. A recent survey of the beef cow-calf industry found that 74 percent of operations fed a protein supplement and 51 percent fed an energy supplement (USDA, 2010). Overall protein was supplemented for an average of 173 days (SE=9.6) and energy for 162 days (SE=12.7), but this was highly variable across regions of the country. Ninety-seven percent of operations in the survey supplemented the cow herd with roughage for an average of 154 days (SE=7.0). The protein supplements were reported as plant protein or urea-based. Corn was reported as the primary energy supplement. The amount of supplement fed per head per day was not included in the report.

Stockers

Stockers graze forage, including wheat pasture, improved pastures, range, and crop residues. Stocker cattle may also receive supplemental protein or energy feeds to increase performance and/or extend pasture forage. Supplements may or may not contain an ionophore. Some stocker calves may be implanted with a growth promoting implant; others are not.

Figure 5-4 Conceptual Model of Beef Production Systems in the United States



Feedlot

Cattle typically enter feedyards between the ages of 100 and 350 days weighing 200 to 350 kg, and go to slaughter weighing between 500 to 700 kg. They are fed high-concentrate or high-byproduct diets for 100 to 200 days. Of the cattle fed, approximately 55 percent are beef steers, 25 to 30 percent are beef heifers, and 12 to 20 percent are dairy steers and heifers. The vast majority of cattle fed are beef breeds of British or Continental breeding. However, many cattle with Brahman genetics are also fed, mostly in the southern plains. In areas with a significant dairy industry, steers and heifers of dairy breeding (mostly Holstein) are also fed.

Typical feedlot diets contain high concentrations of grain (75 percent or more) and/or byproducts such as distillers grains and gluten feed. They are normally balanced for protein, energy, vitamins, and minerals (Vasconcelos and Galyean, 2007). Because many byproducts contain high concentrations of protein and minerals such as phosphorus and sulfur, when these byproducts are fed, dietary concentrations of protein and some minerals may exceed animal requirements. Feeding of ionophores such as monensin is common in the United States, as is the use of growth-promoting implants. The diets fed in feedyards tend to differ between the northern and southern plains. Finishing diets based on dry-rolled corn (DRC) and high-moisture corn (HMC) dominate in the North, whereas diets based on steam-flaked corn (SFC) dominate in the South. The use of bioethanol co-products such as distiller's grains and corn-milling co-products such as corn gluten feed in finishing diets is greater in the northern plains because of the greater availability of these co-products, but their use is increasing in the southern plains.

5.2.2.3 Beef Cattle Housing and Manure Handling

Cow-Calf and Bulls

Cow herds and replacement heifers are most often housed on pasture. Feces and urine are deposited on pastures and rangeland and may be concentrated in areas in which feeding or watering takes place.

Stockers

Stockers are usually housed on pasture and thus no manure handling is used and GHG emissions are a part of the croplands section (see Chapter 3, Quantifying Greenhouse Gas Sources and Sinks in Cropland and Grazing Land Systems). Calves to be used as stockers can be housed for short periods of time in dry-lots.

Feedlot

Housing and manure management at most beef cattle feeding operations differ greatly from those used in other livestock species, with the vast majority being finished in dry-lot pens with soil surfaces. Manure is normally deposited on the pen surface and scraped from the pens after each group of cattle goes to market. Part of the manure may be stacked in the pen to provide mounds that improve pen drainage and assure that cattle have a dry place to lie after rains. Manure removed from the pen may be immediately applied to fields near the feedlot, stockpiled for later use, or composted in windrows. Manure scraped from the pens normally has a moisture content of 30 to 50 percent and may contain some soil from the pen. Because the manure may remain in the pen or in stockpiles for several months before it is applied to the field, a portion of the nitrogen and carbon may be lost before the manure is collected or applied to land. Runoff from pens is normally collected in retention ponds. Settling basins may be used to limit the quantity of manure solids and soil particles that reach the retention pond.

In the Northern United States, and in areas with high rainfall, cattle may be fed in naturally ventilated barns with slotted floors for collection of urine and feces or in deep-bedded barns with concrete floors in which the manure and bedding (normally straw or stalks) are allowed to accumulate during the feeding period (Spiehs et al., 2011). Adding bedding will increase the quantity of carbon (and possibly nitrogen) available to be metabolized by microbes in the pen. These facilities are characterized by the absence of runoff control systems.

5.2.3 Sheep Production Systems

5.2.3.1 Overview of Sheep Production Systems

There are 81,000 sheep and lamb operations in the United States, with an inventory of 5.53 million sheep and lambs as of January 1, 2011 (USDA NASS, 2011). Most breeding flocks are small and consist of less than 100 head of ewes. The lamb feeding industry is also diverse in size, with small feedlots located throughout the farm flock areas and large feeding operations located in close proximity to local grain production capacity (Shiflett, 2011).

5.2.3.2 Diets, Housing, and Manure Handling for Sheep

Lambing season may occur at various times during the year, depending on production objectives, feed resources, environmental conditions, and market targets. When lambing occurs, January through March, ewes are generally housed in bedded barns. Bedding is removed and spread after animals are turned out on pasture. Ewes are generally bred on pasture in September through November and, depending on weather, will be moved into barns prior to lambing—or earlier as forage availability and weather dictate. Diets consist of pasture or grazing crop residue from spring turnout through early- and mid-gestation. When grazed forage is no longer available, ewes are housed or moved to dry-lots and fed hay and/or hay and grain diets as gestation requirements dictate. The primary forage source is alfalfa, and corn is the predominant grain. Diets range from 100 percent hay to 60:40 percent forage:concentrate while lactating. Most lambs are weaned at approximately 90 days and 41 kg and sent to feedlots for finishing.

Pasture lambing is another farm flock production system that is used to maximize nutrients provided by grazed forages. In this case the ewe is bred in November or December to lamb on pasture in April or May. Lambs are weaned at approximately 120 days and 32 kg and may be sent to the feedlot or finished on grass. Ewes are not fed grain, and harvested forage is provided only when growing seasons and weather dictate. These flocks will be housed in bedded barns in areas requiring protection from winter weather conditions. Range production systems include lambing in April or May, where most (and in some cases all) diets are provided by grazed forages. Supplementation with harvested feeds or grains is usually in response to unpredictable weather and environmental conditions.

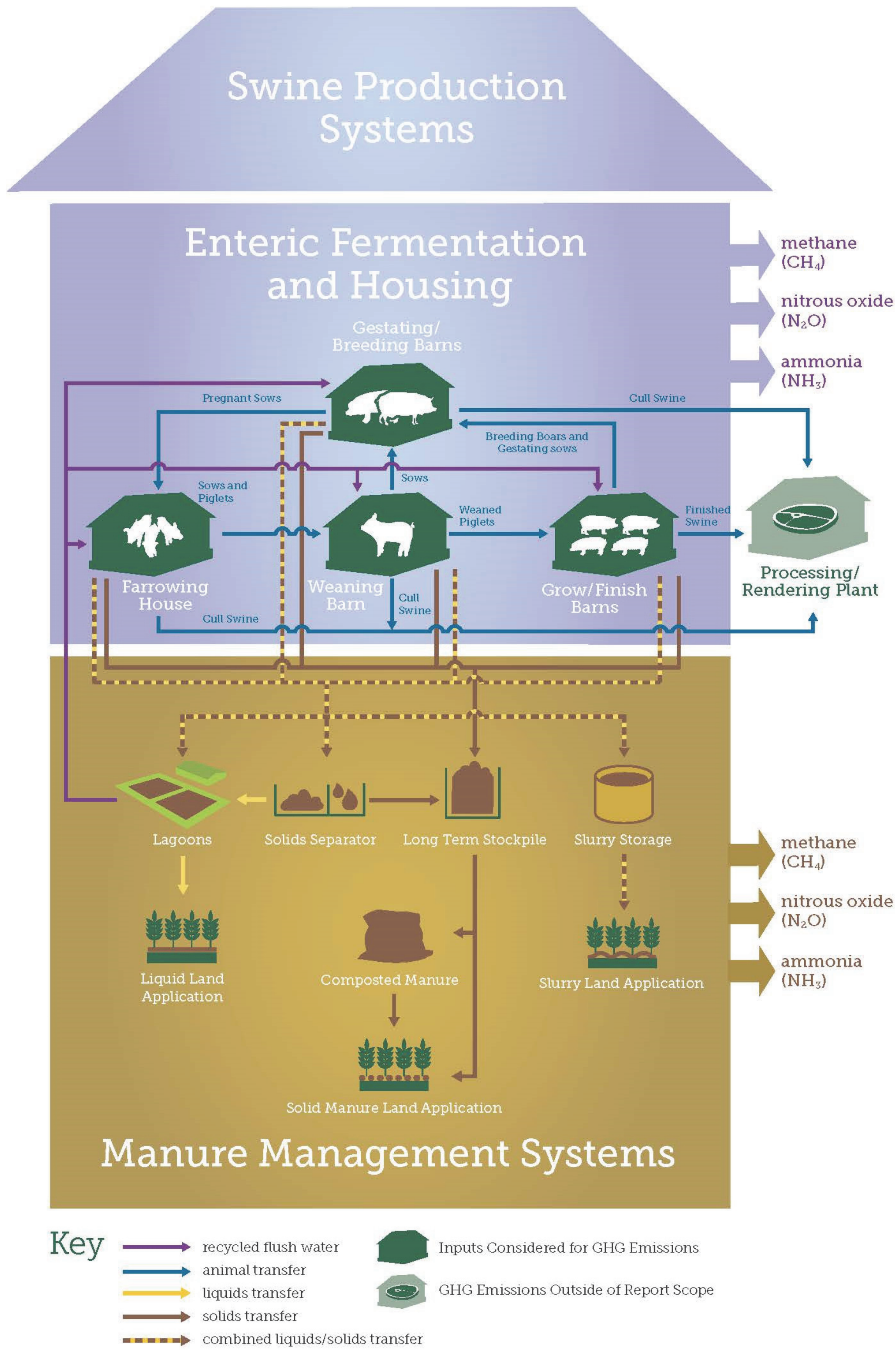
Most lambs are finished in feedlots and fed diets containing 85 to 90 percent grain. Length of feeding periods will range from weeks to months depending on in-weights and time required to reach final weight (industry average final weight = 61 kg). Sheep feedlots are primarily dry-lots, and manure is scraped from the pens similarly to beef cattle feedlots.

5.2.4 Swine Production Systems

5.2.4.1 Overview of Swine Production Systems

The conceptual model (Figure 5-5) of the U.S. swine production system provides an overview of typical production systems, following animals from birth to harvest and following manure from the animal through a management system. Manure is produced during each stage of production occurring in the system, and depending on the location, is managed differently. This has implications on the quantity of GHG emissions and sinks, some of which are discussed in detail in the emissions discussion section (Section 5.3.4).

Figure 5-5: Conceptual Model of Swine Production Systems in the United States



Swine production in the United States remains important to both the nation's diet and economy (Davies, 2011), with significant levels of consumption, imports, and exports. According to the U.S. Department of Agriculture's National Agricultural Statistics Service, the 2011 population was nearly 66 million head (USDA NASS, 2012).

Swine are predominantly grown with production of pork occurring in a two-stage or three-stage system:

- Stage 1: Sow operation, piglets leave at weaning.
- Stage 2 (optional): Nursery operation, weaning (10 days of age/17 lbs) to 42 days of age/45 lbs.
- Stage 3: Several options:
  - A finishing operation (16-week production site where piglets are delivered from a nursery site at approximately 42 days of age/ 45 lbs and stay until 154 days of age (22 weeks) or
  - A wean-to-finish operation (24-week production site where pigs are delivered at weaning directly from a sow operation (10 days of age/17 lbs) and stay until 178 days of age (25.5 weeks)).

The manure management systems associated with these production operations all have the basic elements of collection, storage, treatment, transport, and utilization. Most swine facilities handle manure as a slurry either within the building (deep pit finishing barns or shallow pit nursery, gestation or finishing barns) or in outside storage (pull-plug systems for nurseries, sows, or finishing pigs). Collection and storage is generally accomplished by storage of the waste under the facility, discharge to a separate storage tank, or flushing to an anaerobic lagoon. In the case of in-house manure storage, little water is added to the storage structure, and anaerobic conditions prevail with little biological processing of manure taking place. Outside storage structures that contain slurry with little dilution water offer minimal biological treatment as well. However, lagoon systems where manure is flushed from housing and additional dilution water is added offer more treatment. Dry systems or deep-bedded systems exist to a much lesser extent, primarily for sow or finishing production. In these cases bedding material, often straw, is provided and manure plus bedding is handled as solid material, sometimes composted.

In the Midwest, the system of moving stored swine waste to crop fields is well defined and understood (Hatfield and Pfeiffer, 2005; Malone et al., 2007; Jarecki et al., 2008; Vanotti et al., 2008; Brooks and McLaughlin, 2009; Jarecki et al., 2009; Agnew et al., 2010; Cambardella et al., 2010; Lovanh et al., 2010). Yet these systems continue to evolve to address both old and new issues, such as frozen ground, application timing, and emissions associated with soil application via new equipment. All of the manure management systems result in GHG emissions, but they vary in terms of point and non-point sources.

5.2.4.2 Diet Information for Swine

The swine industry feeds primarily a corn-soybean meal based diet. Dried distillers grains with solubles (DDGS) are often fed to both sows and finishing pigs and, as availability of this feed increases, the amount fed increases to as much as 40 percent of diet dry matter intake (DMI). Similarly, when synthetic amino acid sources price competitively with feed protein sources, the number of synthetic amino acids included in finishing pig diets increases. Two (lysine and methionine) or more (threonine, perhaps tryptophan) synthetic amino acids are fed commonly today with the benefit of reducing total nitrogen fed, and therefore excreted, by swine.

5.2.4.3 Swine Housing and Manure Handling

Most commercially-raised finishing swine are housed indoors to provide a biosecure environment and reduce disease pressures. Manure is handled as slurry with little or no bedding added to the system and minimal addition of water. A small but growing portion of the commercial swine industry house both finishing pigs and sows in hoop barns. In these cases, bedding material, often straw, is provided, and manure plus bedding is handled as solid material.

5.2.5 Poultry Production Systems

5.2.5.1 Overview of Poultry Production Systems

The U.S. poultry production system is comprised of several key processes for poultry, their manure/litter, and their end products (meat, eggs) as depicted in Figure 5-6.

The figure provides an overview of the typical production systems, following both the layer and broiler phases. This conceptual model provides an overview of the typical poultry production systems, following birds from birth to slaughter and following manure from the animal through a management system. Manure is produced during each stage of activities occurring in the system, and depending on the location, is managed differently. The emissions from manure management are discussed in detail in Section 5.3.

The U.S. poultry industry is the world's largest producer and second largest exporter of poultry meat. The U.S. is also a major egg producer. The poultry and egg industry is a major feed grain user, accounting for approximately 45.4 billion kg (100 billion lbs) of feed yearly.

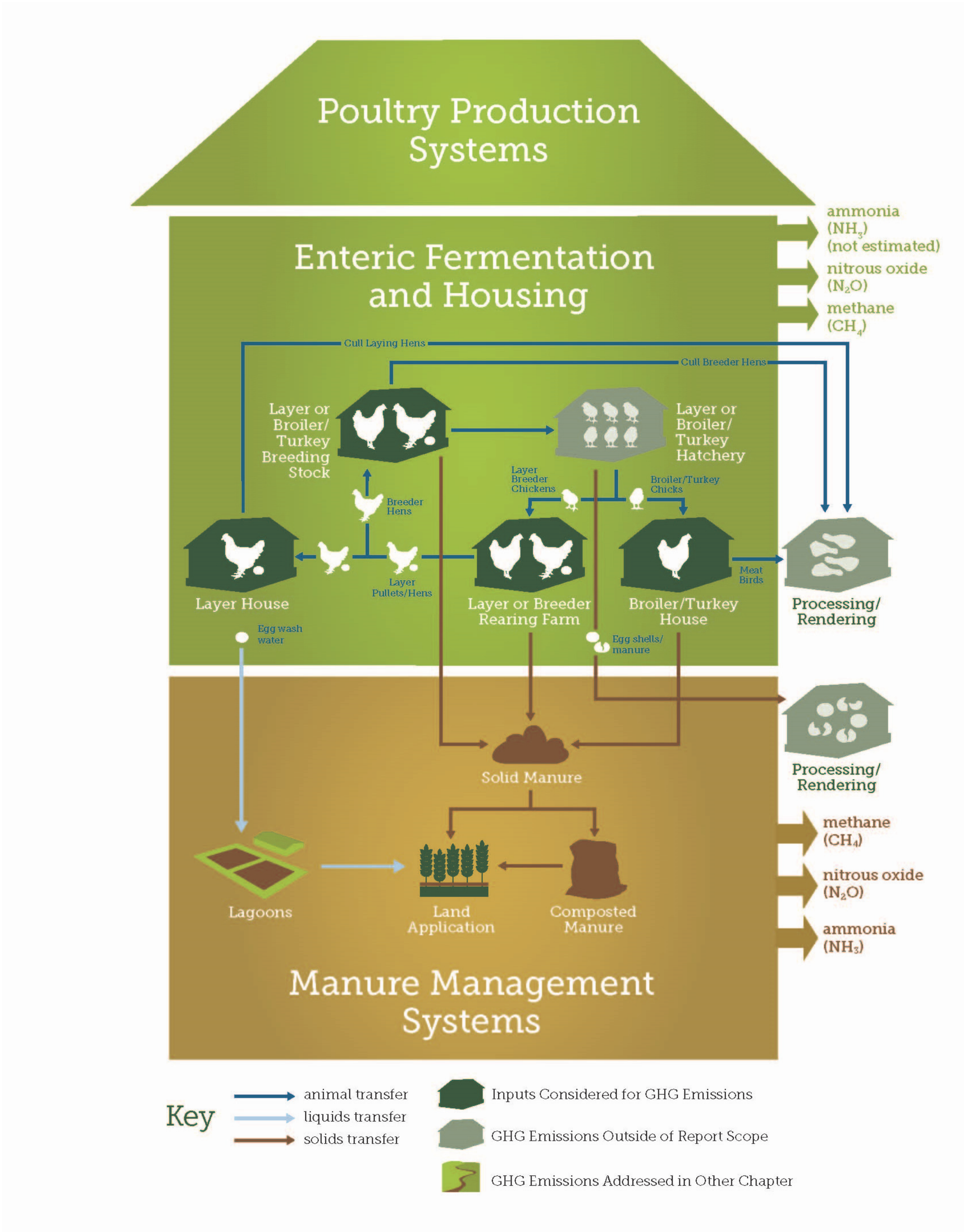
The egg incubation period for a chicken is 21 days. Following hatch, broiler chickens are reared for 42 to 49 days (six to seven flocks per year), depending upon the market intent (e.g., roasters). U.S. egg operations produce more than 90 billion eggs annually. More than 75 percent of egg production is for human consumption (the table-egg market). The remainder of production is for the hatching market. These eggs are hatched to provide replacement birds for the egg-laying flocks and to produce broiler chicks for grow-out operations. Following a 16 to 22 week growth period, hens start laying eggs.

The U.S. turkey industry produces more than one-quarter of a billion birds annually, with the live weight of each bird averaging more than 25 lbs. The egg incubation period for a turkey is 28 days. Following hatch, turkey poults are reared for 15 to 22 weeks (one to three flocks per year) depending on the market intent (e.g., roasters).

5.2.5.2 Diet and Growth Information for Poultry

Diets for meat birds consist largely of corn and soybean meal (commonly 85 to 92 percent of the diet); however, alternate ingredients such as dried distillers grains with solubles (DDGS) and other co-products, and synthetic amino acids are increasingly used. Hen diets are most commonly composed of corn and soybean meal. Other ingredients, such as DDGS, may be included (rarely more than 20 percent of the diet). Ingredient variability is largely in sources of supplemental energy, minerals, and additives to improve animal health and performance. Diets are formulated based on growth rate and egg production and fed as either a mash or a pellet. Bone strength is an important characteristic of meat bird quality therefore provision of minerals such as calcium and phosphorus are carefully considered when diets are formulated. Similarly, eggshell quality is key for laying hens, and as a result, calcium utilization is a key element in diet formulation.

Figure 5-6: Conceptual Model of Poultry Production Systems in the United States



Poultry breeds change rapidly, demonstrating improved production efficiency, and as such, diets are increasingly dense with energy and protein. These changes are due to a combination of genetics and management, including diet formulation.[2] While diet and genetic influences were considered in a study by Havenstein et al. (2007), the results suggest that the diet changes that occurred between 1966 and 2003 interacted with other factors (flock age, ambient temperature) to influence bird growth. Some estimate that 85 percent of the improvement in the growth rate of broiler chickens is attributable to genetics (Havenstein et al., 2003).[3]

In the United States there is no ban, at present, on use of antibiotic growth promoters (AGPs) in poultry production (meat birds). However, the trend is toward consumers wanting products that have not used AGP. Finding replacements for AGP will likely involve the use of multiple products in the diet, each with some of the benefits of AGP, and management changes will play a key role in maintaining animal productivity in their absence. It is unlikely that a single replacement will be found that will prove to be economically viable (Dibner and Richards, 2005).

5.2.5.3 Poultry Housing and Manure Handling

The vast majority of the industry raises birds on litter in mechanically ventilated or naturally ventilated houses. Reuse of litter and number of flocks grown on the same litter is variable across the country, and can range from as low as a single flock to as many as 18 flocks on the same litter source. Litter dry matter content can vary from 40 to 80 percent, depending on management.

Laying hen and pullet housing types range from high-rise houses where hens are in cages and manure accumulates in a basement under the cages and is removed annually, to a manure-belt house where hens are in cages and manure is removed daily or more frequently from the basement to an external shed and stacked before periodic removal for land application (once or twice per year), to aviaries where hens are raised on litter (in large rooms as opposed to cages) that is removed from the aviary annually or more frequently. When manure is removed from the house it may be immediately applied to fields, stockpiled, or composted. Moisture content may vary from 80 percent moisture down to 20 percent moisture (aviaries).

5.3 Emissions from Enteric Fermentation and Housing

Emissions from animal production systems include those from both enteric fermentation and from animal housing (including animal manure in housing areas that may ultimately be flushed or scraped and then transported to an external manure management system). The production of GHGs in livestock systems originates from a variety of sources, including directly from the animals themselves; manure in lots and barns; stockpiled and composting manures; manure slurries or waters in tanks, pits, lagoons, retention ponds, settling cells, etc.; and from soils after manure application. Emissions from these sources depend on animal size and age, diet, manure production, handling and storage system, lot surface and soil characteristics, and ambient weather conditions (i.e., temperature, wind, humidity, and precipitation). For each animal type, this section summarizes

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[2] Havenstein et al. (2007) compared 1966 strains to 2003 strains and observed a 20 percent better cumulative feed conversion ratio in the 2003 tom turkey fed a 2003 diet relative to a 1966 tom fed a diet typical of 1966. Feed efficiency to 11 kg bodyweight was approximately 50 percent better (2.13 at 98 days of age in 2003 toms, compared with 4.21 at 196 days for 1966 toms).

[3] Havenstein et al. (2003) compared the 1957 Athens-Canadian Randombred Control strain and the 2001 Ross 308 strain of broilers when fed representative 1957 and 2001 diets. The 42-day feed conversions for the Ross 308 birds fed the 2001 and 1957 feeds were 1.62 and 1.92, respectively (with average bodyweight of 2,672 and 2,126 g). The 42-day feed conversions for the Athens-Canadian Randombred Control were 2.14 and 2.34 (average bodyweight of 578 and 539 g, respectively).

the current understanding of enteric fermentation and livestock housing emissions and presents recommended models for estimating such emissions, including the rationale for selecting methods.

Actual field measurements of GHGs from enteric fermentation over the past several decades have been instrumental in improving our understanding of the underlying science and the resulting models presented in this section. For dairy animals, most of the emissions estimates available represent the lactating animal. The equations for growing beef animals are likely appropriate for growing dairy animals if diet composition is considered. The text boxes on the following pages summarize several of the key techniques that have been used in measurement studies for both individual animals and groups of animals. Further studies of this type will be needed to address research gaps in Section 5.5.

This section provides the recommended method for estimating GHGs from enteric fermentation and applicable housing emissions. Quantitative methods are provided for dairy, beef, sheep, swine, poultry, and other animals (i.e., goats, American bison, llama, alpacas, and managed wildlife). For each section, background information is provided on the range of emissions and existing models for estimating emissions and the rationale for the method selected. For estimating emissions from enteric fermentation, the activity data is the same for all animal types. Ancillary data includes the properties of the diets (e.g., crude protein (CP), digestible energy (DE), neutral detergent fiber (NDF)). For simplicity, activity data and ancillary data are listed in Table 5-2 and are not repeated below for each animal type.

5.3.1 Enteric Fermentation and Housing Emissions from Dairy Production Systems

Although the dairy industry is primarily composed of three livestock types [growing (i.e., calves, replacement heifers), lactating cows, and dry cows], most of the limited emissions research conducted to date has been targeted at lactating cows, which typically produce at least 50 percent more enteric CH_4 per head than other dairy cattle. Few emissions data exist for calves, heifers, and dry cows. Therefore, the discussion here focuses primarily on lactating cows.

Data needed to estimate emissions include housing system (pasture, barn type, dry-lot), animal characteristics (breed, body weight, growth potential, stage of lactation, milking frequency, and milk production) and population, dietary information (DMI, dietary CP—also NDF, fat, DE, metabolizable energy (ME), net energy (NE), nutrient excretion (N, C, and volatile solids), use of recombinant bovine somatotropin, use of monensin, type of manure handling system, frequency of manure removal, type of bedding, and manure characteristics (total ammonium nitrogen, pH).

Enteric Fermentation

Enteric CH_4 production varies with production stage in dairy cattle, with the highest rates being produced by lactating cows (Table 5-4). This table illustrates, conceptually, the observed variation in cattle at different stages of maturity and activity, but it is not intended to provide a depiction of absolute differences. There are many factors that affect enteric CH_4 production, and therefore altering dairy cattle diets could have an impact on enteric CH_4 production. For an in-depth discussion of dietary effects on enteric CH_4 production, see Section 5.3.7 (*Factors Affecting Enteric Fermentation Emissions*). However, the results in Table 5-4 clearly illustrate the difference in enteric emissions; in particular, emissions from dairy cattle are relatively higher than those from growing (i.e., heifers) and dry cattle.

Table 5-4: Examples of CH_4 Emissions Measured in Dairy Cattle

Animal Type	CH_4 Emission	Method Used to Measure Emissions	Reference
Dairy cattle	260 g animal^{-1} day^{-1}	Calculated Blaxter and Clapperton	Crutzen et al. (1986)

<table>
<tr><th>Animal Type</th><th>CH_4 Emission</th><th>Method Used to Measure Emissions</th><th>Reference</th></tr>
<tr><td>Heifer 6-24 month</td><td>140 g LU^{-1} day^{-1}</td><td>See above</td><td></td></tr>
<tr><td>Dairy cattle, dry period</td><td>139 g LU^{-1} day^{-1}</td><td>Respiration calorimetry</td><td rowspan="2">Holter & Young (1992)</td></tr>
<tr><td>Dairy cattle, lactating</td><td>268 g LU^{-1} day^{-1}</td><td>See above</td></tr>
<tr><td>Dairy cattle</td><td>257 g LU^{-1} day^{-1}</td><td>Respiration calorimetry</td><td>Kirchgessner et al. (1991)</td></tr>
<tr><td>Dairy cattle, lactating</td><td>429 g $animal^{-1}$ day^{-1}</td><td>Wind tunnel</td><td rowspan="2">Sun et al. (2008)</td></tr>
<tr><td>Dairy cattle, dry period</td><td>290 g $animal^{-1}$ day^{-1}</td><td>Wind tunnel</td></tr>
<tr><td>Dairy cattle, lactating</td><td>538 – 648 g $animal^{-1}$ day^{-1}</td><td>Respiration calorimetry</td><td>Aguerre et al. (2011)</td></tr>
</table>

LU, livestock unit = 500 kg

Methods for Measuring CH_4 Emissions from Enteric Fermentation

Individual Animals

The standard method of measuring CH_4 emissions from ruminants is by respiration calorimetry chambers. Other techniques, including head boxes, internal tracers, micrometeorology, isotope dilution, and polyethylene tunnels, have been used (Kebreab et al., 2006; Harper et al., 2011). Several new technologies have been developed to measure individual animal emissions. To address the difficulty in measuring enteric CH_4 from many animals on pasture, alternate methods are sought. As one example, Goopy et al. (2011) has proposed a portable static chamber method to measure daily CH_4 production. Until validated, results using alternate methods should be viewed with caution.

A variety of respiration chambers have been developed to measure enteric CH_4 losses and/or total energy metabolism of the animal. In general, air is pulled from the chamber at a known rate and replaced with outside air. Flow of air and concentrations of CH_4, CO_2, and oxygen (O_2) in the air entering and leaving the chamber are measured to determine total CO_2 and CH_4 production and O_2 consumption. When properly calibrated and used, respiration chambers give highly accurate, precise measurements. However, they are expensive to build and operate, and require significant knowledge, skill, and labor.

Feed intake and production are usually depressed in animals in chambers and the measurements do not necessarily reflect intake and production from typical commercial operations. This limitation can be partially overcome by feeding animals at different levels of intake and measuring the effects of intake level. Head boxes use the same principles as respiration calorimetry, and have many of the same limitations. In-barn chambers using drop-down curtains have been used to measure, at relatively low cost, emissions of NH_3, CH_4, and other gasses from groups of dairy cows (Powell et al., 2007; Powell et al., 2008; Aguerre et al., 2011).

Internal tracer techniques such as the sulfur hexafluoride (SF_6) tracer method (Johnson et al., 1994) were developed to allow measurements from free-ranging animals, such as those managed under pasture situations, or when real-world levels of feed intake are needed. The limitations to this method are the need for trained animals, the need for larger sample sizes (compared with chambers) to detect the influence of mitigation techniques, and concerns about inconsistent releases of tracer gas from SF_6 permeation tubes manufactured for large release rates. Additionally, the SF_6 technique generally results in emission estimates that are lower than chamber measurements; possibly because the SF_6 method does not measure all lower gut CH_4 production (McGinn et al., 2006). The advantages and shortcomings of the SF_6 method have been recently reviewed (Lassey et al., 2011).

Methods for Measuring CH_4 Emissions from Enteric Fermentation

Group of Animals

Micrometeorology methods have been used extensively to measure CH_4 and NH_3 emissions from pastures, whole feed yards, or portions of the feed yard (pens, retention ponds, manure stockpiles, etc.). These methods have been reviewed (Fowler et al., 2001; Flesch et al., 2005; Harper et al., 2011). Lauback et al. (2008) compared the SF_6 method with three micrometeorological methods (integrated horizontal flux, flux gradient, and backward Lagrangian stochastic (bLS)) using steers grazing paddocks. In general, the micrometeorological methods gave higher CH_4 measurements than the SF_6 method, with the difference being greater when animals were within 22 meters of the CH_4 sampler. This effect was especially true for the flux gradient method. The lower values for the SF_6 method could be due in part to the fact that the SF_6 method does not measure emissions from the lower gut or from fermentation of feces on the paddock surface.

Tomkins et al. (2011) compared enteric CH_4 emissions of steers on pasture using the bLS method and respiration chambers. Emissions estimated using the bLS model were slightly greater than with respiration chambers (136.1 vs. 114.3 g head daily^{-1}). However emissions per gram of DMI were similar (29.7 vs. 30.1 g CH_4 kg DMI^{-1}, respectively), suggesting that the bLS model may be suitable for estimating enteric emissions.

Most dispersion models and micrometeorological methods assume that emissions are uniformly distributed over the source area. In some cases, such as for individual cattle in a pen or field, this is not true. Therefore, McGinn et al. (2011) developed a method that used a point-source dispersion model and atmospheric CH_4 concentrations measured using multiple open-path lasers to measure CH_4 emissions from a paddock containing 18 cattle. Measured enteric CH_4 emissions were similar to values measured using other techniques. However, recoveries of known CH_4 releases averaged only 77 percent using this method. The method gave more reliable measurements during the daytime when atmospheric conditions were unstable than at night when atmospheric conditions were stable.

Methods for Measuring Emissions from Manure

Estimating emissions from large open source areas typically associated with both dairy and beef cattle production is very challenging, due to the inability to contain and measure the source area. Instruments and techniques to measure ambient atmospheric gases from these large source areas (i.e., dry-lot beef and dairy cattle yards, freestall dairies with naturally ventilated curtain sidewall barns, and grazing land) must be able to detect lower concentrations than those encountered in typical enclosed confined animal production systems, because of the low concentrations and high variability resulting from high and variable ventilation rates. A larger challenge with measuring emissions from open facilities is the ability to estimate airflow due to the lack of a defined, constant air inlet and air outlet. Reported background NH_3 concentrations typically range from <1.3 to 53.3 parts per billion (ppb) (Todd et al., 2005), background atmospheric N_2O concentrations near feedyards average about 319 ppb (Michal et al., 2010), and background CH_4 concentrations typically run in the area of 1,780 ppb (Michal et al., 2010).

Methods for Measuring Emissions from Manure (Continued)

Numerous factors can affect atmospheric concentrations of NH_3 and GHG near livestock operations including sampling height, atmospheric stability, wind speed, background concentrations, stocking density, sampling site, sampling time, temperature, and wind direction (fetch). Average daily NH_3 concentrations measured at a variety of similar source areas ranged from approximately 100 to 2,000 µg m^{-3}. Measured maximum concentrations rarely exceed 2,000 µg m^{-3}. Ammonia concentrations decrease rapidly downwind of source areas (Miner, 1975), approaching background concentrations in less than 800 meters (McGinn et al., 2003; Sweeten, 2004).

Atmospheric CH_4 concentrations measured at feedlots and dry lot dairies have ranged from 3.3 to 4.7 parts per million (ppm) (Michal et al., 2010), and from background (approximately 1.78 ppm) to 6.20 ppm (Bjorneberg et al., 2009), respectively. Nitrous oxide concentrations measured at feedlots ranged from 319 ppb (background) to 443 ppb and averaged 396 ± 16 ppb (Michal et al., 2010). Nitrous oxide concentrations were highest following a rainfall event. After a rain, CH_4 concentrations averaged 3.7 ± 0.1 ppm. At dry-lot dairies, median N_2O concentrations ranged from 314 ppb to 330 ppb, which are very close to global background values (Bjorneberg et al., 2009).

Small flux chambers and wind tunnels have been used to estimate emissions of NH_3, CH_4, and N_2O from farmlands, pastures, pen surfaces, lagoons, and retention ponds (Hutchinson and Mosier, 1981; Venterea et al., 2009; Venterea, 2010; Harper et al., 2011; Hristov et al., 2011). In general, chambers alter the microenvironment of the surface and may alter emissions. Thus, the accuracy of these methods for determining emission factors for some gases (especially NH_3) has been questioned (Gao and Yates, 1998; Harper, 2005; Venterea et al., 2009; Parker et al., 2010; Venterea, 2010; Harper et al., 2011). Measures of NH_3 emissions using flux chambers and wind tunnels are highly dependent upon air flow and air turnover rates in the chamber (Cole et al., 2007b; Parker et al., 2010). Based on the conventional two-film model used to describe volatilization from a solute-solvent mixture (Parker et al., 2010), many gaseous emissions are controlled by the gas film above the liquid or the upper portion of the liquid (liquid film) defined by the Henry's law constant. If volatilization is inhibited by high concentrations in the gas phase (i.e., gas-film controlled), increases in gaseous concentration—such as with flux chambers—will lead to significant underestimation of true flux. Venterea (2010) reported that emissions of N_2O estimated using static chambers were underestimated by approximately three to 38 percent, depending upon soil water content, type of regression performed (linear vs. quadratic vs. nonlinear), and other factors. The percentage of underestimation tended to be greater with dry soils, probably because N_2O flux is lower when soils are dry. Sommer et al. (2004) reported that GHG emissions from compost stockpiles measured using static chambers were only 12 to 22 percent of values measured using the integrated horizontal flux method.

Because of these factors, flux chambers should be used to examine relative differences, rather than emission factors of NH_3, CH_4, and N_2O emissions from pen surfaces, lagoons, retention ponds, manure stockpiles, or compost windrows. In addition, the surface of pastures and feedlot pens is temporally and spatially heterogeneous, with dry areas, areas with fresh feces, and areas with urine of different ages (Woodbury et al., 2001; Cole et al., 2009a; Cole et al., 2009b). To adequately represent the surface, the number of chamber measurements required (estimated as the coefficient of variation squared/100: Kienbusch, 1986) can be very large (i.e., one chamber/quare meter: Cole et al., 2007b).

Housing

There are a wide variety of dairy cattle housing systems due to variations in herd size and regional practices. In the northeastern United States, herd size tends to be smaller and cattle are housed in freestall and tie-stall barns and on pasture; in the western part of the country, herd sizes tend to be larger and animals are housed in freestall barns or dry-lots with few producers using pasture-based systems. These differences in housing can lead to differences in both GHG and NH_3 emissions. Examples of reported emissions from varying housing systems are presented in Table 5-5.

Table 5-5: Examples of Reported On-Farm Emission Estimates for CH_4, N_2O, and NH_3 from a Variety of Dairy Cattle Housing Systems

Housing	Country	Emissions (g cow^{-1} d^{-1})			Reference
		CH_4	N_2O	*NH_3*	
Barn	Germany	402		*64.8*	Saha et al. (2014)
Tie stall barn	Austria	170-232[a]	0.14-1.2[a]	*4-7.4[a]*	Amon et al. (2001)
Barn	Germany	256	1.8	*14.4*	Jungbluth et al. (2001)
Dry-lot	U.S.			*41-140*	Cassel et al. (2005)
Hardstanding	UK	0.03[b]	0.01	*11*	Ellis et al. (2001)
Open-freestall	U.S.	410	22	*80*	Leytem et al. (2013)
Tie stall barn	Canada	390			Kinsman et al. (1995)
Pasture	NZ	300-427			Laubach & Kelliher (2005)
Dry-lot	U.S.	490	10	*130*	Leytem et al. (2011)
Standoff pad	NZ	1.66[b]	0.03		Luo & Saggar (2008)
Barn	Denmark	256	1.2	*16*	Zhang et al. (Zhang et al., 2005)
Dry-lot	China	397	37		Zhu et al. (Zhu et al., 2014)
Barn	Sweden	216-312[a]		*21-27[a]*	Ngwabie et al. (2009)
Barn	Germany	464	45	*92.4*	Samer et al. (Samer et al., 2011)
Pasture	Uruguay	372			Dini et al. (Dini et al., 2012)

*Denotes measurements in g LU^{-1} d^{-1}, where a LU (livestock unit) = 500 kg.
†Measurements do not include enteric CH_4 production.

Variations in emissions from housing are due to factors such as temperature, diet composition, water consumption, ventilation flow rates, type of manure handling systems, manure removal frequency, feces, and urine characteristics (i.e., pH and total ammoniacal nitrogen (TAN)), and type of bedding used. Although differences can be great between emission rates, there are some emission characteristics that are consistent across most studies. Many studies have reported strong diel trends in emissions of CH_4 and NH_3, with emissions tending to be lower in the late evening and early morning and then higher throughout the day till early evening (Amon et al., 2001; Cassel et al., 2005; Powell et al., 2008; Sun et al., 2008; Bjorneberg et al., 2009; Flesch et al., 2009; Ngwabie et al., 2009; Aguerre et al., 2011; Leytem et al., 2011). This strong diel trend in emissions can be associated with wind speed and temperature, as winds tend to be light in the late evening and early morning and then, in most instances, steadily increase throughout the day to reach a peak in the late afternoon. Temperature also increases from early morning to late afternoon, and then decreases again. Additionally, cattle activity tends to increase from morning to late afternoon as animals wake and begin to eat, drink, ruminate, defecate, and urinate. As these activities increase, one would expect an increase in CH_4 (and NH_3) emissions. There are also seasonal trends in emissions, the most prominent being in NH_3 emissions, with the lowest rates in winter compared with the other seasons (Amon et al., 2001; Powell et al., 2008; Bjorneberg et al., 2009; Flesch et al., 2009; Aguerre et al., 2011; Leytem et al., 2011). Powell et al. (2008), Flesch et al. (2009), and Aguerre et al. (2011) reported that barn emissions of NH_3 in Wisconsin were lowest in winter, with winter rates about one-half to one-third lower than those in the spring and summer, which was

Ammonia Emissions in Dairy Cattle Housing

As mentioned earlier, ammonia is not a greenhouse gas, however, ammonia emissions are estimated as part of the nitrogen balance approach. Emissions of NH_3 from dairy cattle housing systems have been strongly linked to dietary nitrogen intake, as this affects the amount of urea nitrogen excreted in urine. Of the nitrogen in the total crude protein (CP) typically consumed by a dairy cow on commercial dairy farms, 20 to 35 percent is secreted in milk and the remaining nitrogen from CP is excreted about equally in feces and urine. Feed nitrogen (N=CP÷6.25) use efficiency (percentage of feed nitrogen secreted as milk nitrogen) and the 50:50 fecal nitrogen:urinary nitrogen excretion ratio can be influenced greatly, however, by what is fed to the cow. Feeding nitrogen in excess of nutritional requirements has very few significant impacts on milk production or quality; it decreases feed nitrogen use efficiency and increases the relative amount of urea nitrogen excreted in urine. The urea nitrogen contained in cow urine (which is 55 to 80 percent of the nitrogen contained in urine, depending on concentrations of CP in the ration) is the major source of NH_3 emission from dairy farms. Urea is produced when nitrogen-rich proteins and/or non-protein nitrogen sources break down (mainly in the cow rumen), forming NH_3 gas that may be used by ruminal microbes to produce microbial proteins or can be absorbed through the ruminal wall to the blood stream. In the kidney, blood NH_3 from the digestive tract or tissue metabolism is eventually converted to urea before being excreted in the urine. Urease enzymes, which are present in feces and soil, rapidly convert excreted urea to ammonium, which can be hydrolyzed quickly into NH_3 gas and lost to the atmosphere. Thus, the increase in urea nitrogen excretion due to excessive ration CP increases NH_3 emissions during the collection, storage, and land application of manure (Rotz, 2004; Misselbrook et al., 2005; Powell et al., 2008; Arriaga et al., 2010).

Paul et al. (1998) examined the effects of altering dietary CP on NH_3 losses from dairy cows. They reported that NH_3 emissions during the first 24 hours following manure excretion were 38 and 23 percent of the total manure nitrogen from diets with 16.4 and 12.3 percent CP concentrations, respectively, and 22 and 15 percent of total manure nitrogen from diets containing 18.3 and 15.3 percent dietary CP, respectively. Misselbrook et al. (Misselbrook et al., 2005) reported that reducing dietary CP content resulted in less total nitrogen excretion and a smaller proportion of the excreted nitrogen being present in urine; urine nitrogen concentration was 90 percent greater for the high-CP than the low-CP diet.

However, Li et al. (2009) found no effect of lowering dietary CP in lactating dairy cattle on NH_3 emissions from the floor of a naturally ventilated freestall dairy barn at low and moderate temperatures (0 to 20°C). This lack of response to CP is likely due to the fact that urease activity is negligible at temperatures below 10°C (Bluteau et al., 2009). Factors that are essential in determining NH_3 emissions are manure or urine pH and the total ammoniacal nitrogen content, both of which are related to the dietary CP level.

The majority of NH_3 emissions from housing systems are due to the volatilization of NH_3 from urine deposition. As discussed above, nitrogen intake drives the amount of urea that is excreted in the urine. As this urine is deposited on barn floors, pastures, or dry-lots, it mixes with urease from either feces or soil and is then hydrolyzed to ammonium and, via effects of pH, converted to NH_3 and lost to the atmosphere. The loss of NH_3 happens rapidly, with most NH_3 losses occurring within 24 hours following deposition. Therefore, estimation of NH_3 emissions needs to take into account the amount of urea generated by the cow, pH (urine, manure, or soil), temperature, and air flow over the source. Strategies that reduce nitrogen excretion will be very beneficial in reducing NH_3 emissions from housing/pasture systems.

attributed to cold winter temperatures. In general, N_2O emissions from housing were found to be low and showed no discernible diel or seasonal trends (Bjorneberg et al., 2009; Ngwabie et al., 2009; Adviento-Borbe et al., 2010; Leytem et al., 2011), suggesting that these emissions from this sector of the production system are of relatively little concern. There are consistent reports of both diel and seasonal variations in both CH_4 and NH_3 emissions, so it is imperative that these factors be captured in any estimation of emissions for a given production system.

Emissions of CH_4 are dominated by enteric fermentation in housing/pasture systems. Amon et al. (2001) examined CH_4 emissions from a tie-stall dairy barn in Austria using either a slurry-based system or straw-based system. In both systems, about 80 percent of the net CH_4 emissions were due to enteric fermentation, with the remaining amount coming from the manure. Sun et al. (2008) measured CH_4 emission from dairy cows and fresh manure in chambers, and reported that fresh manure alone did not produce noticeable CH_4 fluxes. In some dairy production systems, manure is removed from the animal housing area frequently; therefore, CH_4 emissions from animal housing areas of a dairy can be largely attributed to enteric emissions.

N_2O emissions tend to be negligible from both animals and fresh manure. The majority of N_2O emissions result from manure storage, pasture, and land application of manures. Therefore, the main sources of N_2O emissions from animal housing would be from dry-lot dairies and stand-off pads, because there is potential for deposited nitrogen to be nitrified and denitrified under wet conditions and lost as N_2O. Luo and Saggar (2008) measured N_2O and CH_4 emissions from a dairy farm stand-off pad in New Zealand and reported N_2O fluxes from 0 to 3 g N_2O-N day^{-1}, which they attributed to the concentrations of water and nitrate in the pad materials. Overall, only 54 g of N_2O-N was emitted from the pad over the time of use, representing ~0.01 percent of the excreta nitrogen deposited on the pad.

While there have been overall improvements in milk production with breeding programs, there is no evidence that any breed of dairy cow produces less enteric CH_4. Münger and Kreuzer (Münger and Kreuzer, 2008) measured enteric CH_4 production from Holstein, Simmental, and Jersey cows and found no persistent differences in CH_4 yields, with average enteric CH_4 being approximately 25g CH_4 kg DMI^{-1}.

5.3.1.1 Method for Estimating Emissions from Dairy Production Systems

Method for Estimating CH_4 Emissions from Enteric Fermentation in Dairy Cows

- Mills et al. (2003) developed a series of submodels to estimate enteric CH_4 emissions from dairy and beef cattle. The optimal model appeared to be a nonlinear Mits3 equation, which is utilized by the DairyGEM Model (a subset of IFSM) (Rotz et al., 2011b) and is shown in Equation 5-1 (Mits3 equation) is based primarily on metabolizable energy intake, acid detergent fiber (ADF), and starch content of diet.
- Data sources are user input on dietary intake, as well as dietary data from the Feedstuffs Composition Table (Ewan, 1989; Preston, 2013).
- Use of the DairyGEM/Mits3 equation is recommended over the IPCC Tier 2 equation (IPCC, 2006) because it has proven to be more accurate, in general, for dairy cows.

Equation 5-1: Non-Linear Mits3 Equation

$$\boldsymbol{CH_4} = (\mathbf{E_{max}} - [\mathbf{E_{max}} \exp^{-cx}]) \times \mathbf{0.018}$$

Where:

CH_4 = Enteric methane emissions per day (kg CH_4 head^{-1} day^{-1})

E_{max} = Maximum possible CH_4 emissions (MJ head^{-1} day^{-1})

c = Shape parameter determining emission change with increasing metabolizable energy intake (see Equation 5-2)

x = Metabolizable energy intake (MJ head^{-1} day^{-1})

0.018 = Conversion of MJ to kg of CH_4 (kg CH_4 MJ^{-1})

The E_{max} is constant for all animals at 45.98 MJ/head/day. The shape parameter "c" is calculated from the dietary non-fiber carbohydrate (NFC) to acid detergent fiber (ADF) ratio in Equation 5-2.

Equation 5-2: Calculating Shape Parameter

$$\mathbf{c} = -\mathbf{0.0011} \times \frac{\mathbf{NFC}}{\mathbf{ADF}} + \mathbf{0.0045}$$

Where:

c = Shape parameter determining emission change with increasing metabolizable energy intake (unitless)

NFC = [(100-NDF + CP + EE)/100] x DMI (kg head^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1} day^{-1})

ADF = Acid Detergent Fiber (kg head^{-1} day^{-1})

NDF = Neutral Detergent Fiber (%)

CP = Crude Protein (%)

EE = Ether extract (%)

Mills et al. (2003) noted that nonlinear models have two advantages over linear models: 1) a maximum emission is set; and 2) it is explainable from a biological sense. The feedstuff characteristics needed to calculate emissions from dairy cattle are included in the example below (Ewan, 1989; Preston, 2013). The full table can be found in Appendix 5-B.

Table 5-6: Example Feedstuffs Table[a]

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g	NE_l	DE (% of GE)*	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Alfalfa Cubes	x91	57	57	25	57		18	30	29	36	46	40	2.0	11	1.30	0.23	1.9	0.37	0.33	20
Alfalfa dehydrated 17% CP	92	61	62	31	61	65.16	19	60	26	34	45	6	3.0	11	1.42	0.25	2.5	0.45	0.28	21
Alfalfa fresh	24	61	62	31	61	62.54	19	18	27	34	46	41	3.0	9	1.35	0.27	2.6	0.40	0.29	18

Source: Preston (2013).

[a] Column headings:

DM	= Dry matter	GE	= Gross energy	ASH	= Ash
TDN	= Total digestible nutrients	CP	=Crude protein	Ca	= Calcium
NEm	= Net energy for maintenance	UIP	= Undegradable intake protein	P	= Phosphorous
NEg	= Net energy for growth	CF	= Crude fiber	K	= Potassium
NEl	= Net energy for lactation	ADF	= Acid detergent fiber	Cl	= Chlorine
Mcal	= Megacalories	NDF	= Neutral detergent fiber	S	= Sulfur
cwt	= Centum weight (hundredweight)	eNDF	= effective neutral detergent fiber	Zn	= Zinc
DE	=Digestible energy	EE	= Ether extract	ppm	= parts per million

Method for Estimating Dairy Cows' GHG Emissions from Housing

Methane

- The DairyGEM Model (a subset of IFSM) (Rotz et al., 2011a) calculates CH_4 emissions from housing surfaces.
- DairyGEM uses the IPCC (2006) Tier 2 method to estimate CH_4 emissions when manure is allowed to accumulate in the housing.

Nitrous Oxide

- Nitrogen excreted estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O emissions from manure in housing.

Methane Emissions from Dairy Cows' Housing

The DairyGEM Model (Rotz et al., 2011a) calculates CH_4 emissions from barn floors using an empirical model developed from three freestall barns (Chianese et al., 2009c).

Equation 5-3: Calculating CH_4 Emissions from Barn Floors (Chianese et al., 2009c)

$$\mathbf{CH_4 = \max(0.0, 0.13T) \times \frac{A_{barn}}{1000}}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 head^{-1} day^{-1})

T = Barn temperature (˚C)

A_{barn} = Area of the barn floor covered with manure (m^2)

When manure is allowed to accumulate as a stockpile, on a dry-lot, or in a pit below the animal confinement, the DairyGEM model uses the IPCC (2006) Tier 2 method to estimate CH_4 emissions (Equation 5-4). This is the same equation used for estimating emissions from manure that is managed outside of housing (see Section 5.4.1 *Temporary Stack and Long-Term Stockpile* and 5.4.2 *Composting* for details).

Equation 5-4: IPCC Tier 2 Approach for Estimating CH_4 Emissions in Housing

$$E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}$$

Where:

E_{CH4} = CH_4 emissions per day (kg CH_4 day^{-1})

m = Total dry manure per day [a] (kg dry manure day^{-1})

VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 (kg VS)$^{-1}$)

MCF = CH_4 conversion factor for the manure management system (%)

0.67 = Conversion factor of m^3 CH_4 to kg CH_4

The maximum CH_4 producing capacity (B_0) for manure varies by animal category and is provided in Table 5-19. The CH_4 conversion factors (MCF) for manure deposited on a dry-lot, stored in a deep pit, or from cattle bedding can be found in Table 5-7. The MCFs for manure stored as a stockpile are provided in Table 5-20 through Table 5-22. The MCFs for manure composted within housing are provided in Table 5-24.

Table 5-7: Methane Conversion Factors for Dry-Lots, Pit Storage Below Animal Confinement, and Cattle/Swine Bedding

Temperature		Dry-Lot	Pit Storage Below Animal Confinement and Cattle/Swine Deep Bedding	
			< 1 month	> 1 month
	≤10 °C			17%
	11 °C			19%
Cool	12 °C	1%	3%	20%
	13 °C			22%
	14 °C			25%
	15 °C			27%
	16 °C			29%
	17 °C			32%
	18 °C			35%
	19 °C			39%
Temperate	20 °C	1.5%	3%	42%
	21 °C			46%
	22 °C			50%
	23 °C			55%
	24 °C			60%
	25 °C			65%
	26 °C			71%
Warm	27 °C	2%	30%	78%
	≥28 °C			80%

Source: IPCC (2006).

The Sommer model is used to estimate emissions from any liquid manure (less than 10 percent dry matter) stored in housing. The estimation method for liquid manure can be found in Section 5.4.4 *Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks*.

Nitrous Oxide Emissions from Dairy Cows' Housing

To estimate nitrogen losses from housing, the amount of nitrogen excreted (N_{ex}) by each animal category is first estimated. Equation 5-5, Equation 5-6, and Equation 5-7 are the equations recommended by the American Society of Agricultural and Biological Engineers (ASABE) for estimating N_{ex}.

Equation 5-5: ASABE Approach for Estimating Nitrogen Excretion from Lactating Cows

$$\mathbf{N_{ex} = (Milk \times 2.303) + (DIM \times 0.159) + (DMI \times C_{CP} \times 70.138) + (BW \times 0.193) - 56.632}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

Milk = Milk production per animal per day (kg milk animal^{-1} day^{-1})

DIM = Days in milk (days)

DMI = Dry matter intake (kg animal^{-1} day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed)$^{-1}$)

BW = Average live body weight (kg)

Equation 5-6: ASABE Approach for Estimating Nitrogen Excretion from Dry Cows

$$\mathbf{N_{ex} = (DMI \times 12.747) + (C_{CP} \times 1606.290) - 117.500}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1}day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed) $^{-1}$)

Equation 5-7: ASABE Approach for Estimating Nitrogen Excretion from Heifers

$$\mathbf{N_{ex} = (DMI \times C_{CP} \times 78.390) + 51.350}$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI = Dry matter intake (kg dry feed animal^{-1}day^{-1})

C_{CP} = Concentration of crude protein of total ration (g crude protein (g dry feed) $^{-1}$)

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex}, Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities and animal species as a fraction of N_{ex} (see Table 5-8). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-8: Typical Ammonia Losses from Dairy Housing Facilities (Percent of N_{ex})

Facility Description	% Loss	Facility Description	% Loss
Open dirt lots (cool, humid region)	15 - 30	Roofed facility (shallow pit under floor)	10 - 20
Open dirt lots (hot, arid region)	30 - 45	Roofed facility (bedded pack)	20 - 40
Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	5 - 15	Roofed facility (deep pit under floor - includes storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

N_2O is lost from the excreted nitrogen. A quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Greenhouse Gas Inventory (Equation 5-8). This estimation method is the same as the method present in the *Temporary Stack and Long-Term Stockpile* and the

Composting sections (See Sections 5.4.1 and 0). This equation will over-estimate the emissions from animal housing if some of the nitrogen excreted is managed outside of housing (i.e., the equation accounts for nitrogen loss due to NH_3 emissions but does not account for the quantity of nitrogen that is managed in manure management systems).

Equation 5-8: IPCC Tier 2 Approach for Estimating N_2O Emissions from Housing

$$\mathbf{E_{N_2O,housing} = n \times N_{ex} \times (1 - \%\ NH_3\ loss/100) \times EF_{N2O} \times \frac{44}{28} \times \frac{1}{1000}}$$

Where:

$E_{N2O,\ housing}$	= Nitrous oxide emissions from housing per day (kg N_2O day^{-1})
N	= Number of head of livestock species (animal)
N_{ex}	= Total nitrogen excretion per animal per day (g N $animal^{-1}$ day^{-1})
$\%NH_3$ loss	= Percent of N_{ex} lost as NH_3 in animal housing - see Table 5-8
EF_{N2O}	= N_2O emission factor for manure in housing (kg N_2O-N kg N^{-1})
$\frac{44}{28}$	= Conversion of N_2O-N emissions to N_2O emissions
$\frac{1}{1000}$	= Conversion of grams to kilograms

For manure in deep pits, dry-lots, or mixed with bedding, the emission factors are provided in Table 5-9. The N_2O emission factors for manure in housing that is stored in a stockpile are provided in Table 5-23. The emission factors for manure that is composted within a housing area are provided in Table 5-25.

Table 5-9: N_2O Emission Factors for Manure Stored in Housing

Category	N_2O Emission Factor (kg N_2O-N/ kg N)
Cattle and Swine Deep Bedding (Active Mix)	0.07
Cattle and Swine Deep Bedding (No Mix)	0.01
Pit Storage Below Animal Confinements	0.002
Dry-Lot	0.02

Source: IPCC (2006).

The remaining nitrogen excreted that is not lost as N_2O or volatilized as NH_3 in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure-to-manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

The DairyGEM Model provides daily estimates; users can refer to that model for a more in-depth analysis of their emissions.

Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment

$$\mathbf{TN_{storage} = n \times N_{ex} \times (1 - \%\ NH_3\ loss/100) \times \frac{1}{1000}}$$

Where:

$TN_{storage}$	= Total nitrogen entering manure storage (kg N day^{-1})
N	= Number of head of livestock species (animal)
N_{ex}	= Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})
$\%NH_3$ loss	= Percent of N_{ex} lost as NH_3 in animal housing - see Table 5-8
$\frac{1}{1000}$	= Conversion of grams to kilograms

5.3.1.2 Rationale for Selected Method for Estimating Emissions from Dairy Production Systems

There are a variety of methods and models available to estimate emissions from dairy production systems, ranging from simple carbon footprint models to highly complex process-based models for the determination of NH_3 and GHG emissions. The IPCC Tier 1 methodology provides a simplistic method used for country inventory purposes. When additional data are available, there are a series of equations that can be used to develop IPCC Tier 2 estimates. The data used for these estimates are typically easily obtainable from the production facility or available in a lookup table. While these methods provide estimates for emissions that may be suitable for a rough determination of emissions inventories, they are in some cases based on very limited data and may not be very representative of emissions at the farm level. The development of process-based models has provided a way to obtain a more detailed analysis of emissions at the farm scale.

A wide variety of models applicable to dairy production facilities were identified and evaluated, including: Carbon Accounting for Land Managers; Climate Friendly Food Carbon Calculator; Cool Farm Tool; CPLAN; DairyGEM; Dairy Wise; Farming Enterprise GHG Calculator; Farm GHG; Holos; Integrated Farm System Model (IFSM); Manure And Nutrient Reduction Estimator (MANURE); Manure DeNitrification-DeComposition (Manure DNDC); OVERSEER; and SIMS Dairy.

These models were evaluated to determine their suitability for use to determine emissions estimates for dairy production facilities in the United States. Eleven criteria were used to identify models that could be used to estimate CH_4 from enteric CH_4 production and CH_4, N_2O, and NH_3 from animal housing systems. Two of the criteria were considered critical: the model had to be relevant to U.S. climate and dairy production systems and it had to be publically available. If the models met these two criteria they were further ranked based on the remaining nine criteria. Four of the models considered met the critical criteria: DairyGEM, IFSM, Cool Farm Tool, and MANURE. Although DairyGEM is a subset of IFSM, it was included separately because DairyGEM only estimates emissions from the animal housing and manure storage area. Therefore, it is less cumbersome to use and requires fewer inputs.

Model Evaluation Criteria for Dairy Production Systems

1. The model is based on well-established, scientifically sound relationships among farm management inputs, emissions outputs (process-based/mass-balance model preferable).
2. The model is relevant to U.S. climate and dairy production systems.
3. The model can estimate CH_4, and N_2O, and NH_3 emissions from dairy housing systems (including enteric CH_4 production).
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management).
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates.
6. Model emission estimates for both enteric CH_4 production and emissions associated with cattle housing are easily captured.
7. The model includes some mitigation strategies for reducing emissions and produces realistic changes in emissions values when these changes are made within the production system.
8. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates.
9. The model has been tested/validated with on-farm data.
10. The model works reliably (little to no errors or program crashes).
11. The model is publicly available and accessible.

Out of these four models, DairyGEM had the most flexibility for describing the production system and met all of the specified criteria. In addition, this model implements emission estimate methodologies that are advanced beyond the IPCC Tier 2 determinations. It models CH_4 emissions from enteric fermentation and manure management and the nitrogen balance associated with nitrogen excreted in manure. The underlying methods in the DairyGEM model are recommended for determining CH_4 emissions from enteric fermentation and housing systems for dairy cattle (see further discussion in Appendix 5-E, Table 5-E-1, and subsequent relevant text). The estimates generated from this model could then be modified to account for mitigation strategies that could alter the emissions currently being generated on-farm. Some mitigation strategies are already embedded in the model, such as alternative feeding, manure handling/storage, and the use of bovine somatotropin, while others could be used by developing a table with modifiers based on literature values to determine how on-farm emissions could change with the implementation of these strategies. For N_2O emissions, a nitrogen balance approach (based on the concepts in DairyGEM) using nitrogen excretion equations from ASABE Standard D384.2 is recommended. The use of the ASABE equations takes into account the impact of dietary changes on nitrogen excretion.

5.3.2 Enteric Fermentation and Housing Emissions from Beef Production Systems

Because of differences in the diets, animal physiological state and age, and manure handling, the proportions and sources of GHGs differ among the cow-calf, stocker, and finishing segments of the beef cattle industry. A primary source of GHGs from the beef cattle industry is enteric CH_4, produced primarily in the rumen, although some CH_4 is also produced in the lower gut. In addition, CH_4 and N_2O may be produced from feces and urine on pastures and feedlot pen surfaces. Emissions from housing and manure handling (prior to entering a management system) are discussed, and equations for stockpiled manure (Section 5.4) can be applied for emission estimation.

Phetteplace et al. (2001) estimated GHG emissions from simulated beef and dairy[4] systems in the United States using modifications of the IPCC (1997) methodology. The systems were comprised of a base herd of mature cows plus calves and replacements, stocker calves, a feedlot, and a dairy with 100 lactating cows. They also evaluated emissions from calves that went through the entire cow-calf, stocker and feedlot system (cow-calf to feedlot). Greenhouse gas emissions head^{-1} (CO_2-eq) from Phetteplace et al. (2001) are presented in Table 5-10 (with the exception of the dairy herd).

Table 5-10: Simulated GHG Emissions for Ruminant Systems (kg CO_2-eq/head/year)

Item	Cow-calf	Stocker	Feedlot	Cow-calf Through Feedlot
Dietary TDN, %	62	57	88	62
GHG (kg CO_2-eq/head/year)				
Enteric CH_4	1,140	1,725	743	1,167
Manure CH_4	34	48	12	34
Total CH_4	1,175	1,773	755	1,201
N_2O	1,487	1,721	1,294	1,490
CO_2	127	380	1,245	252
Total CO_2-eq	2,788	3,874	3,294	2,944

Source: Phetteplace et al. (2001).

Elsewhere, Beauchemin et al. (2010) used the Holos model (Little et al., 2008) to conduct a life-cycle assessment of beef production in western Canada. Of total CO_2-eq, 63 percent was from enteric CH_4.[5] These are very similar to values reported by the U.S. Department of Agriculture (2004b). Sixty-one percent of CO_2-eq emissions were from the cow-calf herd, 19 percent were from replacement heifers, eight percent were from backgrounding operations, and 12 percent were from feedlots. Seventy nine percent of enteric CH_4 losses were from the cow herd, three percent from bulls, two percent from calves, seven percent from backgrounders, and nine percent from feedlots. N_2O contributions (CO_2-eq) as a percent of total GHG emissions were as follows: feedlot manure – two percent, feedlot soil – two percent, cow-calf herd soil – two percent, and cow-calf herd manure – 20 percent.

Cow-Calf and Bulls

There is no evidence that any breed of beef cow produces less enteric CH_4 than another. There are a few reports suggesting that efficient cattle (those selected for feed efficiency or residual feed intake (RFI)) may produce less enteric CH_4 (Nkrumah et al., 2006; Hegarty et al., 2007). However, Freetly and Brown-Brandl (2013) reported that cattle with greater feed efficiency actually produced more CH_4; thus raising some questions about the genetic factors associated with feed efficiency and CH_4 emissions. It is unclear whether the changes observed are a result of altered feed intake or are associated with a change in altered ruminal microbial population. Additionally, recent information indicates that there is an interaction between diet quality and feed efficiency on enteric CH_4 emissions, where efficient cows produce less CH_4 when grazing high-quality pasture but not when grazing poor-quality forage (Jones et al., 2011). Residual feed intake is moderately heritable—(0.28 to 0.58; Moore et al., 2009), thus it might be possible to genetically select for animals with lower enteric CH_4 production. An examination of the value for selection for low enteric CH_4 production has been conducted with sheep in New Zealand and Australia. Simulations using published data

[4] Discussion of emissions from dairy production systems can be found in Section 5.3.1.

[5] 5% of emissions were from manure CH_4, 23% from manure N_2O, 4% from soil N_2O, and 5% from energy CO_2.

indicate that without accurate feed intake information and a method by which many animals can be screened for CH_4 emissions, selection for lower enteric CH_4 production is not likely to be economically viable (Cottle et al., 2011).

Measurement of enteric CH_4 from grazing cattle has been conducted primarily from animals grazing improved pastures using micrometeorological methods and tracer techniques. Lassey (2007) summarized much of the CH_4 emissions data that had been collected using the SF_6 tracer technique. Intake was either calculated from a requirements model or from use of markers (Cr_2O_3 or Yb_2O_3). Estimated forage digestibility (in vitro) ranged from 48.7 to 83 percent, which resulted in estimated CH_4 conversion factors [i.e., enteric CH_4 as a percentage of gross energy intake (GEI)] ranging from 3.7 to 9.5 percent. The mean Ym from all of the studies was 6.25 and agrees reasonably well with that used by IPCC (2006) for cattle on pasture. Methane emissions from cows grazing improved pasture, Kentucky fescue, and Bermuda grass in the southern United States were reported by Pavao-Zuckerman et al. (1999) and DeRamus et al. (2003). In both of these studies significant reductions in enteric CH_4 unit^{-1} of animal weight gain resulted from the implementation of best management practices designed to improve pasture quality.

Enteric emissions estimates can be made using micrometeorological methods and tracer techniques. One report in which CH_4 emissions were measured from beef cows grazing native range in October and May illustrated a large variation in enteric emissions. In October, when cows were losing BW, they produced 87 g CH_4 head daily^{-1}, and on the same pasture in May they produced 252 g CH_4 head daily^{-1} (Olson et al., 2000). Westberg et al. (2001) measured CH_4 from cows grazing the same pasture across seasons and found similar results, with higher CH_4 emissions from cows grazing lush spring growth and the lowest emissions from grazing stockpiled fall pasture. These differences are attributable to differences in both DMI and forage quality. In general, as forage quality increases, DMI also increases. Some "rules of thumb" for DMI on pasture include the following:

- Poor quality pasture - DMI = 1 to 1.75 percent of body weight;
- Medium quality forage - DMI = 1.75 to 2.25 percent of body weight;
- High quality forage DMI = 2.25 to 3 percent of body weight.

Stockers

Enteric CH_4 emissions of stockers while grazing have been measured by Laubach et al. (2008), Tomkins et al. (2011), McGinn et al. (2011), and Boadi et al. (2002), using a variety of techniques including the SF_6 tracer, and several micrometeorological approaches. The same factors that affect CH_4 emissions from grazing beef cows are important in stocker cattle. Those factors are level of feed intake, digestibility of forage consumed, supplementation, and the chemical composition of the plants consumed. Enteric emissions estimates can be made using micrometeorological methods or, tracer techniques or can be predicted from IPCC Tier 2 methods (see enteric discussion). Critical variables include measurements or estimations of feed intake and feed quality (chemical composition). Many of the equations currently available may not accurately predict measured enteric emissions from grazing cattle (Tomkins et al., 2011).

Feedlot

Most estimates of enteric methane emission from finishing beef cattle are based on work using animals confined to respiration chambers, although a few studies have used micrometeorological methods in open feedlots. Enteric CH_4 losses from finishing beef cattle normally range from 50 to 200 L head^{-1} daily (Johnson and Johnson, 1995; McGinn et al., 2004; Beauchemin et al., 2008; Loh et al., 2008; Hales et al., 2012; 2013; Hales et al., 2014; Todd et al., 2014a; Todd et al., 2014b). In most studies in the U.S., diets have been based on DRC or SFC; whereas most studies in Canada the diets are based on barley. The IPCC Tier 2 (2006) enteric CH_4 conversion factor (Ym) for feedlot cattle is

3 ± 1 percent of GEI. There are few studies that have measured emissions of CH_4 and N_2O from feedlot pen surfaces and runoff control structures. The primary factors that control enteric methane emissions in feedlot cattle are feed intake, grain type, grain processing method, dietary roughage concentration and characteristics, and dietary fat concentration.

5.3.2.1 Method for Estimating Emissions from Beef Production Systems

Method for Estimating CH_4 Emissions from Enteric Fermentation in Beef Cattle

- IPCC Tier 2 approach, with some adjustment factors, based on diet, animal weight, pregnancy/lactation, activity (IPCC, 2006).
- Data sources are user inputs on dietary intake, lactation and pregnancy rates, animal weight, housing, and the Feedstuffs Composition Table (Ewan, 1989; Preston, 2013).
- Although the equations utilized are the same as existing inventory methods, the method takes into account a large database of feed types (found in Appendix 5-B, Feedstuff Composition Table), as well as reporting at the monthly, rather than annual, temporal scale.

Equation 5-10: IPCC Tier 2 Equation for Calculating Gross Energy Requirements for Beef Cattle

$$GE = \left[\frac{\left(\frac{NE_m + NE_a + NE_l + NE_{work} + NE_p}{REM}\right) + \left(\frac{NE_g}{REG}\right)}{\frac{DE\%}{100}}\right]$$

Where:

GE = Gross energy (MJ day^{-1})

NE_m = Net energy required by the animal for maintenance (MJ day^{-1})[a]

NE_a = Net energy for animal activity (MJ day^{-1})[b]

NE_l = Net energy for lactation (MJ day^{-1})[c]

NE_{work} = Net energy for work (MJ day^{-1})[d]

NE_p = Net energy required for pregnancy (MJ day^{-1})[e]

REM = Ratio of net energy available in a diet for maintenance to digestible energy consumed[f]

NE_g = Net energy needed for growth (MJ day^{-1})[g]

REG = Ratio of net energy available for growth in a diet to digestible energy consumed[h]

DE = Digestible energy expressed as a percent of gross energy (%)

[a] Calculated using Eqn10.3 in IPCC (2006) based on body weight ("BW").
[b] Calculated using Eqn 10.4 in IPCC (2006) based on NE_a and feeding situation.
[c] Calculated using Eqn 10.8 in IPCC (2006) based on milk production ("milk") and milk fat ("fat").
[d] Calculated using Eqn 10.11 in IPCC (2006) based on information on daily hours of work ("work").
[e] Calculated using Eqn 10.13 in IPCC (2006) based on NE_m and pregnancy status.
[f] Calculated using Eqn 10.14 in IPCC (2006) based on DE.
[g] Calculated using Eqn 10.13 in IPCC (2006) based on body weight, mature weight ("MW"), and daily weight gain ("WG").
[h] Calculated using Eqn 10.15 in IPCC (2006) based on DE.

Equation 5-11: IPCC Tier 2 Equation for Calculating Enteric CH_4 Emissions from Beef Cattle

$$\mathbf{DayEmit} = \frac{\mathbf{GE \times Y_m/100}}{\mathbf{55.65}}$$

Where:

DayEmit = Emission factor (kg CH_4 head^{-1} day^{-1})

GE = Gross energy intake (MJ head^{-1} day^{-1})

Y_m = CH_4 conversion factor, which is the fraction of GE in feed converted to CH_4 (%)

55.65 = A factor for the energy content of methane (MJ kg $CH_4$$^{-1}$)

The DE ultimately used in the IPCC Tier 2 equation (in Equation 5-11) will be weighted based on portion of total feed intake from a particular feed type. The DE data for particular feedstuffs can be found in Appendix 5-B. The recommended Ym for beef replacement heifers, steer stockers, heifer stockers, beef cows, and bulls is 6.5 percent for all regions of the country. For feedlot cattle, the IPCC (2006) Ym of 3 percent is adjusted based on diets. All feedlot cattle initially start with a baseline Ym of three percent (IPCC, 2006).The correction factors to Ym for feedlot cattle for different scenarios are provided below (see Appendix 5-A for additional details). The Ym used for calculating emissions for these cattle is modified based on animal diets, as indicated in Table 5-11.

Table 5-11: Determination of Adjusted Methane Conversion Factor (Ym) for Feedlot Cattle

Variable	Ym (as a % of GEI)
Baseline Ym (IPCC, 2006)	**3%**
Ionophore in diet (Tedeschi et al., 2003; Guan et al., 2006):	
▪ Yes	No change
▪ No	Increase Ym by 4% (Ym= 3% x 1.04= 3.12% of GEI)
Fat Content (Zinn and Shen, 1996; Beauchemin et al., 2008; Martin et al., 2010) (For each percent of added fat (as supplemental fat or in byproducts such as distillers grain that contain about 10 percent fat), decrease by four percent to a maximum of a 16 percent decrease)	
▪ 1% supplemental fat	Decrease Ym by 4% (Ym = 3% x 0.96 = 2.88%)
▪ 2% supplemental fat	Decrease Ym by 8% (Ym = 3% x 0.92 = 2.76%)
▪ Four or higher added fat content	Decrease Ym by 16% (Ym= 3% x 0.84=2.52%)
Grain Type (Beauchemin and McGinn, 2005; Archibeque et al., 2006; Hales et al., 2012):	
▪ Grain in animal diet is steam flaked (SF) or high moisture (HM)	No Change
▪ Grain in animal diet is unprocessed (UP) or dry rolled (DR)	Increase Ym 20% (Ym = 3% x 1.2 = 3.6%)
▪ Grain in diet is barley rather than corn or sorghum	Increase Ym 30% (Ym = 3% x 1.3 = 3.9)
Grain Concentration (see Appendix 5-A for details and references):	
▪ Diet contains more than 60 percent grain	No Change
▪ Diet contains 45 to 60 percent grain	Increase Ym 10% (Ym= 3% x 1.1 = 3.3%)
▪ Diet contains less than 45 percent grain	Increase Ym 40% (Ym = 3% x 1.40 = 4.2%)

Method for Estimating Beef Cattle GHG Emissions from Housing

Methane

- The IPCC (2006) Tier 2 method can be used to estimate CH_4 emissions when manure is allowed to accumulate on feedlot pen surfaces as described below.

Nitrous Oxide

- Nitrogen excreted estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O emissions from manure in housing.

Emissions from Feedlot Pen Surfaces

There are few, if any, studies that have measured CH_4 or N_2O emissions from beef cattle feedlot pen surfaces and retention ponds. The study of Todd et al. (2014a; 2014b) suggests there is little CH_4 production from feedlot pen surfaces. The use of the IPCC (2006) methodologies is recommended to estimate emissions from feedlot pens and retention ponds.

In order to estimate CH_4 emissions from beef feedlot pen surfaces, the quantity of volatile solids excreted is first estimated. These can be estimated by lab testing samples from the facility or using values from the ASABE Standard D384.2 (ASABE, 2005).[6] CH_4 emissions from the pen surface can be estimated using the IPCC (2006) Tier 2 approach as outlined in section 5.4.1.2. For cattle feedlots, a maximum CH_4 production capacity (B_0) of 0.33 m^3/ kg volatile solids is assumed (Table 5-19) and the CH_4 conversion factor for pen surfaces ranges from 1 to 2 percent of B_0, depending upon environmental temperature (Table 5-20). Once manure is scraped from the pens and removed, the methods described in section 5.4.1 can be used to estimate CH_4 emission from manure that is composted or stored in stockpiles.

In order to estimate N_2O emissions from the pen surfaces of beef feedlots the quantity of nitrogen excreted on to the pen surface must be known. This can be estimated using Equation 5-12 from the ASABE Standard D384.2. For a beef feedlot, a default value of 0.069 kg of N kg dry manure^{-1} can be used if N_{ex} is not calculated.

[6] Volatile solids values can be estimated from equations (1) or (2) in section 4.3.1 of ASABE D384.2. Default volatile solids values are also presented in Table 5-32 of this document.

Equation 5-12: ASABE Approach for Estimating Nitrogen Excretion from Beef Cattle

$$\mathbf{N_{ex}} = \sum_{x=1}^{n} \frac{\mathbf{DMI_x \times C_{CP-x} \times DOF_x}}{\mathbf{6.25}} - [\mathbf{41.2 \times (BW_F - BW_I)}]$$
$$+ \left[\mathbf{0.243 \times DOF_T} \times \left[\frac{\mathbf{BW_F + BW_I}}{\mathbf{2}}\right]^{\mathbf{0.75}} \times \left[\frac{\mathbf{SRW}}{\mathbf{BW_F \times 0.96}}\right]^{\mathbf{0.75}}\right.$$
$$\left. \times \left[\frac{\mathbf{BW_F - BW_I}}{\mathbf{DOF_T}}\right]^{\mathbf{1.097}}\right]$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

DMI_x = Dry Matter Intake for ration x (kg dry feed animal^{-1}day^{-1})

C_{CP-x} = Concentration of crude protein of total ration (g crude protein g dry feed^{-1})

DOF_x = Days on feed for an individual ration (days)

BW_F = Live body weight at finish of feeding period (kg)

BW_I = Live body weight at the start of feeding period (kg)

DOF_T = Total days on feed from start to finish of feeding periods (days)

SRW = Standard reference weight for expected final body fat (kg)

x = Ration number

n = Total number of rations fed

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex}. Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities as a fraction of N_{ex} (see Table 5-12). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-12: Typical Ammonia Losses from Beef Cattle Housing Facilities Expressed as a Percent of N_{ex}

Facility Description	% Loss	Facility Description	% Loss
Open dirt lots (cool, humid region)	30 – 45	Roofed facility (bedded pack)	20 - 40
Open dirt lots (hot, arid region)	40 – 60	Roofed facility (deep pit under floor, including storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

An alternative approach is to use the equation of Todd et al. (2013) which calculates monthly feedlot NH_3 emissions as a function of dietary crude protein and average monthly temperature (Equation 5-13).

Equation 5-13: Monthly Beef Feedlot NH_3 Emissions as a Function of Dietary Crude Protein and Monthly Temperature

$$\mathbf{Ln(NH_3) = 8.82 - 1629\left[\frac{1}{T}\right] + 0.108 \times CP}$$

Where:

NH_3 = NH_3 emission from housing per day (g NH_3 head^{-1} day^{-1})

T = Average monthly temperature (K)

CP = Dietary crude protein as a fraction of dry matter (%)

N_2O emissions are calculated using the IPCC (2006) Tier 2 method and dry-lot emission factors described in Equation 5-8 and Table 5-9. The quantity of nitrogen that leaves the feedlot pens in manure can then be calculated using Equation 5-9. N_2O-N losses from manure collected and removed from the pens can be determined from manure nitrogen using Equation 5-27 and Equation 5-29 and the emission factors in Table 5-23 and Table 5-25 found in Section 5.4 Manure Management. NH_3 losses from manure nitrogen removed from the pens can be calculated as described in Appendix 5-C.1 and 5-C.3.

5.3.2.2 Rationale for Selected Method for Estimating Emissions from Beef Production Systems

Cow-Calf, Bulls, and Stockers

The most appropriate predictions available for entity scale estimation are IPCC Tier 2 methods for grazing cattle. Critical variables that are important to define in order to generate prediction methods include measurements or estimations of feed intake and feed quality (chemical composition) for pasture or rangelands. If the intake is not known, intake prediction equations/models such as NRC (2000) can be used. The NRC (2000) provides an equation for the calculation of DMI for grazing beef cows and for stocker cattle: NEm intake = $SBW^{0.75}$ * (0.04997 * NEm^2 + 0.04631) where NEm is the estimated Mcal kg^{-1} of the pasture, and SBW is the average shrunk body weight for the period of grazing (kg). The requirement for knowledge of the NEm concentration of the pasture may limit the usefulness of the prediction in some situations.

In situations in which the herd is housed in a dry-lot or barn facility, emission factors for CH_4 and N_2O associated with pen surfaces, manure storage, and animal movement/manure disturbance would be appropriate.

Feedlot

Ellis et al., (2009) reported that several equations appeared to be good predictors of enteric CH_4 losses from feedlot cattle based on Canadian studies. However, many of those equations tend to greatly overestimate enteric losses when compared with data from cattle fed a typical southern plains finishing diet (Hales et al., 2012; 2013; Todd et al., 2014a; Todd et al., 2014b). Although Kebreab et al. (2008) reported that MOLLY and IPCC Tier 2 (2006) gave predicted values similar to measured values with feedlot cattle, there was a large variability in individual animals with errors of 75 percent or greater. Kebreab et al. (2008) noted the average Ym (MJ enteric CH_4 MJ GEI^{-1}) for feedlot cattle based on experimental data was 3.88 percent of GEI (range 3.36 to 4.56), which was higher than the IPCC (2006) value of 3.0 percent and the recently obtained values with typical finishing diets of 2.85 percent (Hales et al., 2012; 2013).

Currently, IPCC Tier 2 may be the most useful methodology for prediction of enteric emissions from feedlot beef cattle. Unfortunately, the Tier 2 method does not allow for estimating changes in enteric emissions related to changes in diet or management.

Therefore, a modified IPCC (2006) method is recommended to estimate enteric CH_4 emissions from beef cattle fed high concentrate finishing diets. The CH_4 conversion factor (Ym) will be adjusted by factors in the animals' diets as described in Section 5.3.2.1. A baseline scenario based on typical U.S. beef cattle feeding conditions is established, and the Ym values are adjusted based on published research. Emission values are modified using correction factors that are based on changes in animal management and feeding conditions from the baseline scenario.

5.3.3 Enteric Fermentation and Housing Emissions from Sheep

GHG emissions associated with sheep production include enteric CH_4 emissions, manure and bedding emissions, and emissions associated with grazing and manure application to land.

The New Zealand Ministry for the Environment (2010) estimated that sheep younger than a year of age emit 5.1 percent of GEI as enteric CH_4, and adult sheep emit 6.3 percent of their GEI as CH_4. These emission factors, when combined with population estimates, result in baseline enteric emissions of 11.60 kg CH_4 head^{-1} year^{-1}. Sheep are also estimated to deposit 15.9 kg N head^{-1} year^{-1}.

Lassey (2007) summarized the enteric emissions measurements from grazing sheep trials from New Zealand and Australia in which the SF_6 tracer technique was used. Forage characteristics ranged from lush (in vitro digestibility estimate of 82 percent) to poor quality (called "dead," with an in vitro digestibility of 54 percent). Intake was measured using complete fecal collection or a marker (n-alkane). Enteric CH_4 emissions ranged from 11.7 g day^{-1} for sheep fed forage of higher quality (6.9 percent of GEI) to 35.2 g day^{-1} for sheep fed forage of lower quality (6.3 percent of GEI). The average enteric emissions were 5.39 percent of GEI, or 23.5 g day^{-1}. In general, lower forage quality resulted in a greater amount of CH_4 emitted as a proportion of the energy intake than did higher forage quality.

New Zealand pastures grazed by sheep had elevated N_2O emissions (7.4 g N_2O-N ha^{-1} day^{-1} vs. 3.4g N_2O-N ha^{-1} day^{-1}) compared with control, but significantly less than that observed when cattle grazed (32.0 g N_2O-N ha^{-1} day^{-1}) (Saggar et al., 2007). The data were used to evaluate the NZ-DNDC model, a process-based New Zealand whole farm model. To our knowledge there are no published estimates of GHG emission from sheep manure systems.

5.3.3.1 Method for Estimating Emissions from Sheep

Method for Estimating Enteric Fermentation CH_4 Emissions from Sheep

- Howden equation (Howden et al., 1994), based on dietary DMI.
- The equation from Howden et al. (1994) estimates emissions based solely on DMI; hence, emission factors not utilized.

Equation 5-14: Equation for Enteric Fermentation Emissions from Sheep (Howden et al., 1994)

$$\mathbf{CH_4 = Intake \times 0.0188 + 0.00158}$$

Where:

CH_4	= Methane emissions (kg CH_4 head^{-1} day^{-1})
Intake	= Dry Matter Intake (kg head^{-1} day^{-1})

The dry matter data for particular feedstuffs can be obtained from Appendix 5-B.

No emissions estimation methods have been provided for housing as most sheep are kept on pasture and minimal emissions are expected.

5.3.3.2 Rationale for Selecting Method for Estimating Emissions from Sheep

Howden et al. (1994) generated an equation from which to predict CH_4 emissions from sheep. Equation 5-7 resulted from a linear extrapolation of DMI to emissions. It has since been evaluated and found to be robust enough to be the equation used in the Australian National Greenhouse Gas Inventory. Klein and Wright (2006) measured CH_4 from sheep in respiration chambers and compared their results to the Howden et al. (1994) equation. Actual CH_4 averaged 1.1 g head^{-1} (SE ± 0.05) and predicted CH_4 was 1.1 g head^{-1} (SE ± 0.02). A potential concern regarding the Howden equation is that much of the data included in the analysis was based on tropical forages. Nonetheless, when intake data are available, the Howden equation presents the best method by which to estimate sheep enteric emissions. When intake is not available, the IPCC Tier 2 method of estimation should be used. Emissions from feedlot sheep should use the Ym values from Blaxter and Clapperton's original paper (1965) in which they measured CH_4 emissions from sheep with respiration calorimetry chambers. Sheep fed highly digestible diets at three times maintenance produced 35 percent less CH_4 (kcal 100K kcal of feed energy^{-1}) than those fed similar diets at maintenance; thus, a reduced Ym value is warranted. The equation is $CH_4 = 1.3 + [0.112 \times (\%\ digestibility/100)] + [ME\ intake/maintenance\ ME\ requirement] \times [2.37 - 0.050 \times (\%digestibility/100)]$.

5.3.4 Enteric Fermentation and Housing Emissions from Swine Production Systems

Sources of GHG emissions include enteric fermentation; manure stored within the animal housing, whether it is stored as a liquid or mixed with bedding; emissions that occur during the transport of manure to an external manure storage structure; the outside manure storage structure; emissions that occur during transport of manure to the field; and emissions following land application of manure. Because GHG mitigation has not been a focus of U.S. research for the swine industry nor a high priority for swine producers, data are not readily available to identify the magnitude of each of the above points of emission within a farm. However, emissions of CH_4 are expected to occur primarily during manure storage, and emissions of N_2O are expected to predominate following land application of manure.[7] Often manure is stored underneath the pig housing in a deep pit. For this reason, emissions discussion in this section includes in-house manure storage and comparison of in-house manure storage systems with systems that store manure externally. Because swine feeds are dry, emissions of GHG from feed storage areas are believed to be negligible.

[7] Greenhouse gas emissions resulting following land application are addressed separately in the sections on Chapter 3: Croplands and Grazing Lands.

Greenhouse gas emission data from swine facilities is somewhat limited. Liu et al. (2011a) reported that grow/finish pigs emitted 42 to 79 mg CH_4 kg BW^{-1} daily from chambers where pigs were housed with manure. Daily emissions of N_2O ranged from 11.4 to 12.4 mg N_2O kg BW^{-1} (Li et al., 2011). These values are somewhat higher than data used by Verge et al. (2009) in calculating GHG emissions from Canadian pork production (43 mg CH_4 kg BW^{-1} and 4 mg N_2O kg BW^{-1}). Philippe et al. (2007) observed GHG emissions in the range reported by Li et al. (2011) though their observations were in European deep litter and slatted floor systems. The reported gaseous emissions from pigs raised on the slatted floor and on the deep litter were, respectively, 0.54 and 1.11 g pig^{-1} day^{-1} for N_2O, and 16.3 and 16.0 g pig^{-1} day^{-1} for CH_4.

Liu et al. (2011a) conducted a meta-analysis to identify factors that contribute to GHG emissions from swine production. Findings, shown in Table 5-13, illustrate that type of emission source (swine buildings or manure storage facilities) was not significant for CH_4 and N_2O emissions. Swine category (stage of production) and geographic location was significant for both of the GHG gases. Neither temperature nor size of operation was significant in the overall analysis.

Table 5-13: P Values of Main Effects on GHG Emissions from Swine Operations

Cause of Variation	CH_4 (n=76)	N_2O (n=53)
Emission source	0.94	0.93
Swine category	0.05	<0.01
Geographic region	0.04	0.02
Temperature	0.20	0.95
Size of operation	0.89	0.24

Source: Liu et al. (2011a).

Within the meta-analysis, Liu et al. (2011a) found that swine buildings with straw-flow systems generated the lowest CH_4 and N_2O emissions of systems compared, while pit systems generated the highest CH_4 emissions, and bedding systems generated the highest N_2O emissions. Emissions from lagoons and slurry storage basin/tanks were compared; lagoons generated significantly higher N_2O emissions than slurry storage basin/tanks, while CH_4 emissions were not different. Straw-based bedding resulted in numerically higher CH_4 but lower N_2O emissions when compared with sawdust or corn stalk bedding systems. Liu et al. (2011a) observed an increasing trend for CH_4 emissions as manure removal frequency decreased ($P = 0.13$). Deep pits and pits flushed using lagoon effluent also generated relatively high CH_4 emissions. Results for N_2O emissions showed very high uncertainties ($P = 0.49$). Deep pits and pits with manure removed every three or four months had relatively higher N_2O emissions. A summary of other findings from the meta-analysis conducted by Liu et al. (2011a) showed that CH_4 emissions from slurry storage facilities without covers were significantly higher than from those with covers.

The highest CH_4 emissions were from farrowing swine, and were significantly higher than those from finishing and nursery swine. Compared with farrowing swine, the gestating swine had significantly lower CH_4 emissions. The highest N_2O emissions were from gestating swine and were significantly higher than those from finishing swine.

North American studies reported significantly higher CH_4 emissions from swine operations than European and Asian studies (Liu et al., 2011a) . This is probably due to the different prevailing manure handling systems and different manure handling practices in different regions. Emissions of CH_4 from lagoons and manure storage facilities increased with increasing temperature. For swine buildings, temperature was not a significant factor.

5.3.4.1 Method for Estimating Emissions from Swine Production Systems

Method for Estimating Enteric Fermentation CH_4 Emissions from Swine

- IPCC Tier 1 approach, using U.S. emission factor of 1.5 kg CH_4/head/year. (IPCC, 2006).
- Sole data source is the IPCC Tier I emission factor for swine. User input is total number of head, regardless of class or weight.

Equation 5-15: Equation for Enteric Fermentation Emissions from Swine (IPCC, 2006)

$$\mathbf{CH_4 = Population \times 0.00411}$$

Where:

CH_4	= Methane emissions per day (kg CH_4 day^{-1})
Population	= Number of swine (head)
0.00411	= Daily CH_4 emissions from each animal (kg head^{-1} day^{-1})

Method for Estimating Swine GHG Emissions from Housing

Methane

- The IPCC (2006) Tier 2 method is used to estimate CH_4 emissions when manure is allowed to accumulate below the animal confinement as described below.

Nitrous Oxide

- Nitrogen intake, retention, and excretion estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O from manure in housing.

Methane Emissions from Swine Housing

The IPCC (2006) Tier 2 equation is used to estimate CH_4 emissions when manure is allowed to accumulate in a pit below the animal confinement. The estimation method is provided in Equation 5-4. The maximum CH_4 producing capacity for swine is provided in Table 5-19. The MCFs for manure stored in a deep pit or from swine bedding is provided in Table 5-7.

Nitrous Oxide Emissions from Swine Housing

To estimate nitrogen losses from swine housing, the amount of nitrogen excreted (N_{ex}) for each animal classes are first estimated. Equation 5-16 describes the relationship between nitrogen intake, retention, and excretion for swine. Equation 5-17, Equation 5-18, Equation 5-19, and Equation 5-20 provide the methods for estimating the nitrogen intake and retention for the different swine classes as recommended by the ASABE.

Equation 5-16: ASABE Approach for Estimating Nitrogen Excretion from Swine

$$\mathbf{N_{ex} = N_{intake} - N_{Retention}}$$

Where:

N_{ex} = Total nitrogen excretion per animal (g animal^{-1})

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

Equation 5-17: ASABE Approach for Estimating Nitrogen Excretion from Grow-Finish Pigs

$$\mathbf{N_{intake} = ADFI_G \times C_{CP} \times \frac{DOF_G}{625}}$$

$$\mathbf{N_{retention} = \frac{BW_F \times DP_F \times FFLP_F}{159.4} - BW_I \times [DP_F - 0.05 \times (BW_F - BW_I)] \times \frac{FFLP_F + 0.07 \times (BW_F - BW_I)}{159.4}}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_G$ = Average daily feed intake over finishing period (g day^{-1})

C_{CP} = Concentration of crude protein of total (wet) ration (%)

DOF_G = Days on feed to finish animal (grow-finish phase) (days)

BW_F = Final (market) body weight (kg)

DP_F = Average dressing percent (yield) at final weight (%)

BW_I = Initial body weight (kg)

$FFLP_F$ = Average fat-free lean percentage at final weight (%)

Equation 5-18: ASABE Approach for Estimating Nitrogen Excretion from Weaning Pigs

$$\mathbf{N_{intake}} = \mathbf{ADFI_G} \times \mathbf{C_{CP}} \times \frac{\mathbf{DOF_N}}{\mathbf{625}}$$

$$\mathbf{N_{retention}} = \mathbf{DOF_N} \times \mathbf{FFLG_G} \times \frac{\mathbf{1 + [0.137 \times (BW_{F-N} + BW_{I-N})]}}{\mathbf{125.8}}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_G$ = Average daily feed intake over finishing period (g day^{-1})

C_{CP} = Concentration of crude protein of total (wet) ration (%)

DOF_N = Days on feed to finish animal (nursery phase) (days)

$FFLG_G$ = Average fat-free lean gain from 20 to 120kg (g day^{-1}) [a]

BW_{F-N} = Final body weight in nursery phase (kg)

BW_{I-N} = Initial body weight in nursery phase (kg)

[a] Recommended values are: 350 g day^{-1} for high lean growth capacity pigs; 325 g day^{-1} for high-moderate lean growth capacity pigs; and 300 g day^{-1} for moderate lean growth capacity pigs.

Equation 5-19: ASABE Approach for Estimating Nitrogen Excretion from Gestating Sows

$$\mathbf{N_{intake}} = \mathbf{ADFI_S} \times \mathbf{C_{CP}} \times \left(\frac{\mathbf{GL}}{\mathbf{625}}\right)$$

$$\mathbf{N_{Retention}} = \mathbf{(GLTG \times 36.8) + (LITTER \times 39.1)}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_S$ = Average daily feed intake during gestation (g day^{-1})

C_{CP} = Concentration of crude protein (%)

GL = Gestation period length (days)[a]

GLTG = Gestation lean tissue gain (kg)[b]

LITTER = Number of pigs in litter (head)

[a] Assumed to be 115 days.
[b] Recommended value from ASABE is 19.205 kg.

Equation 5-20: ASABE Approach for Estimating Nitrogen Excretion from Lactating Sows

$$\mathbf{N_{intake} = ADFI_{LACT} \times C_{CP} \times \left(\frac{LL}{625}\right)}$$

$$\mathbf{N_{Retention} = (38.6 \times LLTG) + (LW_{WEAN} \times 32) - (LW_{BIRTH} \times 36.8)}$$

Where:

N_{intake} = Nitrogen intake per finished animal (g animal^{-1})

$N_{retention}$ = Nitrogen retained per finished animal (g animal^{-1})

$ADFI_{LACT}$ = Average daily feed intake during lactation (g day^{-1})

C_{CP} = Concentration of crude protein (%)

LL = Lactation length (days to weaning) (days)

LLTG = Lactation lean tissue gain (kg)[a]

L_{WEAN} = Litter weight at weaning (kg)

LW_{BIRTH} = Litter weight at birth (kg)

[a] Recommended value from ASABE is -4.20 kg.

Some of the nitrogen excreted is volatilized as NH_3, hence, the estimation of NH_3 losses is necessary to estimate N_2O emissions using a nitrogen balance approach. The NH_3 lost from manure in housing is estimated as a fraction of N_{ex} according to the ranges provided in Table 5-14. A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-14: Typical Ammonia Losses from Swine Housing Facilities (Percent of N_{ex})

Facility Description	% Loss	Facility Description	% Loss
Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	5 - 15	Roofed facility (bedded pack)	20 - 40
Roofed facility (shallow pit under floor)	10 - 20	Roofed facility (deep pit under floor - includes storage loss)	30 - 40

Source: Koelsh and Stowell (2005).

The IPCC (2006) Tier 2 approach is used for N_2O emissions from manure stored in housing. The estimation method is provided in Equation 5-8. The N_2O emission factors can be found in Table 5-9.

The remaining nitrogen excreted that is not lost as N_2O or volatilized as NH_3 in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure to manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

N_2O-N losses from manure collected and removed from housing can be determined from manure nitrogen using equations from Section 5.4 Manure Management for the appropriate manure management system. NH_3 losses from manure nitrogen removed from housing can be calculated using the methodology presented in Appendix 5-C.1 and 5-C.3.

5.3.4.2 Rationale for Selecting Method for Estimating Emissions from Swine

Miles et al. (2006) suggest that a robust model for enteric and housing emissions would include factors such as house management, animal size and age, pH, and manure moisture. Due to the current data limitations, an NH_3 and GHG estimation model should minimally include number of animals, excreta moisture content, diet protein and fiber content, and excreta pH. The challenge is that these criteria may not be readily available to the farm manager.

Liu et al. (2011a) compared literature values with IPCC values and concluded that the variation of the measured CH_4 and N_2O housing emission rates has not been adequately captured by the IPCC approaches. For CH_4 emissions, the differences between the IPCC-estimated emission rates and measured values were significantly influenced by type of emission source, geographic region, and measurement methods. Larger differences between estimated and measured CH_4 emission rates were observed in North American studies than in European studies. In North American studies, the results of meta-analysis indicated an overestimation by the IPCC approaches for CH_4 emissions from lagoons (pooled relative difference: -33.9%; 95% CI: -66.8% to -0.01%), and the discrepancy between the IPCC-estimated emissions and the measured values occurred mainly at lower temperatures. In European studies, the results indicated an overestimation of the IPCC approaches in swine buildings with pit systems. For N_2O emissions from swine operations, an overall underestimation of the IPCC approaches was observed in European studies but not in North American studies. In European studies, the pooled N_2O emission factors for swine buildings with pit systems was 1.6% (95% CI, 0.6% to 2.7%), while the IPCC default emission factor for pit systems is 0.2%. Larger uncertainties were observed for measured N_2O emissions from bedding systems and from straw flow systems.

Model Evaluation Criteria for Swine Production Systems

1. The model is based on well-established scientifically sound relationships between farm management inputs and emissions outputs (process-based model or mass-balance model preferable);
2. The model is relevant to U.S. climate and swine production systems;
3. The model can estimate CH_4, N_2O, and NH_3 emissions from enteric fermentation and swine housing systems;
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management);
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates;
6. The model includes some mitigation strategies for reducing emissions, and produces realistic changes in emissions values when these changes are made within the production system;
7. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates;
8. The model has been tested/validated with on-farm data;
9. The model works reliably (little to no errors or program crashes); and
10. The model is publicly available and accessible.

In order to consider an alternative to the IPCC approach, a wide variety of models applicable to swine production facilities were identified and evaluated, including 1) CAR Livestock, 2) Manure And Nutrient Reduction Estimator (MANURE), 3) COOL Farm Tool, 4) Carbon Accounting for Land

Managers, 5) Farming Enterprise GHG Calculator, 6) CPLAN, and 7) Holos. These models were evaluated by 10 criteria (see box) to determine their suitability for use in determining emissions estimates for swine production facilities in the United States. Two of these criteria were considered to be critical, in that if they were not met by the model, they could not be considered for use (i.e., the model had to be relevant to U.S. climate and swine production systems and had to be publicly available).

The Holos model considered diet (standard, low crude protein, or high-digestibility feeds) and manure handling options (anaerobic digestion, covered or uncovered slurry storage, deep pit, or solid storage). In addition, the Holos model provided an estimate of uncertainty for the model output. The MANURE model (WRI, 2009) collected the most comprehensive data and allowed for easy comparison of the impact of changes in manure handling and use on emissions of NH_3, N_2O (direct and indirect), and CH_4. On the animal side, MANURE based its calculations solely on animal numbers; feeding was not considered. The other models considered, while meeting minimum criteria, lacked any improvements over the IPCC approach. Consequently, the IPCC method was selected (i.e., Holos utilizes the IPCC Tier 1 approach for housing) with nitrogen excretion estimated using ASABE equations that account for diets.

5.3.5 Housing Emissions from Poultry Production Systems

Meat Birds

Greenhouse gas emissions within the farm boundary of a broiler chicken farm will originate almost exclusively from the animal housing, which also serves as the storage location for manure. Liu et al. (2011a) reported that for a 20-week grow-out of turkeys on litter, average daily N_2O emissions were 0.045 g (kg bodyweight)$^{-1}$, and daily CH_4 emissions were 0.08 g (kg bodyweight)$^{-1}$. Emission sources external to the housing include GHG emissions from farm vehicles. If a house is cleaned or decaked (removal of the top, crusted portion of the litter) and stored on the farm, GHG and NH_3 production and emissions could occur; Appendix 5-C provides further discussion on NH_3 emissions from housing. Practices to decake and the timing of land application of cake and litter vary from site to site and may or may not include further composting.

Laying Hens

Greenhouse gas emissions within the farm boundary of an egg farm may originate from the housing or the manure storage location. Emission sources external to the housing include GHG emissions from farm vehicles. External to the farm itself, GHG emissions result from land application of litter or stockpiling of the litter in fields prior to land application.

Laying hen housing systems without litter would likely exhibit greater emissions than litter systems, but comparison of estimates are sparse. Laying hen houses typically store excreta in a basement or may move excreta out of the house frequently (daily or more often); this would relocate emissions to a storage shed rather than change the cumulative emissions unless some form of processing (drying) took place prior to storage. Li et al. (2010) reported daily CH_4 emissions of 39.3 to 45.4 mg hen^{-1} and N_2O emissions of 58.6 mg hen^{-1} (hen bodyweight average = 1.9 kg) in a basement-type system. This compares to a litter system for a 20-week grow-out of turkeys where average daily N_2O emissions were 0.045 g kg^{-1} bodyweight and daily CH_4 emissions were 0.08 g kg^{-1} bodyweight (Liu et al., 2011a). Based on the comparison of these two studies, differences in GHG emissions from dry litter systems and wetter, stacked laying hen systems would be expected.

Management practices to reduce litter moisture offer the most promise for reducing emissions of CH_4 and N_2O. Quantitative estimates of how emissions vary with litter moisture are not available, but would likely follow similar dynamics as soil moisture content. Reuse of litter and decaking procedures might also be used as strategies to reduce emissions in the future. However, data are not available at present to use as part of a systems model.

Ammonia Emissions in Poultry Housing

As mentioned earlier, ammonia is not a greenhouse gas, however, ammonia emissions are estimated as part of the nitrogen balance approach. Meat birds are typically raised in litter systems. Litter temperature, pH, and moisture, along with the ammonium content and house ventilation rate are recognized as major factors controlling NH_3 loss from broiler litter (Elliot and Collins, 1982; Carr et al., 1990; Moore et al., 2010). There are seasonal variations in emissions, with losses tending to be greater in summer (warmer months) than in winter (Coufal et al., 2006). Bird age/size can affect litter temperature, which may influence seasonal effects on emissions (Miles et al., 2008). In addition, the formation of cake in the house can have a large impact on emissions. Miles et al. (2008) reported that extremely caked areas of the house had virtually no fluxes of NH_3. Areas of litter where anaerobic conditions develop suppress NH_3 formation and release (Carr et al., 1990). Moore et al. (2011) determined that NH_3 emissions from broiler houses averaged 37.5 g bird^{-1}, or 14.5 g kg bird marketed^{-1} (50-day old birds). The same authors estimated that of the total nitrogen output from typical broiler houses, approximately 22 percent can be associated with NH_3 emissions, 56 percent from harvested birds, and 21 percent from litter plus cake. The addition of aluminum sulfate (alum) at a rate equivalent to five to 10 percent by weight (alum/manure) reduces NH_3 emission from broiler houses by 70 percent (Moore et al., 2000) and results in heaver birds, better feed conversion, and lower mortality (Moore, 2013). Emissions of N_2O and CH_4 are dependent upon litter conditions that favor an anaerobic environment. Limited data are available documenting litter moisture content effects on N_2O and CH_4 emissions. Miles et al. (2011) demonstrated that incremental increases in litter moisture content increased NH_3 volatilization. Similarly, Cabrera and Chiang (1994) demonstrated a range in NH_3 volatilization of 32 percent to 139 percent of initial ammonium content as litter water content increased. Litter temperature is another factor that may influence GHG emissions. Miles et al. (2006) demonstrated that litter temperature affected NH_3 flux, but the study did not measure other gases. Miles et al. (2011) observed that organic bedding materials generated the least amount of NH_3 at the original moisture content when compared with the inorganic materials. The influence of bedding material at increased moisture levels was not clear across the treatments tested. But the findings suggest that choice of bedding material may also influence N_2O and/or CH_4 emissions and could potentially be used as a mitigation strategy.

5.3.5.1 Method for Estimating Emissions from Poultry Production Systems

Method for Estimating Emissions from Poultry Production Systems

Methane

- IPCC Tier 1 approach, utilizing barn capacity and manure CH_4 emissions factors per poultry type.
- IPCC emission factor for poultry enteric CH_4 production is 0. Emissions from hindgut fermentation are small and generally considered part of housing emissions.

Nitrous Oxide

- Nitrogen excretion estimated using equations provided in ASABE D384.2.
- IPCC (2006) Tier 2 approach for N_2O from manure in housing.

Equation 5-21: Methane Emissions from Poultry Housing (IPCC, 2006)

$$\mathbf{CH_4 = Rate \times Barn\_Capacity}$$

Where:

CH_4 = Methane emissions per year (kg CH_4 $year^{-1}$)

Rate = Manure methane emissions (kg CH_4 $head^{-1}$ $year^{-1}$)

Barn_Capacity = Capacity of barn (head)

Nitrous Oxide and Ammonia Emissions from Poultry Housing

To estimate nitrogen losses from housing, the amount of nitrogen excreted (N_{ex}) by each animal category is first estimated. Equation 5-22 and Equation 5-23 are the equations recommended by the American Society of Agricultural Engineers (ASABE) for estimating N_{ex} from broilers, turkeys, ducks, and laying hens.

Equation 5-22: ASABE Approach for Estimating Nitrogen Excretion from Broilers, Turkeys, and Ducks

$$\mathbf{N_{ex} = \sum_{x=1}^{n} \left[FI_x \times \frac{C_{CP-x}}{6.25} \right] \times (1 - N_{RF})}$$

Where:

N_{ex} = Total nitrogen excretion per finished animal (g N (finished animal)$^{-1}$)

FI_x = Feed intake per phase (g feed (finished animal)$^{-1}$)

C_{CP-X} = Concentration of crude protein of total ration in each phase (g crude protein (g (wet) feed)$^{-1}$)

N_{RF} = Retention factor for nitrogen (fraction)

Equation 5-23: ASABE Approach for Estimating Nitrogen Excretion from Laying Hens

$$\mathbf{N_{ex}} = \left(\mathbf{FI} \times \frac{\mathbf{C_{CP}}}{\mathbf{6.25}}\right) - \left(\mathbf{0.0182} \times \mathbf{Egg_{wt}} \times \mathbf{Egg_{pro}}\right)$$

Where:

N_{ex} = Total nitrogen excretion per animal per day (g N animal^{-1} day^{-1})

FI = Feed intake (g feed (finished animal)$^{-1}$)

C_{CP} = Concentration of crude protein of total ration (g crude protein (g (wet) feed)$^{-1}$)

Egg_{wt} = Egg weight[a] (g)

Egg_{pro} = Fraction of eggs produced each day[b] (eggs hen^{-1} day^{-1})

[a] Default egg weight is 60 g for light layer strains and 63 g for heavy layer strains.
[b] Default fraction is 0.80.

The NH_3 lost from manure for meat and egg-producing birds is estimated as a fraction of N_{ex}. Koelsch and Stowell (2005) provide estimates on the typical NH_3 loss from different housing facilities as a fraction of N_{ex} (see Table 5-15). A range of values has been provided for each facility type; the lower values should be used during the winter, the higher values should be used during the summer, and intermediate values should be used for the spring and autumn.

Table 5-15: Typical Ammonia Losses from Poultry Housing Facilities (Percent of N_{ex})

Facility Description	Applicable Species	% Loss	Facility Description	Applicable Species	% Loss
Roofed facility (litter)	Meat producing birds	25 - 50	Roofed facility (stacked manure under floor - includes storage loss)	Egg-producing birds	25 - 50

Source: Koelsh and Stowell (2005).

N_2O can also be lost from the excreted nitrogen. A quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Greenhouse Gas Inventory (Equation 5-8). This estimation method is the same as the method present in the Temporary Stack and Long-Term Stockpile and the Composting sections (see sections 5.4.1 and 5.4.2 for more details). The N_2O emission factors for poultry manure in housing is 0.001 (kg N_2O-N/ kg N) for poultry manure with or without bedding IPCC (2006).

The remaining nitrogen excreted that is not volatilized as NH_3 or lost as N_2O in housing then enters manure storage and treatment. If data are not available to track the nitrogen that is transferred along with the manure to manure storage and treatment, the nitrogen can be estimated as described in Equation 5-9. However, this equation is overestimating the nitrogen transferring to manure storage and treatment as some nitrogen will be lost in housing. This remaining total nitrogen value is an input into the N_2O equations for manure stored or treated.

5.3.5.2 Rationale for Selecting Method for Estimating Emissions from Poultry Production Systems

Miles et al. (2006) suggest that a robust model would include factors such as house management, bird size and age, cake management, pH, and litter moisture. Due to current data limitations, an NH_3 and GHG estimation model should minimally include number of animals, litter/excreta moisture content, dietary protein and fiber content, and litter/excreta pH. A variety of models applicable to poultry production facilities were identified and evaluated, including Carbon Accounting for Land Managers; CFF Carbon Calculator; CPLAN; and 4) Holos. These models were evaluated with respect to 10 criteria (see box) to determine their suitability for use in determining emissions estimates for poultry production facilities in the United States.

Model Evaluation Criteria for Poultry Production Systems

1. The model is based on well-established scientifically sound relationships between farm management inputs and emissions outputs (process-based model or mass-balance model preferable).
2. The model is relevant to U.S. climate and production systems.
3. The model can estimate CH_4, N_2O, and NH_3 emissions from poultry housing systems.
4. There is flexibility in the model to describe the production system (animals, feed, housing, and in-house manure management).
5. The model is easy to use and is designed to use easily obtainable farm information to determine emissions estimates.
6. The model includes some mitigation strategies for reducing emissions and produces realistic changes in emissions values when these changes are made within the production system.
7. There is transparency in the model calculations, and technical guidelines are available to elaborate the methodologies used to obtain the emissions estimates.
8. The model has been tested/validated with on-farm data.
9. The model works reliably (little to no errors or program crashes).
10. The model is publicly available and accessible.

Two of these criteria were considered to be critical, in that if they were not met by the model, they could not be considered for use (i.e., the model had to be relevant to U.S. climate and poultry production systems and had to be publicly available). The Holos model did consider wet or dry manure handling for laying hen operations. For all poultry types, the Carbon Accounting for Land Managers model requested information related to burning of manure and time birds spend in a free-range system. This information was then used to calculate the mass of manure available for direct and indirect emissions. No model requested information on diet or in-house litter management practices. For CH_4 emissions, only the Holos model provided an estimate of confidence of model output. Specific to estimates of poultry manure CH_4 emissions, the model had an uncertainty under 20 percent for broilers, turkeys, layers in wet manure handling systems, and layers in dry manure handling systems. Consequently, the IPCC method was selected (i.e., Holos utilizes the IPCC Tier 1 approach for housing). For N_2O emissions, the IPCC Tier 2 was used with nitrogen excretion estimated using ASABE equations that account for diets.

5.3.6 Enteric Fermentation and Housing Emissions from Other Animals

Although the majority of emissions from livestock in the United States are from cattle, sheep, swine, and poultry, emissions from other animals can also be important to consider, particularly at the entity level. Overall, populations of the animals discussed in this section (goats, American bison, llamas, alpacas, and managed wildlife) are much fewer than those of the animals discussed in prior sections. However, the availability of research on emissions from these animals allows us to explore them at least at an introductory level. At the entity level, populations of these animals may be significant enough to warrant calculating their emissions. This report recommends methods for estimating CH_4 emissions from goats and American bison (Equation 5-24 and Equation 5-25).

5.3.6.1 Goats

Enteric emissions from goat production systems were estimated by U.S. EPA (U.S. EPA, 2011) using IPCC (2006) methods to be 16 Gg CH_4 (of a total of 6,655 Gg). Emissions of manure CH_4 and N_2O from goat production were made using IPCC (2006) methods. Goats were associated with 1 Gg of manure CH_4 (of a total of 2,356 Gg) and less than 0.5 Gg of N_2O.

The impact of diet on Japanese goat enteric CH_4 emissions was measured by Bhatta et al. (2007). Goats fed a range of diets from 100 percent forage to 80 percent concentrate produced from 16.4 to 22 g CH_4 day^{-1} (5.0 to 8.2 percent of GEI).

The IPCC (2006) Tier 1 equation, presented in Equation 5-24, for estimating enteric fermentation emissions from goats is the best option for calculating emissions at the entity level.

Equation 5-24: Tier 1 Equation for Calculating Methane Emissions from Goats

$$\mathbf{CH_4 = Pop \times EF_G}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 day^{-1})

Pop = Population of goats (head)

EF_G = Emission factor for goats (0.0137 kg CH_4 head^{-1} day^{-1})

5.3.6.2 American Bison, Llamas, Alpacas, and Managed Wildlife

Galbraith et al. (1998) measured enteric CH_4 from growing bison (n=5), wapiti (n=5), and white-tailed deer (n=8) fed alfalfa pellets in the winter-spring (February-March) and spring (April-May) using respiration calorimetry chambers. The bison produced an average of 86.4 g day^{-1} (6.6 percent GEI), the wapiti, 62.1 g day^{-1} (5.2 percent GEI), and the deer 23.6 g day^{-1} CH_4 (3.3 percent GEI). Using a detailed method of calculation to estimate historical bison emissions, Kelliher and Clark (2010) estimated that grazing bison would produce 72 kg CH_4 year^{-1} or 197g CH_4 day^{-1}. Hristov (2012) estimated present day bison produce 21 g CH_4 (kg DMI)$^{-1}$ day^{-1}, eat approximately 12.8 kg DM day^{-1}, and produce 268 g CH_4 day^{-1}. The differences between these estimates are differences in animal weights, DMI, limited measurements of bison emissions, and assumed CH_4 conversion factors. The U.S. EPA uses IPCC Tier 1 methodologies to estimate bison emissions, and currently Tier 1 is the best option to estimate enteric emissions.

The IPCC (2006) Tier 1 equation for estimating enteric fermentation emissions from American bison is based on the emission factor for buffalo and has been modified as recommended by IPCC to account for average weight as seen in Equation 5-25.

Equation 5-25: Tier 1 Equation for Calculating Methane Emissions from American Bison

$$\mathbf{CH_4 = Pop \times EF_{AB}}$$

Where:

CH_4 = Methane emissions per day (kg CH_4 head^{-1} day^{-1})

Pop = Population of American bison (head)

EF_{AB} = Emission factor for American bison (kg CH_4 head^{-1} day^{-1})

EF_{AB} is the IPCC emission factor for *buffalo* (0.15 kg CH_4 head^{-1} day^{-1}), adjusted for American bison based on the *ratio* of live weights of American bison (513 kg) to buffalo (300 kg) to the 0.75 power.

$$\mathbf{EF_{AB} = 55\ kg \times \left(\frac{513\ kg}{300\ kg}\right)^{0.75}}$$

The New Zealand Ministry for the Environment (2010) uses a factor of 6.4 percent of GEI to predict enteric CH_4 emissions from farmed red deer and projects an emission rate per year of 23.7 kg CH_4 head^{-1} year^{-1}. Deer are also estimated to excrete 31.0 kg N head^{-1} year^{-1} contributing toward N_2O production. The values used to make these calculations are from measurements of deer CH_4 emissions using the SF_6 tracer method. Elk, white-tailed, and mule deer enteric emissions were estimated by Hristov (2012) to be 86.4, 16, 17 g CH_4 head^{-1} day^{-1} respectively. IPCC Tier 1 is the recommended method by which these emissions should be estimated.

Adult llamas fed oat hay in a study designed to define energy requirements were found to lose 7.1 percent of GEI as enteric CH_4 (Carmean et al., 1992). Pinares-Patino et al. (2003) compared enteric CH_4 emissions measured with respiration calorimetry chambers from alpaca and sheep fed alfalfa diets and found the alpaca produced 14.9 g CH_4 day^{-1} (5.1 percent of GEI) and the sheep produced 18.8 g CH_4 day^{-1} (4.7 percent of GEI). When grazing a perennial ryegrass/white clover pasture, the alpaca produced 22.6g CH_4 day^{-1} (9.4 percent GEI) compared to 31.1 g CH_4 day^{-1} (7.5 percent GEI) for sheep. The authors attribute the high conversion of GEI to CH_4 from the alpaca to grazing selectivity on pasture; the alpaca were observed to select "more structural plant parts."

5.3.7 Factors Affecting Enteric Fermentation Emissions

A number of factors may influence enteric fermentation and resulting CH_4 emissions. A thorough review of such factors is outside the scope of this document, but key factors have been reviewed by others (Monteny et al.; (2006), Beauchemin et al.; (2008), Eckard et al.; (2010), and Martin et al.; (2010)) and are discussed briefly below.

Benchaar et al. (2001) used the rumen digestion model of Dijkstra et al. (1992), as modified by Benchaar et al. (1998), and the CH_4 prediction system of Baldwin (1995) to estimate the effects of dietary modifications on the enteric CH_4 production of a 500 kg dairy cow. The model predicted enteric CH_4 production based on a ruminal H balance. Inputs into the model included the following: daily DMI; chemical composition of the diet; solubility and degradability of protein and starch in the diet; degradation rates of protein, starch, and NDF; ruminal volume; and fractional passage rates of solids and liquid fractions from the rumen. Values modified in the simulations were DMI, dietary forage, concentrate ratio, starch availability (barley vs. corn), stage of maturity of forage, form of forage (hay or silage), particle size of alfalfa, and ammonization of cereal straw. The modeled effects of dietary changes on enteric CH_4 emissions in diets fed to dairy cows are presented in Table 5-16.

There are many factors that affect enteric CH_4 emissions but the most critical factors are the level of dry matter intake, the composition of the diet, and the digestibility of the dry matter, as illustrated in Table 5-16.

Table 5-16: Summary of Effects of Various Dietary Strategies on Enteric CH_4 Production in Dairy Cows using Modeled Simulations

Strategy	CH_4 Variation (per unit of GEI)	CH_4 Variation (per unit of DE)
Increasing DMI	-9 to -23%	-7 to -17%
Increasing concentrate proportion in the diet	-31%	-40%
Switching from fibrous concentrate to starchy concentrate	-24%	-22%
Increased forage maturity	+15%	-15%
Alfalfa vs. timothy hay	+28%	-21%
Method of forage preservation (ensiled vs dried)	-32%	-28%
Increased forage processing (smaller particle size)	-21%	-13%
Ammoniated treatment of poor quality forage(straw)[a]	x 5	x 2
Protein supplementation of poor quality forage (straw)	× 3	× 1.5

Source: Benchaar et al., (2001), Table 12.
[a] Effects are due to significant increase in hay digestibility with no change in DM intake.

Dietary Fat: Many studies have demonstrated that supplemental fat can decrease enteric CH_4 emissions in ruminants. In a review of studies, Beauchemin et al. (2008) noted that enteric CH_4 emissions (g [kg DMI]$^{-1}$) decreased by approximately 5.6 percent for each one percent increase in fat added to the diet. In a larger review, Martin et al. (2010) reported a decrease of 3.8 percent (g [kg DMI]$^{-1}$) with each one percent addition of fat. Lovett et al. (2003) reported that total daily emissions decreased from 0.19 to 0.12 kg CH_4 head^{-1} (reported as 260 to 172 L CH_4 head^{-1}) (6.6 and 4.8 percent of GEI) from steers fed diets containing 0 or 350 g of coconut oil, respectively. This effect was consistent regardless of dietary forage concentration (65, 40, and 10 percent of DM).

Although added fat may reduce enteric CH_4 emissions, ruminants have a low tolerance for dietary fat. Thus, total fat level in the diet must be maintained below eight percent of dietary DM. Some sources of fat appear to have some protection against biohydrogenation by ruminal microbes and thus may be better tolerated (Corrigan et al., 2009; Vander Pol et al., 2009).

Grain Source, Grain Processing, Starch Availability: Grain source and grain processing method can also affect enteric CH_4 losses. In general, the greater the ruminal starch digestibility, the lower the enteric CH_4 emissions. At constant energy intake (2 x maintenance), Hales et al. (2012) reported approximately 20 percent lower (2.5 vs. 3.0 percent of GEI) enteric CH_4 emission in cattle fed typical high-concentrate (75 percent corn) steam flaked corn (SFC) based finishing diets than in steers fed dry-rolled corn (DRC) based finishing diets. Based on the rumen stoichiometry of Wolin (1960), Zinn and Barajas (1997) estimated that CH_4 production per unit of glucose equivalent fermented in the rumen also decreased with more intensive grain processing (i.e., coarse, medium, or fine flakes). Similar responses were noted with the feeding of high-moisture corn compared with DRC (Archibeque et al., 2006). Somewhat in contrast, Beauchemin and McGinn (2005) reported lower enteric CH_4 emissions from feedlot cattle fed DRC-based diets (2.81 percent of GEI) than from cattle fed steam-rolled barley-based diets (4.03 percent of GEI), possibly the result of lower ruminal pH on the corn-based diet (5.7 vs. 6.2, respectively; (Van Kessel and Russell, 1996) and/or higher NDF in the barley diet. Enteric CH_4 emissions were 38 percent (barley) to 65 percent (corn) lower in high-concentrate (nine percent silage) finishing diets than on grower (70 percent silage) diets.

Feeding Coproduct Ingredients: Distillers grains with solubles (DGS) and other coproducts of the milling and ethanol industries are widely used as cattle feeds. The effects of feeding 30 to 35 percent DGS (DM basis) on enteric CH_4 emission have been variable, ranging from a significant decrease of 25 to 30 percent (McGinn et al., 2009) to no effect (Hales et al., 2012), to an increase (Hales et al., 2013). These differing results were probably due to differences in forage and fat intake. In the study by McGinn et al. (2009) the diet contained 65 percent silage, and dietary fat intake increased by approximately three percentage units[8] when dried DGS were added to the diet. In contrast, Hales et al. (2012; 2013) fed diets that contained only 10 percent forage and were equal in total fat concentration.

Roughage Concentration and Form: The concentration and form of roughage in the diet will affect both enteric and manure CH_4 production (Hales et al., 2014). Using a ruminal volatile fatty acids (VFA) stoichiometry model, Dijkstra et al. (2007) suggested that CH_4 losses from carbohydrates substrates (g kg^{-1} substrate) in a concentrate diet with ruminal pH variation and a pH of 6.5 were 2.11, 3.18, 3.38, and 3.10 for starch, soluble sugars, hemicellulose, and cellulose, respectively. Similarly, with dairy cows, Moe and Tyrrell (1979) reported that enteric CH_4 production per unit carbohydrate digested was three times greater for cellulose than for hemicellulose. Aguerre et al. (2011) found that lactating dairy cattle emitted more CH_4 when the forage:concentrate ratio was changed from 47:53 to 68:32, 0.54 kg CH_4 day^{-1} vs. 0.65 kg CH_4 day^{-1} respectively.

In general, as the concentration of forage in the diet increases, enteric CH_4 production increases and the quantity of volatile solids excreted increases. Using a micrometeorological mass difference method, Harper et al. (1999) reported CH_4 emissions of 230 g $animal^{-1}$ daily (7.7 to 8.1 percent of GEI) in feeder cattle on pasture, but only 70 g $head^{-1}$ daily (1.9 to 2.2 percent of GEI) in cattle fed high-concentrate diets. Measured CH_4 losses for pasture cattle were higher than values predicted using the IPCC (1997; 2006) CH_4 conversion factors (MCF or Ym), or Australian methodology (NGGIC, 1996). In contrast, measured CH_4 losses for feedlot cattle were about 67 percent of those estimated using the IPCC (2006) Ym of three percent of GEI or the Australian methodology (NGGIC, 1996), but were similar to values reported by Branine and Johnson (1990), Blaxter and Wainman (1964), and Hales et al. (2012; 2013; Hales et al., 2014).

Enteric fermentation of tropical grasses and legumes may also be different than predicted by IPCC or national GHG inventory methods. Kennedy and Charmley (2012) measured enteric CH_4 production of cattle fed Australian tropical grasses and legumes to be 5.0 to 7.2 percent of GE intake which is similar to IPCC (2006) Tier 2 estimates (5.5 to 7.5 percent of GE intake) of cattle fed forage diets but somewhat lower than the Australian National Greenhouse Accounts National Inventory Report (2007) of 8.7 to 9.6 percent of GE intake.

Blaxter and Wainman (1964) compared the effects of feeding diets with six varying hay: flaked corn ratios (100:0, 80:20, 60:40, 40:60, 20:80, 5:95) on enteric CH_4 emissions when fed at two times the maintenance level of intake. CH_4 emissions as a percentage of GEI increased slightly between the 100:0 diet (7.44 percent) and the 60:40 diet (8.17 percent), then decreased to the 5:95 diet (3.4 percent).

In Ireland, Lovett et al. (2003) reported total daily enteric CH_4 emissions of 0.15, 0.19, and 0.12 kg $head^{-1}$ (reported as 207, 270, and 170 L $head^{-1}$) for heifers fed diets containing 65, 40, and 10 percent forage (the remainder as concentrate), respectively. As a percentage of GEI, losses were 6.1, 6.6, and 4.4 percent, respectively.

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[8] The term "percentage units" in this document is used to refer to changes in diets or emissions that are *not* proportional to their baselines. For example, a reduction in emissions from three percent to one percent is a 2 "percentage unit" reduction or a *67 percent reduction*.

Using steers fed all-forage diets, Ominski et al. (2006) reported that, within the range of forage qualities tested (alfalfa-grass silage containing 61, 53, 51, or 46 percent NDF, DM basis), enteric CH_4 emissions of steers, as a percentage of GEI, were not significantly affected by NDF content (5.1 to 5.9 percent), although daily CH_4 production tended to be highest for the 53 percent NDF diet (0.12, 0.15, 0.13, and 0.14 kg head^{-1} day^{-1}, respectively). Similarly, using grazing sheep, Milano and Clark (2008) reported no effect of forage quality (rye grass – 52 or 47 percent NDF, 77 or 67 percent organic matter [OM] digestibility) on enteric CH_4 emissions.

Although dietary forage quality may sometimes not affect enteric CH_4 emissions, it will affect forage digestibility and thus fecal excretion of volatile solids. Thus, feeding more digestible forages or concentrates may decrease GHG emissions from manure.

Level of Intake: Blaxter and Wainman (1964) compared the effects of feeding six diets at two levels of intake. Enteric CH_4 emissions, as a percent of GEI, were 23 percent greater in steers fed at maintenance than in steers fed at 2X maintenance (8.1 vs. 6.6 percent of GEI, respectively). However, in a study evaluating emissions from cattle fed ryegrass diets, Milano and Clark (2008) reported that as DMI increased from 0.75 percent of maintenance to 2X maintenance, enteric CH_4 emissions (g day^{-1}) increased linearly (r^2 = 0.80 to 0.84). Emissions as a percentage of GEI were not affected by DMI, and ranged from 4.9 to 9.5 percent of GEI (15.9 to 30.4 g [kg DMI]$^{-1}$).

Using a high-forage (70 percent barley silage) or medium-forage (30 percent silage) diet fed at levels from 1X to approximately 1.8X maintenance, Beauchemin and McGinn (2006b) noted that enteric CH_4 emissions, as a percent of GEI, decreased by approximately 0.77 percentage units for each unit increase in feed intake (expressed as level of feed intake above maintenance). This was less than the estimate using the Blaxter and Clapperton (1965) equation (0.93 to 1.28 percent percentage units) or the 1.6 percentage units suggested by Johnson and Johnson (1995).

Feed Additives and Growth Promoters: Cooprider et al. (2011) noted that the daily CH_4 and manure N_2O production of cattle fed through a “natural” program with no use of antibiotics, ionophores, or growth promoters were similar to cattle fed in more traditional systems that used anabolic implants and diets that contained ionophores and beta-agonists. However, typical cattle had greater average daily weight gain (1.85 vs. 1.35 kg day^{-1}) and thus took 42 fewer days to reach the same end point (596 kg body weight [BW]). Thus, overall, cattle fed using modern growth technologies had 31 percent lower GHG emissions per head. CH_4 emissions kg of BW gain^{-1} was 1.1 kg greater for the “natural” cattle (5.02 vs. 3.92 CO_2-eq kg BW gain^{-1}) than the traditional cattle.

Monensin decreases enteric CH_4 emissions in finishing cattle by 10 to 25 percent (Tedeschi et al., 2003; McGinn et al., 2004). However, in feedlot cattle the effects appear to be transitory, lasting for 30 days or less (Guan et al., 2006). In contrast, Odongo et al. (2007) reported that monensin (24 ppm) in dairy diets decreased enteric CH_4 by seven to nine percent for up to six months. Waghorn et al. (2008) found no effect of monensin controlled-release capsules on CH_4 production of pasture-fed dairy cows, and Hamilton et al. (2010) also found no change in enteric CH_4 production from monensin fed to dairy cows offered a total mixed ration.

A number of studies have demonstrated that a variety of halogenated analogues have the potential to dramatically decrease ruminal CH_4 production (Johnson, 1972; Trei et al., 1972; Johnson, 1974; Cole and McCroskey, 1975; Tomkins and Hunter, 2004; Tomkins et al., 2009). In general the effect was greater in cattle fed high-forage diets than in cattle fed high-concentrate diets. When CH_4 losses were dramatically reduced, a significant quantity of hydrogen could be lost (one to two percent of GEI) via eructation, suggesting an alternative electron sink is also needed. In general, the compounds did not improve production efficiency significantly. In addition, the potential toxicity of these compounds made them impractical for routine use.

A number of nitrocompounds (nitropropanol, nitroethane, nitroethanol) have also significantly decreased ruminal CH_4 production in vitro (Anderson et al., 2003), with a concomitant increase in hydrogen production/release. The effect appeared to be enhanced when a nitrate reducing bacterium was added to the culture (Anderson and Rasmussen, 1998).

Several studies have suggested that feeding of condensed tannins can decrease enteric CH_4 production by 13 to 16 percent; either through a direct toxic effect on ruminal methanogens or indirectly via a decrease in feed intake and diet digestibility (Eckard et al., 2010). Tannins may also shift nitrogen excretion away from urine to feces and inhibit urease activity in feces, which could potentially decrease NH_3 and N_2O emissions from manure (Powell et al., 2009; Powell et al., 2011).

Feeding yeast cultures, enzymes, dicarboxylic acids (fumarate, malate, acrylate), and plant secondary compounds, such as saponins, may decrease enteric CH_4 emissions under some feeding conditions (McGinn et al., 2004; Beauchemin and McGinn, 2006a; Ungerfeld et al., 2007; Beauchemin et al., 2008; Eckard et al., 2010; Martin et al., 2010).

Novel Microorganisms and their Products: Klieve and Hegarty (1999) noted that enteric CH_4 production may be biocontrolled directly by use of viruses and bacteriocins. Lee et al. (2002) reported that a bacteriocin (Bovicin HC5) from *Streptococcus bovis* reduced in vitro CH_4 production by up to 50 percent. It appeared, that in contrast to results with monensin, the ruminal microorganisms did not adapt to the bacteriocin.

Australian researchers have suggested that vaccinating against methanogens can decrease CH_4 emissions. However, the results have not been consistent (Wright et al., 2004; Eckard et al., 2010) because efficacy is dependent on the specific methanogen population and that is dependent on diet, location, and other factors.

Genetics: As previously noted, several studies have suggested that cattle selected for lower RFI (i.e., increased feed use efficiency) tend to have lower ruminal enteric CH_4 production (Nkrumah et al., 2006; Hegarty et al., 2007), although the effect may depend on stage of production (lactation vs. dry and pregnant) and/or quality of the diet consumed (Jones et al., 2011). RFI is moderately heritable (0.28 to 0.58) (Moore et al., 2009), thus it might be possible to genetically select for animals with lower enteric CH_4 production. However, Freetly and Brown-Brandl (2013) found higher CH_4 emissions from more efficient animals. Thus, more information is needed to define under what conditions CH_4 emissions are related to feed efficiency or to genetics.

Factors Affecting Emissions from Sheep

Sheep, like cattle, are ruminant animals and thus the same dietary factors will positively or negatively affect emissions from enteric fermentation.

Factors Affecting Emissions from Swine

Dietary modifications can effectively reduce nitrogen excretions and mitigate air emissions (especially NH_3, a precursor for N_2O) from livestock operations (Sutton et al., 1996; Canh et al., 1998b). Feeding strategies to reduce nitrogen excretions include reduced CP diets supplemented with synthetic amino acids (AA) (Panetta et al., 2006), and modifying the dietary electrolytes to reduce urinary pH (Canh et al., 1998a). In both hog *and* poultry operations, reductions in NH_3 emissions have been reported by supplementing with AA and reducing CP in diets.

Reducing dietary CP content has been shown to be an effective way to reduce the amount of nitrogen excreted (Lenis, 1993; Hartung and Phillips, 1994). This can be achieved without any negative effect on animal performance by supplementing with an improved synthetic AA balance, resulting in a reduction of excess CP excreted (Canh et al., 1998b; Ferket et al., 2002). In U.S.-type diets (corn-soybean meal based) the most limiting amino acids are Lysine, Methionine, Threonine, and Tryptophan, followed by Isoleucine, Valine, and Histidine (Outor-Monteiro et al., 2010). Sutton et al. (1996) reported that nitrogen excretion decreased by 28 percent when diet CP content decreased from 13 percent to 10 percent (corn-soybean meal) for growing-finishing pig diets supplemented with Lys, Met, Thr, and Trp. Several studies reported reductions in nitrogen excretion and subsequent decreases in NH_3 emissions in non-ruminants (swine and poultry) (Hartung and Phillips, 1994; Canh et al., 1997; Canh et al., 1998a; Canh et al., 1998b; Hayes et al., 2004). Powers et al. (2007) observed that, as a result of feeding reduced CP diets with increased amounts of synthetic AA, NH_3 emissions were reduced by 22 percent (three AA) and 48 percent (five AA) compared with the control diet containing only one AA, and diet had no effect on pig performance.

Canh et al. (1998b) and Ndegwa et al. (2008) reported that some nitrogen excretion could be shifted from urine to feces by increasing dietary fiber content, or by reducing dietary nitrogen content, with no significant differences in animal performance or growth. Urinary nitrogen is predominantly inorganic in nature and fecal nitrogen is mostly organic. The conversion of urea from urine to NH_3 is a fast process, while conversion of organic nitrogen to volatile NH_3 in feces is a slow process.

The reduction in NH_3 emission associated with lower CP diets not only comes from reduction in nitrogen excretion, but also from lower manure pH. Portejoie et al. (2004) reported that slurry pH decreased by 1.3 units when dietary CP decreased from 20 to 12 percent, and slurry from pigs fed the lower CP diet had a higher DM content and lower TAN and TKN contents. Le et al. (2008), Hanni et al. (2007), and Canh et al. (1998b) also reported that lower manure pH resulted from feeding lower CP in diets. It should be noted that water intake was often restricted in earlier studies.

Aarnink and Verstegen (2007) summarized four dietary strategies to reduce NH_3 emissions: 1) lowering CP intake in combination with the addition of limiting AA; 2) shifting nitrogen excretion from urine to feces by including fermentable carbohydrates in the diet; 3) lowering urinary pH with the addition of acidifying salts to the diet; and 4) lowering feces pH with the inclusion of fermentable carbohydrates in the diet. They claimed that by combining these strategies, NH_3 emissions in growing-finishing pigs could be reduced by a total of 70 percent. To reduce odor from pig manure, Le et al. (2007) suggest that dietary sulfur-containing AA should be minimized to just meet the recommended requirements.

Current research has concentrated on farm production efficiency and reducing NH_3 emissions; little has focused on GHG emissions mitigation (Bhatti et al., 2005). Ball and Möhn (2003) showed that low CP diets can reduce total GHG emissions from growing pigs by 25 to 30 percent (directly from the animals as well as from the manure after excretion) and from sows by 10 to 15 percent. Atakora et al. (2003) reported a 27.3 percent decrease in CH_4 emissions in pigs fed 16 percent CP (supplemented with AA) diets, compared with 19.0 percent CP diets. Atakora et al. (2004) reported that the CO_2 equivalents emitted by finishing pigs and sows fed wheat-barley-canola diets were reduced by 14.3 to 16.5 percent when feeding the reduced CP, AA-supplemented diets, and were similar for finishing pigs and sows. The reduction was only 7.5 percent when feeding the corn-soybean meal-based reduced CP diet. Misselbrook et al. (1998) found that CH_4 emissions during storage were less at low than at a high dietary CP content. The emission of CH_4 was significantly related to content of dry matter, total carbon, and VFA in the manure. Misselbrook et al. (1998) claimed that the 50 percent reduction in CH_4 emission from the slurry observed when pigs were fed the lower CP diet was probably the result of the reduced volatile fatty acids (VFA) content of the slurry, and CH_4 emissions were more closely related to VFA content than to total carbon content. There appears to be a close relationship between fermentable carbohydrates in the diet and CH_4 production (Kirchgessner et al., 1991). Manure pH also influences CH_4 production. Kim et al. (2004) noted a 14 percent reduction in CH_4 emission when ideal pH was reduced one unit through addition of acidogenic calcium and phosphorus sources to pig diets. Increasing fermentable carbohydrate levels in the diet to lower the pH of manure, with the goal of reducing NH_3 emissions, might increase CH_4 production (Aarnink and Verstegen, 2007). Canh et al. (1998a) observed that for each 100-g increase in the intake of dietary non-starch polysaccharide (NSP), the slurry pH decreased by approximately 0.12 units and the NH_3 emission from slurry decreased by 5.4 percent when dietary NSP ranged from 150 to 340 g NSP kg DM^{-1}.

Feeding of dried distillers grains with solubles (DDGS) has become common practice in the swine industry. Li et al. (2011) demonstrated that feeding diets containing 20 percent DDGS increased emissions of CH_4 but not N_2O when compared to control diets without DDGS. Observed increases in CH_4 emissions approximated 18 percent. Ammonia emissions resulting from feeding 20 percent DDGS were either higher or lower than diets without DDGS, depending on the form of trace minerals included in the diet. Diets including inorganic forms of trace minerals had seven percent greater NH_3 emissions, while feeding organic forms of trace minerals decreased NH_3 emissions almost 20 percent compared to control diets (Liu et al., 2011a).

In a recent meta-analysis, Liu et al. (2011a) used 32 data points in a subgroup of studies that included diet CP information to analyze the effect of diet CP on GHG emissions. Three factors (diet CP, geographic region, and swine production phase) were considered in the regression analysis. Diet CP was not a significant factor. Emissions of CH_4 are positively correlated with diet crude protein in swine production, most significantly for lagoon and slurry storage systems (Liu et al., 2011a). Clark et al. (2005) determined that reducing dietary CP may actually increase CH_4 emissions, so results are varied. It had been expected that a lower CP diet may result in lower nitrogen excretion, and thus might be able to reduce N_2O emissions from manure. However, this hypothesis was not supported by the results of the meta-analysis.

Diet formulation at each stage of the life cycle influences nutrients excreted in manure, as well as emissions that result from that manure during storage and potentially following land application. From a modeling perspective, the focus needs to be on management factors, including diet formulation and manure handling practices.

Feed efficiency improvements can reduce emissions throughout the entire food production cycle by reducing the amount of feed needed for meat production, thereby reducing inputs into feed production as well as reducing manure nutrients that must be managed. Feed efficiency is the product of genetics and environment (management). Genetic differences are difficult to assess, because this information is retained by companies. Genetic improvements are not insignificant over time and may in fact be a larger contributor to gains than management. However, from a modeling perspective, the focus needs to be on management factors, including diet formulation and in-house manure/litter practices. Feed efficiency could be a model component in the future once more data on the impacts of feed efficiency on GHG emissions are available.

Factors Affecting Emissions from Meat Birds

Emissions of both N_2O and NH_3 can be restricted by reducing the litter nitrogen content through diet modification. Ferguson et al. (1998a; 1998b) fed reduced dietary protein diets to broiler chickens. Although performance was hindered in both studies, NH_3 concentration and litter nitrogen content were reduced significantly as a result of the low-protein diets. Applegate et al. (2008) reported similar litter nitrogen effects when turkey toms were fed reduced-protein diets. No performance differences were observed. These diets were then fed to turkey toms by Liu et al. (2011a), who observed a 12 percent reduction in NH_3 emissions as a result of reducing cumulative nitrogen intake by 9 percent. Feeding specific AA allowed for similar nitrogen intakes across treatments, but reduced NH_3 emissions by 25 percent (Liu et al., 2011a) and nitrogen in litter by 12 percent (Liu et al., 2011b), because nitrogen was better utilized by the birds. Across all diets, N_2O

emissions made up less than one percent of nitrogen output (Liu et al., 2011b), suggesting that reducing dietary nitrogen may have less influence on N_2O emissions than other factors.

Factors Affecting Emissions from Laying Hens

Diet factors can alter air emissions from laying hen facilities. Much of the work to date has focused on reducing NH_3 emissions. Roberts et al. (2007) showed that inclusion of dietary corn DDGS, wheat middlings, or soy hulls lowered the seven-day cumulative manure NH_3 emission from 3.9 g kg of dry manure^{-1} for the control to 1.9, 2.1, and 2.3 g kg of dry manure^{-1}, respectively; it also lowered the daily NH_3 emission rate. Reducing the CP content by one percent had no measurable effect on NH_3 emission. Wu-Haan et al. (2007b) fed a reduced-emissions diet containing 6.9 percent of a $CaSO_4$-zeolite mixture and slightly reduced protein to 21-, 38-, and 59-week-old Hy-Line W-36 hens; they observed that daily NH_3 emissions from hens fed the reduced-emissions diets (185.5, 312.2, and 333.5 mg bird^{-1}) were less than emissions from hens fed the control diet (255.1, 560.6, and 616.3 mg bird^{-1}) for trials 1, 2, and 3, respectively. Total nitrogen excretion from hens fed the control and reduced-protein diets was not different (Wu-Haan et al., 2007a). Because of the acidifying nature of the diets, the mass of nitrogen remaining in excreta following a three-week storage period was less from hens fed the control diet than from hens fed the reduced-protein diet (Wu-Haan et al., 2007a). Li et al. (2010) found that feeding corn DDGS decreased the mass of NH_3 emitted daily by 80 mg hen^{-1}(592 vs. 512 mg hen^{-1} day^{-1} for zero percent and 20 percent DDGS, respectively), and by 14 percent per egg produced, and daily CH_4 emissions by 13 to 15 percent (39.3 vs. 45.4 mg hen^{-1} day^{-1}; and 0.70 vs. 0.82 mg g egg^{-1} day^{-1}).

5.3.8 Limitations and Uncertainty in Enteric Fermentation and Housing Emissions Estimates

At the entity level, uncertainty in enteric CH_4 production in cattle typically results from, lack of precision in estimating energy intake, feed type and intake, characteristics of particular feedstuffs (i.e., acid detergent fiber, starch, etc.), DE, maximum possible CH_4 emissions, CH_4 conversion factors (Ym), synergies or countereffects between mitigation options, and net energy expenditure by the animal. The assumptions about implications of dietary changes on enteric CH_4 production are based on literature values (including empirical field studies) and may not be indicative of true changes in emissions for particular animal types, as this will vary depending on an individual animal's health, management practices, animal activities, and baseline diet. For swine, goats, American Bison, llamas, alpacas, and managed wildlife, the recommended estimation methods for emissions from enteric fermentation are based on the IPCC Tier 1 approach, which has an uncertainty of 30 to 50 percent.

Methane emissions from dairy cattle housing areas are estimated using equations from DairyGEM (IFSM). In predicting emissions, uncertainty will result from a lack of precision in estimating excreted volatile solids and nitrogen excreted, pH, temperature, air velocity, and surface area of exposed manure, bedding pack, CH_4 conversion factors (MCFs), and maximum CH_4-producing capacity for manures. Comparison of modeled values with on-farm evaluations has found the model predicts on-farm emissions within five to 20 percent (unpublished data).

Methane emissions from poultry housing areas are estimated using the IPCC Tier 1 method. Uncertainty in predictions of emissions result from a lack of precision in estimating feed intake, nitrogen excreted and volatile solids, MCF, volatilization fraction, and in some instances emission factors that were chosen in the model. Unfortunately there is a lack of published information related to GHG emissions from poultry and to the best of our knowledge this model has not been validated/tested using on-farm data.

Much of the published uncertainty information in inventory guidance, such as IPCC Good Practice Guidance (IPCC, 2000) and in the U.S. National GHG Inventory (U.S. EPA, 2013), focus on uncertainties present in calculating inventories at the regional or national scale, many of which do not translate to the entity level. Some of the sources of uncertainty at the regional or national scale included variability in native vegetation eaten by grazing animals, assumptions about the types of feed farmers provide for animals (including the practice of including nutritional supplements), management practices such as housing options and daily animal activity, average animal weights, and animal populations. The quantity of uncertainty at larger scales is difficult to define, dependent on both the accuracy in reporting practices and experts' understanding of the implications of management practices and the accuracy of particular estimation methodologies. Consistent improvement in reporting practices can help remove some of this uncertainty.

Available default values and uncertainty information is included in Table 5-17.

Table 5-17: Available Uncertainty Data for Emissions from Housing and Enteric Fermentation

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Daily Milk Production	Milk	kg milk/animal/day		3%	5%			Expert Assessment
Supplemental Fat (feedlot)	S.Fat	Percent				2	4	Expert Assessment
Maximum daily emissions for dairy cows	E_{max}	MJ/head	45.98					Mills et al. (2003)
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (cool, humid region)	NH_3 loss	Percent of N_{ex}				15%	30%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (hot, arid region)	NH_3 loss	Percent of N_{ex}				30%	45%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	NH_3 loss	Percent of N_{ex}				5%	15%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (shallow pit under floor)	NH_3 loss	Percent of N_{ex}				10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (cool, humid region)	NH_3 loss	Percent of N_{ex}				30%	45%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (hot, arid region)	NH_3 loss	Percent of N_{ex}				40%	60%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	%NH_3 loss	Percent of N_{ex}				5%	15%	Koelsh and Stowell (2005)

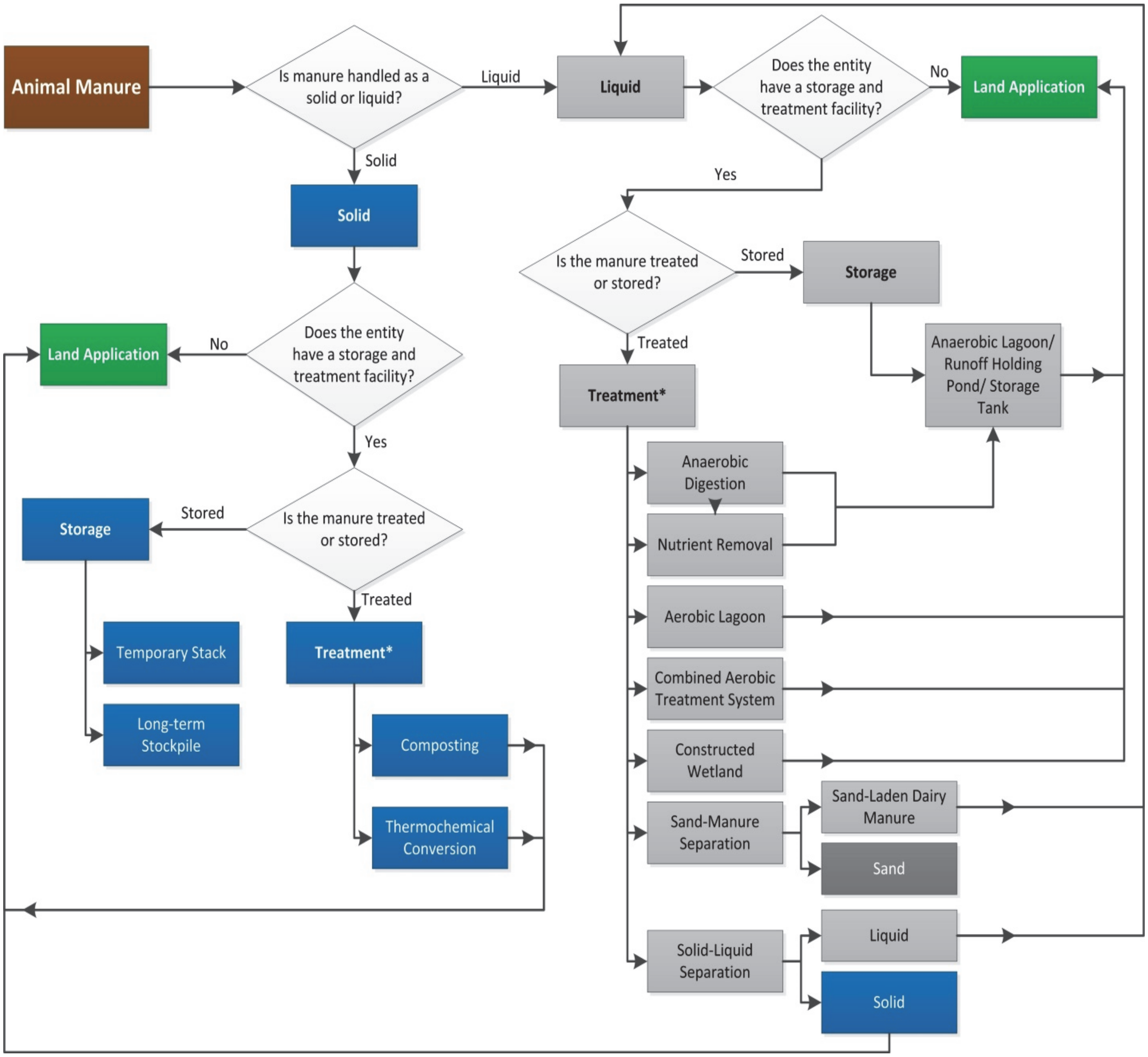
Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (shallow pit under floor)	$\%NH_3$ loss	Percent of N_{ex}				10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (bedded pack)	$\%NH_3$ loss	Percent of N_{ex}				20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	$\%NH_3$ loss	Percent of N_{ex}				30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Poultry Housing – Roofed facility (litter) (Meat Producing birds)	$\%NH_3$ loss	Percent of N_{ex}				25%	50%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Poultry Housing – Roofed facility (stacked manure under floor, includes storage loss) (Egg-producing birds)	$\%NH_3$ loss	Percent of N_{ex}				25%	50%	Koelsh and Stowell (2005)
Methane Emissions from Goats – Emission factor for goats	EF_G	kg CH_4/head/day	0.0137					IPCC (2006)

5.4 Manure Management

Use of manure as a source of plant nutrients reduces the need for purchased commercial fertilizer. Manure storage allows for manure applications to land to be synchronized with crop cultural needs. This practice reduces the potential for soil compaction due to poor timing of manure application (wet soil conditions) and makes more efficient use of farm labor. Many animal manure storage or treatment structures create anaerobic conditions that result in the production and release of GHGs and odors. Manure that is recycled to the land base can have potential negative effects on water quality (both surface and ground water).

Manure storage and treatment, as a component of manure management systems, plays a critical role in GHG emissions. At the entity level, various manure storage and treatment approaches will lead to different amounts of GHG emission. Animal manure can be classified into two categories based on their physical properties: *solid*, defined as dry matter above 15 percent; and *liquid*, defined as dry matter of less than 15 percent (including liquid manure with a dry matter of less than 10 percent and *slurry* manure with a dry matter between 10 and 15 percent). Three solid manure storage/treatment practices (temporary stack/long-term stockpile, composting, and thermo-chemical conversion) and eight liquid manure storage/treatment practices (aerobic lagoon, anaerobic lagoon/runoff holding pond/storage tanks, anaerobic digestion, combined aerobic treatment system, sand-manure separation, nutrient removal, solid-liquid separation, and constructed wetland) were evaluated and the emission estimation methods are presented. At the farm entity level, several practices are often strategically combined to treat manure. In order to provide tools to evaluate these scenarios, activity data (i.e., mass flow data and chemical and physical characteristics of influent and effluent, environmental temperature, pH, and total nitrogen) from individual practices will be used to link practices in the combined system for individual farm entities. A schematic structure of possible combinations of manure storage and treatment practices at the entity level is presented in Figure 5-7. As illustrated in the figure, manure can be handled as a solid or liquid. For each stream, the manure can be applied directly to land, stored, or treated before storage or land application. In some practices, solids are separated from the liquid manure stream and treated using a solids handling system.

Figure 5-7: Schematic Structure of Possible Combination of Manure Storage and Treatment Practices



Note: Individual practices could be combined together to treat manure based on the need at the entity level.

Each manure management practice is described as an individual unit practice in this document. The references for estimation of GHG emission for individual practice are listed in Table 5-18.

Table 5-18: List of Individual Manure Storage and Treatment Practices

Section	Storage and Treatment Practices	Major References for GHG Estimation
Solid manure		
5.4.1	Temporary and long-term storage	IPCC (2006); U.S. EPA (2011)
0	Composting	IPCC (2006); U.S. EPA (2011)
Liquid manure		
5.4.3	Aerobic lagoon	IPCC (2006); U.S. EPA (2011)
5.4.4	Anaerobic lagoon/runoff holding ponds/storage tanks	Sommer et al. (2004)
5.4.5	Anaerobic digestion with biogas utilization	IPCC (2006); CDM (2012)

Section	Storage and Treatment Practices	Major References for GHG Estimation
5.4.6	Combined aerobic treatment system	Vanotti et al. (2008)
5.4.7	Sand–manure separation	
5.4.8	Nutrient removal	
5.4.9	Solid–liquid separation	Ford and Fleming (2002)
5.4.10	Constructed wetland	Stein et al. (2006; 2007b) Stone et al. (2002; 2004)
5.4.11	Thermo-chemical conversion	

The remainder of this section presents the method for estimating GHGs from the sources listed in Table 5-18. For each source of GHGs with an estimation method, the following information is provided:

- **Overview of the GHG Source and the Resulting GHGs.** This section provides an overview of manure management technology, the resulting GHG emissions, and the methodology proposed for estimating the emissions.
- **Rationale for Selected Method**. This section presents the reasoning for the selection of the method recommended in this report.
- **Activity Data**. This section lists the activity data required for estimating GHGs at the entity level.
- **Ancillary Data.** This section lists ancillary data such as CH_4 conversion factors (MCF) and maximum CH_4 production capacity (B_0).
- **Method**. This section provides detailed descriptions, including equations for the selected methods.
- For each source of GHGs without an estimation method, a qualitative overview is provided. Methods for estimating NH_3 emissions are provided in Appendix 5-C.

5.4.1 Temporary Stack and Long-Term Stockpile

5.4.1.1 Overview of Temporary Stack and Long-Term Stockpiles

Method for Estimating Emissions from Manure Storage and Treatment – Temporary Stack and Long-Term Stockpile

Methane

- IPCC Tier 2 approach using IPCC and U.S. EPA Inventory emission factors, utilizing monthly data on volatile solids and dry manure. Volatile solids content can be obtained from sampling and lab testing.
- Method is only readily available method.

Nitrous Oxide

- IPCC Tier 2 approach using U.S.-based emission factors and monthly data on volatile solids, total nitrogen, and dry manure.
- No specific models exist; method is the only readily available method.

Management methods for stored manure are differentiated by the length of time they are stockpiled (i.e., temporary stack and long-term storage). Temporary stack is a short-term manure storage method that is used to temporarily hold solid manure when bad weather prohibits land application, and/or when there is limited availability of cropland for manure application. With temporary stack, the manure is removed and applied to land within a few weeks of piling. Temporary storage is not a preferred method to store manure because it requires the manure to be handled twice.

Long-term storage is a permanent manure storage method in which solid manure is piled on a confined area or stored in a deep pit for longer than six months. In low-rainfall areas, the stockpile can be piled on the field with the installation of nutrient runoff control. In higher rainfall areas, a concrete pad and wall are constructed to store solid manure and prevent nutrient runoff from heavy rain.

Greenhouse gases generated from both storage methods have a pattern similar to that of enteric fermentation. Carbon and nitrogen compounds in manure are broken down by microbes to CH_4, and N_2O. The main factors influencing GHG emissions from storage are temperature and storage time. Due to the longer storage time, long-term stockpile solid manure storage generates a significant amount of GHGs. Temporary stack, as a short-term manure storage method, generates less GHGs than the long-term stockpile solid storage. However, it is still necessary to quantitatively delineate the emissions in order to assist livestock and poultry farms in evaluating their manure management operations. Temporary stack and long-term stockpiles of manure also produce NH_3; proposed methods to estimate NH_3 emissions are presented in Appendix 5-C.

The IPCC Tier 2 methodology is provided for estimating CH_4 emissions from temporary stacks or long-term stockpiles. This methodology uses a combination of IPCC and country-specific emission factors from the U.S. EPA GHG Inventory. The amount of manure, volatile solids content, and temperature are specific to the entity. The method for calculating N_2O emissions is the same as the equation presented in the U.S. GHG Inventory.

Rationale for Selected Method

The IPCC equations are the only available methods for estimating $CH_{4,}$ and N_2O emissions from temporary stack and long-term stockpiles. These methodologies best describe the quantitative relationship among activity data at the entity level.

Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:[9]

- Animal type
- Total dry manure
- Volatile solids of dry manure[10]
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the daily N_2O emission, the following information is needed:

- Total dry manure
- Total nitrogen content of the manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

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[9] Although daily estimates for the activity data are optimal, tracking this level of detail would be burdensome. Annual estimates don't allow for seasonal variation in diets and climate. Consequently, disaggregation of the data by season or by periods of major shifts in animal population is suggested.

[10] Volatile solids, total nitrogen content, and ammonia-nitrogen content should be obtained through sampling and lab testing.

Ancillary Data

The ancillary data used to estimate CH_4 emission for temporary storage and long term stockpiles are: maximum CH_4 producing capacities (B_0) and MCFs. The B_0 values for solid manure storage are obtained from the IPCC and listed in Table 5-19. Methane conversion factors for different manure management systems (including temporary storage of solid manure) are also obtained from the IPCC and listed in Table 5-20 and 5-16.

The ancillary data used to estimate N_2O emissions for temporary storage and long term stockpiles are the N_2O emission factors for solid manure storage systems are presented in Table 5-23 (U.S. EPA, 2011).

5.4.1.2 Method

Methane Emissions from Temporary Stack and Long-Term Stockpile

The Tier 2 approach by the IPCC model is recommended to estimate CH_4 emissions and is described in Equation 5-26 (IPCC, 2006). Daily CH_4 emission is estimated as a function of the volatile solids in manure placed into the storage and the animal-specific MCF.

Equation 5-26: IPCC Tier 2 Approach for Estimating CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}}$$

Where:

E_{CH4} = CH_4 emissions per day (kg CH_4 day^{-1})

m = Total dry manure per day [a] (kg dry manure day^{-1})

VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 (kg VS)$^{-1}$)

MCF = CH_4 conversion factor for the manure management system (%)

0.67 = Conversion factor of m^3 CH_4 to kg CH_4

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-19: Maximum CH_4 Producing Capacities (B_0) from Different Animals

Animal	Maximum CH_4 Producing Capacity (B_0) (m^3/kg VS)	Animal	Maximum CH_4 Producing Capacity (B_0) (m^3/kg VS)
Beef replacement heifers	0.33[b]	Breeding swine	0.48
Dairy replacement heifers	0.17[b]	Layer (dry)	0.39
Mature beef cows	0.33[b]	Layer (wet)	0.39
Steers (>500 lbs)	0.33[b]	Broiler	0.36
Stockers (All)	0.17[b]	Turkey	0.36
Cattle on feed	0.33[b]	Duck	0.36
Dairy cow	0.24[b]	Sheep	0.19[b]
Cattle	0.19[b]	Feedlot sheep	0.36[b]
Buffalo	0.1[a]	Goat	0.17[b]
Market swine	0.48	Horse	0.3
		Mule/Ass	0.33

[a] There are no data for North America region; the data from Western Europe are used to calculate the estimation.
[b] Numbers are from the EPA U.S. Inventory: 1990-2009 (U.S. EPA, 2011). Other numbers are from IPCC (2006).

Table 5-20: Methane Conversion Factors for Temporary Storage of Solid Manure from Different Animals

Animal	Methane Conversion Factor (%)		
	Temp = 10-14°C	Temp = 15-25°C	Temp = 26-28°C
Dairy cow	1	1.5	2
Cattle	1	1.5	2
Buffalo	1	1.5	2
Market swine	1	1.5	2
Breeding swine	1	1.5	2
Layer (dry)	1.5	1.5	1.5
Broiler	1.5	1.5	1.5
Turkey	1.5	1.5	1.5
Duck	1	1.5	2
Sheep	1	1.5	2
Goat	1	1.5	2
Horse	1	1.5	2
Mule/Ass	1	1.5	2

Source: IPCC (2006).

Table 5-21: Methane Conversion Factors for Long-Term Stock Storage of Solid Manure from Different Animals

Animal	Methane Conversion Factor (%)		
	Temp = 10-14°C	Temp = 15-25°C	Temp = 26-28°C
Dairy cow	2	4	5
Cattle	2	4	5
Buffalo	2	4	5
Market swine	2	4	5
Breeding swine	2	4	5
Layer (dry)	1.5	1.5	1.5
Broiler	1.5	1.5	1.5
Turkey	1.5	1.5	1.5
Duck	1	1.5	2
Sheep	1	1.5	2
Goat	1	1.5	2
Horse	1	1.5	2
Mule/Ass	1	1.5	2

Source: IPCC (2006).

Table 5-22: Methane Conversion Factors for Long-Term Storage of Slurry Manure from Buffalo

Temperature (°C)	Methane Conversion Factor (%)
10	17
11	19
12	20
13	22
14	25
15	27
16	29
17	32
18	35
19	39

Temperature (°C)	Methane Conversion Factor (%)
20	42
21	46
22	50
23	55
24	60
25	65
26	71
27	78
28	80

Source: IPCC (2006).

Nitrous Oxide Emissions from Temporary Stack and Long-Term Stockpile

Nitrous oxide emissions are dependent on nitrification and denitrification. Manure storage is one of the main sources of U.S. overall N_2O emissions. The only quantitative method for estimating N_2O emissions from solid manure is the IPCC Tier 2 approach, which is also used for the U.S. Inventory. This approach is based on the use of emission factors from the most recent IPCC Guidelines and total nitrogen values are estimated according to Equation 5-9. Equation 5-27 presents the equation to estimate the N_2O emissions for solid manure.

Equation 5-27: IPCC Tier 2 Approach for Estimating N_2O Emissions

$$\mathbf{E_{N_2O} = m \times EF_{N2O} \times TN \times \frac{44}{28}}$$

Where:

E_{N2O} = Nitrous oxide emission per day (kg N_2O day^{-1})

m = Total dry manure per day[a] (kg dry manure day^{-1})

EF_{N2O} = N_2O emission factor (kg N_2O-N kg N^{-1})

TN = Total nitrogen at a given day (kg N (kg dry manure)$^{-1}$)

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-23: N_2O Emission Factors for Solid Manure Storage

Type of Storage	N_2O Emission Factor (kg N_2O-N/kg N)
Temporary storage of solid/slurry manure	0.005
Long-term storage of solid manure	0.002
Long-term storage of slurry manure	0.005

Source: U.S. EPA (2011).

5.4.2 Composting

5.4.2.1 Overview of Composting

Method for Estimating Emissions from Manure Storage and Treatment –Composting

Methane

- IPCC Tier 2 approach, utilizing monthly data on volatile solids and dry manure. Volatile solids content can be obtained from sampling and lab testing.
- Method is the only readily available method.

Nitrous Oxide

- IPCC Tier 2 approach, utilizing data on a nitrous oxide emission factor, total initial nitrogen, and dry manure.
- Method depends on whether the system is in vessel, static pile, intensive windrow, or passive windrow.
- Method is only readily available method.

Composting is the controlled aerobic decomposition of organic material into a stable, humus-like product (USDA NRCS, 2007). Animal manure may be composted in a variety of different systems, including in-vessel systems, windrows, or static piles. In-vessel systems handle compost in a closed system such as a rotary drum or box that incorporates regular movement to ensure proper aeration. The largest composting operations divide up the compost into long heaps for windrow composting or into one large pile for aerated static pile composting. In the former method, proper oxygen flow can be maintained via manual turning or pipe systems, whereas in the latter method, it is maintained through pipe systems. Composting has become a popular method in some regions to decrease the volume and weight of livestock manure and to produce a product that is often more acceptable to farmers as a fertilizer. During a 100- to 120-day composting period, the weight and volume of manure may be decreased by 15 to 70 percent (Eghball et al., 1997; Inbar et al., 1993; Lopez-Real & Baptista, 1996). Furthermore, the heat generated through the composting process can kill parasites, pathogens, and weed seeds found in animal waste, creating a safer product for crop application.

The quantity of GHG emissions is affected by the composting method employed. Hao et al. (2001) reported that GHG emissions from cattle manure compost increased about twofold when the compost was actively composted rather than passively composted in windrows. Active windrows were turned six times (days 14, 21, 29, 50, 70, and 84). Passive windrows were never turned, but air was introduced into the windrows by a series of open-ended perforated steel pipes. To the extent that the rate of GHG formation depends on oxygen saturation in the pore space, aeration method (i.e., forced-air vs. passive/convective) and rate (or turning frequency) will affect the magnitude of GHG emissions during the composting process.

Eghball et al. (1997) reported that 19 to 45 percent of the nitrogen present in manure was lost during composting, with the majority of this presumably as NH_3. Using changes in the nitrogen:phosphorus ratio of feedlot manure that was placed in compost windrows and the nitrogen:phosphorus ratio of "finished" compost, Cole et al. (2011) estimated that 10 to 20 percent of nitrogen was lost during composting. The U.S. EPA currently assumes that one to 10 percent of nitrogen entering compost systems is lost as N_2O (IPCC, 2006; U.S. EPA, 2009).

The IPCC Tier 2 methodology is provided for estimating CH_4 and N_2O emissions from composting. This methodology uses country-specific emission factors from the U.S. EPA GHG Inventory. The amount of manure, volatile solids content, and temperature are specific to the entity. The GHG estimation method for manure composting does not consider other organic carbon sources that might be added into manure composting.

Rationale for Selected Method

The IPCC equations are the only available methods for estimating CH_4 and N_2O emissions from composting. These methodologies best describe the quantitative relationship amongst activity data at the entity level.

5.4.2.2 Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:

- Animal type
- Total dry manure
- Volatile solids of dry manure
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the daily N_2O emissions, the following information is needed:

- Total dry manure in the storage

- Total nitrogen in manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9: Total Nitrogen Entering Manure Storage and Treatment The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.2.3 Ancillary Data

The ancillary data used to estimate CH_4 emissions for manure composting are: maximum CH_4 producing capacities (B_0) and MCFs. The B_0 values are obtained from the IPCC (2006) and listed in Table 5-19. The MCF values are obtained from EPA (U.S. EPA, 2011) and listed in Table 5-24.

The ancillary data used to estimate N_2O emission for manure composting are the N_2O emission factors (Table 5-25).

5.4.2.4 Method

Methane Emissions from Composting

The Tier 2 approach in the IPCC model is adapted with country-specific factors to estimate CH_4 emissions from composting of solid manure. Daily CH_4 emissions are estimated as a function of the volatile solids in manure placed into the storage and the MCF.

Equation 5-28: IPCC Tier 2 Approach for Calculating Methane Emissions from Composting Solid Manure

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{MCF}{100}}$$

Where:

E_{CH4} = Methane emissions per day (kg CH_4 day^{-1})

m = Total dry manure[a] (kg dry manure day^{-1})

VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure (m^3 CH_4 (kg VS)$^{-1}$) (see Table 5-24)

MCF = Methane conversion factor for the manure management system (%)

0.67 = Conversion factor of m^3 CH_4 to kg CH_4

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

The B_0 values for composting solid manure are obtained from the IPCC (2006) and are listed in Table 5-19. Methane conversion factors for different approaches of composting solid manure are obtained from IPCC (2006).

Table 5-24: Methane Conversion Factors for Composting Solid Manure

Animal	Methane Conversion Factor (%)		
	Cool Climate	Temperate Climate	Warm Climate
Manure composting – in vessel	0.5	0.5	0.5
Manure composting – static pile	0.5	0.5	0.5
Manure composting – intensive windrow	0.5	1	1.5
Manure composting – passive windrow	0.5	1	1.5

Source: IPCC (2006).

Nitrous Oxide Emissions from Composting

A Tier 2 IPCC model is adapted to estimate N_2O emissions from composting of solid manure. Equation 5-29 presents the equation for estimating N_2O emissions from composting of solid manure. Emission factors for different composting methods are listed in Table 5-25 and total nitrogen is estimated according to Equation 5-9.[11]

Equation 5-29: IPCC Tier 2 Approach for Estimating N_2O Emissions from Composting of Solid Manure

$$\mathbf{E_{N_2O} = m \times EF_{N2O} \times TN \times \frac{44}{28}}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

m = Total dry manure[a] (kg day^{-1})

EF_{N2O} = N_2O emission (loss) relative to total nitrogen in manure (kg N_2O-N (kg TN)$^{-1}$)

TN = Total nitrogen in the initial (fresh) manure (kg TN (kg dry manure)$^{-1}$)

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

Table 5-25: N_2O Conversion Factors (EF_{N2O}) for Composting Solid Manure

Category	N_2O Emission Factor (kg N_2O-N/ kg TN)
Cattle and Swine Deep Bedding (Active Mix)	0.07
Cattle and Swine Deep Bedding (No Mix)	0.01
Pit Storage Below Animal Confinements	0.002

Source: IPCC (2006).

[11] Some studies have been conducted on the rate of N_2O emissions for swine (Fukummoto et al., 2003; Szanto et al., 2006) but this data is limited and further research is necessary. See Section 0 Research Gaps for further discussion.

5.4.3 Aerobic Lagoon

5.4.3.1 Overview of Aerobic Lagoons

Method for Estimating Emissions from Manure Storage and Treatment – Aerobic Lagoon

Methane

- The MCF for aerobic treatment is negligible and is designated as zero percent in accordance with the IPCC Guidance.

Nitrous Oxide

- IPCC Tier 2 method utilizing IPCC emission factors.
- Method takes into account the volume of the lagoon and the total nitrogen content of the manure.
- Method is the only readily available method.

Aerobic lagoons are man-made outdoor basins that hold animal wastes. The aerobic treatment of manure involves the biological oxidation of manure as a liquid, with either forced or natural aeration. Natural aeration is limited to aerobic lagoons with photosynthesis and is consequently shallow to allow for oxygen transfer and light penetration. These systems become anoxic during low-sunlight periods. Due to the depth limitation, naturally aerated aerobic lagoons have large surface area requirements and are impractical for large operations.

The IPCC Tier 2 methodology is provided for estimating CH_4 and N_2O emissions from aerobic lagoons. This methodology uses a combination of IPCC and country-specific emission factors from the U.S. EPA GHG Inventory. Aerobic conditions result in the oxidation of carbon to CO_2, not the reduction of carbon to CH_4, thus CH_4 emissions from aerobic lagoons is considered negligible and is designated as zero in accordance with IPCC. The method for calculating N_2O emissions accounts for the volume of the lagoon as well as the total nitrogen content of the manure.

5.4.3.2 Rationale for Selected Methods

The IPCC equations are the only available methods for estimating CH_4, and N_2O emissions from aerobic lagoons. These methodologies best describe the quantitative relationship among activity data at the entity level.

5.4.3.3 Activity Data

No activity data are needed (MCF=0) for the estimation of CH_4 gas emissions.

In order to estimate the daily N_2O emissions, the following information is needed:

- Surface area of lagoon
- Volume of the material in the lagoon
- Total nitrogen content of the manure

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9. The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.3.4 Ancillary Data

The ancillary data used to estimate N_2O emissions for aerobic lagoon are N_2O emission factors (U.S. EPA, 2011).

5.4.3.5 Method

Methane Emissions from Aerobic Lagoon

The MCF for aerobic treatment is negligible and was designated as zero percent in accordance with the IPCC (2006). The solids from the bottom of the lagoon have significant volatile solids and B_0 associated with livestock type; the characteristics of the solids should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.

Nitrous Oxide Emissions from Aerobic Lagoon

The Tier 2 approach in the IPCC model is adapted to estimate N_2O emissions from aerobic lagoons. The N_2O conversion factors for different aeration system are listed in Table 5-26. The estimation method for N_2O emissions is provided in Equation 5-30.

Table 5-26: N_2O Conversion Factors (EF_{N2O}) for Aerobic Lagoons

Aeration Type	N_2O Conversion Factor (kg N_2O-N/kg N)
Natural aeration	0.01
Forced aeration	0.005

Source: IPCC (2006).

Equation 5-30: Calculating N_2O emissions from Aerobic Lagoons

$$\mathbf{E_{N_2O} = V \times EF_{N2O} \times TN \times \frac{44}{28}}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

V = Total volume of the lagoon liquid (m^3 day^{-1})

EF_{N2O} = Nitrous oxide emission (loss) relative to total nitrogen in the lagoon liquid (kg N_2O-N (kg TN)$^{-1}$)

TN = Total nitrogen in the lagoon liquid (kg TN m^{-3})

$\frac{44}{28}$ = Conversion of N_2O-N emissions to N_2O emissions

5.4.4 Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks

5.4.4.1 Overview of Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks

Method for Estimating Emissions from Manure Storage and Treatment – Anaerobic Lagoons, Runoff Holding Ponds, Storage Tanks

Methane

- Sommer model (Sommer et al., 2004) is used with degradable and nondegradable fractions of volatile solids from Møller et al. (2004).
- This method was selected as it accounts for manure temperature and total volatile solids content of manure. Volatile solids content can be obtained from sampling and lab testing.

Nitrous Oxide

- Emissions are a function of the exposed surface area and U.S.-based emission factors.
- Method is the only readily available option.

The most frequently used liquid manure storage systems are anaerobic lagoons (in the Southern portion of the United States), earthen or earthen-lined storages (in the Northern portion of the country), runoff holding ponds, and above-grade storage tanks. Anaerobic lagoons are earthen basins that provide an environment for anaerobic digestion and storage of animal waste. Both the American Society of Agricultural and Biological Engineers and U.S. Department of Agriculture Natural Resources Conservation Service have engineering design standards for construction and operation of anaerobic lagoons. In most feedlots a holding pond is constructed to collect runoff for short-term storage. Storage tanks range from lower-cost earthen basins to higher-cost, glass-lined steel tanks. The manure that enters these systems is usually diluted with flush water, water wasted at stalls, and rainwater.

All of these storage systems (without aeration) are biologically-anaerobic lagoons, which mean that they have similar potential, as with enteric fermentation, to produce CH_4 and N_2O. Due to the large quantity of liquid manure produced in the United States, liquid manure storage can be a major source of GHG emissions from animal operations. In terms of estimation of GHG emission from anaerobic lagoon/runoff holding pond/storage tanks, these storage systems are classified into four categories: 1) covered storage with a crust formed on the surface; 2) covered storage without a crust formed on the surface; 3) uncovered storage with a crust formed on the surface; and 4) uncovered storage without a crust formed on the surface.

The algorithms for calculating CH_4 emissions described by Sommer et al. (2004) are recommended for estimating emissions at the entity-level. The model considers volatile solids to be the main factor influencing emissions from manure and relates emissions to the content of degradable volatile solids. Nitrous oxide is estimated as a function of the exposed surface area of the manure storage and whether a crust is present on the surface.

Rationale for Selected Methods

The Sommer algorithms link carbon turnover, volatile solids, temperature, and storage time to CH_4 emissions estimates and is the best available method for estimating CH_4 emissions at the entity level. The method provided for N_2O is the only available method for estimating emissions. These methodologies best describe the quantitative relationship among activity data at the entity level.

5.4.4.2 Activity Data

In order to estimate the daily CH_4 emissions, the following information is needed:

- Animal type
- Total dry manure
- Volatile solids in the storage
- Temperatures (local ambient temperature and manure temperature)

In order to estimate the N_2O emission, the following information is needed:

- Total dry manure
- Total nitrogen content of the manure
- The exposed surface area of the manure storage

The total nitrogen content of the manure entering storage systems can be estimated according to the nitrogen balance method as described in Equation 5-9. The fraction of nitrogen excreted by an animal that is not emitted as a gas is the portion that enters storage.

5.4.4.3 Ancillary Data

The ancillary data used to estimate CH_4 emissions for anaerobic lagoons, runoff holding ponds, and storage tanks are the maximum CH_4 producing capacities (B_0), potential CH_4 yield ($E_{CH4,\,pot}$), rate correcting factors (b_1 and b_2), Arrhenius constant (A), activation energy (E), gas constant (r), and collection efficiency (η) for liquid manure storage from different animals. These data are available from the IPCC (2006) and Sommer et al. (2004) and are listed in Table 5-27.

The ancillary data used to estimate N_2O emissions for anaerobic lagoons, runoff holding ponds, and storage tanks is the N_2O emission factor from Table 5-29 (U.S. EPA, 2011).

5.4.4.4 Method

Methane Emissions from Anaerobic Lagoons, Runoff Holding Ponds, Storage Tanks

The Sommer model (Sommer et al., 2004) is used as the estimation method for CH_4 emission (Rotz et al., 2011b). Daily CH_4 emissions are estimated as a function of manure temperature and the volatile solids in manure placed into liquid storages. The parameters for the estimation are listed in Table 5-28.

Equation 5-31: Using the Sommer Model to Calculate Daily CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times 0.024 \times (VS_d \times b_1 + VS_{nd} \times b_2) \times e^{\ln(A) - \frac{E}{RT}} \times (1 - \eta)}$$

Where:

E_{CH4}	= Methane emission per day (kg CH_4 day^{-1})
m	= Total dry manure per day (kg dry manure day^{-1})[a]
0.024	= Dimensionless factor to modify the Sommer model based on VS
VS_d and VS_{nd}	= Degradable and nondegradable VS in the manure, respectively (kg (kg dry manure)$^{-1}$)
b_1 and b_2	= Rate correcting factors (dimensionless)
A	= Arrhenius parameter (g CH_4 (kg VS)$^{-1}$ hr^{-1})
E	= Activation energy (J mol^{-1})
R	= Gas constant (J K^{-1} mol^{-1})
T	= Storage temperature (K)
η	= Collection efficiency of different liquid storage categories

[a] Dry manure refers to material remaining after removal of water. It is determined through the evaporation of water from the manure sample at 103-105°C. Forced air oven is the most common equipment to measure the dry matter.

The degradable fraction of the volatile solids is dependent on the potential CH_4 yield and the maximum CH_4 producing capacities and can be calculated using Equation 5-32. The fraction of nondegradable volatile solids (material that is not broken down by microorganisms) is calculated from the total volatile solids content and degradable fraction of the volatile solids, as described by Equation 5-33. The B_0 values are obtained from the IPCC (2006) and are listed in Table 5-19.

Equation 5-32: Calculating the Degradable Fraction of the Volatile Solids

$$\mathbf{VS_d = VS_T \times \frac{B_0}{E_{CH4,pot}}}$$

Where:

VS_d = Degradable VS fractions in the manure on a given day (kg (kg dry manure)$^{-1}$)

VS_T = Volatile solids content in the storage on a given day (kg (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacities (kg CH_4 (kg VS)$^{-1}$)

$E_{CH4,\,pot}$ = Potential CH_4 yield of the manure (kg CH_4 (kg VS)$^{-1}$)

Equation 5-33: Calculating the Non-Degradable Fraction of the Volatile Solids

$$\mathbf{VS_{nd} = VS_T - VS_d}$$

Where:

VS_d and VS_{nd} = Degradable and nondegradable VS fractions in the manure on a given day (kg (kg dry manure)$^{-1}$), respectively

VS_T = Volatile solids content in the storage on a given day (kg (kg dry manure)$^{-1}$)

The collection efficiency (η) depends on different liquid storage categories of: 1) covered storage with a crust formed on the surface; 2) covered storage without a crust formed on the surface; 3) uncovered storage with a crust formed on the surface; and 4) uncovered storage without a crust formed on the surface. A crust allows air and CH_4 to be retained on the surface of the manure storage and increases the potential for oxidation of CH_4 (Hansen et al., 2009; Nielsen et al., 2010). When a crust does not form, CH_4 is directly emitted without rapid oxidation. For cattle slurry and pig slurry, degradable and nondegradable volatile solids (as a fraction of VS_T) are given in Table 5-28.

Table 5-27: Parameters for Estimating CH_4 Emission from Liquid Manure Storage

Parameters		Cattle	Swine
Arrhenius constant (ln(A)) – g CH_4 (kg VS)$^{-1}$ hr^{-1}		43.33	43.21
Activation energy (E) – J mol^{-1}		1.127×10^5	1.127×10^5
Gas constant (R) – J K^{-1} mol^{-1}		8.314	8.314
Rate correction factor for VS_d (b_1)		1	1
Rate correction factor for VS_{nd} (b_2)		0.01	0.01
Potential methane yield of the manure ($E_{CH4,\,pot}$) (kg CH_4/ kg VS)		0.48	0.50
Collection efficiency (η)	Covered storage with a crust form on the surface[a]	1	1
	Covered storage without a crust form on the surface[a]	1	1
	Uncovered storage with a crust form on the surface[b]	0	0
	Uncovered storage without a crust form on the surface[c]	-0.4	-0.4

Source: Sommer et al. (2004) and IPCC (2006).
[a] CH_4 gas from covered storage with a crust form on the surface is collected and flared.
[b] Uncovered storage with a crust form on the surface is used for the derivation of Equation 5-22.
[c] The emission for uncovered storage without a crust is 40 percent greater than uncovered storage with a crust, so the collection efficiency for this case is -40 percent.

Table 5-28: Degradable and Nondegradable Volatile Solids for Cattle and Swine Manure

Type of Manure	VS_d/VS_T	VS_{nd}/VS_T
Cattle liquid manure	0.46	0.54
Swine liquid manure	0.89	0.11

Source: Møller et al. (2004).

Nitrous Oxide Emissions from Anaerobic Lagoon, Runoff Holding Pond, Storage Tanks

Nitrous oxide emissions from liquid manure storage typically represent a relatively small portion of the N_2O emissions from farms. Most studies indicate the criticality of the crust for the formation and emission of N_2O (Petersen and Sommer, 2011). Therefore, N_2O emissions from liquid manure storage are estimated as a function of the exposed surface area of the manure storage and the presence of a crust on the surface.

Equation 5-34: Calculating N_2O Emissions from Liquid Manure Storage

$$E_{N_2O} = EF_{N2O} \times \frac{A_{surface}}{1000}$$

Where:

E_{N2O} = Nitrous oxide emissions per day (kg N_2O day^{-1})

EF_{N2O} = Emission rate of N_2O (g N_2O m^{-2} day^{-1})

$A_{surface}$ = Exposed surface area of the manure storage (m^2)

1,000 = Conversion factor for grams to kilograms $\left(\frac{1\text{ kg}}{1000\text{ g}}\right)$

The emission factor of N_2O is dependent on crust formation on the liquid storage. The crust allows air to be retained on the surface of the manure storage and increases the potential for nitrification and denitrification (Hansen et al., 2009; Nielsen et al., 2010). When a crust does not form, oxygen is not retained on the liquid surface with nitrogenous compounds, and therefore no N_2O is formed and emitted. The emission factors of N_2O for different liquid storage methods are listed in Table 5-29.

Table 5-29: Emission Factor of N_2O for Liquid Storage with Different Crust Formation

Type of Liquid Storage	$EF_{N2O,man}$ (g $N_2O/m^2/day$)
Uncovered liquid manure with crust	0.8
Uncovered liquid manure without crust	0
Covered liquid manure	0

Source: Rotz et al. (2011a).

5.4.5 Anaerobic Digester with Biogas Utilization

5.4.5.1 Overview of Anaerobic Digester with Biogas Utilization

Method for Estimating Emissions from Manure Storage and Treatment – Anaerobic Digester with Biogas Utilization

Methane

- IPCC Tier 2 using Clean Development Mechanism EFs for digester types to estimate CH_4 leakage from digesters.
- Anaerobic digester systems convert organic matter in manure into CH_4 and subsequently combust CH_4 into CO_2.
- Gas leakage from digesters is the main source of GHG emission.
- Leakage of CH_4 from the anaerobic digester system is estimated.

Nitrous Oxide

- N_2O leakage from digesters is fairly small and negligible.

One of the most commonly discussed waste management alternatives for GHG reduction and energy generation is anaerobic digestion. Anaerobic digestion is a natural, biological conversion process that has been proven effective at converting wet organic wastes into biogas (approximately 60 percent CH_4 and 40 percent CO_2). Biogas can be used as a fuel source for engine-generator sets, producing relatively clean electricity while also reducing some of the environmental concerns associated with manure. The digester can be as simple as a covered anaerobic lagoon (Gould-Wells and Williams, 2004) or as sophisticated as thermophilic or media matrix (attached growth) digesters (Cantrell et al., 2008a). There are a wide variety of anaerobic digestion configurations, such as continuous stirred tank reactor (CSTR), covered lagoon, plug-flow, temperature phased, upflow anaerobic sludge blanket (UASB), packed-bed, and fixed film. The digestion is also categorized based on culture temperature: thermophilic digestion in which manure is fermented at a temperature of around 55°C, or mesophilic digestion at a temperature of around 35°C. Among these technologies, CSTR, plug-flow, and covered lagoon, all under mesophilic conditions, are the most often-used methods.

During anaerobic digestion, a group of microbes work together to convert organic matter into CH_4, CO_2, and other simple molecules. The main advantages of applying anaerobic digestion to animal manures are odor reduction, electricity generation, and the reduction of GHG emissions and manure-borne pathogens. Anaerobic digestion is also an excellent pre-treatment process for subsequent manure treatment to remove organic matter and concentrate phosphorus. Considering the small amount of N_2O existing in biogas, N_2O emissions are not estimated for the anaerobic digestion of liquid manure.

The challenges associated with anaerobic digestion relate to initial capital cost, operation, and maintenance and other gases that may be generated (e.g., nitric oxides). The economics relate to access to the electrical grid and sufficient green-electricity offsets to make the operation profitable. Profitable conditions are relatively scarce. Finally, the digester sludge must be managed. Another conversion alternative with energy creation potential is thermochemical conversion (Cantrell et al., 2008a). Systems that use thermochemical conversions to syngases, bio-oil, and biochar for electricity and fuel are emerging, but are not yet established.

Since an anaerobic digestion system converts organic carbon in manure into CH_4 and subsequently combusts CH_4 into CO_2, the GHG emissions from manure anaerobic digestion operation are mainly from the leakage of digesters. The leakage of CH_4 can be estimated based on the IPCC Tier 2 approach in combination with technology-specific emission factors.

5.4.5.2 Rationale for Selected Method

The IPCC equation is the only available method for estimating CH_4 emission from digesters. This methodology best describes the quantitative relationship among activity data at the entity level and takes into account the specific technology employed.

5.4.5.3 Activity Data

In order to estimate the CH_4 leakage from anaerobic digestion, the following information is needed:

- Animal type
- Total dry manure into the digester
- Volatile solids in the manure
- Digester temperatures

5.4.5.4 Ancillary Data

Ancillary data for anaerobic digestion effluent are needed for further estimation of CH_4 and N_2O emissions from post-treatment approaches such as aerobic or anaerobic lagoons, nutrient removal operations, etc. Thus, the necessary data for the effluent include effluent flow rate, total solids, volatile solids, chemical oxygen demand, effluent temperature, environmental temperature, liquid/solid separation methods, and total nitrogen.

5.4.5.5 Method

Equation 5-35 describes the IPCC Tier 2 approach for estimating CH_4 emissions for anaerobic digesters. The CH_4 generated from digesters is assumed to be flared or used as a biogas; the only emissions from digesters are from system leakage.

Equation 5-35: IPCC Tier 2 Approach for Estimating CH_4 Emissions

$$\mathbf{E_{CH_4} = m \times VS \times B_0 \times 0.67 \times \frac{EF_{CH_4,leakage}}{100}}$$

Where:

E_{CH4} = CH_4 emissions per day (kg CH_4 day^{-1})

m = Total dry manure per day (kg day^{-1})

VS = Volatile solids (kg VS (kg dry manure)$^{-1}$)

B_0 = Maximum CH_4 producing capacity for manure from different animal (m^3 CH_4 (kg VS)$^{-1}$)

0.67 = Conversion factor from weight to volume of methane (kg CH_4 m^{-3})

$EF_{CH4,\ leakage}$ = Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester (%)

The B_0 values are obtained from the IPCC (2006) and are listed in Table 5-19. The emission factors for the amount of CH_4 leakage by technology are listed in Table 5-30.

Table 5-30: Emission Factors for the Fraction of Methane Leaking from Digesters

Digester Configurations	$EF_{CH4,\ leakage}$ (%)
Digesters with steel or lined concrete or fiberglass digesters with a gas holding system (egg shaped digesters) and monolithic construction	2.8
UASB type digesters with floating gas holders and no external water seal	5
Digesters with unlined concrete/ferrocement/brick masonry arched type gas holding section; monolithic fixed dome digesters	10
Other digester configurations	10

Source: CDM (2012).

5.4.6 Combined Aerobic Treatment Systems

Method for Estimating Emissions from Combined Aerobic Treatment Systems

- Method is to utilize 10 percent of the emissions resulting from estimation of emissions from Liquid Manure Storage and Treatment – Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks.
- Method based on research findings that systems avoid 90 percent of the GHG emissions from standard anaerobic lagoon treatment.

Dealing with the total treatment of wastewater from either swine or dairy is complex, because the liquid and solid phases must be treated. In municipal sewage treatment systems, the wastewater is very dilute so the treatment of the biochemical oxygen demand by aeration is a fundamental process. In contrast, the solids content of livestock wastewater is quite high, as is the biochemical oxygen demand. Consequently, the cost of stabilizing the biochemical oxygen demand with aeration has proven to be uneconomical. A successful solution to this problem was developed by Vanotti et al. (2007), who used polyacrylamide flocculation to remove more than 90 percent of the solids (Vanotti and Hunt, 1999; Vanotti et al., 2002). The solid fraction was then composted (Vanotti, 2006). The remaining liquid was transferred to a separated water tank where it was subsequently aerated (Vanotti and Hunt, 2000; Vanotti et al., 2007; Vanotti and Szogi, 2008). During these two phases of treatment, more than 90 percent of the GHG emissions from standard anaerobic lagoon treatment were avoided (Vanotti et al., 2008). The avoidance was achieved by aerobic treatment of the solids via composting and nitrification/denitrification in the liquid effluent.

After nitrification/denitrification, the treated effluent moves to the settling tank and subsequently into the phosphorus treatment chamber. Here the wastewater, which has low alkalinity, is amended with liquid lime, and the pH is raised to approximately 10. In the presence of high pH and calcium, the phosphorus is precipitated and the pathogens are killed (Vanotti et al., 2003; Vanotti et al., 2005; Vanotti et al., 2009). The treated wastewater is then recycled into the houses. This process provides a healthier environment for the pigs (Vanotti et al., 2009). The system must be operated to ensure proper and timely flushing of the house. The polyacrylamide addition and the solids separation units must be operated properly. Aeration of the nitrification tank must be maintained, as must the addition of liquid lime. The pumps that maintain the internal recycling must also be maintained and operated correctly. This system is the only treatment system to meet and be certified for expansion of swine production in North Carolina.

To estimate emissions for combined aerobic treatment systems, the methodology for anaerobic lagoons, runoff holding ponds, and storage tanks is applied to the system. Gas emissions of CH_4 and N_2O are estimated using 10 percent of the values for emissions from anaerobic lagoon treatment.

5.4.7 Sand-Manure Separation

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Sand/Manure Separation

- No method is provided as GHG emissions are negligible from the sand/manure separation process. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured and used as the inputs to estimate emissions of GHGs for subsequent storage and treatment operations.

Sand is one of the standard materials for dairy cow bedding. It provides superior cow comfort, environment for udder health (and consequently better milk quality), and traction when compared with organic bedding materials. Sand separation systems can be classified as mechanical separation and sedimentation separation. Sedimentation separation uses dilution water and gravity to allow sand to passively settle in sand traps. Due to the high organic material content contained in the settled sand, the sand recovered from the sand trap needs to be drained multiple times and dried prior to reuse. Mechanical sand-manure separation systems use recycled liquid manure and aeration to suspend manure solids, settle sand at the bottom of the separator, and recover the sand using a heavy duty auger. Sand is generally discharged with less than two percent organic matter. The mechanically separated sand can be reused for bedding.

Since sand-manure separation is relatively quick (compared with other storage and treatment methods), GHG emissions from the operation are minimal. The process of separating sand and manure is not assumed to contribute to GHG emissions. After sand-manure separation, the separated liquid manure is treated as the influent for the next step of storage and treatment operations. The various storage and treatment operation options are shown in Figure 5-7. The parameters of volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid manure should be measured, and used as the inputs to estimate emissions of GHGs.

5.4.8 Nutrient Removal

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Nutrient Removal

- Not estimated due to limited quantitative information on GHGs from nitrogen removal processes.

Nitrogen and phosphorus are the primary elements that cause eutrophication in surface waters. With increased Federal, State and local attention on non-point waste sources, more and more animal operations will likely use nutrient removal approaches to treat liquid manure before land application and other uses. Compared to phosphorus, nitrogen in manure contributes to N_2O emission; removing it can significantly alleviate emissions. Nitrogen in manure comprises NH_3, particulate organic nitrogen, and soluble organic nitrogen. Five main nitrogen removal approaches—Biological Nitrogen Removal (BNR), Anamox, NH_3 stripping, ion exchange, and struvite crystallization—have been applied for municipal and industrial wastewater, as well as for animal waste streams. Because N_2O originates from nitrogen sources, quantification of nitrogen removal is important to estimate emissions from animal manure.

Because most nitrogen removal methods for liquid manure are currently in the research and development stage, very little quantitative information is available on the nitrogen removal methods mentioned above for animal manure under different operation conditions. The suggested estimation method is to consider the liquid manure after nutrient removal as the influent for storage and treatment approaches that entities will use to further treat liquid manure. Measurements of volatile solids, total nitrogen, organic nitrogen, and manure temperature of the treated liquid manure are needed to estimate CH_4 and N_2O emissions.

5.4.9 Solid–Liquid Separation

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Solid– Liquid Separation

- No method is provided as GHG emissions are negligible. However, resulting volatile solids, total nitrogen, organic nitrogen, and manure temperature of the separated liquid and solid manure should be measured and used as the inputs to estimate emissions of GHGs and NH_3 for subsequent storage and treatment operations.

Solid–liquid manure separation has been used widely by dairy farms. One purpose of solid–liquid separation is to physically separate and remove the larger solids from liquid manure in order to store and treat them separately. The available commercial methods include gravity sedimentation and mechanical separation (with or without coagulation flocculation). Sedimentation and mechanical separation without coagulation flocculation are the most popular methods used by animal farms. Similar to sand–liquid manure separation, GHG emissions from the operation are minimal; however, separation has an impact on nutrient distribution in separated solid and liquid manure, which will influence GHG emissions from the next stage of manure storage and treatment for solid and liquid manure. The separated liquid manure is treated as the influent for the next step of storage and treatment operations. The possible storage and treatment options are delineated in Figure 5-7.

The parameters of total solids (dry manure), total nitrogen, organic nitrogen, and manure temperature of the separated liquid and solid manure should be measured, and used as the inputs to estimate GHGs emission in the subsequent storage and treatment operations. The distribution of total solids after solid–liquid separation for typical mechanical separators are listed in Table 5-317 (Ford and Fleming, 2002).

Table 5-31: Efficiency of Different Mechanical Solid-Liquid Separation

Separation Technique	Manure Type	Screen Size (mm)	Influent (% DM)	Total Solid Removal Efficiency (%)	Source
Screen					
Stationary inclined screen	Swine	1.0	0.0-0.7	35.2	Shutt et al. (1975)
	Beef	0.5	0.97-4.41	1-13	Hegg et al. (1981)
	Dairy	1.5	3.83	60.9	Chastain et al. (2001)
Vibrating screen	Swine	0.39	0.2-0.7	22.2	Shutt et al. (1975)
	Beef	0.52-1.91	5.5-7.4	4-44	Gilbertson and Nienaber (1978)

Separation Technique	Manure Type	Screen Size (mm)	Influent (% DM)	Total Solid Removal Efficiency (%)	Source
	Beef	0.64-1.57	1.55-3.19	6-16	Hegg et al. (1981)
	Dairy	0.64-1.57	0.95-1.9	8-16	Hegg et al. (1981)
	Swine	0.64-1.57	1.55-2.88	3-27	Hegg et al. (1981)
	Swine	0.10-2.45	1.5-5.4	11-67	Holmberg et al. (1983)
Rotating screen	Beef	0.75	1.56-3.68	4-6	Hegg et al. (1981)
	Dairy	0.75	0.52-2.95	0-14	Hegg et al. (1981)
	Swine	0.75	2.54-4.12	4-8	Hegg et al. (1981)
In-channel flighted conveyor screen	Dairy	3	7.1	4.22	Møller et al. (2000)
	Swine	3	5.66	25.8	Møller et al. (2000)
Centrifugal					
Centrifuge	Beef		7.5	25	Glerum et al. (1971)
Centrisieve	Swine		5-8	30-40	Glerum et al. (1971)
Decanter centrifuge	Beef		6.9	64	Chiumenti et al. (1987)
	Beef		6.0	45	Chiumenti et al. (1987)
	Swine		7.58	66	Glerum et al. (1971)
	Swine		1.9-8.0	47.4-56.2	Sneath et al. (1988)
Liquid cyclone	Swine			26.5	Shutt et al. (1975)
Filtration/pressing					
Roller press	Swine		5.2	17.3	Pos et al. (1984)
	Dairy		4.8	25	Pos et al. (1984)
	Beef		4.5	13.3	Pos et al. (1984)
Belt press	Dairy	1-2	7.1	32.4	Møller et al. (2000)
	Swine	1-2	5.7	22.3	Møller et al. (2000)
Screw press	Swine		5	16	Chastain et al. (1998)
	Swine		1-5	15-30	Converse et al. (1999)
	Dairy		1-10	15.8-47	Converse et al. (1999)
	Dairy		2.6	23.8	Converse et al. (1999)
	Dairy		4.9	33.4	Converse et al. (1999)
Fournier rotary press[a]	Swine			85	Ford and Fleming (2002) Fournier (2010)
Rotary vacuum filter	Swine		7.5	51	Glerum et al. (1971)
Pressure filter	Beef		7	76	Chiumenti et al. (1987)
Continuous Belt Microscreening Unit					
	Swine		2-8	40-60	Fernandes et al. (1988)

[a] With polymer addition.

5.4.10 Constructed Wetland

Globally, constructed wetlands are used for the treatment of wastewaters, capture of sediments,

Method for Estimating Emissions from Liquid Manure Storage and Treatment – Constructed Wetland

- Currently no method is provided to estimate gas emission from constructed wetland of animal manure, although GHG sinks are noted to likely be greater than CH_4 and N_2O emissions, which are considered negligible.

and drainage water abatement (Hammer, 1989; Kadlec and Knight, 1996; Tanner et al., 1997; Hunt et al., 2002; Hunt et al., 2003; Picek et al., 2007; Harrington and McInnes, 2009; Mustafa et al., 2009; Soosaar et al., 2009; Elgood et al., 2010; Harrington and Scholz, 2010; VanderZaag et al., 2010; Chen et al., 2011; Locke et al., 2011; Tanner and Headley, 2011; Tanner and Sukias, 2011; Vymazal, 2011). Constructed wetlands are generally classified as sub-surface or surface flow wetlands (Kadlec and Knight, 1996). The sub-surface wetlands typically consist of wetland plants growing in a bed of highly porous media, such as gravel or wood chips. They are commonly used to improve drainage water quality. These wetlands are generally rectangular in shape and one to two meters in depth. There is lack of agreement about the relative impact of microbial and plant processes in the function of subsurface wetlands, including GHG production and emissions. However, it is accurate to say that plants and microbes are typically interdependently involved (Picek et al., 2007; Zhu et al., 2007; Wang et al., 2008; Faubert et al., 2010; Lu et al., 2010; Tanner and Headley, 2011). The microbial community advances biogeochemical processes (Tanner et al., 1997; Hunt et al., 2003; Zhu et al., 2007; Dodla et al., 2008; Faulwetter et al., 2009), while the plant community advances transported oxygen into the depth of the wetlands, provides root surfaces for rhizosphere reactions, and vents gases to the atmosphere. The plant processes are significantly affected by plant community composition and weather conditions (Towler et al., 2004; Stein and Hook, 2005; Stein et al., 2006; Zhu et al., 2007; Wang et al., 2008; Taylor et al., 2010).

Surface flow wetlands have a much more direct interchange with the atmosphere for the supply of oxygen and nitrogen, as well as the emissions of GHGs. They can be variable in shape and are generally less than 0.5 meters deep. Surface wetlands minimize clogging problems, but they can have significant loss of treatment as a result of channel flow. There are reasonably functional models for wetland design optimized for either carbon or nitrogen removal (Stone et al., 2002; Stone et al., 2004; Stein et al., 2006; Stein et al., 2007a). The management of GHGs (principally CH_4 and N_2O) from treatment wetlands is somewhat similar to managing GHGs in rice (Freeman et al., 1997; Tanner et al., 1997; Fey et al., 1999; Johansson et al., 2003; Mander et al., 2005a; Mander et al., 2005b; Teiter and Mander, 2005; Picek et al., 2007; Maltais-Landry et al., 2009; Wu et al., 2009).

Of particular importance is the maintenance of wetland oxidative/reductive potential conditions sufficiently positive to avoid CH_4 production (Tanner et al., 1997; Insam and Wett, 2008; Seo and DeLaune, 2010). This requires higher levels of oxygen and lower levels of available carbon. It has been reported that the fluxes of N_2O and CH_4 from treatment wetlands are generally below 10 mg N_2O-N m^{-2} d^{-1} and 300 mg CH_4-C m^{-2} d^{-1} (Mander et al., 2005a; Søvik et al., 2006). The management of N_2O emissions is complicated by the fact that nitrates are often present in the wastewaters or drainage waters. This nitrate will be denitrified under the prevailing anaerobic condition of the treatment wetlands—it is one of treatment wetland's critical functions. However, it is important that the preponderance of denitrification proceeds to completion, with the ultimate production of inert di-nitrogen gas. Complete denitrification requires higher carbon/nitrogen ratios

(Klemedtsson et al., 2005; Hwang et al., 2006; Hunt et al., 2007). Thus, there is an important balance between sufficient carbon for complete denitrification and copious carbon that can drive wetlands into the low reduction/oxidation conditions associated with CH_4 production.

Estimation methods are very complicated and case-based. In an approximate estimation manner that considers wetlands very similar to cropland, treatment wetlands of animal manure are GHG sinks more than sources. The CH_4 and N_2O emission from wetland treatment of animal manure could be negligible. The critical activity data include hydraulic load; inflow water composition, especially carbon and nitrogen; pretreatments such as solids removal or nitrification; amendments; and drying cycles. Critical ancillary data include rainfall, temperature, wind speed, storm events, changes in livestock stocking rates, cropping/tillage systems, and fertilization timing/rates.

5.4.11 Thermo-Chemical Conversion

Method for Estimating Emissions from Solid Manure Storage and Treatment – Thermochemical Conversion

- No method is provided as CH_4 and N_2O emissions are considered negligible.

Combustion, the most primitive and exothermic form of thermochemical treatment of livestock waste, has been in use since antiquity; however, its use for large-scale livestock waste treatment has generally been hampered by economic, health, and environmental quality issues (Florin et al., 2009). Principal among these issues has been components that degrade air quality, including GHGs (mainly CO_2). Nonetheless, thermochemical treatment of livestock manure has attributes that continue to attract efforts to make it economically and environmentally effective (Raman et al., 1980; He et al., 2000; He et al., 2001; Ocfemia et al., 2006; Ro et al., 2007; Cantrell et al., 2008a; Cantrell et al., 2008b; Powlson et al., 2008; Cantrell et al., 2009; Dong et al., 2009; Jin et al., 2009; Ro et al., 2009; Xiu et al., 2009; Cantrell et al., 2010a; Cantrell et al., 2010b; Stone et al., 2010; Wang et al., 2011; Xiu et al., 2011).

Recently, pyrolysis/gasification has received much interest for its treatment of livestock waste. There have also been advances in the cleaning of exhaust gases (He et al., 2001; Ro et al., 2007; Cantrell et al., 2008a; Dong et al., 2009; Xiu et al., 2009; Xiu et al., 2011). Pyrolysis/gasification offers three principal end products: syngas, bio-oil, and biochar (Cantrell et al., 2008a; Xiu et al., 2011). The quality and quantity of end products will vary with feedstock, exposure time, and pyrolysis/gasification temperature. The syngas can be used for direct combustion or to run an electrical generator (Ro et al., 2010). It can also be used via Fischer-Tropsch conversion for production of liquid fuel (Cantrell et al., 2008a). Pyrolysis/gasification for syngas and eventual liquid fuel production is a very attractive potential business model for specific agricultural fuels.

In terms of GHG emission, treatment of flue gas from combustion and utilization of syngas from pyrolysis/gasification are critical. The thermal processes with a flue gas clean-up unit and syngas utilization unit should minimize the GHG emission from the thermal conversion processes.

In order to estimate the daily emissions of CH_4 and N_2O the following information is needed: type of thermal conversion processes; detailed information on the process, such as with/without flue gas clean-up unit or syngas utilization unit; inflow composition, such as moisture, carbon, and nitrogen; and mass flow through the process, including mass in, flue gas/syngas, and ash/biochar. The measurements can be based on dietary changes or seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure piles and the rapid changes that can occur in chemical and physical composition, frequent measurements are recommended to ensure accuracy of the estimation. The total energy balance of the system should also be known. For instance, the carbon credits of biochar cannot be claimed while ignoring the energy required to create the biochar. The effectiveness of the exhaust gas cleaning process in removing air quality degrading components must be certified.

Due to the nature of thermal conversion, much lower emissions (CH_4 and N_2O,) are generated from the thermal conversion compared with other storage or treatment methods. The CH_4 and N_2O emissions from complete thermal conversion processes are relatively small and negligible.

5.4.12 Limitations and Uncertainty in Manure Management Emissions Estimates

For temporary and long-term storage, composting, and aerobic lagoons, the IPCC Tier 2 methodology is used to estimate CH_4 emissions. The maximum CH_4 production capabilities (B_0) for ruminant animals are U.S. specific values from the U.S. EPA Inventory of U.S. GHG Emissions and Sinks. IPCC estimates that the uncertainty associated with these country-specific factors is ±20 percent. B_0 values for other animal values are IPCC defaults and have an associated uncertainty of ±30 percent. The MCFs provided in the Guidelines for solid, slurry, and solid/slurry manure are from the IPCC Guidance and have an estimated uncertainty of ±30 percent. The B_0 and MCF values provided are intended for use at the national level, thus application of these factors at the entity level may result in higher uncertainty.

A modified Tier 2 approach is provided for estimating CH_4 emissions from anaerobic digesters. The leak rates for different digester types is taken from the Clean Development Mechanism's methodological tool for project and leakage emissions from anaerobic digesters (CDM, 2012). The Clean Development Mechanism's leak rates are based on IPCC (2006), Flesch et al. (2011), and Kurup (2003). The leakage rate taken from Flesch et al. (2011) is based on measurements taken from an Integrated Manure Utilization System installed in Alberta, Canada. The system processes 100 metric tons of manure daily and was the most technologically advanced system available at the time of the study. The studies performed by Kurup (2003) were based on a system located in Kerala, India. No uncertainty estimates are provided for these leak rates; however, the actual leak rate of an entity may differ due to differences in technology, maintenance, or other factors.

The Sommer model (Sommer et al., 2004) is recommended for estimating CH_4 emissions from anaerobic lagoons, runoff holding ponds, and storage tanks. Similar to the IPCC Tier 2 methods used for stockpiles, composting, and aerobic lagoons, the Sommer model requires B_0 values from IPCC. The degradable and nondegradable volatile solids can be calculated using the B_0 and potential CH_4 yield or a default value from Møller at al. (2004). The default values presented are based on typical concentrations on Danish cattle and pig slurries; values do not differentiate between type of cattle or diet of the animal and thus there is higher relative uncertainty associated with using the default values.

Sommer et al. (2004) performed an analysis to determine the sensitivity of emission estimates towards different factors. One factor considered is the effect of slurry storage temperature on CH_4 emissions. Sommer et al. (2004) applied average monthly temperatures for seven different locations (all Nordic countries) at constant volatile solids and management. When compared to the model results for Denmark (which are calibrated to correspond with IPCC methodology), the emissions estimates varied from -1 to +36 percent for pig slurry and -23 to +1 percent for cattle slurry. Given that the climatic conditions of the United States differs from Nordic countries, the variation as a result of slurry storage temperature is expected to be greater.

IPCC methodology or modified methodology is used to estimate the N_2O emissions from temporary stack and long-term storage, composting, and aerobic lagoons. IPCC reports large uncertainties with the default emission factors applied (-50 percent to +100 percent). These emission factors were intended for use at the national level and do not take into account varying temperature, moisture content, aeration, manure nitrogen content, metabolizable carbon, duration of storage, and other aspects of treatment for different entities, thus the uncertainty is expected to be higher than reported by IPCC.

The methods recommend that the user send manure samples to a laboratory to obtain an estimate of the volatile solids, NH_3, and nitrogen content of manure. A measurement of manure characteristics can help minimize uncertainty by providing an entity-specific value that takes into account animal and diet characteristics. If laboratory-tested volatile solids values are not available, default values from the American Society of Agricultural and Biological Engineers (ASABE) can be applied. ASABE provides default manure characteristics based on data from published and unpublished information. These values are arithmetic averages and may not represent the differences in animal age, diet, usage, productivity, and management. There is a higher amount of uncertainty associated with the use of ASABE values but there is no quantified uncertainty provided for these values. Note that within the standard cited below there are equations provided that allow for farm-specific values to be determined based on animal characteristics and diet composition. The table below is intended to provide 'average' values, but where farm data are available, equations should be used in order to provide more estimates that better reflect farm conditions and practices.

Available default values and uncertainty information is included in Table 5-32.

Table 5-32: Available Uncertainty Data for Emissions from Manure Management

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Beef Finishing Cattle		kg dry manure/animal/ day	2.4	-20	20			ASABE (2005)
Total Dry Manure – Beef Cow (confinement)		kg dry manure/animal/ day	6.6	-20	20			ASABE (2005)
Total Dry Manure – Beef Growing calf (confinement)		kg dry manure/animal/ day	2.7	-20	20			ASABE (2005)
Total Dry Manure – Dairy Lactating cow		kg dry manure/animal/ day	8.9	-20	20	8.7	11.3	ASABE (2005)
Total Dry Manure – Dairy Dry cow		kg dry manure/animal/ day	4.9	-20	20	8.8	11.2	ASABE (2005)
Total Dry Manure – Dairy Heifer		kg dry manure/animal/ day	3.7	-20	20			ASABE (2005)
Total Dry Manure – Dairy Veal 118 kg		kg dry manure/animal/ day	0.12	-20	20			ASABE (2005)
Total Dry Manure – Horse Sedentary 500 kg		kg dry manure/animal/ day	3.8	-20	20			ASABE (2005)
Total Dry Manure – Horse Intense exercise 500 kg		kg dry manure/animal/ day	3.9	-20	20			ASABE (2005)
Total Dry Manure – Poultry Broiler		kg dry manure/animal/ day	0.03	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (male)		kg dry manure/animal/ day	0.07	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (females)		kg dry manure/animal/ day	0.04	-20	20			ASABE (2005)
Total Dry Manure – Poultry Duck		kg dry manure/animal/ day	0.04	-20	20			ASABE (2005)
Total Dry Manure – Layer		kg dry manure/animal/ day	0.02	-20	20			ASABE (2005)
Total Dry Manure – Swine Nursery pig (12.5 kg)		kg dry manure/animal/ day	0.13	-20	20			ASABE (2005)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Swine Grow finish (70 kg)		kg dry manure/animal/ day	0.47	-20	20			ASABE (2005)
Total Dry Manure – Swine gestating sow 200 kg		kg dry manure/animal/ day	0.5	-20	20			ASABE (2005)
Total Dry Manure – Swine Lactating sow 192 kg		kg dry manure/animal/ day	1.2	-20	20			ASABE (2005)
Total Dry Manure – Swine Boar 200 kg		kg dry manure/animal/ day	0.38	-20	20			ASABE (2005)
Volatile solids – Beef Finishing cattle	VS	kg VS/kg dry manure	0.81	-25	25			ASABE (2005)
Volatile solids – Beef Cow (confinement)	VS	kg VS/kg dry manure	0.89	-25	25			ASABE (2005)
Volatile solids – Beef Growing calf (confinement)	VS	kg VS/kg dry manure	0.85	-25	25			ASABE (2005)
Volatile solids – Dairy Lactating cow	VS	kg VS/kg dry manure	0.84	-25	25			ASABE (2005)
Volatile solids – Dairy Dry cow	VS	kg VS/kg dry manure	0.85	-25	25			ASABE (2005)
Volatile solids – Dairy Heifer	VS	kg VS/kg dry manure	0.86	-25	25			ASABE (2005)
Volatile solids – Dairy Veal 118 kg	VS	kg VS/kg dry manure		-25	25			ASABE (2005)
Volatile solids – Horse Sedentary 500 kg	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Horse Intense exercise 500 kg	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Poultry Broiler	VS	kg VS/kg dry manure	0.73	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (male)	VS	kg VS/kg dry manure	0.8	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (females)	VS	kg VS/kg dry manure	0.79	-25	25			ASABE (2005)
Volatile solids – Poultry Duck	VS	kg VS/kg dry manure	0.58	-25	25			ASABE (2005)
Volatile solids – Layer	VS	kg VS/kg dry manure	0.73	-25	25			ASABE (2005)
Volatile solids – Swine Nursery pig (12.5 kg)	VS	kg VS/kg dry manure	0.83	-25	25			ASABE (2005)
Volatile solids – Swine Grow finish (70 kg)	VS	kg VS/kg dry manure	0.8	-25	25			ASABE (2005)
Volatile solids – Swine gestating sow 200 kg	VS	kg VS/kg dry manure	0.9	-25	25			ASABE (2005)
Volatile solids – Swine Lactating sow 192 kg	VS	kg VS/kg dry manure	0.83	-25	25			ASABE (2005)
Volatile solids – Swine Boar 200 kg	VS	kg VS/kg dry manure	0.89	-25	25			ASABE (2005)
Total nitrogen at a given day – beef finishing cattle		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – beef cow (confinement)		kg N/kg dry manure	0.03					ASABE (2005)
Total nitrogen at a given day – beef growing calf (confinement)		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy lactating cow		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy dry cow		kg N/kg dry manure	0.05					ASABE (2005)
Total nitrogen at a given day – dairy heifer		kg N/kg dry manure	0.03					ASABE (2005)
Total nitrogen at a given day – dairy veal 118 kg		kg N/kg dry manure	0.13					ASABE (2005)
Total nitrogen at a given day – Horse Sedentary 500 kg		kg N/kg dry manure	0.02					ASABE (2005)
Total nitrogen at a given day – Horse Intense Exercise		kg N/kg dry manure	0.04					ASABE (2005)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total nitrogen at a given day – poultry, broiler		kg N/kg dry manure	0.04					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (male)		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (females)		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – poultry, duck		kg N/kg dry manure	0.04					ASABE (2005)
Total nitrogen at a given day – layer		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – swine nursery pig (12.5 kg)		kg N/kg dry manure	0.09					ASABE (2005)
Total nitrogen at a given day – swine grow finish (70 kg)		kg N/kg dry manure	0.08					ASABE (2005)
Total nitrogen at a given day – swine gestating sow 200 kg		kg N/kg dry manure	0.06					ASABE (2005)
Total nitrogen at a given day – swine lactating sow 192 kg		kg N/kg dry manure	0.07					ASABE (2005)
Total nitrogen at a given day – swine boar 200 kg		kg N/kg dry manure	0.07					ASABE (2005)
Methane Conversion Factor (MCF) [a]– Dairy Cow	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Cattle	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Buffalo	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Market Swine	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Breeding Swine	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Layer (Dry)	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Broiler	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Turkey	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Duck	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Sheep	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Goat	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Horse	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Mule/Ass	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Buffalo	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – In vessel manure composting	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Static pile manure composting	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Intensive windrow	MCF	%		-30	30			IPCC (2006)
Methane Conversion Factor[a] – Passive windrow	MCF	%		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Beef Replacement Heifers	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Replacement	B_o	m^3 CH_4/kg VS	0.17	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Mature Beef Cows	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Steers (>500 lbs)	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Maximum Methane Producing Capacities – Stockers (All)	B_o	m^3 CH_4/kg VS	0.17	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle on Feed	B_o	m^3 CH_4/kg VS	0.33	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Cow	B_o	m^3 CH_4/kg VS	0.24	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle	B_o	m^3 CH_4/kg VS	0.19	-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Buffalo[b]	B_o	m^3 CH_4/kg VS	0.1					IPCC (2006)
Maximum Methane Producing Capacities – Market Swine	B_o	m^3 CH_4/kg VS	0.48	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Breeding Swine	B_o	m^3 CH_4/kg VS	0.48	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (dry)	B_o	m^3 CH_4/kg VS	0.39	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (wet)	B_o	m^3 CH_4/kg VS	0.39	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Broiler	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Turkey	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Duck	B_o	m^3 CH_4/kg VS	0.36	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Sheep	B_o	m^3 CH_4/kg VS	0.19	-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Feedlot sheep	B_o	m^3 CH_4/kg VS	0.36	-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Goat	B_o	m^3 CH_4/kg VS	0.17	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Horse	B_o	m^3 CH_4/kg VS	0.3	-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Mule/Ass	B_o	m^3 CH_4/kg VS	0.33	-30	30			IPCC (2006)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with steel or lined concrete or fiberglass digesters with a gas holding system (egg shaped digesters) and monolithic construction	$EF_{CH4,\ leakage}$	%	2.8					CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – UASB type digesters with floating gas holders and no external water seal	$EF_{CH4,\ leakage}$	%	5					CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with unlined concrete/ferrocement/brick masonry arched type gas holding section; monolithic fixed dome digesters	$EF_{CH4,\ leakage}$	%	10					CDM (2012)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Other digester configurations	$EF_{CH4, leakage}$	%	10					CDM (2012)
Temporary storage of liquid/slurry manure –N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			U.S. EPA (2011)
Long-term storage of solid manure – N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.002	-50	100			U.S. EPA (2011)
Long-term storage of slurry manure – N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			U.S. EPA (2011)
Cattle and Swine Deep Bedding (Active Mix)- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.07					IPCC (2006)
Cattle and Swine Deep Bedding (No Mix)- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.01					IPCC (2006)
Pit Storage Below Animal Confinements- N_2O emission factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.002					IPCC (2006)
Natural aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.01	-50	100			IPCC (2006)
Forced aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	kg N_2O-N/kg N	0.005	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure with a crust[c]	EF_{N2O}	kg N_2O-N/kg N	0.8	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure without a crust[c]	EF_{N2O}	kg N_2O-N/kg N	0	-50	100			IPCC (2006)
N_2O emission factor for liquid storage – covered liquid manure[c]	EF_{N2O}	kg N_2O-N/kg N	0	-50	100			IPCC (2006)
Manure Management – Multiple Sources – collection efficiency, covered storage (with or without crust)	η	Percentage	1					Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage with crust formation	η	Percentage	0					Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage without crust formation	η	Percentage	-0.40					Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_1)	b_1	Dimensionless	1					Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_2)	b_2	Dimensionless	0.01					Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, cattle	A	g CH_4 /kg VS/hr	43.33					Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, swine	A	g CH_4 /kg VS/hr	43.21					Sommer et al. (2004)
Potential methane yield of the manure cattle	$E_{CH4, pot}$ -	kg CH_4/kg VS	0.48					Sommer et al. (2004)
Potential methane yield of the manure - swine	$E_{CH4, pot}$	kg CH_4/kg VS	0.5					Sommer et al. (2004)

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - cattle liquid manure	VS_d/ VS_T	Unitless	0.46					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - swine liquid manure	VS_d/ VS_T	Unitless	0.89					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio Non-degradable volatile solids to total volatile solids - cattle liquid manure	VS_{nd}/ VS_T	Unitless	0.54					Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio non-degradable volatile solids to total volatile solids – swine liquid manure	VS_{nd}/ VS_T	Unitless	0.11					Møller et al. (2004)

[a] The values for methane conversion factor (MCF) vary depending on the temperature and the manure management system. IPCC (2006) provides estimated uncertainty ranges for these MCFs.
[b] There are no data for North America region; the data from Western Europe are used to calculate the estimation. There is no reported uncertainty for this adapted value.
[c] IPCC (2006) reports large uncertainties with default N_2O emission factors. The N_2O EF values vary depending on the animal species and temperature of the manure management system.

5.5 Research Gaps

Research gaps have been identified for animal production systems, covering activity data, as well as key areas that would facilitate more accurate estimation of emissions from enteric fermentation and manure management systems. Recommendations are discussed below.

5.5.1 Enteric Fermentation

Cattle

Future research related to improving emissions estimates should be aimed at expanding the options within existing models to better describe an individual farm system and incorporate more options for mitigation strategies to see how emissions might change with implementation of these strategies as well as consider the interactive effects of multiple strategies.

Beef Cow-Calf, Bulls, Stocker, and Sheep

Key data needs include measurement/prediction of feed intake on pasture, measurement/prediction of CH_4 from grazing animals (larger numbers of animals), and methods by which to characterize range forage and intake under production conditions.

Feedlot

There is a need for equations and models to accurately predict enteric CH_4 emissions from cattle and sheep fed high-concentrate finishing diets.

Dairy

One of the largest research gaps is the lack of basic data related to emissions from calves, heifers, and dry cow housing systems. In addition, there is a need for equally consistent and reliable methods for measuring relative differences in emissions associated with the implementation of a variety of management practices. Further model development for estimating emissions should include an expansion of options to describe the production facility and inclusion of management practices that can be adopted to mitigate emissions.

Swine

Future research related to improving emissions estimates should be aimed at expanding the options within these models to better describe an individual farm system and incorporate more options for management and mitigation strategies to see how emissions might change with implementation of different practices. Minimally, the diet considerations in Holos need to be incorporated into the MANURE model and expanded to reflect production phase.

Poultry

Future research related to improving emissions estimates should be aimed at expanding the options within these models to better describe an individual farm system and incorporate more options for management and mitigation strategies to see how emissions might change with implementation of different practices.

5.5.2 Manure Management

Greenhouse gas emissions from a variety of manure management systems have been developed from a limited number of studies and a limited number of potential variations in management and the environmental conditions around a particular manure management system. The largest deficiency in the current GHG studies is the lack of characterization of the temporal variation in the GHG emissions from different systems and the spatial variation in GHG emissions induced by meteorological conditions among specific locations. In general, the research needed to develop a more complete understanding of the GHG emissions can be summarized as:

- Develop data bases from research observations of commercial facilities that characterize the storage system, time in storage, environmental conditions and location, and the attributes of the manure source, e.g., type of animal, diet, loading rate.
- Utilize the databases to derive simulation models to quantify the GHG emissions from different manure management systems.
- Validate the models using independent observations from manure management systems distributed around the United States.
- Develop operational models capable of being applied to production scale systems which utilize simple parameters as input variables and produce results in agreement with the more complex simulation models.
- Utilize these models to develop potential strategies which could be employed to mitigate GHG emissions from manure management systems.

Temporary Stack and Long-Term Stockpile

Methane emission data from solid storages in different regions under different climates are limited. In order to develop a more accurate model to estimate the CH_4 emission from solid manure storages, in-depth studies are needed to integrate temperature, storage time, storage method, and mass flow with CH_4 emission in different regions. As for N_2O emission, systematically collecting more intense data (a variety of spatial and temporal scales) from different regions will be a good first step toward accurate N_2O emission models. Once these data are collected and used to develop/validate models, work will likely be needed to develop farmer-friendly models using simple farm parameters as input variables, resulting in emissions estimates that are correlated with those of more complex models. For example, these models, if synchronized, could form part of a comprehensive manure stewardship toolkit.

There is a paucity of data on CH_4 and N_2O emissions from open lot (beef feedlots and dairies) pen surfaces and runoff control structures and on the chemical and physical factors controlling those emissions.

Composting

Greenhouse gas emissions data from composting in different regions under different operational conditions are limited. A good first step toward an accurate GHG emissions model would be to collect more data from different regions and different operational conditions. Consequently, in-depth studies integrating compost pile size/surface area, pile shape, aeration rate, storage time, composting temperature, etc., with GHG emissions need to be conducted to develop complex models describing GHG emissions from composting. Furthermore, work will likely be needed to develop farmer-friendly models using simple farm parameters as input variables, resulting in emission estimations that are correlated with those of more complex models.

There have been some studies performed to estimate the emission factors for N_2O from composting manure in different systems and for different livestock categories. (Fukummoto et al., 2003; Szanto et al., 2006) have conducted studies on composting swine manure at specific ambient temperatures. Factors have been presented in the studies but there is significant uncertainty due to the limited data available. Further research is needed to refine these emission factors as well as develop factors for other animals.

Aerobic Lagoon

In-depth studies are needed to integrate lagoon depths, aeration rate, pH, temperature, and nutrient conditions of manure with GHG emissions, which will facilitate the development of comprehensive models to predict GHG emissions under different operational and climate conditions. Simplified and farm-friendly models using farm operational parameters as inputs should be developed to help farms estimate the GHG emissions at the entity level.

Anaerobic Lagoon, Runoff Holding Pond, and Storage Tanks

All models to estimate GHG emissions from liquid manure storage are relatively inaccurate, due to the complexity and variety of livestock manure operations. In order to develop a more accurate model to estimate emissions from liquid manure storages, in-depth studies are needed to integrate manure storage configuration, temperature, storage time, storage method, mass flow, and surface turbulence with emissions in different regions. In addition, systematically collecting more data from different regions will be very helpful to develop more statistically accurate models to estimate GHG emissions.

Anaerobic Digestion

Changes in chemical oxygen demand, volatile solids, total solids, and nitrogen in the anaerobic digestion process are indirectly linked to GHG emissions from post-treatment of anaerobic digestion effluent. The effectiveness of anaerobic digestion at mitigating GHG emissions has been studied intensively. However, anaerobic digestion effluent can lead to GHG emissions. More in-depth studies are needed to develop integrated models that can accurately predict the overall GHG emission from the combination of anaerobic digestion and post-treatment approaches.

Combined Aerobic Treatment Systems

Methods and techniques to reduce the capital and operating costs are needed. There is also a need to develop better ways to conserve and derive energy from the waste material. There is a paucity of data on GHG emissions from these systems and development of emission models will require integration of data characterizing these systems and the climatic conditions in order to develop these models. These models will need to be validated against observed data.

Nutrient Removal

Various methods of nitrogen removal, such as biological nitrogen removal, Anamox, NH_3 stripping, ion exchange, and struvite crystallization, should be investigated at commercial-scale animal operations under different climate conditions. Characteristics of manure, mass flow, and gas emissions should be closely monitored in order to provide the data needed to construct relatively precise estimation models. In addition, further research is needed to pilot innovative beef and dairy GHG emission reduction strategies in feedlots and dairies.

Constructed Wetland

Although there are numerous papers published about various aspects of treatment wetland effectiveness and emissions, there currently is not an established method for calculation of GHG emissions from any of the treatment wetland types. Moreover, there are not sufficient unifying publications to suggest that a reliable method could be established within the scope of this report. A more robust and extensive database on GHG emissions from treatment wetlands is needed. Concomitantly, there is a need for better predictive equation and models.

Thermo-Chemical Conversion

More studies are needed on the effects of thermal conversion of animal manure on GHG emission in order to conclude detailed emission profiles corresponding to different type of manure. These studies would entail detailed observations of the manure conversion system along with GHG emissions and information on the environmental conditions.

Appendix 5-A: Enteric CH_4 from Feedlot Cattle – Methane Conversion Factor (Ym)

As noted in the Beef Production Systems section (Section 5.3.2.2), a modified IPCC (2006) method is proposed to estimate enteric CH_4 emissions from finishing beef cattle. For this report, a baseline scenario based on typical U.S. beef cattle feeding conditions was established and baseline values were set based on published research. To estimate methane emissions, emission values are modified using adjustment factors that are based on changes in animal management and feeding conditions from the baseline scenario. This appendix presents background information on the baseline scenario and adjustment factors.

The following baseline scenarios are established for beef cattle in U.S. feedlots:

1. Medium to large frame steer (or heifer) yearlings are fed a high concentrate finishing diet containing <=10 percent forage in diet dry matter (= to 8 to 18 percent NDF) in dry-lot, soil-surfaced pens.
2. The grain portion of the diet is at least 70 percent of diet dry matter.
3. The grain source is steam flaked (SFC) or high moisture corn (HMC).
4. The dietary crude protein concentration is 12.5 to 13.5 percent of diet dry matter (Vasconcelos and Galyean, 2007).
5. The dietary ruminally degradable protein (DIP or RDP) concentration is 7.5 to 9 percent of diet dry matter (Vasconcelos and Galyean, 2007).
6. The diet contains monensin (Rumensin, Elanco Animal Health) at recommended concentrations (Vasconcelos and Galyean, 2007).
7. Diets for heifers contain melengestrol acetate (MGA) at the recommended concentrations (Vasconcelos and Galyean, 2007).
8. Cattle are implanted with an estrogenic implant throughout the feeding period (Vasconcelos and Galyean, 2007).
9. No beta-agonist is fed.
10. The diet contains no supplemental fat (vegetable oil, yellow grease, etc.) and has a total fat concentration of less than 4.5 percent of diet dry matter.
11. Enteric CH_4 emission is three percent of gross energy intake (GEI: (IPCC, 2006).
12. The dietary forage is chopped alfalfa, sorghum, or grass hay at seven to 10 percent of diet dry matter.
13. The diet contains minerals and vitamins at the recommended level (NRC, 2000).
14. Temperatures are mild/moderate during the feeding period.
15. Cattle are slaughtered at an average body weight of approximately 582 kg (1,280 lb.) (KSU, 2012).
16. Average dressing percent is 61 percent.
17. Cattle are fed 150 days.

The Ym adjustment factors for feedlot cattle fed high-concentrate diets in Table 5-11 were determined based on the following literature reviews and analyses.

Ionophores: On average, the feeding of ionophores decreases DMI by about five percent (Delfino et al., 1988; Vogel, 1995; Robinson and Okine, 2001; Tedeschi et al., 2003) and decreases ADG by about two percent (Delfino et al., 1988; Tedeschi et al., 2003). Feeding ionophores decreases enteric methane emissions approximately 20 percent for the first two to four weeks on feed (Tedeschi et al., 2003; Guan et al., 2006). Therefore, over a 150-day feeding period, overall enteric methane emissions are decreased approximately 4 percent. Because of an increase in the gain:feed ratio, enteric methane emissions per unit of production are decreased when ionophores are fed.

Supplemental Fat: For each one percent increase in supplemental fat (up to a maximum of four percent added fat), enteric methane emissions (as a percentage of gross energy intake) decrease approximately 3.8 to 5.6 percent (Zinn and Shen, 1996; Beauchemin et al., 2008; Martin et al., 2010). A conservative value of four percent per one percent increase in supplemental fat is recommended because many fat sources used in the industry are partially saturated and may have less effect on enteric CH_4 production than the highly unsaturated fats used in most studies. For example if three percent supplemental fat is added to the diet, then CH_4 production is decreased 12 percent (three percent added fat times four percent is equivalent to a 12 percent decrease). The revised enteric CH_4 emission is 2.64 percent of GEI (three percent baseline * 0.88 = 2.64 percent of GEI). Many distiller's grains contain approximately 8 to 12 percent fat. Addition of distiller's grain may serve as a source of supplemental fat, and thus decrease enteric CH_4 (McGinn et al., 2009). However Hales et al. (2013) noted that feeding increasing concentrations of Wet Distillers Grains with Solubles (WDGS) in equal-fat diets increased enteric CH_4, likely due to the increased NDF intake.[12]

Grain processing & Grain source: Grain processing directly affects enteric CH_4 production via its effects on ruminal fermentation. Enteric CH_4 emissions, as a percent of GEI, are 20 percent greater with diets based on DRC than in diets based on steam-flaked corn (SFC) or high moisture corn (HMC) (Archibeque et al., 2006; Hales et al., 2012). More extensive grain processing may also improve the gain:feed ratio about 10 percent (Owens et al., 1997; Zinn and Barajas, 1997) and may, decrease manure CH_4 emissions via decreased fecal starch excretion (Zinn and Barajas, 1997; Hales et al., 2012). Enteric CH_4 emissions are 20 to 40 percent greater with finishing diets based on barley than diets based on corn; presumably because of the lower starch and higher fiber content of barley (Benchaar et al., 2001; Beauchemin and McGinn, 2005). A mean (30%) for these studies is recommended for a barley adjustment factor.

Dietary Forage and Grain Concentration effects: Limited data exists to evaluate effects of dietary forage and grain concentration on enteric methane production from beef cattle that are fed typical U.S-based, high concentrate finishing diets. Equations from Ellis et al. (2007; 2009) illustrate the effects of dietary forage, NDF, and starch on enteric CH_4 production. In particular, the following 10 equations illustrate the relationships:

- CH_4 (MJ/day) = 3.96 + 0.561 × DMI (kg/day)
- CH_4 (MJ/day) = 4.79 + 0.0492 × Forage (%)
- CH_4 (MJ/day) = 5.58 + 0.848 × NDF (kg/day)
- CH_4 (MJ/day) = 5.70 + 1.41 × ADF (kg/day)
- CH_4 (MJ/day) = 2.29 + 0.670 × DMI (kg/day)
- CH_4 (MJ/day) = 4.72 + 1.13 × Starch (kg/day)
- CH_4 (MJ/day) = -1.01 + 2.76 × NDF (kg/day) +0.722 × Starch (kg/day)
- CH_4 (MJ/day) = 2.68 – 1.14 (Starch:NDF) + 0.786 × DMI (kg/day)
- CH_4 (MJ/day) = 2.50 = 0.367 × Starch (kg/day) + 0.766 × DMI (kg/day)
- CH_4 (MJ/day) = 2.70 + (1.16 × DMI (kg/day)) – (15.8 × ether extract (kg/day))

[12] Ym is adjusted for distiller grains by changes in fat content and grain concentration. For example, a 30 percent concentration of distiller grains in the finishing diet will typically increase the dietary fat level by 2 to 3 percent and decrease the grain content by 25 to 30 percent. The resulting change in Ym is a decrease by 8 percent to account for increase in fat content and an increase of 10 percent to account for a decrease in grain content (i.e., Ym = 3% x 0.92 x 1.10 = 3.036 %).

To develop adjustment factors for grain concentrations in diets, artificial data sets were created that varied in forage (range of 5 to 25 percent), NDF (range 10 to 20 percent), fat (range of 3 to 6 percent), and starch (range of 30 to 60 percent of diet dry matter) content. Using these data sets, enteric CH_4 emissions were estimated using the appropriate equation(s) of Ellis et al. (2007; 2009). Effects of dietary changes on enteric CH_4 were then determined by linear regression analysis. On average, enteric CH_4 production (MJ/day) increased five percent for each one percent increase in dietary forage concentration; increased 13 percent for each one kg increase in dietary NDF intake, increased five percent for each one kg increase in starch intake and decreased five percent for each one unit increase in the dietary starch:NDF ratio. Small increases in forage concentration from the baseline value had small effects on Ym; whereas, greater increases had a larger effect (Hales et al., 2012; Hales et al., 2014). An evaluation of these factors indicated an enteric CH_4 Ym adjustment factor of 10% for small increases in forage (and decreases in grain concentration) and a larger correction factor of 40 percent for greater changes (diet concentrate less than 45 percent). These factors are recommended for accounting for the grain concentration in finishing diets.

No Ym adjustment factor was explicitly modeled to account for the following dietary management factors:[13]

- Beta-agonists: Beta-agonists do not directly affect the Ym (i.e., enteric CH_4 emissions per unit of gross energy intake), therefore no adjustment factor is recommended. However, because of a 4 percent increase in feed efficiency, a 2.5 to 3.5% increase in hot carcass weight (HCW), and an increase in live body weight (Vasconcelos et al., 2008; Elam et al., 2009; Montgomery et al., 2009; Delmore et al., 2010; Radunz, 2011), enteric CH_4 emissions per unit of production are decreased when beta-agonists are fed.
- Melengestrol acetate (MGA: heifers only): Feeding MGA to heifers does not directly affect enteric CH_4 emissions. However, because of a nine percent increase in the gain:feed ratio (Hill et al., 1988; Kreikemeier and Mader, 2004) enteric CH_4 emissions per unit of production are decreased when MGA is fed.
- Direct Fed Microbials: Most direct fed microbials do not appear to directly affect enteric CH_4 emissions and effects on animal performance are somewhat variable (Krehbiel et al., 2003). No adjustment factor is recommended for the feeding of direct fed microbials.
- Dietary Crude Protein and Ruminal Degradable Protein (RDP): Dietary protein may potentially affect animal performance and enteric CH_4 emissions via effects on ruminal fermentation. However, there is no readily available data with modern feedlot diets with which to compare (Berger and Merchen, 1995; Robinson and Okine, 2001; Gleghorn et al., 2004; Cole et al., 2006; Wagner et al., 2010). There is no recommended Ym adjustment factor for dietary protein. Dietary protein may affect emissions of manure greenhouse gases (N_2O) and definitely affects NH_3 emissions (Todd et al., 2013).
- Implanting regimens: Implants do not directly affect enteric CH_4 emissions. However because of an increase in feed efficiency, live body weight, and HCW (Herschler et al., 1995; Robinson and Okine, 2001; Wileman et al., 2009), enteric CH_4 emissions per unit of production are decreased when implants are used.
- Ambient temperature: Cold and hot temperatures may potentially affect enteric CH_4 emission due to effects on feed intake, ruminal digestion and rate of passage (Young, 1981); however, the actual effects are not clear. Therefore no adjustment factor for environmental temperature is used. Cold temperatures may decrease CH_4, N_2O and NH_3 losses from pen

[13] Although these management factors are not modeled to impact Ym, some of them do impact enteric CH_4 per unit of production. Hence, in evaluating methane intensity per unit of production, these factors would have an impact.

surfaces via effects on microbial activity in the manure. Conversely, warm temperatures may increase emissions from manure via increased microbial activity.

The IPCC Tier 2 model is currently the most useful for predicting emissions from cow-calf and stocker production, as well, as noted in the earlier cow-calf and stocker Sections (5.3.2.2). Enteric emissions from all cattle other than dairy cows and dairy heifers are estimated using the IPCC Tier 2 equation or the modified IPCC Tier 2 previously discussed for feedlot cattle. To use these equations, it is necessary to make sure the inputs to the equations are as accurate as possible. For DE (as a percentage of GE), we recommend using the feedstuffs composition table provided in NRC (1989) and Ewan (1989). Several feedstuffs from the table are included in Table 5-C-1. After review of the models, their strengths and limitations, models based on the Mills equations (e.g., DairyGEM, COWPOLL, IFSM) appear to be the most useful for predicting emissions from dairy cattle. The Mits3 equation recommended for calculating enteric CH_4 emissions from dairy cows and dairy heifers (used in DairyGEM/IFSM) requires different dietary input information than that required for the IPCC Tier II equation. Specifically, DairyGEM/IFSM requires the starch and ADF content of feeds. Because starch is nearly equivalent to NFC (which is starch + sugar + pectin) in high forage diets (dairy diets), we use NFC in the Mits3 equation (NFC = 100 – (NDF + CP + EE + Ash)). These values can be found in Appendix 5-B.

Appendix 5-B: Feedstuffs Composition Table

This table provided data inputs for enteric fermentation emissions calculations for cattle and sheep.

Table 5-B-1: Feedstuffs Composition Table (Preston 2013, except where noted for digestible energy)

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g	NE_l	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Alfalfa Cubes	x91	57	57	25	57		18	30	29	36	46	40	2.0	11	1.30	0.23	1.9	0.37	0.33	20
Alfalfa Dehydrated 17% CP	92	61	62	31	61	65.16	19	60	26	34	45	6	3.0	11	1.42	0.25	2.5	0.45	0.28	21
Alfalfa Fresh	24	61	62	31	61	62.54[b]	19	18	27	34	46	41	3.0	9	1.35	0.27	2.6	0.40	0.29	18
Alfalfa Hay Early Bloom	90	59	59	28	59	63.72	19	20	28	35	45	92	2.5	8	1.41	0.26	2.5	0.38	0.28	22
Alfalfa Hay Midbloom	89	58	58	26	58	61.79	17	23	30	36	47	92	2.3	9	1.40	0.24	2.0	0.38	0.27	24
Alfalfa Hay Full Bloom	88	54	54	20	54	55.71	16	25	34	40	52	92	2.0	8	1.20	0.23	1.7	0.37	0.25	23
Alfalfa Hay Mature	88	50	50	12	49	54.18	13	30	38	45	59	92	1.3	8	1.18	0.19	1.5	0.35	0.21	23
Alfalfa Seed Screenings	91	84	92	61	87		34		13	15			10.7	6	0.30	0.67				
Alfalfa Silage	30	55	55	21	55	60.71[c]	18	19	28	37	49	82	3.0	9	1.40	0.29	2.6	0.41	0.29	26
Alfalfa Silage Wilted	39	58	58	26	58	60.71[d]	18	22	28	37	49	82	3.0	9	1.40	0.29	2.6	0.41	0.29	26
Alfalfa Leaf Meal	89	60	60	30	60		26	15	16	24	34	35	3.0	10	2.88	0.34	2.2		0.32	39
Alfalfa Stems	89	47	47	7	46		11	44	44	51	68	100	1.3	6	0.90	0.18	2.5			
Almond Hulls	89	56	56	23	56	59.90	3	60	16	29	36	100	3.1	7	0.24	0.10	2.0	0.03	0.07	20
Ammonium Chloride	99	0	0	0	0		163	0	0	0	0	0	0.0		0.00	0.00	0.0	66.00	0.00	0
Ammonium Sulfate	99	0	0	0	0		132	0	0	0	0	0	0.0						24.15	
Apples	17	70	73	44	71		3	10	7	9	25	10	2.2	2	0.06	0.60	0.8			
Apple Pomace Wet	20	68	70	41	69		5	10	18	27	36	27	5.2	3	0.13	0.12	0.5		0.04	11
Apple Pomace Dried	89	67	69	40	68	56.69	5	15	18	28	38	29	5.2	3	0.13	0.12	0.5		0.04	11
Artichoke Tops (Jerusalem)	27	61	62	31	61		6		18	30	41	40	1.1	10	1.62	0.11	1.4			
Avocado Seed Meal	91	52	52	16	51		20		19	24			1.2	16						
Bahiagrass Hay	90	53	53	18	53	54.85	6	37	32	41	72	98	1.8	7	0.47	0.20	1.4		0.21	
Bakery Product Dried	90	90	100	68	94	81.31	11	30	3	9	30	0	11.5	4	0.16	0.27	0.4	2.25	0.15	33
Bananas	24	84	92	61	87		4		4	5			0.8	3	0.03	0.11	1.5			8
Barley Hay	90	57	57	25	57	60.89	9		28	37	65	98	2.1	8	0.30	0.28	1.6		0.19	25
Barley Silage	35	59	58	26	58		12	22	34	37	58	61	3.0	9	0.46	0.30	2.4		0.22	28
Barley Silage Mature	35	58	58	26	58		12	25	30	34	50	61	3.5	9	0.30	0.20	1.5		0.15	25
Barley Straw	90	44	44	1	43	43.98	4	70	42	55	78	100	1.9	7	0.32	0.08	2.2	0.67	0.16	7
Barley Grain	89	84	92	61	87		12	28	5	7	20	34	2.1	3	0.06	0.38	0.6	0.18	0.16	23
Barley Grain, Steam Flaked	85	90	100	70	100		12	39	5	7	20	30	2.1	3	0.06	0.35	0.6	0.18	0.16	23

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Barley Grain Steam Rolled	86	84	92	61	87		12	38	5	7	20	27	2.1	3	0.06	0.41	0.6	0.18	0.17	30
Barley Grain 2-row	87	84	92	61	87		12		6	8	24	34	2.3	2	0.05	0.31	0.6	0.18	0.17	
Barley Grain 6-row	87	84	92	61	87		11		6	8	24	34	2.2	3	0.05	0.36	0.6	0.18	0.15	
Barley Grain Lt. Wt. (42-44 lb/bu)	88	78	83	54	80		13	30	9	12	30	34	2.3	4						
Barley Feed Pearl Byproduct	90	74	78	49	76		15	25	12	15			3.9	5	0.05	0.45	0.7		0.06	
Barley Bran	91	59	59	28	59		12	28	21	27	36	6	4.3	7						
Barley Grain Screenings	89	71	74	46	73		12		9	11			2.6	4	0.35	0.33	0.9		0.15	
Beans Navy Cull	90	84	92	61	87	84.52	24	25	5	8	20	0	1.4	5	0.15	0.60	1.4	0.06	0.26	45
Beet Pulp Wet	17	77	82	53	79	75.09	9	35	20	25	45	30	0.7	5	0.65	0.08	0.9	0.40	0.22	21
Beet Pulp Dried	91	76	81	52	78	79.81	9	44	21	26	46	33	0.7	5	0.65	0.08	0.9	0.40	0.22	21
Beet Pulp Wet with Molasses	24	77	82	53	79		11	25	16	21	39	33	0.6	6	0.60	0.10	1.8		0.42	11
Beet Pulp Dried with Molasses	92	77	82	53	79	82.52	11	34	17	23	40	33	0.6	6	0.60	0.10	1.8		0.42	11
Beet Root (Sugar)	23	80	86	56	83		4		5	7	16		0.4	3						
Beet Tops (Sugar)	19	58	58	26	58		14		11	14	25	41	1.3	24	1.10	0.22	5.2	0.20	0.45	20
Beet Top Silage	25	52	52	16	51		12		12				2.0	32	1.38	0.22	5.7		0.57	20
Bermudagrass Coastal Dehydrated	90	62	63	33	63		16	40	26	29	40	10	3.8	7	0.40	0.25	1.8	0.72	0.23	18
Bermudagrass Coastal Hay	89	56	56	23	56	53.05	10	20	30	36	73	98	2.1	6	0.47	0.21	1.5	0.70	0.22	16
Bermudagrass Hay	89	53	53	18	53	50.79	10	18	29	37	72	98	1.9	8	0.46	0.20	1.5	0.70	0.25	31
Bermudagrass Silage	26	50	50	12	49		10	15	28	35	71	48	1.9	8	0.46	0.20	1.5	0.72	0.25	31
Birdsfoot Trefoil Fresh	22	66	68	38	67		21	20	21	31	47	41	4.4	9	1.78	0.25	2.6		0.25	31
Birdsfoot Trefoil Hay	89	57	57	25	57		16	22	31	38	50	92	2.2	8	1.73	0.24	1.8		0.25	28
Biuret	99	0	0	0	0		248	0	0	0	0	0	0.0	0	0.00	0.00	0.0	0.00	0.00	0
Blood Meal, Swine/Poultry	91	66	68	38	67		92	82	1	2	10	0	1.4	3	0.32	0.28	0.2	0.30	0.70	22
Bluegrass KY Fresh Early Bloom	36	69	71	43	70	75.62	15	20	27	32	60	41	3.9	7	0.37	0.30	1.9	0.42	0.19	25
Bluegrass Straw	93	45	45	3	44		6		40	50	78	90	1.1	6	0.20	0.10				
Bluestem Fresh Mature	61	50	50	12	49	56.82	6		34				2.5	5	0.40	0.12	0.8		0.05	28
Bread Byproduct	68	90	100	68	94		14	24	1	2	3	0	3.0	3	0.10	0.18	0.2	0.76	0.15	40
Brewers Grains Wet	23	85	93	62	88	62.66	26	52	13	21	45	18	7.5	4	0.30	0.58	0.1	0.15	0.32	78
Brewers Grains Dried	92	84	92	61	87	60.43	25	54	14	24	49	18	7.5	4	0.30	0.58	0.1	0.15	0.32	78
Brewers Yeast Dried	94	79	85	55	81		48		3				1.0	7	0.10	1.56	1.8		0.41	41
Bromegrass Fresh Immature	30	64	65	36	65	78.57	15	22	28	33	54	40	4.1	10	0.45	0.34	2.3		0.21	20

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g	NE_l	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Bromegrass Hay	89	55	55	21	55	62.19[e]	10	33	35	41	66	98	2.3	9	0.40	0.23	1.9	0.40	0.19	19
Bromegrass Haylage	35	57	57	25	57		11	26	36	44	69	61	2.5	8	0.38	0.30	2.0		0.20	19
Buckwheat Grain	88	75	79	50	77	72.27	12		13	17			2.8	2	0.11	0.36	0.5	0.05	0.16	10
Buttermilk Dried	92	88	98	65	91		34	0	5	0	0	0	5.0	10	1.44	1.00	0.9		0.09	44
Cactus, Prickly Pear	23	61	62	31	62		5		16	20	28		2.1	18	4.00	0.10	1.5		0.20	
Calcium Carbonate	99	0	0	0	0		0		0	0	0	0	0.0	99	38.50	0.04	0.1		0.00	0
Canarygrass Hay	91	53	53	18	53		9	26	32	34	67	98	2.7	8	0.38	0.25	2.7		0.14	18
Canola Meal, Solv. Ext.	90	72	75	47	74		41	30	11	19	29	23	2.0	8	0.74	1.14	1.1	0.07	0.78	68
Carrot Pulp	14	62	63	33	63		6		19	23	40	0	7.8	9						
Carrot Root Fresh	12	83	90	60	86	92.29	10		9	11	20	0	1.4	10	0.55	0.32	2.5	0.50	0.17	
Carrot Tops	16	73	77	48	75		13		18	23	45	41	3.8	15	1.94	0.19	1.9			
Cattle Manure Dried	92	38	40	0	36	30.58	15		35	42	55	0	2.5	14	1.15	1.20	0.6		1.78	240
Cheatgrass Fresh Immature	21	68	70	41	69		16		23				2.7	10	0.60	0.28				
Citrus Pulp Dried	90	78	83	54	80		7	38	13	20	21	33	2.9	7	1.81	0.12	0.8	0.04	0.08	14
Clover Ladino Fresh	19	69	71	43	70	73.22	25	20	14	33	35	41	4.8	11	1.27	0.38	2.4		0.20	20
Clover Ladino Hay	90	61	62	31	61	63.40	21	25	22	32	36	92	2.0	9	1.35	0.32	2.4	0.30	0.20	17
Clover Red Fresh	24	64	65	36	65		18	21	24	33	44	41	4.0	9	1.70	0.30	2.0	0.60	0.17	23
Clover Red Hay	88	55	55	21	55	58.33	15	28	30	39	51	92	2.5	8	1.50	0.25	1.7	0.32	0.17	17
Clover Sweet Hay	91	53	53	18	53		16	30	30	38	50	92	2.4	9	1.27	0.25	1.8	0.37	0.46	
Coconut Meal, Mech. Ext.	92	76	81	52	78	79.66	21	56	13	21	56	23	6.8	7	0.40	0.30	1.0	0.33	0.04	
Coffee Grounds	88	20	36	0	16		13		41	68	77	10	15.0	2	0.10	0.08				
Corn Whole Plant Pelleted	91	63	64	34	64		9	45	21	24	40	6	2.4	6	0.50	0.24	0.9		0.14	
Corn Fodder	80	65	66	37	66		9	45	25	29	48	100	2.4	7	0.50	0.25	0.9	0.20	0.14	
Corn Stover Mature (Stalks)	80	54	54	20	54		5	30	35	43	70	100	1.3	7	0.45	0.15	1.2	0.30	0.14	22
Corn Silage, Milk Stage	26	65	66	37	66		8	18	26	32	54	60	2.8	6	0.40	0.27	1.6		0.11	20
Corn Silage, Mature Well Eared	34	72	75	47	74	72.88	8	28	21	27	46	70	3.1	5	0.28	0.23	1.1	0.20	0.13	22
Corn Silage, Sweet Corn	24	65	66	37	66		11		20	32	57	60	5.0	5	0.24	0.26	1.2	0.17	0.16	39
Corn Grain, Whole	88	88	98	65	91	88.85	9	58	2	3	9	60	4.3	2	0.02	0.30	0.4	0.05	0.14	18
Corn Grain, Rolled	88	88	98	65	91		9	54	2	3	9	34	4.3	2	0.02	0.30	0.4	0.05	0.14	18
Corn Grain, Steam Flaked	85	93	104	71	97	95.44	9	59	2	3	9	40	4.1	2	0.02	0.27	0.4	0.05	0.14	18
Corn Grain, High Moisture	74	93	104	71	97	91.64	10	42	2	3	9	0	4.0	2	0.02	0.30	0.4	0.06	0.14	20
Corn Grain, High Oil	88	91	102	69	95		8	54	2	3	8	60	6.9	2	0.01	0.30	0.3	0.05	0.13	18

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Corn Grain, Hi-Lysine	92	87	96	64	90		12	58	4	4	11	60	4.4	2	0.03	0.24	0.4	0.05	0.11	18
Corn and Cob Meal	87	82	89	59	85	83.15	9	52	9	11	26	56	3.7	2	0.06	0.27	0.5	0.05	0.13	16
Corn Cobs	90	48	48	9	47	53.18	3	70	36	39	88	56	0.6	2	0.12	0.04	0.8		0.27	5
Corn Screenings	86	91	102	69	95		10	52	3	4	9	20	4.3	2	0.04	0.27	0.4	0.05	0.12	16
Corn Bran	91	76	81	52	78		11		10	17	51	0	6.3	3	0.04	0.15	0.1	0.13	0.08	18
Corn Germ, Full-fat	97	135	198	160	198		12	55	6	11	36	20	44.9	2	0.02	0.28	0.1	0.02	0.17	60
Corn Gluten Feed	90	80	86	56	83	78.47	22	25	9	12	38	36	3.2	7	0.11	0.84	1.3	0.25	0.47	84
Corn Gluten Meal 41% CP	91	85	93	62	88		46	63	5	9	32	23	3.2	3	0.13	0.55	0.2	0.07	0.62	35
Corn Gluten Meal 60% CP	91	89	99	67	93	75.29	67	65	3	6	11	23	2.5	2	0.06	0.54	0.2	0.10	0.90	40
Corn Cannery Waste	29	68	70	41	69		8	15	28	36	59	0	3.0	5	0.10	0.29	1.0		0.13	25
Cottonseed, Whole	91	95	107	73	99		23	38	27	37	47	100	19.4	5	0.16	0.64	1.0	0.06	0.24	34
Cottonseed, Whole, Delinted	90	95	107	73	99		24	39	19	28	40	100	22.9	5	0.12	0.54	1.2		0.24	36
Cottonseed, Whole, Extruded	92	87	98	67	91		26	50	32	44	53	33	9.5	5	0.17	0.68	1.3		0.24	38
Cotton Gin Trash (Burrs)	91	42	43	0	40		9		35	50	70	100	2.0	14	1.40	0.18	1.9		0.14	25
Cottonseed Hulls	90	45	45	3	44	44.30	5	45	48	70	87	100	1.8	3	0.15	0.08	1.0	0.02	0.05	10
Cottonseed Meal, Solv. Ext. 41% CP	90	77	82	53	79	72.85	47	42	13	18	25	23	1.5	7	0.22	1.23	1.6	0.05	0.44	66
Cottonseed Meal, Mech. Ext. 41% CP	92	79	85	55	81	71.71	46	50	13	19	31	23	5.0	7	0.21	1.18	1.6	0.05	0.42	64
Crab Waste Meal	91	29	37	0	30		32	65	11	13			3.0	43	15.00	1.88	0.5	1.63	0.27	107
Crambe Meal, Solv. Ext.	91	81	88	58	84		31	45	25	35	47	23	1.4	8	1.27	0.86	1.1	0.70	1.26	44
Crambe Meal, Mech. Ext.	92	88	98	65	91		28	50	24	33	42	25	17.0	7	1.22	0.78	1.0	0.65	1.18	41
Cranberry Pulp Meal	88	49	49	11	48		7		26	47	54	33	15.7	2						
Crawfish Waste Meal	94	25	36	0	29		35	74	12	15				42	13.10	0.85				
Curacao Phosphate	99	0	0	0	0		0		0	0	0	0	0.0	95	34.00	15.00				
Defluorinated Phosphate	99	0	0	0	0		0		0	0	0	0	0.0	95	32.60	18.07	1.0			100
Diammonium Phosphate	98	0	0	0	0		115	0	0	0	0	0	0.0	35	0.52	20.41	0.0		2.16	
Dicalcium Phosphate	96	0	0	0	0		0		0	0	0	0	0.0	94	22.00	18.65	0.1		1.00	70
Distillers Grains, Wet	25	91	102	69	95		28	52	8	18	40	4	9.6	5	0.10	0.70	1.0	0.20	0.60	95
Distillers Grain, Barley	90	75	79	50	77		30	56	16	20	44	4	8.5	4	0.15	0.67	1.0	0.18	0.43	50
Distillers Grain, Corn, Dry	91	95	106	72	99	76.86	30	58	8	16	44	4	9.5	4	0.09	0.75	0.9	0.14	0.70	65

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Distillers Grain, Corn, Wet	36	97	109	74	102		30	47	8	16	44	4	9.5	4	0.09	0.75	0.9	0.14	0.70	65
Distillers Grain, Corn with Solubles	89	98	111	76	103	81.50	30	54	8	16	38	4	11.9	6	0.20	0.75	0.9	0.18	0.80	85
Distillers Dried Solubles	93	87	96	64	91	79.45	31	47	4	7	22	4	13.0	8	0.35	1.20	1.8	0.28	1.10	91
Distillers Corn Stillage	7	92	103	70	96		22	55	8	10	21	0	8.1	5	0.14	0.72	0.2		0.60	60
Distillers Grain, Sorghum, Dry	91	84	92	61	87	72.85	33	62	13	20	44	4	10.0	4	0.20	0.68	0.3		0.50	50
Distillers Grain, Sorghum, Wet	35	86	95	63	89		33	55	13	19	43	4	10.0	4	0.20	0.68	0.3		0.50	50
Distillers Grain, Sorghum with Solubles	92	85	93	62	88		33	53	12	18	42	4	10.0	4	0.23	0.70	0.5		0.70	55
Elephant (Napier) Grass Hay, Chopped	92	55	55	21	54		9		24	46	63	85	2.0	10	0.35	0.30	1.3		0.10	
Fat, Animal, Poultry, Vegetable	99	195	285	230	285	80.08[f]	0		0	0	0	0	99.0	0	0.00	0.00	0.0			
Feather Meal Hydrolyzed	93	67	69	40	68		87	68	1	14	42	23	7.0	3	0.48	0.45	0.1	0.20	1.82	90
Fescue KY 31 Fresh	29	64	65	36	65		15	20	25	32	64	40	5.5	9	0.48	0.37	2.5		0.18	22
Fescue KY 31 Hay Early Bloom	88	60	60	30	60	53.57	18	22	25	31	64	98	6.6	8	0.48	0.36	2.6		0.27	24
Fescue KY 31 Hay Mature	88	52	52	16	51		11	30	30	42	73	98	5.0	6	0.45	0.26	1.7		0.14	22
Fescue (Red) Straw	94	43	44	0	41		4		41				1.1	6	0.00	0.06				
Fish Meal	90	74	78	49	76		66	60	1	2	12	10	9.0	20	5.55	3.15	0.7	0.76	0.80	130
Flax Seed Hulls	91	38	40	0	36		9		32	39	50	98	1.5	10						
Garbage Municipal Cooked	23	80	86	56	83		16		9	50	59	30	20.0	10	1.20	0.43	0.6	0.67		
Glycerol (Glycerin)	88	90	100	68	94		0	0	0	0	0	0	0.0	6				4.00		
Grain Screenings	90	65	66	37	66		14		14				5.5	9	0.25	0.34				30
Grain Dust	92	73	77	48	75		10		11				2.2	10	0.30	0.18				42
Grape Pomace Stemless	91	40	42	0	38	27.50	12	45	32	46	54	34	7.6	9	0.55	0.07	0.6	0.01		24
Grass Hay	88	58	58	26	58		10	30	33	41	63	98	3.0	6	0.60	0.21	2.0		0.20	28
Grass Silage	30	61	62	31	61		11	24	32	39	60	61	3.4	8	0.70	0.24	2.1		0.22	29
Guar Meal	90	72	75	47	74		39	34	16				3.9	5						
Hominy Feed	90	89	99	67	93		11	48	5	8	21	9	6.5	3	0.04	0.55	0.6	0.06	0.10	32
Hop Leaves	37	49	49	11	48		15		15				3.6	35	2.80	0.64				
Hop Vine Silage	30	53	53	18	53		15		21	24			3.1	20	3.30	0.37	1.8		0.22	44
Hops Spent	89	35	39	0	33		23		26	30			4.6	7	1.60	0.60				
Kelp Dried	91	32	38	0	29	54.67	7		7	10			0.5	39	2.72	0.31				
Kenaf Hay	92	48	48	9	47		10		31	44	56	98	2.9	12						
Kochia Fresh	29	55	55	21	55	65.11	16		23				1.2	18	1.10	0.30				
Kochia Hay	90	53	53	18	53		14		27				1.7	14	1.00	0.20				
Kudzu Hay	90	54	54	20	54		16		33				2.6	7	3.00	0.23				

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		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Lespedeza Fresh Early Bloom	25	60	60	30	60		16	50	32				2.0	10	1.20	0.24	1.1		0.21	
Lespedeza Hay	92	54	54	20	54		14	60	30				3.0	7	1.10	0.22	1.0		0.19	29
Limestone Ground	98	0	0	0	0		0	0	0	0	0	0	0.0	98	34.00	0.02			0.03	
Limestone Dolomitic Ground	99	0	0	0	0		0	0	0	0	0	0	0.0	98	22.30	0.04	0.4			
Linseed Meal, Solv. Ext.	91	77	82	53	79		38	36	10	18	25	23	1.7	6	0.43	0.91	1.5	0.04	0.47	60
Linseed Meal, Mech. Ext.	91	82	89	59	85		37	40	10	17	24	23	6.0	6	0.42	0.90	1.4	0.04	0.46	59
Meadow Hay	90	50	50	12	49	63.37	7	23	33	44	70	98	2.5	9	0.61	0.18	1.6		0.17	24
Meat Meal, Swine/Poultry	93	71	74	46	73		56	64	2	7	48	0	10.5	24	9.00	4.42	0.5	1.27	0.48	190
Meat and Bone Meal, Swine/Poultry	93	72	75	47	74		56	24	1	5	34	0	10.0	29	13.50	6.50				
Milk, Dry, Skim	94	87	96	64	90		36	0	0	0	0	0	0.9	8	1.36	1.09	1.7	0.96	0.34	41
Mint Slug Silage	27	55	55	21	55		14		24				1.8	16	1.10	0.57				
Molasses Beet	77	75	79	50	77	91.95	8	0	0	0	0	0	0.2	12	0.14	0.03	6.0	1.64	0.60	18
Molasses Cane	77	74	78	49	76	86.63	6	0	0	0	0	0	0.5	14	0.95	0.09	4.2	2.30	0.68	15
Molasses Cane Dried	94	74	78	49	76	82.12	9	0	2	3	7	0	0.3	14	1.10	0.15	3.6	3.00		30
Molasses, Cond. Fermentation Solubles	43	69	71	43	70		16	0	0	0	0	0	1.0	26	2.12	0.14	7.5	2.73	0.93	30
Molasses Citrus	65	75	79	50	77	84.11	9	0	0	0	0	0	0.3	8	1.84	0.15	0.2	0.11	0.23	137
Molasses Wood, Hemicellulose	61	70	73	44	71		1	0	1	2	4	0	0.6	7	1.10	0.10	0.1		0.05	
Monoammonium Phosphate	98	0	0	0	0		70	0	0	0	0	0	0.0	24	0.30	24.70	0.0		1.42	81
Mono-Dicalcium Phosphate	97	0	0	0	0		0		0	0	0	0	0.0	94	16.70	21.10	0.1		1.20	70
Oat Hay	90	54	54	20	54	59.36	10	25	31	39	63	98	2.3	8	0.40	0.27	1.6	0.42	0.21	28
Oat Silage	35	60	60	30	60	64.00[g]	12	21	31	39	59	61	3.2	10	0.34	0.30	2.4	0.50	0.25	27
Oat Straw	91	48	48	9	47	49.64	4	40	41	48	73	98	2.3	8	0.24	0.07	2.5	0.78	0.22	6
Oat Grain	89	76	81	52	78	75.63	13	18	11	15	28	34	5.0	4	0.05	0.41	0.5	0.11	0.20	40
Oat Grain, Steam Flaked	84	88	98	65	91		13	26	11	15	30	32	4.9	4	0.05	0.37	0.5	0.11	0.20	40
Oat Groats	91	91	102	69	95	88.29	18	15	3				6.6	2	0.08	0.47	0.4	0.10	0.20	
Oat Middlings	90	91	102	69	95		16	20	4	6			6.0	3	0.07	0.48	0.5		0.23	
Oat Mill Byproduct	89	33	38	0	30		7		27	37			2.4	6	0.13	0.22	0.6		0.24	
Oat Hulls	93	38	40	0	36	38.39	4	25	33	41	75	90	1.6	7	0.16	0.15	0.6	0.08	0.14	31
Orange Pulp Dried	89	79	85	55	81		9		9	16	20	33	1.8	4	0.71	0.11	0.6		0.05	
Orchardgrass Fresh Early Bloom	24	65	66	37	66	60.13	14	23	30	32	54	41	4.0	9	0.33	0.39	2.7	0.08	0.20	21
Orchardgrass Hay	88	59	59	28	59	64.29[h]	10	27	34	40	67	98	3.3	8	0.32	0.30	2.6	0.41	0.20	26
Pea Vine Hay	89	59	59	28	59		11		32	50	62	92	2.0	7	1.25	0.24	1.3		0.20	20

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		TDN %	NEm (Mcal/cwt.)	NEg	NEl	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Pea Vine Silage	25	58	58	26	58		16		29	44	55	61	3.3	8	1.25	0.28	1.6		0.29	32
Pea Vine Straw	89	51	51	14	50	49.62	7		41	49	72	98	1.4	7	0.75	0.13	1.1		0.15	
Peas Cull	88	85	93	62	88		23	22	7	9	12	0	1.4	4	0.14	0.46	1.1	0.06	0.26	30
Peanut Hulls	91	22	36	0	18	23.17	7		63	65	74	98	1.5	5	0.20	0.07	0.9			
Peanut Meal, Solv. Ext.	91	77	82	53	79	71.90	51	27	9	16	27	23	2.5	6	0.26	0.62	1.1	0.03	0.30	38
Peanut Skins	92	0	0	0	0		17		13	20	28	0	22.0	3	0.19	0.20				
Pearl Millet Grain	87	82	89	59	85	68.04	13		2	6	18	34	4.5	3	0.03	0.36	0.5			
Pineapple Greenchop	17	47	47	7	46		8		24	35	64	41	2.4	7	0.28	0.08				
Pineapple Bran	89	71	74	46	73	72.43	5		20	33	66	20	1.5	3	0.26	0.12				
Pineapple Presscake	21	71	74	46	73		5		24	35	69	20	0.8	3	0.25	0.09				
Potato Vine Silage	15	59	59	28	59		15		26				3.7	19	2.10	0.29	4.0		0.37	
Potatoes Cull	21	80	86	56	83		10	0	2	3	4	0	0.4	5	0.03	0.24	2.2	0.30	0.09	
Potato Waste Wet	14	82	89	59	85		7	0	9	11	18	0	1.5	3	0.16	0.25	1.2	0.36	0.11	12
Potato Waste Dried	89	85	93	62	88	95.85	8	0	7	9	15	0	0.5	5	0.16	0.25	1.2	0.39	0.11	12
Potato Waste Wet with Lime	17	80	86	56	83		5	0	10	12	16	0	0.3	9	4.20	0.18				
Potato Waste Filter Cake	14	77	82	53	79		5	0	2				7.7	3	0.10	0.19	0.2			
Poultry Byproduct Meal	93	79	85	55	81		62	49	2				14.5	17	4.00	2.25	0.5	0.58	0.56	129
Poultry Manure Dried	89	38	40	0	36	67.83	28	22	13	15	35	0	2.1	33	10.20	2.80	2.3	1.05	0.20	520
Prairie Hay	91	50	50	12	49	55.53	7	37	34	47	67	98	2.0	8	0.40	0.15	1.1	0.06	0.06	34
Pumpkins, Cull	11	80	86	56	83		15		14	21	30	0	8.9	9	0.24	0.43	3.3			
Rice Straw	91	40	42	0	38	51.16	4		38	47	72	100	1.4	13	0.23	0.08	1.2		0.11	
Rice Straw Ammoniated	87	45	45	3	44		9		39	53	68	100	1.3	12	0.25	0.08	1.1		0.11	
Rice Grain	89	79	85	55	81	83.86	8	30	10	12	16	34	1.9	5	0.07	0.32	0.4	0.09	0.05	17
Rice Polishings	90	90	100	68	94		14		4	5			14.0	9	0.05	1.34	1.2	0.12	0.19	28
Rice Bran	91	71	74	46	73	66.64	14	30	13	18	24	0	16.0	11	0.07	1.70	1.8	0.09	0.19	40
Rice Hulls	92	13	35	0	8	15.91	3	45	44	70	81	90	0.9	20	0.12	0.07	0.5	0.08	0.08	24
Rice Mill Byproduct	91	39	41	0	37		7		32	50	60	0	5.7	19	0.25	0.48	2.2		0.30	31
Rye Grass Hay	90	58	58	26	58	66.07[i]	10	30	33	38	65	98	3.3	8	0.45	0.30	2.2		0.18	27
Rye Grass Silage	32	59	59	28	59		14	25	22	37	59	61	3.3	8	0.43	0.38	2.9	0.73	0.23	29
Rye Straw	89	44	44	1	43	33.72	4		44	55	71	100	1.5	6	0.24	0.09	1.0	0.24	0.11	
Rye Grain	89	80	86	56	83	84.83	14	20	3	9	19	34	2.5	3	0.07	0.55	0.5	0.03	0.17	33
Safflower Meal, Solv. Ext.	91	56	56	23	56	57.72	24		33	41	57	36	1.3	6	0.35	0.79	0.9	0.21	0.23	65
Safflower Meal Dehulled, Solv. Ext.	91	75	79	50	77	70.55	47		11	20	27	30	0.8	7	0.38	1.50	1.2	0.18	0.22	36
Safflower Hulls	91	14	35	0	34		4		58	73	90	100	3.7	2						
Sagebrush Fresh	50	50	50	12	49	59.04[j]	13		25	30	38		9.2	10	1.00	0.25			0.22	
Sanfoin Hay	88	61	62	31	62		14	60	24				3.1	9						

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NE_m (Mcal/cwt.)	NE_g (Mcal/cwt.)	NE_l (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Shrimp Waste Meal	90	48	48	9	47		50	60	11				5.5	25	8.50	1.75		1.15		
Sodium Tripolyphosphate	96	0	0	0	0		0		0	0	0	0	0.0	96	0.00	25.98	0.0		0.00	
Sorghum Stover	87	54	54	20	54		5		33	41	65	100	1.8	10	0.50	0.12	1.2			
Sorghum Silage	32	59	59	28	59	65.58	9	25	27	38	59	70	2.7	6	0.48	0.21	1.7	0.45	0.11	30
Sorghum Grain (Milo), Ground	89	82	89	59	85		11	55	3	6	15	5	3.1	2	0.04	0.32	0.4	0.10	0.14	18
Sorghum Grain (Milo), Flaked	82	90	100	68	94		11	62	3	6	15	38	3.1	2	0.04	0.28	0.4	0.10	0.14	18
Soybean Hay	89	52	52	16	51	54.10	16		33	40	55	92	3.5	8	1.28	0.29	1.0	0.15	0.24	24
Soybean Straw	88	42	43	0	40	45.98	5		44	54	70	100	1.4	6	1.59	0.06	0.6		0.26	
Soybeans Whole	88	92	103	70	96		41	28	8	11	15	100	18.8	5	0.27	0.64	1.9	0.03	0.34	56
Soybeans Whole, Extruded	88	93	104	71	97		40	35	9	11	15	100	18.8	5	0.27	0.64	2.0	0.03	0.34	56
Soybeans Whole, Roasted	88	93	104	71	97		40	48	9	11	15	100	18.8	5	0.27	0.64	2.0	0.03	0.34	56
Soybean Hulls	90	77	82	52	79	66.86	13	28	39	48	62	28	2.3	5	0.60	0.19	1.3	0.02	0.12	38
Soybean Meal, Solv. Ext. 44% CP	89	84	92	61	87	79.50	49	35	7	10	15	23	1.5	7	0.36	0.70	2.2	0.07	0.41	62
Soybean Meal, Solv. Ext. 49% CP	89	87	96	64	90		54	36	4	6	8	23	1.3	7	0.28	0.71	2.2	0.08	0.45	61
Soybean Mill Feed	90	50	50	12	49		15		36	46			1.9	6	0.46	0.19	1.7		0.07	
Spelt Grain	88	75	79	50	77	77.18	13	27	10	17	21	34	2.1	4	0.04	0.40	0.4		0.15	47
Sudangrass Fresh Immature	18	70	73	44	71	73.27	17		23	29	55	41	3.9	9	0.46	0.36	2.0		0.11	24
Sudangrass Hay	88	57	57	25	57	62.67	9	30	36	43	67	98	1.8	10	0.50	0.22	2.2	0.80	0.12	26
Sudangrass Silage	31	58	58	26	58	60.29	10	28	30	42	64	61	3.1	10	0.58	0.27	2.4	0.52	0.14	29
Sunflower Meal, Solv. Ext.	92	65	66	37	66	44.89	40	27	18	22	36	23	2.8	8	0.44	0.97	1.1	0.15	0.33	55
Sunflower Meal with Hulls	91	57	57	25	57		31	35	27	32	44	37	2.4	7	0.40	1.03	1.0		0.30	85
Sunflower Seed Hulls	90	40	42	0	38		4	65	52	63	73	90	2.2	3	0.00	0.11	0.2		0.19	200
Sugar Cane Bagasse	91	39	41	0	37	52.15	1		49	60	86	100	0.6	4	0.90	0.29	0.5		0.10	
Tapioca Meal, Cassava Byproduct	89	82	89	59	85		1		5	8	34		0.8	3	0.03	0.05				
Timothy Fresh Pre-Bloom	26	64	65	36	65		11	20	31	36	59	41	3.8	7	0.40	0.28	1.9	0.57	0.15	28
Timothy Hay Early Bloom	88	59	59	28	59	60.75	11	22	32	39	63	98	2.7	6	0.58	0.26	1.9	0.51	0.21	30
Timothy Hay Full Bloom	88	57	57	25	57	58.68	8	30	34	40	65	98	2.6	5	0.43	0.20	1.8	0.62	0.13	25
Timothy Silage	34	59	59	28	59	59.32	10	25	34	45	70	61	3.4	7	0.50	0.27	1.7		0.15	
Tomatoes	6	69	71	43	70		16		9	11			4.0	6	0.14	0.35	4.2			
Tomato Pomace Dried	92	64	65	36	65	53.98	23		26	50	55	34	10.6	6	0.43	0.59	3.6			
Triticale Hay	90	56	56	23	56		10		34	41	69	98			0.30	0.26	2.3			25

Feedstuff	DM %	Energy					Protein		Fiber				EE %	ASH %	Ca %	P %	K %	Cl %	S %	Zn ppm
		TDN %	NEm (Mcal/cwt.)	NEg (Mcal/cwt.)	NEl (Mcal/cwt.)	DE (% of GE)[a]	CP %	UIP %	CF %	ADF %	NDF %	eNDF %								
Triticale Silage	34	58	58	26	58		14		30	39	56	61	3.6		0.58	0.34	2.7		0.28	36
Triticale Grain	89	85	93	62	88	83.82	14	25	4	5	22	34	2.4	2	0.07	0.39	0.5		0.17	37
Turnip Tops (Purple)	18	68	70	41	69		18		10	13			2.6	14	3.10	0.40	3.0	1.80	0.27	
Turnip Roots	9	86	95	63	89	92.94	12	0	11	34	44	40	1.6	9	0.65	0.31	3.1	0.65	0.43	40
Urea 46% N	99	0	0	0	0		288	0	0	0	0	0	0.0	0	0.00	0.00	0.0	0.00	0.00	0
Vetch Hay	89	58	58	26	58	59.44	18	14	30	33	48	92	1.8	8	1.25	0.34	2.4		0.13	
Wheat Fresh, Pasture	21	71	74	46	73	76.07	20	16	18	30	50	41	4.0	13	0.35	0.36	3.1	0.67	0.22	
Wheat Hay	90	57	57	25	57	62.73	9	25	29	38	66	98	2.0	8	0.21	0.22	1.4	0.50	0.19	23
Wheat Silage	33	59	59	28	59	63.99	12	21	28	37	62	61	3.2	8	0.40	0.28	2.1	0.50	0.21	27
Wheat Straw	91	43	44	0	41	45.77	3	60	43	57	81	98	1.8	8	0.17	0.06	1.3	0.32	0.17	6
Wheat Straw Ammoniated	85	50	50	12	49		9	25	40	55	76	98	1.5	9	0.15	0.05	1.3	0.30	0.16	6
Wheat Grain	89	88	98	65	91	86.45 [k]	14	23	3	4	12	0	2.3	2	0.05	0.43	0.4	0.09	0.15	40
Wheat Grain Hard	89	88	98	65	91	88.54 [l]	14	28	3	6	14	0	2.0	2	0.05	0.43	0.5		0.16	45
Wheat Grain Soft	89	88	98	65	91	89.96 [m]	12	23	3	4	12	0	2.0	2	0.06	0.40	0.4		0.15	30
Wheat Grain, Steam Flaked	85	91	102	69	95		14	29	3	4	12	0	2.3	2	0.05	0.39	0.4		0.15	40
Wheat Grain Sprouted	86	88	98	65	91		12	18	3	4	13	0	2.0	2	0.04	0.36	0.4		0.17	45
Wheat Bran	89	70	73	44	71	71.16	17	28	11	14	46	4	4.4	7	0.13	1.32	1.4	0.05	0.24	96
Wheat Middlings	89	80	86	56	83		18	22	8	11	36	2	4.7	5	0.14	1.00	1.3	0.05	0.20	98
Wheat Mill Run	90	76	81	52	78	79.11	17	28	9	12	37	0	4.5	6	0.11	1.10	1.2	0.07	0.22	90
Wheat Shorts	89	78	83	54	80		19	25	8	10	30	0	5.3	5	0.10	0.93	1.1	0.08	0.20	118
Wheatgrass Crested Fresh Early Bloom	37	60	60	30	60	79.78	11	25	26	28	50	41	1.6	7	0.46	0.32	2.4			
Wheatgrass Crested Fresh Full Bloom	50	55	55	21	55	65.89	10	33	33	36	65	41	1.6	7	0.39	0.28	2.1			
Wheatgrass Crested Hay	92	54	54	20	54	56.51	10	33	33	36	65	98	2.4	7	0.33	0.20	2.0			32
Whey Dried	94	82	89	59	85	91.47 [n]	14	15	0	0	0	0	0.9	10	0.98	0.88	1.3	1.20	0.92	10
Yeast, Brewer's	92	79	85	55	81	73.76	47	30	3	4		0	0.9	7	0.13	1.49	1.8			

DM = Dry matter
TDN = Total digestible nutrients
NEm = Net energy for maintenance
NEg = Net energy for growth
NEl = Net energy for lactation
Mcal = Megacalories
cwt = Centum weight (hundredweight)
DE =Digestible energy
GE = Gross energy
CP =Crude protein
UIP = Undegradable intake protein
CF = Crude fiber
ADF = Acid detergent fiber
NDF = Neutral detergent fiber
eNDF = effective neutral detergent fiber
EE = Ether extract
ASH = Ash
Ca = Calcium
P = Phosphorous
K = Potassium
Cl = Chlorine
S = Sulfur
Zn = Zinc
ppm = parts per million

[a] DE (% of GE) values from Ewan (1989)
[b] Average of fresh, late vegetative; fresh, early bloom; fresh, midbloom; fresh, full bloom
[c] Average of silage wilted – early bloom; silage wilted – midbloom; silage wilted – full bloom
[d] Average of silage wilted – early bloom; silage wilted – midbloom; silage wilted – full bloom

[e] Average of hay – sun-cured, late vegetative; hay – sun-cured, late bloom
[f] Average of fat, animal poultry; oil, vegetable
[g] Average of silage, late vegetative; silage, dough stage
[h] Average of hay, sun-cured, early bloom; hay, sun-cured, late bloom
[i] Average of ryegrass, Italian *Lolium multiflorum*: hay, sun-cured, late vegetative; hay, sun-cured, early bloom; average of ryegrass, perennial *Lolium perenne*: hay, sun-cured
[j] Average of sagebrush, big *Artemisia* tridentate: browse, fresh, stem-cured; sagebrush, bud *Artemisia spinescens*: browse, fresh, early vegetative; browse, fresh, late vegetative; and sagebrush, fringed *Artemisia frigida*: browse, fresh, midbloom; browse, fresh, mature
[k] Average of wheat, *Durum Triticum* durum and wheat *Triticum aestivum* grain
[l] Average of grain, hard red spring; grain, hard winter
[m] Average of grain, soft red winter; grain, soft white winter; grain, soft white winter, pacific coast
[n] Average of dehydrated (cattle) and low lactose, low lactose, dehydrated (dried whey product)(cattle)

Appendix 5-C: Estimation Methods for Ammonia Emissions from Manure Management Systems

This appendix presents methods for estimating NH_3 from manure management systems. NH_3, although not a GHG, is emitted in large quantities from animal housing and manure management systems and is an indirect precursor to nitrous oxide (N_2O) emissions as well as an environmental concern.

5-C.1 Method for Estimating Ammonia Emissions Using Equations from Integrated Farm System Model

Ammonia

- Method is a function of the surface area of the storage unit, resistance to mass transfer, ambient air velocity, total NH_3 and organic nitrogen content, rate of organic nitrogen transformation to total ammoniacal nitrogen, and manure temperature as defined by Rotz et al. (2011b).
- Ammonia and organic nitrogen content can be obtained from sampling and lab testing.

Ammonia emissions from manure storage are mainly from total ammoniacal nitrogen (TAN). For many animal confinement systems, it has been reported that most of the urea in manure has been converted to TAN and lost as NH_3 by the time manure is transferred to storage (Rotz et al., 2011b); therefore, only organic nitrogen in the manure at the storage stage, which is mineralized to TAN, is used to estimate NH_3 release. There are four main steps related to NH_3 release to the atmosphere: diffusion, dissociation, aqueous to gas partitioning, and mass transport away from the manure surface (Rotz et al., 2011b). For solid manure, diffusion through the manure is a main constraint to the emission rate. For liquid manure, NH_3 emissions are a function of the overall mass transfer rate and the difference in the NH_3 concentration between the lagoon and the surrounding atmosphere.

5-C.1.1 Rationale for Selected Method

Ammonia emissions from temporary stack and long term stockpiles, aerobic lagoons, anaerobic lagoons, runoff holding ponds, and storage tanks can be calculated using equations from the DairyGEM Model (a subset of the Integrated Farm System Model) (Rotz et al., 2011b). The equations from Rotz et al. are the only available methods for estimating NH_3 emissions from these systems and best describes the quantitative relationship amongst activity data at the entity level.

5-C.1.2 Activity Data

In order to estimate the daily NH_3 emission from temporary stack, long-term stockpiles, anaerobic lagoons, runoff holding ponds, and storage tanks, the following information is needed:

- Total nitrogen content of manure
- Manure total NH_3-N content
- Surface area of manure pile
- Temperatures (local ambient temperature and manure temperature)
- Local ambient air velocity
- For aerobic lagoons, the pH of the lagoon is also needed.

The timing of measurements can be based on dietary changes or seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure piles causing the changes of the variables, frequent measurements of manure characteristics are recommended to ensure accuracy of the estimation.

5-C.1.3 Ancillary Data

The ancillary data used to estimate NH_3 emission for temporary storage are kinematic viscosity of air, mass diffusivity of NH_3, and resistance to mass transfer. The kinematic viscosity of air at standard atmospheric pressure is listed in Table 5-C-1. The mass diffusivity of NH_3 is obtained from references (Paul and Watson, 1966; Baker, 1969) and listed in Table 5-C-2. The resistance to mass transfer for different solid manure storages are obtained from the DairyGEM model (Rotz et al., 2011a).

5-C.2 Method for Ammonia Emissions from Temporary Stack, Long-Term Stockpile, Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks, and Aerobic Lagoons

Temporary Stack, Long-Term Stockpile, and Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks

As indicated in Equation 5-C-1, NH_3 emissions are a function of the overall mass transfer rate and the difference in NH_3 concentration between the manure and surrounding atmosphere. The mean ambient air NH_3 concentration is 1.3 µg/m³ based on passive measurements from 35 locations across 24 States in the U.S. with one year or more of measurements (Ammonia Monitoring Network, National Atmospheric Deposition Program). The Henry's Law constant is used to define the ratio of NH_3 concentration in a solution in equilibrium with gaseous NH_3 concentration in air and is exponentially related to temperature.

Equation 5-C-1: Ammonia Emissions from Temporary Stack, Long Term Stockpiles, and Anaerobic Lagoons/Runoff Holding Ponds/Storage Tanks

$$\mathbf{E_{NH_3} = 24 \times 3600 \times A_{surface} \times K \times (TAN_m - H \times TAN_a)}$$

Where:

E_{NH3} = NH_3 emissions per day (kg NH_3 day^{-1})

24 = Hours per day (hr day^{-1})

3,600 = Seconds per hour (s hr^{-1})

$A_{surface}$ = Footprint of manure storage (m^2) × shape factor[b]

K = Overall mass transfer coefficient (m s^{-1}) as defined in Equation 5-C-3

TAN_m = Total ammoniacal nitrogen in the manure (kg m^{-3})

TAN_a = NH_3 concentration in ambient air[a] (kg m^{-3})

H = Henry's Law constant as defined in Equation 5-C-2

[a] Ammonia concentration in ambient air can be obtained from National Atmospheric Deposition Program (nadp.sws.uiuc.edu/amon/).
[b] Shape factors (ℝ) are listed in Appendix 5-D.

Equation 5-C-2 describes the calculation for Henry's Law Constant. The manure temperature is calculated as the average ambient temperature over the previous 10 days.

Equation 5-C-2: Calculation Henry's Law Constant

$$\mathbf{H = \frac{T}{0.2138} \times 10^{\left(\frac{1825}{T} - 6.123\right)}}$$

Where:

H = Henry's Law constant for NH_3 (aqueous to gas)

T = Manure temperature (Kelvin degree)

The overall mass transfer coefficient is expressed as the reciprocal of the overall effective resistance of the manure. The mass transfer coefficient through gaseous phase on the top of manure is calculated using Equation 5-C-3. The resistance to mass transfer is calculated in Equation 5-C-5. It has been reported that the mass transfer coefficient through manure has relatively little effect on the mass transfer of NH_3 (Ni, 1999) and thus the $1/K_l$ is considered negligible in the following equation.

Equation 5-C-3: Overall Mass Transfer Coefficient

$$\mathbf{K = \frac{1}{\left(\frac{H}{K_g} + \frac{1}{K_l} + R_m\right)}}$$

Where:

K = Overall mass transfer coefficient ($m\ s^{-1}$)

H = Henry's Law constant for NH_3 (aqueous to gas)

R_m = Resistance to mass transfer ($s\ m^{-1}$)

K_g = Mass transfer coefficient through gaseous phase on the top of manure ($m\ s^{-1}$)

K_l = Mass transfer coefficient through manure ($m\ s^{-1}$)

The mass transfer coefficient through gaseous phase (Equation 5-C-4) is estimated from the air friction velocity and Schmidt number of air. The Turbulent Schmidt number is dependent on the characteristics of the gas and the scales of atmospheric turbulence. Since turbulence is highly dependent on many complex interactions, the Turbulent Schmidt number was approximated by only accounting for the gas characteristics. These characteristics are expressed in the molecular Schmidt number, defined as $SC = \nu/D$, where ν is the kinematic viscosity of air ($m^2\ s^{-1}$), and D is the mass diffusivity of NH_3 ($m^2\ s^{-1}$). In order to calculate Schmidt number, the dynamic viscosity of air, the density of the air, and the mass diffusivity of NH_3 are given based on air temperature in Table 5-C-1 and Table 5-C-2.

Equation 5-C-4: Calculating Mass Transfer Coefficient through Gaseous Phase

$$\mathbf{K_g = 0.001 + 0.0462 \times (0.02 \times V_a^{1.5}) \times (SC)^{-0.67}}$$

Where:

K_g = Mass transfer coefficient through gaseous phase on the top of manure (m s^{-1})

V_a = Ambient air velocity (m s^{-1}) that can be obtained from National Weather Service by searching the target location

SC = Turbulent Schmidt number of NH_3 in the air above manure surface (dimensionless)

Table 5-C-1: Kinematic Viscosity of Air at Different Temperature at Standard Atmospheric Pressure

Temperature (°C)	Kinematic Viscosity (m^2/s) x 10^{-5}
-40	1.04
-20	1.17
0	1.32
5	1.36
10	1.41
15	1.47
20	1.51
25	1.56
30	1.60
40	1.66
50	1.76

Source: White (1999).

Table 5-C-2: Mass Diffusivity of Ammonia at Standard Atmospheric Pressure

Temperature (°C)	Diffusivity of Ammonia (m^2/s) x 10^{-4}
-40	0.106
0	0.110
30	0.200
40	0.209
50	0.233

Source: Paul and Watson (1966) and Baker (1969).

The mass transfer coefficient through manure has little effect on the mass transfer of NH_3, so it is negligible. The resistance to mass transfer is the sum of the resistance through the manure and the resistance of cover materials over the manure (Equation 5-C-5). The values for resistance to mass transfer through the manure and resistance to mass transfer through the cover are listed in Table 5-C-3 for temporary stack and long-term stockpile and in Table 5-C-4 for anaerobic lagoons, runoff holding ponds, and storage tanks.

Equation 5-C-5: Calculation of Resistance to Mass Transfer

$$\mathbf{R_m = R_S + R_C}$$

Where:

R_m = Resistance to mass transfer (s m^{-1})

R_s = Resistance to mass transfer through the manure (s m^{-1})

R_c = Resistance to mass transfer through the cover (s m^{-1})

Table 5-C-3: Resistance to Mass Transfer for Solid Manure Storage

Type of Manure Storage	R_s (s m^{-1})	R_c (s m^{-1})
Uncovered solid manure (dry matter >15%)	3×10^5	0
Covered solid manure (dry matter >15%)	3×10^5	2×10^5
Uncovered slurry manure (dry mater, 10-15%)	2×10^5	0

Type of Manure Storage	R_s (s m^{-1})	R_c (s m^{-1})
Covered slurry manure (dry mater, 10-15%)	2×10^5	2×10^5

Source: Rotz et al. (2011b).

Table 5-C-4: Resistance to Mass Transfer For Anaerobic Lagoons, Runoff Holding Ponds, and Storage Tanks

Type of Cover	R_s (s m^{-1})	R_c (s m^{-1})
Uncovered liquid manure	0	0
Covered liquid manure	0	2×10^5

Source: Rotz et al. (2011a).

Aerobic Lagoons

The method for estimating NH_3 emissions from aerobic lagoons (Equation 5-C-6) is similar to that for stockpiles and anaerobic lagoons but accounts for the concentration of NH_3 in the liquid.

Equation 5-C-6: Resistance to Mass Transfer for Solid Manure Storage (Rotz et al., 2011b)

$$\mathbf{E_{NH_3} = 24 \times 3600 \times K \times A_{surface} \times NH_3}$$

Where:

E_{NH3} = NH_3 emissions per day (kg day^{-1})

24 = Hours per day (hr day^{-1})

3,600 = Seconds per hour (s h^{-1})

K = Overall mass transfer coefficient (m s^{-1})

$A_{surface}$ = Surface area of lagoon (m^2)

NH_3 = Concentration in the liquid (kg m^{-3})

The overall mass transfer coefficient is calculated using Equation 5-C-3 with resistance to mass transfer assumed to be zero. Henry's Law Constant is calculated using Equation 5-C-2 and the mass transfer coefficient through a gaseous phase is calculated using **Equation 5-C-4**. The mass transfer through the liquid film layer is calculated using Equation 5-C-7.

Equation 5-C-7: Calculating the Mass Transfer Coefficient through the Liquid Film Layer

$$K_l = 1.417 \times 10^{-12} \times T^4$$

Where:

K_l = Mass transfer coefficient through the liquid film layer (m s^{-1})

T = Manure temperature (Kelvin)

Equation 5-C-8 describes the estimation method for NH_3 concentration in the liquid. The NH_3 fraction of TAN in the lagoon liquid is a function of pH and a dissociation constant according to Equation 5-C-9.

Equation 5-C-8: Calculating the Ammonia Concentration in the Liquid

$$\mathbf{NH_3 = F \times TAN}$$

Where:

NH_3 = Concentration in the liquid (kg m^{-3})

F = NH_3 of TAN in the lagoon liquid

TAN = Total ammonia nitrogen in the manure liquid (kg m^{-3})

Equation 5-C-9: Calculating the Ammonia Fraction of TAN in the Lagoon Liquid

$$\mathbf{F = \frac{1}{1 + \frac{10^{-pH}}{K_a}}}$$

Where:

F = NH_3 of TAN in the lagoon liquid

pH = Hydrogen ion concentration

K_a = Dissociation constant, where $K_a = 10^{\left(0.05 - \frac{2788}{T}\right)}$

T = Temperature (Kelvin)

5-C.3 Method for Estimating Ammonia Emissions from Composting Using IPCC Tier 2 Equations

Ammonia

- IPCC Tier 2 approach adjusted to estimate NH_3 emissions utilizing data on an NH_3 emission factor, total initial nitrogen, and dry manure.
- The NH_3 emission factor is obtained from a study of composting mixture of cattle and swine manure by Hellebrand and Kalk (2000).
- Nitrogen content can be obtained from sampling and lab testing.
- Method is the only readily available method.

Composting is the controlled aerobic decomposition of organic material into a stable, humus-like product (USDA NRCS, 2007). Eghball et al. (1997) reported that 19 to 45 percent of the nitrogen present in manure was lost during composting, with the majority of this presumably as NH_3.

5-C.3.1 Rationale for Selected Method

The IPCC method is adapted for estimating NH_3 emissions and incorporates NH_3 emission factors from a study of composting cattle and swine manure (Hellebrand and Kalk, 2000). The IPCC equation is the only available method for estimating NH_3 emissions from composting. This methodology best describes the quantitative relationship amongst activity data at the entity level.

5-C.3.2 Activity Data

In order to estimate the daily NH_3 emissions, the following information is needed:

- Total dry manure in the storage
- Total nitrogen in manure

The timing of measurements can be based on dietary changes or on a seasonal timeframe, which is decided by individual farm entity. However, due to the dynamic nature of manure storage causing changes in the variables, frequent measurements of manure characteristics (e.g., volatile solids, temperature, total dry manure) are recommended to improve accuracy of the estimation.

5-C.3.3 Ancillary Data

The ancillary data used to estimate NH_3 emission for manure composting is NH_3 emission factor (Hellebrand and Kalk, 2000).

5-C.4 Method for Ammonia Emissions from Composting

Ammonia emissions from composting are dependent on volatilization and mineralization after nitrification, decomposition of organic nitrogen compounds, or urea hydrolysis. An IPCC Tier 2 approach for estimating N_2O emissions is adapted to estimate NH_3 emissions from composting of solid manure. The NH_3 emission factor of 0.05 is obtained from a study of composting mixture of cattle and swine manure (Hellebrand and Kalk, 2000). Equation 5-C-10 provides the equations for estimating NH_3 emissions.

Equation 5-C-10: IPCC Tier 2 Approach for Calculating NH_3 Emissions from Composting of Solid Manure

$$\mathbf{E_{NH3} = m \times EF_{NH3} \times TN \times \frac{17}{14}}$$

Where:

E_{NH3} = NH_3 emissions per day (kg NH_3 day^{-1})

m = Total dry manure (kg day^{-1})

EF_{NH3} = NH_3 emission (loss) relative to total nitrogen in manure (kg NH_3-N (kg TN)$^{-1}$; =0.05)

TN = Total nitrogen in the initial (fresh) manure (kg TN (kg dry manure)$^{-1}$)

$\frac{17}{14}$ = Conversion of NH_3 to nitrogen

5-C.5 Uncertainty in Ammonia Emissions Estimates

Estimation methods from Rotz et al. (2011b) are used to estimate NH_3 emissions from temporary stack and long-term stockpiles and aerobic lagoons. Rotz et al. takes into account the amount of emissive surface area of the pile or lagoon. Given the difficulty of measuring the surface area of a manure pile, shape factors have been developed to approximate surface area based on general shape and footprint. These shape factors provide an estimate total surface area only; there is associated uncertainty based on the accurracy of the footprint measurements and how well the shape of the pile matches the shape factors defined.

The Rotz et al. equations require the NH_3 concentration in the ambient air on site. National data on ambient NH_3 concentrations are available from the National Atmospheric Deposition Program. The

Program provides ambient NH_3 concentrations from approximately 60 active monitoring sites across the country. Given the dearth of monitoring sites and the potentially long distances between the entity and the nearest measurement, there can be a large amount of uncertainty associated with the ambient air NH_3 concentrations used for estimating NH_3 emissions.

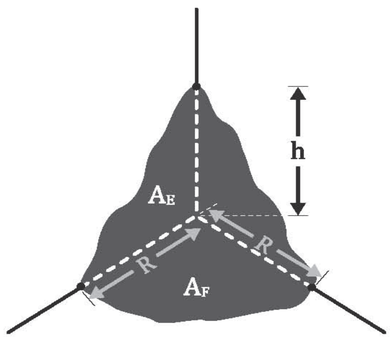
Table 5-C-5: Available Uncertainty Data for Ammonia Emissions Estimates

Parameter	Abbreviation/Symbol	Data Input Unit	Estimated Value	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
pH	pH	-	7.5			6.5	8.5	Expert Assessment
Total ammonia nitrogen in the manure – beef earthen lot	TAN	kg NH_3/m^3	0.1			0	0.02	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, leghorn pullets	TAN	kg NH_3/m^3	0.85			0.66	1.04	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, leghorn hen	TAN	kg NH_3/m^3	0.88			0.54	1.22	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, broiler	TAN	kg NH_3/m^3	0.75					ASABE (2005)
Ammonia concentration in the liquid – dairy lagoon effluent	NH_3	kg NH_3/m^3	0.08					ASABE (2005)
Ammonia concentration in the liquid – dairy slurry (liquid)	NH_3	kg NH_3/m^3	0.14					ASABE (2005)
Ammonia concentration in the liquid – Swine Finisher-Slurry wet-dry feeders	NH_3	kg NH_3/m^3	0.5					ASABE (2005)
Ammonia concentration in the liquid – Swine Slurry storage-dry feeders	NH_3	kg NH_3/m^3	0.34			0.19	0.49	ASABE (2005)
Ammonia concentration in the liquid – Swine flush building	NH_3	kg NH_3/m^3	0.14					ASABE (2005)
Ammonia concentration in the liquid – Swine agitated solids and water	NH_3	kg NH_3/m^3	0.05					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon surface water	NH_3	kg NH_3/m^3	0.04					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon sludge	NH_3	kg NH_3/m^3	0.07					ASABE (2005)
Composting – Ammonia emission (loss) relative to total nitrogen in manure	EF_{NH3}	kg NH_3-N/kg N	0.05					Hellebrand and Kalk (2000)

Appendix 5-D: Manure Management Systems Shape Factors ($\mathbb{R}$)

Factors can be applied to account for the differences in emissive surface areas for different shapes of manure piles. The equations provided below provide estimates for the surface area for common pile shapes; these estimates are applied for calculating NH_3 emissions from temporary stacks.

Figure 5-D-1: Equations for Calculating the Shape Factor for a 2-Sided Storage Bin with Quarter-Cone Pile



$$\mathbb{R} = \sqrt{\frac{h^2}{R^2} + 1}$$

Emitting Area, A_E

$$A_E = \frac{\pi R h}{4} \sqrt{1 + \left(\frac{R}{h}\right)^2}$$

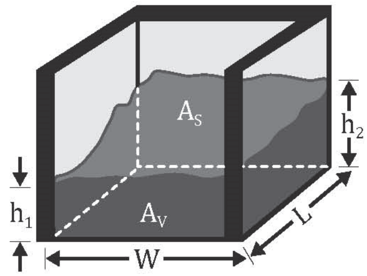
Footprint Area, A_F

$$A_F = \frac{\pi R^2}{4}$$

Limiting Cases

- If $h = R$, $\mathbb{R} = 1.4$
- If $R \gg h$, $\mathbb{R} = 1.0$

Figure 5-D-2: Equations for Calculating the Shape Factor for a 3-Sided Storage Bin



$$\mathbb{R} = \frac{A_E}{A_F} = \frac{h_1 + \sqrt{L^2 + (h_2 - h_1)^2}}{L}$$

Emitting Area, A_E

$A_E = A_V + A_S$

$A_V = h_1 W$

$A_S = \sqrt{L^2 + (h_2 - h_1)^2}\ W$

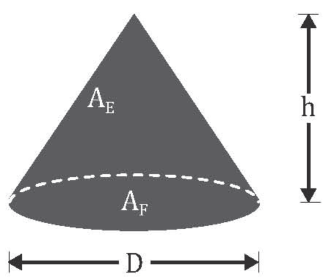
Footprint Area, A_F

$A_F = LW$

Limiting Cases

- If $h_1 = 0$, $\mathbb{R} = \dfrac{\sqrt{L^2 + h_2^2}}{L}$
- If $h_1 = h_2$, $\mathbb{R} = \dfrac{h_1 + L}{L} = 1 + \dfrac{h_1}{L}$
- If $0 < h_1 < h_2$, $\mathbb{R} = \dfrac{h_1 + \sqrt{L^2 + (h_2 - h_1)^2}}{L}$

Figure 5-D-3: Equations for Calculating the Shape Factor for a Conical Manure Pile



$$\mathbb{R} = \sqrt{\left(\frac{4h^2}{D^2} + 1\right)}$$

Emitting Area, A_E

$$A_E = \pi \frac{Dh}{2}\left[1 + \frac{D^2}{4h^2}\right]^{1/2}$$

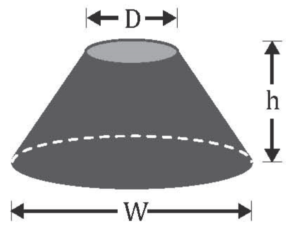
Footprint Area, A_F

$A_F = \pi D^2/4$

Limiting Cases

- If $D \gg h$, $\mathbb{R} = 1.0$
- If $D = 2h$, $\mathbb{R} = 1.4$

Figure 5-D-4: Equations for Calculating the Shape Factor for a Free-Standing, Truncated Conical Stack



$$\mathbb{R} = \frac{A_E}{A_F} = \frac{\left\{2\,h\,(W+D)\sqrt{1+\frac{(W-D)^2}{4h^2}}\right\} + D^2}{W^2}$$

Emitting Area, A_E

$$A_E = \left\{\frac{\pi\,h\,(W+D)}{2}\sqrt{1+\frac{(W-D)^2}{4h^2}}\right\} + \left\{\frac{\pi\,D^2}{4}\right\}$$

Footprint Area, A_F

$$A_F = \pi\,\frac{W^2}{4}$$

Limiting Cases

- If $D = 0$, see "free-standing conical stack"
- If h is very small and $D \approx W$, $\mathbb{R} = 1.0$

Figure 5-D-5: Equations for Calculating the Shape Factor for a Windrow with Triangular Cross Section

$$\mathbb{R} = \sqrt{\left(\frac{4h^2}{W^2} + 1\right)}$$

Emitting Area, A_E

$$A_E = 2h\left[\frac{\pi W}{4} + L\right]\sqrt{\left(1+\frac{W^2}{4h^2}\right)}$$

Footprint Area, A_F

$$A_F = W\left[\frac{\pi W}{4} + L\right]$$

Limiting Cases

- If $2h = W$, $\mathbb{R} = 1.4$
- If $W \gg h$, $\mathbb{R} = 1.0$

Note: Ratio $\mathbb{R} = \frac{A_E}{A_F}$

Does not depend on Windrow Length, L.

Appendix 5-E: Model Review: Review of Enteric Fermentation Models

A number of empirical and mechanistic models have been developed to estimate enteric CH_4 production (Table 5-E-1). Two of the factors that affect enteric CH_4 production to the greatest extent are diet composition and level of intake. Prediction equations and models constructed to predict enteric CH_4 are generally based on these factors. Most statistical equations developed to estimate enteric CH_4 emissions have been developed using data sets of animals fed high-forage diets or mixed diets; few studies have fed high-concentrate diets typical of today's U.S. feedlots.

Table 5-E-1: Models Potentially Useful in Estimating Enteric CH_4 Emissions from Typical U.S. Ruminant Animals

Reference	Variable modeled	Inputs/Comments
Empirical Models		
IPCC (2006)	Enteric CH_4	No. of animals, animal species, animal type, emission factor for each animal type (Tier 2 CH_4 conversion factor; Ym)
Kriss (1930)	Enteric CH_4	Dry matter intake (DMI)
Axelsson (1949)	Enteric CH_4	DMI
Bratzler & Forbes (1940)	Enteric CH_4	Digested carbohydrate
Mills et al. (2003)	Enteric CH_4	Metabolizable energy (ME) intake, starch and acid detergent fiber (ADF) intake
Blaxter & Clapperton (1965)	Enteric CH_4	Digestible energy (DE) (%) at maintenance intake, gross energy intake (GEI), feeding level (multiple of maintenance)
Moe & Tyrrell (1979)	Enteric CH_4	Digestible soluble carbohydrates, digestible hemicellulose, digestible cellulose
Holter & Young (1992)	Enteric CH_4	Digestible soluble carbohydrates, cellulose, hemicellulose, fat intake
Yan et al. (2009)	Enteric CH_4	Digestible energy, silage, and total DMI, silage, and diet ADF
Ellis et al. (2007)	Enteric CH_4	Metabolizable energy intake, ADF, lignin intake
Ellis et al. (2009)	Enteric CH_4	Metabolizable energy intake, cellulose, hemicellulose, and fat intake; non-fiber carbohydrate, neutral detergent fiber (NDF), and DMI
Mills et al. (2001)	Enteric CH_4	DMI
Holos (Little et al., 2008)	Enteric CH_4, manure CH_4	Based on IPCC (2006)
CNCPS (2010)	Enteric CH_4, DMI, nutrient excretion, urine nitrogen excretion;	Uses equation of Mills et al. (2003) for dairy and Ellis et al. (2007) for beef. Animal characteristics, diet nutrient composition, feed protein fractions, animal performance, animal management, in situ degradability of feeds
Integrated Farm System Model (Rotz et al., 2011b)	Enteric CH_4, nutrient excretion, urine nitrogen, DMI, manure NH_3, CH_4, and N_2O	Uses the Mits3 equation of Mills et al. (2003) for enteric CH_4, IPCC (2006) for manure CH_4, and either DAYCENT (Chianese et al., 2009d) or IPCC (2006) for manure N_2O
Phetteplace et al. (2001)	Enteric CH_4, manure CH_4	Animal class, animal age and body weight, quantity of meat/mile produced, feed type, feed intake, manure management
Process-based Models		

Reference	Variable modeled	Inputs/Comments
Kebreab et al., (2004; 2009)	Enteric CH_4, nutrient excretion	DMI, NDF, degradable NDF, total starch, degradable starch, soluble sugars in diet, diet nitrogen, NHx-N in diet, indigestible protein, rate of degradation of starch, and protein
COWPOLL (Dijkstra et al., 1992; Mills et al., 2003; Bannink et al., 2006; Kebreab et al., 2008)	Enteric CH_4	DMI, NDF, degradable NDF, total starch, degradable starch, soluble sugars in diet, diet nitrogen, NHx-N in diet, indigestible protein, rate of degradation of starch, and protein
MOLLY (Baldwin, 1995)	Enteric CH_4	Similar to COWPOLL

Prediction models for enteric emissions. The following is a brief summary of the models evaluated and their strengths and limitations.

Simple Regression Model Based on Digestible Energy. Blaxter and Clapperton (1965)developed a simple regression equation to estimate enteric CH_4 based on digestible energy, feed intake as a percentage of maintenance and GEI. The data set used to create this empirical model was composed mostly of data from sheep fed low-concentrate diets in respiration chambers, which may account for its limited accuracy in predicting CH_4 emissions across ruminant diets (Johnson et al., 1991).

Empirical Model. Moe and Tyrrell (1979) developed an empirical model to estimate enteric CH_4 emission from dairy cows based on diet composition. This empirical model was developed with high-forage diets in dairy cows fed in respiration chambers; its use for estimating beef cattle enteric emissions is therefore limited.

Regression Model. Yan et al. (2000) developed regression equations to predict enteric CH_4 emissions from beef and dairy cattle fed diets based on grass silage. Concentrates represented from 0 to 81.5 percent of the DMI, with a mean of 46.7 percent of diet DMI. When corrected to equal feed intakes, animal body weight had no effect on enteric CH_4 emissions. (Yan et al., 2000) validated their equations using data from the literature, mostly dairy studies with all diets based on grass silage.

Regression Equations. Ellis et al. (2009) developed regression equations to estimate enteric CH_4 production from beef cattle based on studies in which cattle were fed high-concentrate or moderate-concentrate (50 percent) diets. These equations were compared with 14 equations developed earlier by Ellis et al. (2007), seven developed by Mills et al. (2003), the Blaxter and Clapperton (1965) equation, and the Moe and Tyrrell (1979) equation. The mean enteric CH_4 production (MJ day^{-1} and percent of GEI) in all 12 of the studies was greater than values noted more recently (Hales et al., 2012), possibly because of differences in dietary grain content and fat supplementation. However, some of the Ellis (2007; 2009) equations estimated CH_4 emissions similar to those reported by Todd et al. (2014a; 2014b) in open lot feedlots.

The linear model with the lowest residual mean square prediction error (RMSPE) was Equation 5-E-1 as follows:

Equation 5-E-1: Linear Model with the Lowest RMSPE

$$\mathbf{CH_4 = 2.72 + (0.0937 \times ME\ intake) + (4.31 \times CELL) - (6.49 \times HC) - (7.44 \times Fat)}$$

Where:

CH_4	= Methane per day (MJ day^{-1})
ME intake	= ME intake in (MJ day^{-1})
CELL	= Cellulose intake (kg day^{-1})
HC	= Hemicellulose intake (kg day^{-1})
Fat	= Fat intake (kg day^{-1})

A possible advantage to using this equation, compared with other empirical equations, is that the variables required for the calculations can be readily obtained with some training in nutrition. Another is that the independent variables in the model (energy, fiber, and fat intake) are the primary differences that would occur in various beef and dairy cattle diets. However, a major concern with their use for finishing cattle is that a number of the studies used to develop the equations were high-forage diets and/or did not use either supplemental fat or monensin in the diet. As previously noted, when compared with emissions from cattle fed typical finishing diets based on steam-flaked corn (SFC) or dry-rolled corn (DRC), this equation greatly overestimated CH_4 emissions (Hales et al., 2012). Linear equations using nutrient ratios (starch:NDF, etc.) were also developed, but all had greater RMSPE than the previous equation (Ellis et al., 2009). Nonlinear equations were also developed. Despite being more biologically defendable, the nonlinear equations all had greater RMSPE than the linear equation.

In a later study, Yan et al. (2009) developed additional equations using a database of 108 measurements for beef steers of varied breeding in respiration chambers and fed diets that ranged from 100 to 30 percent roughage. They also compared a number of equations developed elsewhere. Equations were "validated" using one-third of the original data set. Emissions were highly correlated to live body weight, DMI, and GEI, but live body weight was a poor predictor of enteric CH_4 emissions. The ability of a number of equations to predict enteric CH_4 measured in the study was varied (eight percent overpredicted, to 33 percent underpredicted). The poorest results were with four linear equations developed by Ellis et al. (2007) that used DMI, MEI, and/or forage intake as independent variables. They attributed the poor response to the fact that a good portion of the data for Ellis et al. (2007) was from grazing animals using the SF_6 technique, which would not include CH_4 from the lower gut. The Blaxter and Clapperton (1965) equation did a respectable job (93 percent of actual with $R^2 = 0.69$; mean prediction error = 0.12; and 63 percent of means square prediction error due to random effects, and 29 percent due to a mean bias).

Empirical and Mechanistic Model. The IFSM Model (and its subset DairyGEM) (Rotz et al., 2005; Chianese et al., 2009b; 2009c; 2009a; 2009d) is a combination empirical and mechanistic model of whole farm nutrient management. The submodel to estimate enteric CH_4 emissions from beef or dairy cattle uses the Mits3 equation of Mills et al. (2003). Ellis et al. (2007) reported that the Mills et al. (2003) equations were poor at predicting CH_4 from beef cattle, probably because they were developed from dairy data. In fact, one equation that worked well with dairy cows actually predicted negative CH_4 emissions from beef cattle fed high-concentrate, low-forage diets. Thus, the current IFSM may not be appropriate to estimate enteric CH_4 emissions from beef cattle, especially feedlot cattle.

Mechanistic Models. MOLLY (Baldwin et al., 1987; Baldwin, 1995) is a mechanistic model that estimates ruminal CH_4 production based on a hydrogen balance within the rumen. Input parameters to the model are daily DMI, chemical composition of the diet, solubility of protein and starch, degradability, ruminal passage rates, ruminal volume, and ruminal pH. COWPOLL (Dijkstra et al., 1992; Mills et al., 2001) is another mechanistic model. Input parameters to the model are similar to MOLLY. MOLLY and COWPOLL both use an H-balance to estimate enteric CH_4 production. However, they use different VFA stoichiometry submodels. Both models require significant inputs that are probably beyond the scope of typical producers. However, they are excellent research tools.

The Cornell Net Carbohydrate and Protein System model (CNCPS, 2010) calculates nutrient requirements, nutrient inputs, animal production (weight gain and/or milk production), and nutrient excretion in beef and dairy cattle. It recently added a submodel (VanAmburgh et al., 2010) to calculate enteric CH_4 emissions. The submodel uses an equation of Mills et al. (2003) to estimate enteric emissions from dairy cows and an equation of Ellis et al. (2007) to estimate enteric emissions from beef cattle. At present, to our knowledge there are no comparisons or independent validations of the new submodels that have been published, and the extent to which the model is responsive to mitigation strategies is unclear.

Comparative Analyses using Independent Data Sets. Several studies have attempted to evaluate the predictive ability of enteric CH_4 models by using an independent data set. Benchaar et al. (1998) compared two mechanistic (Baldwin et al., 1987; Dijkstra et al., 1992; Baldwin, 1995); and two linear (Blaxter and Clapperton, 1965; Moe and Tyrrell, 1979) models with a data set of 32 diets from 13 publications in the literature. They noted that the mechanistic models were better predictors than the regression equations. The linear regression models could only explain 42 to 57 percent of the variation in predicted values, whereas the mechanistic models explained more than 70 percent of the variation. The model of Dijkstra et al. (1992) tended to underestimate actual CH_4 production (mean error = 0.30 Mcal day^{-1}), with the error being greater at higher CH_4 productions. The model of Baldwin (Baldwin et al., 1987; Baldwin, 1995) overestimated CH_4 production by about 0.93 Mcal day^{-1}, primarily due to a high intercept. The equations of Moe and Tyrrell (1979) and Blaxter and Clapperton (1965) tended to overestimate CH_4 production, especially at low production rates.

Comparative Analysis/Lactating and Nonlactating Cows. Wilkerson et al. (1995) compared several published equations (Kriss, 1930; Bratzler and Forbes, 1940; Axelsson, 1949; Blaxter and Clapperton, 1965; Moe and Tyrrell, 1979; Holter and Young, 1992) for their ability to predict enteric CH_4 production from lactating and nonlactating Holstein cows. In general, equations that were based on total DMI or on intake of digested cellulose, hemicellulose, and nonfiber carbohydrates, provided the highest correlation and lowest errors of prediction. Prediction equations that used a quadratic function of DMI were poor at predicting enteric CH_4. In general, the equations predicted emissions from nonlactating cows more accurately than from lactating cows.

Comparative Analysis Linear Models. Kebreab et al. (2006) compared two linear models (Moe and Tyrrell, 1979; Mills et al., 2003), a nonlinear model (Mills et al., 2003), the IPCC Tier 1 and Tier 2 models (IPCC, 1997), and a dynamic mechanistic model (Kebreab et al., 2004) using data from studies conducted in North America. They recommended that the linear models be used when there is limited information on nutrient intake and when the expected emissions are within the range of data from which the model was developed. The nonlinear model of Mills et al. (2003) could be used for extrapolating beyond the range of data used to develop the equation, but the mechanistic model was recommended for evaluation of mitigation options. The IPCC Tier 1 model was found to be adequate for general inventory purposes. The predictive ability of the Tier 2 model, while most useful, was limited.

Comparative Analysis Mechanistic Models. Kebreab et al. (2008) also compared two mechanistic models, MOLLY (Baldwin et al., 1987; Baldwin, 1995) and COWPOLL (Dijkstra et al., 1992; Mills et al., 2001; Bannink et al., 2006), to the IPCC Tier 2 (2006) and linear equation of Moe and Tyrrell (1979). Using a beef cattle data set, MOLLY and IPCC tended to be more accurate than the other models, although MOLLY was more precise. MOLLY and IPCC Tier 2 had minimal mean bias, whereas COWPOLL and the Moe and Tyrrell (1979) equation greatly overpredicted average emissions. COWPOLL, which is based on the enteric CH_4 prediction equations of Mills et al. (2001) and the updated rumen stoichiometry for lactating cows (Bannink et al., 2006), had the poorest ability to predict enteric CH_4 emission from feedlot cattle and tended to overpredict CH_4 emissions (MJ day^{-1}) by as much as 50 percent. Although on average MOLLY and IPCC Tier 2 (2006) gave predicted values similar to measured values, there was a large variability in individual animals, with errors of 75 percent or greater. The large variability in predicted values indicates that there can be large animal-to-animal variation in enteric CH_4 production, even when animals are fed the same diets at similar feed intakes.

Comparative Analysis/Feedlots. McGinn et al. (2008) compared measured (using bLS model) CH_4 emissions (enteric plus pen surface) from feedlots in Australia and Canada with estimates using the IPCC Tier 1, IPCC Tier 2, Blaxter and Clapperton (1965), and Moe and Tyrrell (1979) equations. The Tier 2 method underestimated CH_4 at both locations. Estimates using the IPCC Tier 1 methods were close to measured values in Australia; however, Tier 1 underestimated values for the Canada feedlot. Estimates made using the Blaxter and Clapperton (1965) and Moe and Tyrrell (1979) equations were close to measured values in Canada, but overestimated values in Australia. Methane emissions had a significant diel pattern indicating that short-term measurement of CH_4 emissions at feedlots may overestimate or underestimate daily emissions.

Comparative Analysis of Stoichiometric Models. Alemu et al. (2011) compared enteric CH_4 emissions from dairy cows using a variety of stoichiometric models of ruminal fermentation (Murphy et al., 1982; Bannink et al., 2006; Sveinbjornsson et al., 2006; Nozière et al., 2010), and noted that mechanistic models such as Bannink et al. (2006) are more accurate for predicting enteric CH_4 from dairy cows than the IPCC Tier 2 (2006) method. However, these models required a considerable quantity of data regarding the animals and their diet.

Comparative Analysis Measurement Data and Models. Tomkins et al. (2011) measured enteric CH_4 emissions of steers on pasture using a micrometeorological method and respiration chambers. Emissions estimated using an Ellis (2009) equation (CH_4, MJ day^{-1} = 3.272 +0.736 (DMI, kg day^{-1})) were similar (112.7 g day^{-1}) to measured emissions. Estimates using the equations of Kurihara et al. (1999) as modified by Hunter (2007) (109.1 g day^{-1}), Yan et al. (2009) (105.6 g day^{-1}), and Charmley et al. (2008) (2008:NABCEMS; 100.2 g day^{-1}) were slightly lower, but not as low as the IPCC (2006) model (82.7 g day^{-1}).

Comparative Analyses/Models. Legesse et al. (2011) compared enteric CH_4 emission estimates using MOLLY, COWPOLL, IPCC Tier 2, and one equation of Ellis et al. (2007) under various Canadian beef cow-calf management systems. Differences among the models (26 to 35 percent) were much greater than differences among management systems (three to five percent). The authors suggested that these differences limited the model's utility in predicting CH_4 emission from beef cow systems.

Evaluation of Models. Yan et al. (2000; 2009) noted that CH_4 production (percent of GEI or digestible energy) decreased with increasing DMI (as multiples of maintenance) and with increasing forage in the diet. Thus, they suggested that models that do not consider feeding level will underpredict CH_4 at low planes of nutrition and overpredict enteric CH_4 at high levels of feeding. Similarly, Kebreab et al. (2006) noted that linear models tend to give unrealistically high emission values when DMI increases, whereas nonlinear models gave values approaching the theoretical maximum emission, which is biologically reasonable.

Although several equations of Ellis et al. (2009) appeared to be good predictors of enteric CH_4 losses from feedlot cattle based on Canadian studies, when compared with data from cattle fed a typical corn-based finishing diet (Hales et al., 2012) most tended to greatly overestimate enteric losses. At the present time, the IPCC Tier 2 model with some modifications may be the most useful for prediction of enteric emissions from feedlot beef cattle.

Chapter 5 References

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Chapter 6 Quantifying Greenhouse Gas Sources and Sinks in Managed Forest Systems

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Acronyms, Chemical Formulae, and Units

BA	Basal area
C	Carbon
CH_4	Methane
cm	Centimeters
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
COLE	CarbonOnLineEstimator
CRM	Component ratio method
DBH	Diameter at breast height
DDW	Down dead wood
DOE	Department of Energy
EPA	Environmental Protection Agency
FFE	Fire and Fuels Extension
FIA	Forest Inventory and Analysis
FIADB	Forest Inventory and Analysis Database
FIDO	Forest Inventory Data Online
FOFEM	First Order Fire Effects Model
FVS	Forest Vegetation Simulator model
ft	Feet
g	Gram
GHG	Greenhouse gas
H	Height
ha	Hectare
hp	Horse power
hr	Hour
HW	Hardwood
HWP	Harvested wood products
in	Inches
lbs	Pounds
IPCC	Intergovernmental Panel on Climate Change
m	Meters
mm	Millimeters
Mcf	Thousand cubic feet
N_2O	Nitrous oxide
NO_x	Mono-nitrous oxides
O_2	Oxygen
PW	Pulpwood
SL	Sawlogs
SOC	Soil organic carbon
SSURGO	Soil Survey Geographic database
STATSGO	State Soil Geographic database
SW	Softwood
Tg	Teragrams
UFORE	Urban Forest Effects model
UNFCCC	United Nations Framework Convention on Climate Change
USDA	U.S. Department of Agriculture

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6 Quantifying Greenhouse Gas Sources and Sinks in Managed Forest Systems

This chapter provides guidance for reporting greenhouse gas (GHG) emissions associated with entity-level fluxes from the forestry sector. In particular, it focuses on methods for estimating carbon stocks and stock change from managed forest systems. Section 6.1 provides an overview of the sector. Section 6.2 describes the methods for forest carbon stock accounting. Section 6.3 describes the methods for estimating carbon stocks and stock change from establishing and clearing forest. Section 6.4 describes methods for estimating carbon stocks and stock change from forest management. Section 6.5 describes methods for estimating carbon stocks and stock change from harvested wood products. Section 6.6 describes methods for estimating carbon stocks and stock change from urban forests (i.e., trees outside of forests). Finally, Section 6.7 describes methods for estimating emissions from natural disturbances including forest fires.

6.1 Overview

A summary of proposed methods and models for estimating GHG emissions from managed forest systems is provided in Table 6-1.

Table 6-1: Overview of Managed Forest Systems Sources, Method and Section

Section	Source	Method
6.2.3	Forest Carbon Accounting	Range of options dependent on the size of the entities' forest land including: Forest Vegetation Simulator model with Fire and Fuels Extension (FVS-FFE) (entities that fit the large landowner definition); and default lookup tables (entities fitting the small landowner definition).
6.3.3	Establishing, Re-establishing, and Clearing Forests	Intergovernmental Panel on Climate Change (IPCC) algorithms developed by Aalde et al. (2006). These options use: allometric equations from Jenkins et al. (2003a), or FVS with the Jenkins et al. equations where applicable; and default lookup tables from Smith et al. (2006; GTR NE-343)—default regional values based on forest type and age class developed from FIA data.
6.4.4	Forest Management	Range of options dependent on the size/management intensity/data availability of the entity's forest land including: FVS-FFE with Jenkins (2003a) allometric equations; Default lookup tables of management practice scenarios; and FVS may be used to develop a supporting product providing default lookup tables of carbon stocks over time by region; forest type categories, including species group (e.g., hardwood, softwood, mixed); regeneration (e.g., planted, naturally regenerated); management intensity (e.g., low, moderate, high, very high); and site productivity (e.g., low, high).
6.5.2	Harvested Wood Products	Method uses U.S.-specific harvested wood products (HWPs) tables. The HWPs tables are based on WOODCARB II model used to estimate annual change in carbon stored in products and landfills (Skog, 2008). The entity uses these tables to estimate the average amount of HWP carbon from the current year's harvest that remains stored in end uses and landfills over the next 100 years.
6.6.3	Urban Forests	Range of options depends on data availability of the entity's urban forest land. These options use: i-Tree Eco model (http://www.itreetools.org) to assess carbon from field data on tree populations; and i-Tree Canopy model (http://www.itreetools.org/canopy/index.php) to assess tree cover from aerial images and lookup tables to assess carbon. Quantitative methods are also described for maintenance emissions and altered building energy use and included for information purposes only.

Section	Source	Method
6.7.3	Natural Disturbance—Wildfire and Prescribed Fire	Range of options depends on the data availability of the entity's forest land including: First Order Fire Effects Model (FOFEM) entering measured biomass; and FOFEM model using default values generated by vegetation type. These options use Reinhardt et al. (1997).

6.1.1 Overview of Management Practices and Resulting GHG Emissions

6.1.1.1 Description of Sector

Forestry activities represent significant opportunities to manage GHGs (Caldeira et al., 2004; Pacala and Socolow, 2004). There are many kinds of forestry activities that may be considered by entities as a means to reduce GHGs, such as establishing new forests, agroforestry, improved forest management, and avoided forest clearing. Cost is a major factor guiding decisions about which activities in forestry to pursue (Lewandrowski et al., 2004; Stavins and Richards, 2005; U.S. EPA, 2005). In the annual GHG inventory reported by the U.S. Department of Agriculture (USDA) and the U.S. Environmental Protection Agency (EPA), forests and forest products sequester an average of 790 million metric tons carbon dioxide (CO_2) per year on 253 million hectares (ha) of forest land, making it the main land category sequestering carbon (U.S. EPA, 2012b; USDA, 2011). Most of the carbon sequestered (89 percent) is in the forest ecosystem, with the remainder added to the pool of carbon in wood products.

6.1.1.2 Resulting GHG Emissions

Forests remove carbon from the atmosphere and store it in vegetative tissue such as stems, roots, barks, and leaves. Through photosynthesis, all green vegetation removes CO_2 and releases oxygen (O_2) to the atmosphere. The remaining carbon is used to create plant tissues and store energy. During respiration, carbon-containing compounds are broken down to produce energy, releasing CO_2 in the process. Any remaining carbon is sequestered until the natural decomposition of dead vegetative matter or combustion releases it as CO_2 to the atmosphere. The net carbon stock in forests increases when the amount of carbon withdrawal from the atmosphere during photosynthesis exceeds the release of carbon to the atmosphere during respiration. The net carbon stock decreases when biomass is burned.

Other GHGs, such as nitrous oxide (N_2O) and methane (CH_4), are also exchanged by forest ecosystems. N_2O may be emitted from soils under wet conditions or after nitrogen fertilization; it is also released when biomass is burned. CH_4 is often absorbed by the microbial community in forest soils but may also be emitted by wetland forest soils. When biomass is burned in either a prescribed fire/control burn or in a wildfire, precursor pollutants that can contribute to ozone and other short-lived climate forcers as well as CH_4 are emitted. A wildfire is an unplanned ignition caused by lightning, volcanoes, unauthorized activity, accidental human-caused actions, and escaped prescribed fires. A prescribed fire/control burn is any fire intentionally ignited by management under an approved plan to meet specific objectives.

Some of the carbon in forests is released to the atmosphere after the harvest of timber. However, the amount of the carbon released, and when, depends on the fate of the harvested timber. If the timber is used to make wood products, a portion of the sequestered carbon will remain stored for up to several decades or longer. If the harvested trees are burned and used to produce energy, carbon will be released through combustion but may also prevent carbon emissions that would have been released through the burning of fossil fuels. Such emissions from biomass energy use are typically combusted with higher efficiency as compared to open biomass burning as would occur in a wildfire situation netting lower carbon emissions.

6.1.1.3 Forest Sector Schematic

Figure 6-1 is a simplified representation of the key forest carbon pools, carbon transfers, and GHG fluxes for the forest system. At this time, CO_2 is the main GHG represented comprehensively. Emissions of non-CO_2 GHGs interact with other sectors; at this time, potential fluxes of non-CO_2 GHGs are represented in a general manner on the schematic. The proportion of total system carbon in each pool can vary over time depending on a variety of factors; rates of carbon transfer are also variable.

6.1.1.4 Management Interactions

Forestry practices typically trigger ecosystem responses that change over time. For example, a newly established forest will take up carbon at a low rate initially, and then pass into a period of relatively rapid carbon accumulation. The carbon uptake rate will then typically decline as heterotrophic and autotrophic respiration increase and growth is balanced against mortality in the older forest. From this point in time, standing live tree biomass may not increase, but evidence suggests that carbon may continue to flow into other forest carbon pools until the forest is removed by harvest or a natural disturbance event.

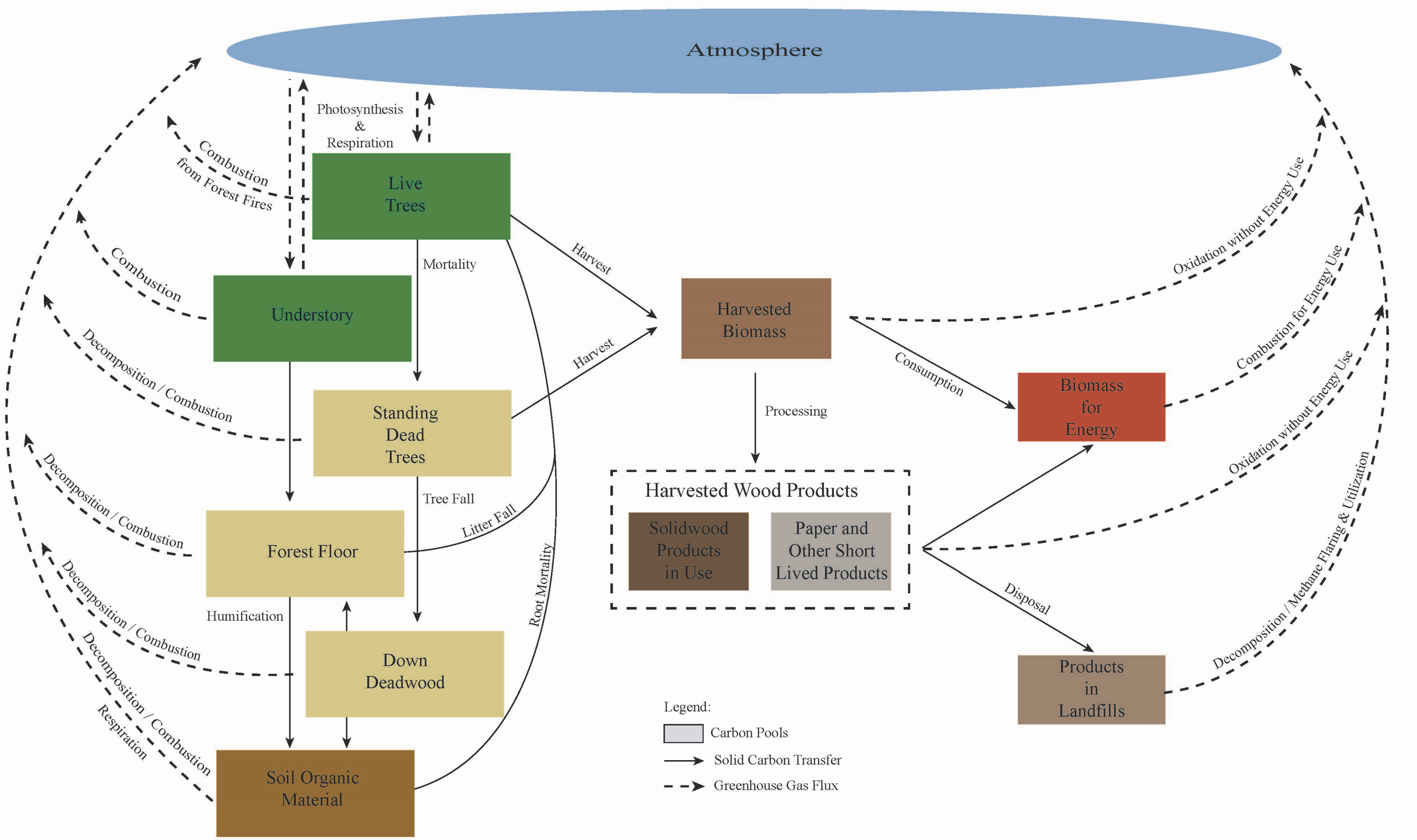
The net effects of management activities on carbon flows in forest ecosystems include changes in many different pools of carbon (such as aboveground biomass, belowground biomass, litter, soil, etc.). Carbon accounting should be comprehensive, addressing the net effects of activities on all carbon flows. Forestry activities cause carbon to move between the various pools and to/from the atmosphere. For example, forest management may be very effective at increasing the accumulation of biomass in commercially valuable forms—that is, in the trunks of commercial tree species. This increased growth may simply result from reducing competition from other types of trees, causing a transfer of carbon uptake from one group of trees to another. Forestry activities can also have effects on forest soils, woody debris, and the amount of carbon in wood products. The net carbon flow effects of any activity will be the sum of all the individual effects on the different carbon pools.

In addition, there may be interactions between biological and physical processes that are affected by forest management treatments or natural disturbances (e.g., changes in albedo during forest regeneration, after wildfires). While these interactions occur, research in this field is in the early stages and such interactions are beyond the scope of this guidance.

6.1.1.5 Risk of Reversals

Carbon that is sequestered in soils, vegetation, or wood products is not necessarily permanently removed from the atmosphere. Forestry activities intended for one purpose may be changed by a different landowner or a change in management objectives. Landowners may change their practices, causing the release of stored carbon, or natural disturbances may cause the loss of stored carbon to the atmosphere. Insect epidemics, drought, or wildfire may happen at any time and may affect all or only a portion of the land area within activity or entity boundaries. Natural disturbances may be rare events, in which case the effects on estimated carbon flows may be small when averaged over large forested areas or long periods of time. Catastrophic disturbances such as wind storms may cause obvious and easily estimated changes in carbon stocks, while in other cases, such as a one-year period of insect defoliation, it may be difficult after a few years to separate the effects of the natural disturbance from other factors. It should be noted that GHG registries generally require entities to calculate carbon stocks and fluxes and generally require entities to conduct an assessment of risk of reversal of projected carbon values. Such assessments generally

Figure 6-1: Schematic of Forest Carbon Pools, Carbon Transfers, and Greenhouse Gas Flux



include risk of natural disturbances such as fire, drought, insect and disease mortality, wind throw (hurricane, tornado, high wind events), as well as financial risks, management risks, and social political risks. These risk assessments are commonly used to generate a value that discounts the projected carbon value of management activities and to provide an “insurance policy” against reversals that may be used to ensure that a program’s climate benefits are realized. Many forest management practices can reduce these natural hazard risks (such as fuel hazard reduction, forest thinning for growth or resilience to droughts, climate change, insect or disease agents, and use of prescribed fire to reduce risk of fires). Reducing the risk of reversal through management may lead to reduced emissions, long-term net increase in carbon stocks, and improved results in a risk assessment.

6.1.2 System Boundaries and Temporal Scale

For this report, the nominal system boundaries are the extent of the landowner’s property. Estimation methods presented in this section are for the forest sector; however, where the forest sector may interact with the animal agriculture or croplands and grazing lands sectors, these instances are noted and landowners should refer to the relevant sector guidance. A landowner may need to use estimation methods for several sectors to achieve a comprehensive report of GHG sources and sinks for their property, ensuring that double counting does not occur. In addition, if land-use transitions occur within the property, these must be accounted for so that apparent changes in carbon stocks or fluxes are “real” and not the result of an unrecorded transfer from one sector to another. While GHG fluxes will occur across the system boundary, these are generally not estimated except in the instance of harvested wood products (HWPs).

The forest sector presents an accounting challenge related to temporal scale that may not occur in other sectors. While many farms operate on an annual cycle, forestry operations, by their nature, occur over multiple years and decades. While annual estimation and reporting are required, annual measurements of forest carbon pools are not economically feasible, nor are changes in carbon stocks generally detectable within acceptable error levels on an annual basis. This necessitates the use of models and projections to assess the carbon consequences of management practices and evaluate the possible GHG benefits of a change in management practices. Throughout the forest guidance, references will be made to several types of estimates that may be generated. A Type I estimate is the estimate of the carbon stock in the current year (or a recent past year) based on field measurements and other data. To assess the carbon impacts of a practice over time, a necessary step to generate an annual estimate, projections of future carbon stocks must be made. This will be referred to as a Type II estimate and will require the use of lookup tables, simulation models, or other tools. A Type III estimate is used to assess the change in the GHG footprint as a result of a change in management practice. To generate a Type III estimate, a landowner will need to produce Type II estimates for the current practice and the practice under consideration and compare the two. While some landowners may require only an estimate of current carbon stocks (Type I estimate), many will be interested in generating estimates of the rate of carbon storage over time (Type II estimate), which necessitates the use of models to project forest growth. The overall goal of this guidance is to enable a landowner to develop an estimate of their GHG footprint and to assess the potential effects of changes in management practices or land use on this footprint (for forest systems, this will be dominated by carbon). Type II estimates can be generated and compared for the current management scheme and multiple alternatives (which may include a “no action” scenario). Comparing the estimates permits landowners to evaluate the potential impacts of a wide range of possible factors, including foregone growth, land-use change, and changes in management practices.

Generally, entities report annually for the life of a project. Since forests may last indefinitely, there is no biological ending, although events such as land-use change, a natural disturbance, or biome shift from climate change may effectively end the life of a specific forest or forest type. Various programs may impose time limits for reporting, or the entity may choose a project length that is consistent with management objectives. The accounting methods are not affected by project or reporting period length; therefore no specific recommendations are made in this guidance.

6.1.3 Summary of Selected Methods/Models

6.1.3.1 Field Measurements of Carbon Pools and Fluxes

Methods for estimating the key forest carbon pools are well developed and fairly standard. Pools are defined in Section 6.2, although detailed methods are not given. Methods for measuring forest carbon stocks are described in a variety of publications, including the IPCC Good Practice Guidance for Land Use, Land Use Change, and Forestry (IPCC, 2003), Pearson et al. (2007), and Hoover (2008), among others. As the Forest Inventory and Analysis (FIA) program of the USDA Forest Service is the Federal program tasked with providing national-scale estimates of the U.S. forest carbon stocks/flux (Heath et al., 2011), documented inventory procedures from this program (USDA Forest Service, 2010a; 2010b) serve as a basis for many facets of entity level carbon reporting prescribed in this document.

6.1.3.2 Lookup Tables and Regional Estimates

The most comprehensive collection of tables of carbon stock estimates is Smith et al. (2006). Estimation methods are described, and estimates for each carbon pool are provided by forest type for each region of the conterminous United States. The volume includes methods and tables to estimate carbon in HWPs.

6.1.3.3 Models

A variety of models may be used to assist in the estimation of forest carbon stocks and stock changes. Models will be described in more detail in the sections that follow, but for reference purposes, brief summaries of the most commonly used models are provided below. Some of these models are complex and may require a substantial time investment. Interacting with some of these models often requires specialist knowledge or training or both. For such models, an online estimation tool could be developed so that landowners would not need to learn each individual model, but would interact with them through the interface of an estimation tool, while the components operate in the background. While all models have strengths and limitations, the models recommended for use in each section of this report were selected because of their nationwide coverage, history of performance, and suitability for this task.

Forest Vegetation Simulator and Fire and Fuels Extension Carbon Reports. The Forest Vegetation Simulator (FVS) is a national system of growth and yield models, with multiple regional variants, that can be used to simulate growth and yield for U.S. forests. FVS is a stand-level model and can simulate nearly any type of forest management practice. The Fire and Fuels Extension (FFE) to FVS can be used to generate reports of all carbon pools except soil but including HWPs; non CO_2 GHGs are not included.[1] A number of geographic variants are available, each with regionally specific equations and default values.[2]

i-Tree. Two of the tools in i-Tree estimate carbon storage within urban trees, annual carbon sequestration, and carbon emissions avoided through energy conservation due to urban trees. One tool, the Urban Forest Effects (UFORE) model, focuses on an entire urban forest. The other tool,

[1] See http://www.fs.fed.us/fmsc/fvs/index.shtml
[2] Suggested variants may be found here : http://www.fs.fed.us/fmsc/fvs/whatis/index.shtml

STRATUM, focuses on street tree populations. Tree sample (e.g., from random field plots) or inventory data are required to run the model. Models to estimate future carbon effects based on local field data and user-defined mortality and planting rates have also been developed.[3]

First Order Fire Effect Model. The First Order Fire Effects Model (FOFEM) is a national level model with geographic variants, designed to predict tree mortality, fuel consumption, smoke production, and soil heating caused by prescribed fire or wildfire.[4]

COMSUME. CONSUME is a decision-making tool designed to assist resource managers in planning for prescribed fire and impacts of wildfire. CONSUME predicts fuel consumption, pollutant emissions, and heat release based on fuel loadings, fuel moisture, and other environmental factors.[5] It allows estimation of GHG emissions and consumption from post-harvest and thinning activities.

6.1.4 Sources of Data

Sources of available data that may be appropriate for use in developing estimates of GHG emissions and carbon sequestration vary by carbon pool (or flux). In all cases, field collection of data is possible, and may be the only available approach for those instances where credible default values have not been developed and/or lookup tables are not available; this may be particularly relevant for agroforestry and urban forestry applications. In the case of many of the non-living forest carbon pools, regional default values are available for down dead wood (DDW), forest floor, and standing dead wood through the FIA program, as well as a number of documents developed in support of official U.S. government estimates. All FIA data are available through a number of portals, including the FIA database tools—Forest Inventory Date Online (FIDO) and EVALIDator—and the CarbonOnLineEstimator (COLE),[6] which interacts directly with the FIA database. See Table 6-2 for a partial list of potential data sources.

Currently, values for soil organic carbon (SOC) stocks are drawn from the State Soil Geographic (STATSGO) database, and are of coarse spatial resolution. A limited amount of field-sampled SOC data are also available through the FIA database as part of the Forest Health Monitoring portion of the inventory process. Carbon in live tree biomass is also available from FIA and like other variables can be retrieved at the county level. The FIA sampling design is intended to meet a specified error target at large areas of forest land; so FIA data may not be appropriate for use at smaller spatial scales. Estimates based on a small number of plots may present an unacceptable error level. COLE and EVALIDator provide error estimates for all variables; these values should be carefully considered before the data are used to develop estimates for a particular site.

Data for emissions of other GHGs from forests are not widely available, although estimates and calculation methods are better developed for N_2O than CH_4. The U.S. EPA and IPCC provide estimation methods and emissions factors for both gases from wildfires, and for N_2O from forest fertilization (IPCC, 2006; U.S. EPA, 2011). The U.S. EPA publishes a National Emissions Inventory every three years, which provides estimates for wildfire as well as prescribed fire for criteria pollutants as well as hazardous air pollutants, including some GHG species (U.S. EPA, 2012a).

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[3] See http://www.itreetools.org/
[4] See http://www.firelab.org/science-applications/fire-fuel/111-fofem
[5] See http://www.fs.fed.us/pnw/fera/research/smoke/consume/index.shtml
[6] See http://www.ncasi2.org/COLE/index.html. COLE was developed through USDA Forest Service financial support, but is currently hosted by NCASI.

6.1.5 Organization of Chapter/Roadmap

This chapter provides guidance on estimating carbon sequestration and GHG emissions for the forest sector. In cases where a landowner's holdings involve multiple land uses, guidance for the other sectors should be consulted. In this chapter, attempts to note areas where cross-sector interactions are likely to occur have been made. Wetlands and hydrologically managed soils are important in several sectors, and for this reason guidance for estimating GHG emissions and sequestration from wetland systems is covered in a separate section, outside of the croplands/grazing lands and forest sectors.

The chapter is organized to provide an overview of the elements of forest carbon accounting, including definitions of the key carbon pools and basic methods for their estimation. Next is a section relating to estimation methods in cases where forests have been established, re-established, and/or cleared. The forest management section considers the GHG implications of a variety of commonly employed management practices, and is followed by guidance on the estimation of carbon in HWPs. While agroforestry systems and urban forests may not be considered as traditional forest landscapes, the working group recognizes the importance of trees located outside of forests. Since the most important component in these systems is often the live biomass, urban systems have been included in the forest sector. Agroforestry is a complex topic, combining aspects of forestry, cropland agriculture, and animal agriculture. Since agroforestry is most likely to be practiced on lands primarily used for agriculture, the estimation guidance is provided in the croplands and grazing lands section of the document. It is important to note that agroforestry has many cross-sector linkages, and a complete estimate of the GHG implications of agroforestry practices may necessitate consultation of the forest methods provided here. As noted above, natural disturbance is one of the important risks of reversal in the forest sector, and the final section provides guidance on estimating the impacts from natural disturbance in forested systems.

The remainder of this chapter is organized as follows:

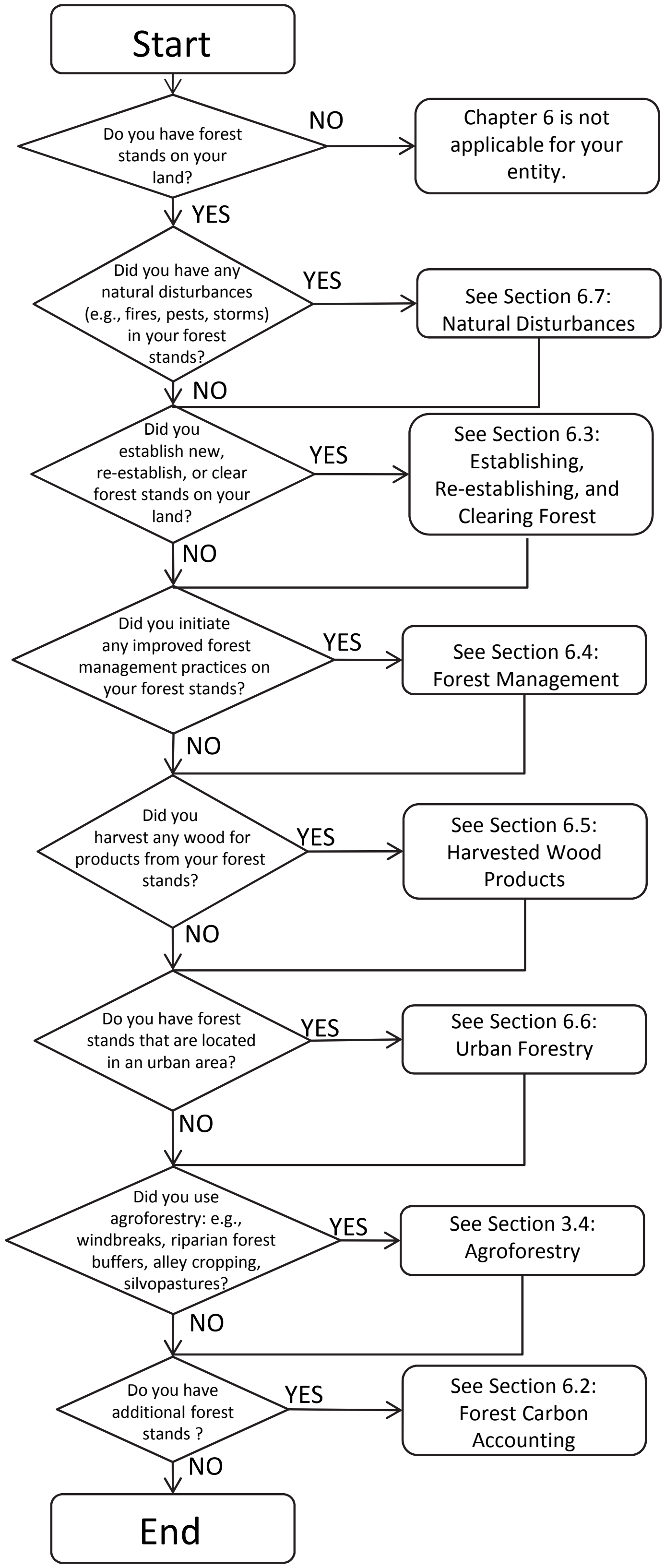
- Section 6.2: Forest Carbon Accounting
- Section 6.3: Establishing, Re-establishing, and Clearing Forest
- Section 6.4: Forest Management
- Section 6.5: Harvested Wood Products
- Section 6.6: Urban Forests
- Section 6.7: Natural Disturbances

Table 6-2 shows internet sites available for information on carbon estimation. Figure 6-2 shows a decision tree for the forest sector showing which forest chapter sections (i.e., source categories) are relevant depending on which forest activities are taking place for an entity.

Table 6-2: Internet Sites Available for Information on C Estimation

Internet site	Organization	Relevant Content
http://fia.fs.fed.us/	USDA Forest Service, Forest Inventory and Analysis	▪ Forest statistics by state, including carbon estimates ▪ Sample plot and tree data ▪ Forest inventory methods and basic definitions
http://www.fhm.fs.fed.us/	USDA Forest Service, Forest Health Monitoring	▪ Forest health status ▪ Regional data on soils and dead wood stocks ▪ Forest health monitoring methods
http://www.usda.gov/oce/climate_change/greenhouse.htm	USDA GHG Inventory	▪ State-by-State forest carbon estimates
http://unfccc.int/ http://www.ipcc.ch/	UNFCCC and IPCC	▪ International guidance on carbon accounting and estimation
http://soildatamart.nrcs.usda.gov/	USDA Natural Resources Conservation Service	▪ Soil Data Mart: access to a variety of soil data
http://www.nrs.fs.fed.us/carbon/tools/	USDA Forest Service, Northern Research Station	▪ Accounting and reporting procedures ▪ Software tools for carbon estimation
http://www.eia.gov/oiaf/1605/gdlins.html	U.S. Energy Information Administration, Voluntary GHG Reporting	▪ Methods and information for calculating sequestration and emissions from forestry; see Part I, Appendix
http://www.epa.gov/climatechange/emissions/usinventoryreport.html	U.S. Environmental Protection Agency	▪ Methods and estimates for GHG emissions and sequestration
http://www.comet2.colostate.edu/	USDA Natural Resources Conservation Service and Colorado State University Natural Resources Ecology Lab	▪ Web-based tool for estimating carbon sequestration and net GHG emissions from soils and biomass for U.S. farms and ranches

Figure 6-2: Decision Tree for Forest Sector Showing Relevant Chapter Sections Depending on Applicable Source Categories



6.2 Forest Carbon Accounting

Methods for Forest Carbon Accounting Utilized in this Guidance

- Range of options dependent on the size of the entities' forest land including:
  - FVS-FFE module (entities that fit the large landowner definition), and
  - Default lookup tables (entities fitting the small landowner definition).
- These options use:
  - Allometric equations from Jenkins et al. (2003a), and
  - Default lookup tables from Smith et al. (2006; GTR NE-343)—default regional values based on forest type and age class developed from FIA data.
- These methods were selected because they provide a range of options dependent on the size of the entities' forest land.

6.2.1 Description of Forest Carbon Accounting

The basic question inherent within the broader context of forest carbon estimation is: "How much carbon is in this forest?" Any discussion of forests or forestry activities in the context of GHGs depends on quantifying forest carbon. Forest ecosystems are generally recognized as significant stocks of carbon, and aggrading, or growing, forests can be strong carbon sinks. Disturbances and forest management influence the size and rates of change of these stocks. It is important to note that forest carbon generally is not measured directly (e.g., collecting forest biomass samples for laboratory determination of carbon content). It is usually quantified indirectly from standard forest inventories and associated carbon models (e.g., litter carbon dependent on forest type and stand age). For live tree pools, forest inventories often only measure limited dimensional attributes (e.g., diameter and height) of individual trees and use biomass component models (e.g., bole and crowns) and wood density values to convert these values into an estimate of total tree biomass. Once an estimate of biomass is attained, a standard carbon conversion constant is applied to produce a carbon stock estimate. Carbon conversions vary slightly, but 50 percent of dry weight is a useful round value applicable to all vegetation and sound wood (IPCC, 2006). For other pools, such as litter layers and soil organic matter, specific carbon content per unit volume depends on decay and composition of the material and is generally less than 50 percent carbon. Given the diversity of estimation procedures and carbon pool definitions, a reasonable selection of methodologies should be available for entities wishing to assess their forest carbon.

A major attribute of carbon "accounting" is to explicitly document and define accounting procedures such that forest carbon reports are comparable across ownerships and forest ecosystems. Absolute quantities of carbon, or carbon mass, are not only a function of a specific forest but also dependent on how pools are defined and how the mass of carbon within the pool is estimated. For example, both remotely sensed images and ground-based tree measurements can provide separate estimates of the same forest. These two techniques are unlikely to provide identical estimates due to methodological differences, including the fact that each approach may define different populations of interest and thus account for different sets of trees. Identifying and resolving such issues is an objective of forest carbon research. Not all forest carbon assessments or management plans need to encompass all carbon (or GHGs) pools if the carbon is properly identified. Measuring the current state of a forest's carbon stocks and recent changes is a part of developing a baseline, which can then be used for additional analysis. A baseline of past carbon stocks and change can be constructed and used with modeling to determine projections of likely future carbon. Similarly, a baseline is necessary for analysis of alternate management options to evaluate potential for sequestration/emission. The technical specifications of baselines (e.g., starting year and included stock categories) are often a social/political decision, and are beyond the purview of this document. However, to standardize forest carbon accounting options for the purpose of entity reporting (e.g., woodland owners), this document will propose a single set of forest carbon pool definitions. The specific recommendations included here are intended to direct landowners to tools and data sources specially developed for quantifying forest carbon. Note that these listed processes are not intended to exclude alternative data summaries that may be available to entities. Details are discussed below in the discussion of the respective forest carbon pools, but the general options listed in decreasing accuracy (and cost) include the following:

(1) Measure/sample your forest and estimate carbon from these data (reduce sample data so as to then apply available biomass equations or other carbon conversion factors);

(2) Characterize your forest according to classifications (i.e., lookup tables) based on stand or site attributes derived from records in the nation's forest inventory database (FIADB) (Woodall et al., 2010; Woudenberg et al., 2010); or

(3) Use associated models (FIDO, COLE, etc.), which base your forest's carbon estimates on representative data sampled by others with critical dependent user variable input (e.g., stand age).

Note that the above three options are not necessarily mutually exclusive. For example, FIADB data or similar models (Option 2) are based on permanent inventory plot sampling and carbon conversion (Option 1), and lookup tables (Option 3) are based on the FIADB (Option 2). The recommended forest carbon inventory options involve tradeoffs in costs and level of information unique to the entities' forest land.

The process of obtaining forest carbon estimates depends on circumstances unique to each entity, but mostly depends on the intended audience and the resources available for forest inventory. For this guidance, a two-tier system is in place. The goal is to be as inclusive as possible while not creating a measurement burden. Smaller holdings that are not actively managed are unlikely to be inventoried; a two-tier approach permits owners of such holdings to estimate their footprint and the potential changes from changes in practices applied without incurring the costs of measurement. Smaller landowners who have inventory data or who wish to acquire it should use the tools and protocols described for large landowners.

Landowner size classes are defined as follows:

Landowners who hold 200 or more acres (80.9 hectares [ha]) of forest land should follow the methods for large landowners. Also, landowners who hold less than 200 acres (80.9 ha) of forest land should follow the methods for large landowners if three or more of the following are true:

- Landowner owns or manages more than 50 forested acres (20.2 ha)
- Landowner's forest is certified
- Landowner has developed a forest management plan
- Landowner's forested property has a history of timber harvesting
- Landowner participates in State forest tax abatement programs

Landowners not meeting the definition of large landowner should follow the methods for small landowners.

Recommended methods depend on forest landowner size. Small landowners may use generalized lookup tables based on region, forest type, and age class to estimate carbon stocks. Large landowners should collect standard forest inventory data and use the FVS-FFE module with Jenkins et al. (2003a) allometric equations. It should be noted that FVS and the FFE are large and complicated models; any tool that implements these methods will require development of a simplified user interface that interacts with FVS and FFE.

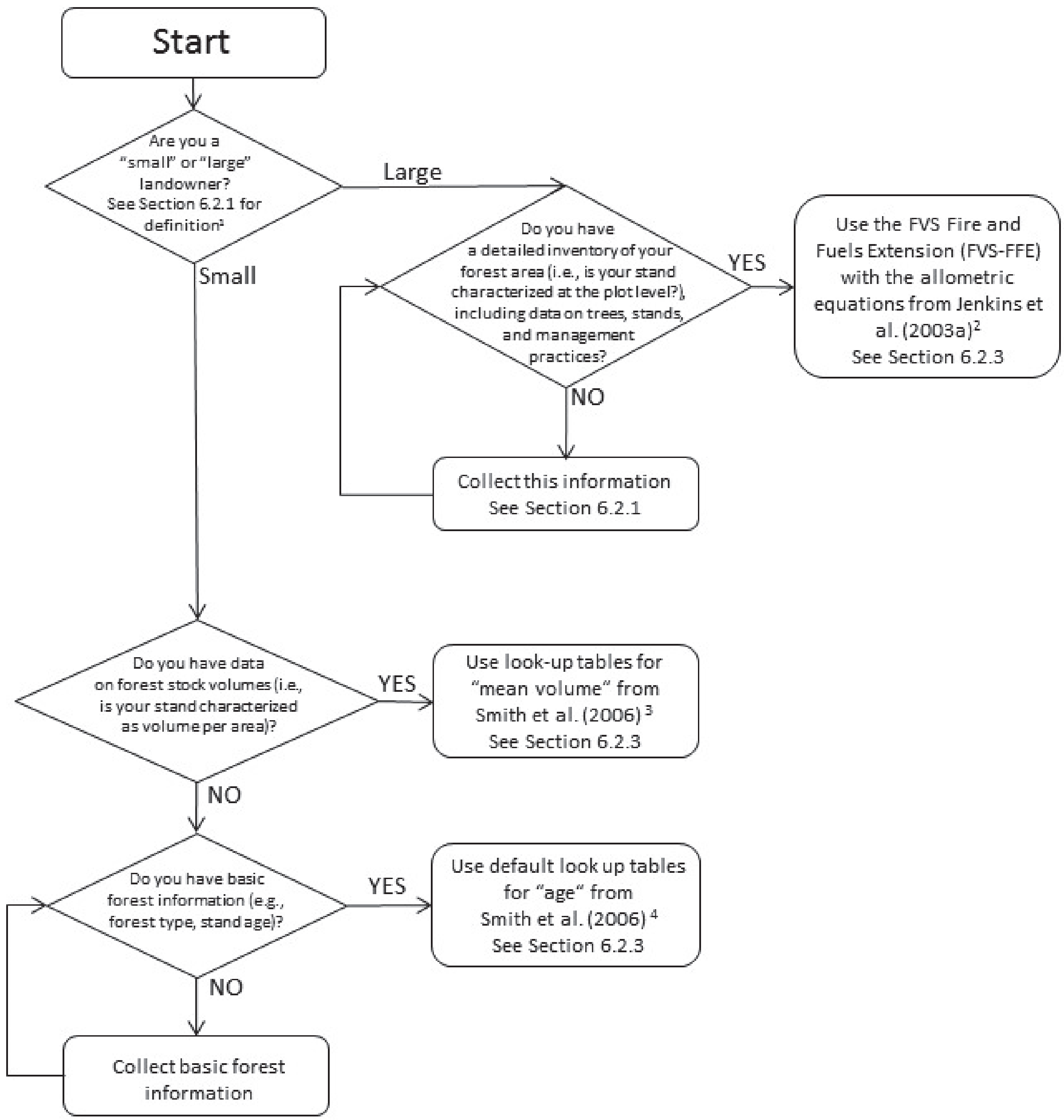
At this time, the Jenkins et al. (2003a) equations are specified since they are nationally consistent. Future development is likely to include the implementation of a more recent FIA biomass estimation method in FVS, enabling the production of estimates that match the official U.S. forest carbon estimates. While local volume or biomass equations may be more accurate for a given location, use of such equations will result in additional inconsistencies in results, so no other equations are approved for use at this time under this methodology.

Although carbon reporting beyond that of the entity level (e.g., major timberland owner or national forest) may use refined measurement protocols, expanded carbon pool definitions, and/or ancillary data (e.g., remotely sensed imagery), the proposed pools and inventory methodologies in this document serve as a starting point. Classification of carbon estimates within multi-tiered systems, and links to models to project future change under alternate scenarios are addressed at the end of Section 6.2.

To facilitate accounting, forest carbon is typically classified into a few discrete pools, which should be comprehensive (all organic carbon) with no gaps and no overlap. The purpose of establishing these separate pools, or bins, of forest carbon is twofold: (1) to align appropriate data with ecosystem/product components (e.g., tree inventories and live tree carbon pool), or alternatively to identify gaps; and (2) as a part of the accounting process, not all reported stock or change necessarily needs to include all of the carbon pools, but what is included must be unambiguously identified. Note that the carbon pools (or bins or classifications) focus on carbon from phytomass. Strictly speaking, total carbon stocks within a forest include a non-plant (not originating from the plant kingdom) percentage, but such pools are not defined because this is generally an insignificant proportion. Exceptions are the forest floor and soil pools, which include decomposers and soil fauna. A sometimes significant amount of carbon is removed from forests as wood is harvested and used in wood products. Some of that carbon remains sequestered for long periods of time, depending on the products. Thus, harvested wood should be included in forest carbon estimates.

Figure 6-3 is a decision tree for the forest carbon accounting source category showing which carbon accounting assumptions (e.g., simulation models, allometric equations, biomass expansion factors, lookup tables) are recommended for an entity depending on the type of activity data available. However, it should be noted that for national reporting—i.e., the annual GHG inventory reported by USDA and U.S. EPA—where individual tree measurements from FIA's inventory plots are available, the component ratio method (CRM) for estimating biomass (Woodall et al., 2011) is currently used. Again, future development will likely bring these methods into alignment.

Figure 6-3: Decision Tree for Forest Carbon Accounting Showing Methods Appropriate for Estimating Forest Carbon Stocks



[1] Small landowners (as defined in Section 6.2.1) may use generalized lookup tables based on region, forest type, and age class to estimate carbon stocks. Large landowners should collect standard forest inventory data and use allometric equations to estimate live tree biomass carbon (other carbon pools may be obtained from lookup tables).

[2] Jenkins et al. (2003a).

[3] Note that volume equations used by landowners should align with "mean volume" specifications (e.g., rotten/cull deductions) of Smith et al. (2006). Different volume equations and deductions will produce volume estimates that differ from those used in the tables.

[4] Smith et al. (2006).

Another aspect of a carbon accounting framework is consistent or comparable representation of change, which goes beyond the identification of carbon pools. Change is affected by processes of recruitment and growth as well as disturbance, mortality, and harvest. In the most basic sense, change can be the difference between two successive stock estimates. This is common for GHG reporting based on standard forest inventories. Some components of change can be measured with intensive sampling at small scales, but in general change is estimated from measurements at two successive inventory times (e.g., total stock change, or growth/removals/mortality estimates, or remotely sensed data), or based on models of ecosystem or biogeochemical change. A basic approach to quantifying change in forest carbon is based on the quantities defined for forest carbon stocks. Net annual carbon stock changes are calculated by taking the difference between the inventories and dividing by the number of years between the inventories for a selected forest or forest area (e.g., Δ stock = ($stock_2$ – $stock_1$)/time). This stock-change approach (IPCC, 2006) is the change method applied to FIA strategic-scale inventories for the stock-change values reported in the U.S. National GHG Inventories (e.g., U.S. EPA, 2011).

Six Steps to Forest Entity Carbon Estimation

The approach to estimation of carbon stocks and fluxes in the forest sector is as follows:

Step 1: **Determine landowner size class based on forest area.** Based on the acreage under consideration, landowners are divided into two groups: “small” landowners and “large” landowners as defined in Section 6.2.1.

Step 2: **Collect forest data.** For both size classes of landowners, some level of forest inventory (i.e., field survey) data is required. However, there are differing data requirements for small landowners and large landowners.

Small landowners should collect basic data on species mix (i.e., type of forest) and stand age (or time since last major disturbance) within their forest. Greater inventory detail can lead to more precise estimates of carbon, but even broad generalizations about the region, age (and/or mean volume), and type of forest can lead to a carbon estimate. The objective is to obtain reasonable and consistent estimates over time at the lowest cost. If a small landowner wishes to conduct an inventory and follow the recommended guidance for large landowners, they are free to choose this option. The principal tradeoff is between cost and accuracy; collecting inventory data increases the cost of developing estimates but increases accuracy.

Large landowners should gather more extensive data about forest and stand characteristics. A thorough forest inventory is created using industry standards and practices of the type described in GTR NRS-18: *Measurement Guidelines for the Sequestration of Forest Carbon.* Variables considered must include dominant species, dominant age class, stand density, and site class. Inclusion of additional variables, while not required, will improve accuracy of carbon estimates.

Step 3: **Estimate initial forest carbon stock and annual fluxes.** Quantities of carbon change over time. Forest carbon estimates are divided into six discrete, mutually exclusive pools, including live trees, standing dead trees, understory vegetation, down dead wood, forest floor, and soil organic carbon. A number of pool-specific carbon conversion methods are available; these methods use the inventory data gathered in Step 2 to quantify carbon for each pool. However, the specific methods to be used differ depending on the landowner size class.

(Continued)

(Continued)

Small landowners, after collecting observational data, can use lookup tables from Smith et al. (2006) (also known as GTR-NE-343: *Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States*) to estimate carbon stocks and carbon stock changes. The lookup tables are categorized by region, forest type, previous land use, and in some cases, management activity. Users must identify the categories for their forests and estimate the area of forestland. To facilitate use of the data from GTR-NE-343, a tool could incorporate the data such that, in most cases, landowners would be able to select their stand characteristics from a drop-down menu of defaults. Based on the landowner's selections from the default menus, the tool would produce estimates of carbon stocks in each of the six carbon pools.

Large landowners should use the data collected in their forest surveys to perform model runs using the FVS model. FVS will use the site- and stand-specific data to provide more accurate estimates of carbon stocks in each of the carbon pools (excluding soil carbon, which FVS does not estimate). Soil carbon estimates can be determined from a range of methods including sampling or existing forest soil carbon estimate datasets depending on a specific entity's circumstances.

Though the methods differ for small landowners and large landowners, both calculate initial carbon stocks and expected annual rates of accumulation under average conditions (repeating the field survey at prescribed intervals will help calibrate or validate the stock change estimates).

The methods also allow for adjustments due to HWPs (Step 4), forest management practices (Step 5), and natural disturbances (Step 6).

<u>Step 4</u>: **Adjust carbon estimates due to HWPs.** Harvesting activities can have considerable impact on carbon quantity across the six forest carbon pools. In terms of emissions, the fate of the harvested material must be considered as well, including whether the material is used in HWPs or for energy. As above, the methods for estimating these impacts differ depending on the landowner size class.

For HWPs, *small landowners* should rely on data provided in lookup tables in GTR-NE-343, which provides factors for calculation of carbon in HWPs based on region, timber type, and industrial roundwood category. The lookup tables divide the harvested forest materials pool into four distinct fates: products in use, landfill, emitted with energy capture, and emitted without energy capture. Carbon emissions differ depending on the fate, which in turn depends on the region and harvest material characteristics. By using the lookup tables, landowners can adjust carbon estimates accordingly.

Large landowners should rely on FVS to model forest management practices, resulting in estimates of the carbon impact of these practices (e.g., harvesting). For example, FVS can consider the type of harvest (e.g., clear cut versus strategic thinning) and project the results of this harvest on carbon stocks, thus allowing users to quantify the carbon impact of various harvesting activities, as well as adjusting for the ultimate fate of harvested materials. The harvested forest material pool is divided by FVS into the same four distinct fates as for GTR-NE-343: products in use, landfill, emitted with energy capture, and emitted without energy capture. Harvests also impact forest growth over time, which is modeled by FVS.

(Continued)

(Continued)

Step 5: **Adjust carbon estimates due to improved forest management.** Forest management practices, such as thinning or fertilization, may impact carbon fluxes as well. As above, the methods for estimating these impacts differ depending on the landowner size class.

FVS allows *large landowners* to quantify the impact of various forest management practices. For example, using keywords (or combinations of keywords) provided by FVS, users can generate estimates for the impact of stand density management, site preparation methods, vegetation controls, various densities of planting stock, fertilization, rotation length management, prescribed fire/control burns and fuel load management, and pest and disease control. With given stand and tree-list data, users can develop a baseline, which can then be compared to alternative management strategies. This allows for assessment of carbon impact of implementing those management practices. It should be noted that FVS is the recommended method, even if a large landowner has its own custom inventory and modeling system, which might be considered superior to regional models such as FVS. The adoption of a single, recommended method for landowners allows for transparent, consistent, comparable, and complete estimates across landowners appreciating that there will be a likely trade off in the accuracy, cost effectiveness, and ease of use of the method for those landowners with custom systems. Future development may include a means for large landowners to use custom models in this framework, but this option is not available at this time.

Unfortunately, the lookup tables do not allow for estimates associated with improved forest management. If prescribed fire/control burning is used by either landowner type, it is recommended that the emissions for the activity be calculated as guided in Step 6.

Step 6: **Adjust carbon estimates due to forest fires and other natural disturbances.** Natural disturbances, such as forest fires, storms, wind, drought, or pest/insect infestation, can also have considerable impact on carbon quantities across the six forest carbon pools. Landowners should estimate the carbon impact of natural disturbances.

For forest fires, wildfires, and prescribed/controlled burns, both small and large landowners should rely on FOFEM to generate carbon estimates. FOFEM input requirements include basic forest type, site location, and dominant species data, but also allows users to input additional information, depending on a specific entity's circumstances, on amount of duff, moisture content, and other variables associated with fire. The severity of the fire can be categorized by percent of the land affected. The resulting output includes estimates of carbon emissions.

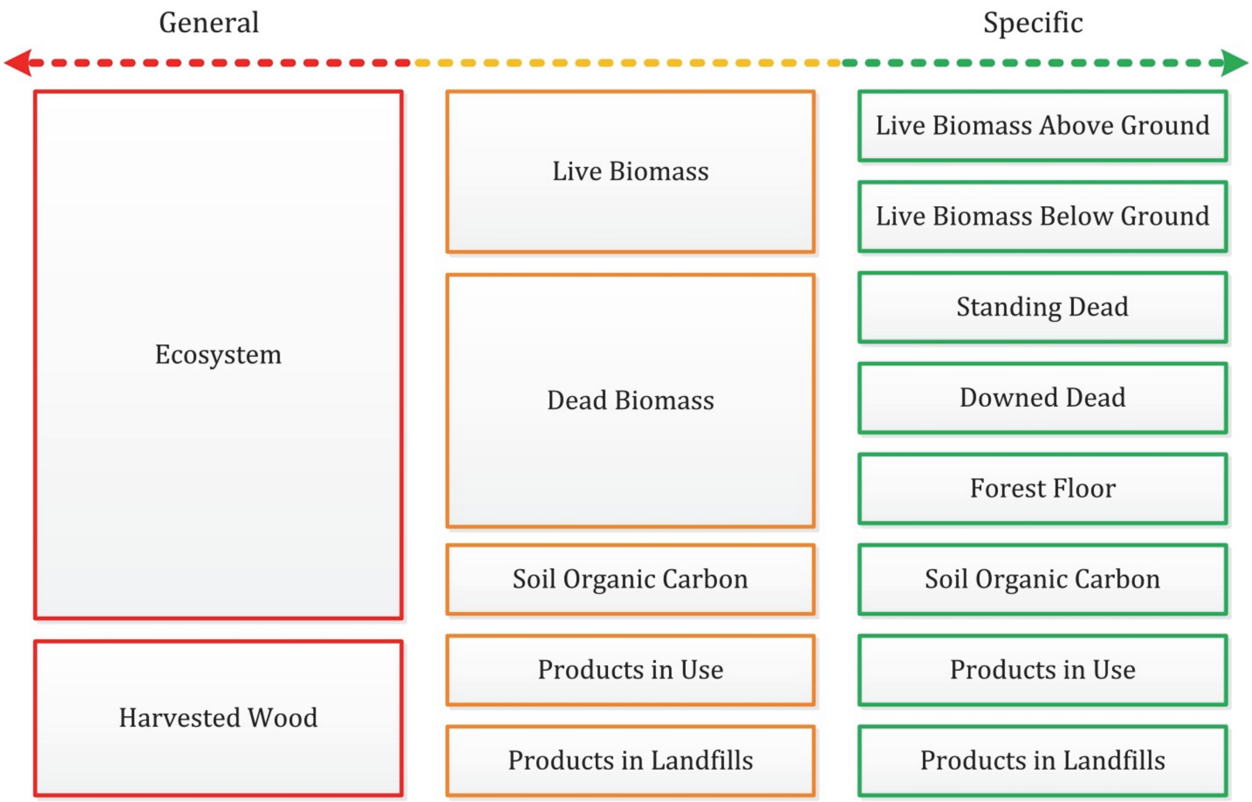
The methods assume s*mall landowners* can provide observational estimates for the impacts of natural disturbances such as pests, based on the percentage of forestland affected by the disturbance. *Large landowners* may model impacts of pests through available keywords and extensions provided by FVS.

The philosophy behind these six steps is that they allow the entity to assess what carbon stocks they have under any present conditions and what stocks they might expect given implementation of a particular harvesting regime, change in forest management practices, and/or a variety of natural disturbances.

6.2.1.1 Forest Carbon Pools

Carbon reporting—such as for the U.S. reporting commitment to the United Nations Framework Convention on Climate Change (UNFCCC), which is met by the U.S. EPA's official GHG inventory (e.g., U.S. EPA, 2011)—provides a framework for the pools described here. However, the pools are modified to more closely correspond to types of forest inventory data. For example, forest carbon can be easily categorized according to aboveground versus belowground, or living versus dead plant material. In practice, classifications of carbon pools depend on the forest data and how they are used. As such, the pools described below are jointly defined by UNFCCC reporting requirements and the use of FIA forest inventory as the primary data source. In other words, the pools defined below are a convenient set, but definitions and boundaries around pools can vary according to specific carbon estimation procedures/capabilities and reporting needs (see Figure 6-4).

Figure 6-4: Forest Carbon Pool Hierarchy Showing How Forest Carbon Pools Can Be Delineated into Even Smaller Pools Dependent on the Entity Needs and Inventory Capabilities



Live trees: A large woody perennial plant (capable of reaching at least 15 feet (4.6 m) in height) with a diameter at breast height (DBH) or at root collar (if multistemmed woodland species) greater than 1 inch (2.5 centimeters [cm]). Includes the carbon mass in roots (i.e., live belowground biomass) with diameters greater than 0.08 in (2 millimeters [mm], stems, branches, and foliage.

Understory: Roots, stems, branches, and foliage of tree seedlings, shrubs, herbs, forbs, and grasses.

Standing dead trees: Dead trees of at least 1 inch (2.5 cm) DBH that have not yet fallen, including carbon mass of coarse roots, stems, and branches, but that do not lean more than 45 degrees from vertical (Woudenberg et al., 2010), including coarse nonliving roots more than 0.08 in (2 mm) in diameter.

Down dead wood (also known as coarse woody debris): All nonliving woody biomass with a diameter of at least 3 inches (7.6 cm) at transect intersection, lying on the ground. This pool also includes some less-than-obvious components of DDW: (1) debris piles, usually from past logging; and (2) previously standing dead trees that have lost enough height or volume, or lean greater than 45 degrees from vertical, so they do not qualify as standing dead trees.

Forest floor: The litter, fulvic, and humic layers, and all fine woody debris with a diameter less than 3 inches (7.6 cm) at transect intersection, lying on the ground above the mineral soil.

Soil organic C: All organic material in soil to a depth of generally 3.3 feet (1 meter [m]), including the fine roots (e.g., less than 0.08 in (2 mm) in diameter) of the live and standing dead tree pools, but excluding the coarse roots of the pools mentioned earlier.

Harvested wood: Wood removed from the forest ecosystem for processing into products, not including logging debris (slash) left in the forest after harvesting.

These pool definitions are developed around a common set in use by a number of publications (e.g., Smith et al., 2006) and at the forest stand level, which in turn differ from stock definitions used by the United States to meet UNFCCC national reporting requirements.

Also notable (in the reporting list) is the inclusion of HWP (covered in detail in Section 6.5), which assumes that a measurable portion of wood removed at harvest remains sequestered from reemission to the atmosphere for a period of time that can be estimated. Pools and estimation of stocks are organized primarily according to data collection and estimation with FIA's permanent inventory plots (phase two (P2), the standard inventory measurements; and phase three (P3), the forest health measurements). Note that pool definitions are not independent of related estimators; details related to estimation are not addressed until subsequent sections of this guidance.

6.2.2 Data Collection for Forest Carbon Accounting

Forest carbon is typically estimated indirectly, through applying conversion constants to a standard forest inventory, using a localized biogeochemical model, or simply looking up specific forest attributes (e.g., stand age, forest type) in a lookup table (e.g., Smith et al., 2006). For the purposes of this documentation, a standard set of carbon pool definitions that are part of FIA's national inventory are delineated that correspond to available lookup tables (Smith et al., 2006).

6.2.2.1 Live Trees

The tree carbon pools include aboveground and belowground (coarse root) carbon mass of live trees. Separate estimates are made for full-tree and aboveground-only biomass to estimate the belowground component. Tree carbon estimates within the FIADB (USDA Forest Service, 2012; Woudenberg et al., 2010) are based on Woodall et al. (2011) and Jenkins et al. (2003a). The per-tree carbon estimates are a function of tree species, diameter, height, and volume of wood. Belowground biomass is calculated as a varying proportion of aboveground biomass. Again, this is dependent on species and size of individual trees. The pool of live trees within the FIADB is defined as trees, or woody biomass with greater or equal to 1 inch (2.5 cm) DBH. However, trees less than 5 inches (12.7 cm) DBH are sampled differently than those that are 5 inches (12.7 cm) or more. These differences should not affect precision in the overall amount of tree carbon or stand level density. Saplings are trees at least 1 inch (2.5 cm) but less than 5 inches (12.7 cm) DBH. The "sapling" versus larger tree distinction is based on sampling differences on the FIA plots. This illustrates that pool classification is dependent on both the obvious physical and spatial separation in a stand as well as data sources.

6.2.2.2 Understory

Understory vegetation is a minor component of biomass or the live plant component. Understory vegetation is defined as all biomass of undergrowth plants in a forest, including woody shrubs and trees less than 1 inch (2.5 cm) DBH. In FIADB-based carbon inventory, it is assumed that 10 percent of understory carbon mass is belowground. This general root-to-shoot ratio (0.11) is near the lower range of temperate forest values provided in IPCC (2006) and was selected based on two general assumptions: ratios are likely to be lower for light-limited understory vegetation compared with larger trees, and a greater proportion of all root mass will be less than 0.08 in (2 mm) in diameter. Estimates of carbon density are based on information in Birdsey (1996), which was applied to FIA permanent plots.

6.2.2.3 Standing Dead

The standing dead tree carbon pools include aboveground and belowground (coarse root) mass. Estimates and allometry are essentially similar to those for live trees, with some additional considerations for decay and mechanical/structural damage (Domke et al., 2011; Harmon et al., 2011). Carbon conversions vary slightly, but 50 percent is a useful round value for dead wood. However, specific carbon content is less for the litter and organic layers of the forest floor. There is not a dead plant material pool corresponding to understory; it is assumed these very quickly become litter or small woody debris. Pairing pool definitions (boundaries) with data sources is also very important with the pools of dead plant material, because measurements specific to estimates are much less likely for DDW, forest floor, etc. In the FIADB the distinction between "standing" and "down" dead wood is based on angle of lean and is applied to P2 (phase two, "standard" forest inventory plot) and P3 (phase three, a smaller number of plots that include additional measurements such as soils and forest floor) data; other definitions may vary. For small diameter standing dead trees, estimates exist but are problematic: FIA data only provide samples of standing dead trees at 5 inches (12.7 cm) DBH or larger. Estimates of saplings (1–5 inch (2.5—12.7 cm) DBH trees) necessarily will be modeled (Woodall et al., 2012).

6.2.2.4 Down Dead Wood

DDW is defined as pieces of dead wood no longer a part of standing dead or snags, yet distinct from smaller or advanced decayed wood of the forest floor. The definition largely corresponds to the P3 down woody material pool, and represents a slight change from the past definition. This pool also includes some less-than-obvious components of DDW: (1) debris piles, usually from past logging; (2) previously standing dead trees that have lost enough height or volume or lean greater than 45 degrees from vertical so they do not qualify as standing dead; (3) stumps with coarse roots (as previously defined); and (4) nonliving vegetation that otherwise would fall under the definition of understory.

6.2.2.5 Forest Floor or Litter

The forest floor is the layers of litter, often classified as the fibric (O_i), hemic (O_e), and sapric (O_a) organic layers above the mineral soil and smaller than DDW. This classification represents a change from the past definition, which also included the small woody debris from the DDW pool. Organic soils present additional challenges when delimiting this pool.

6.2.2.6 Forest Soil Organic Carbon (SOC)

This pool is organic carbon within the soil but excluding coarse roots as defined for live trees, understory, standing dead trees, and stumps—all as defined above. By convention, large pieces of woody material that are separately and independently estimated through sampling and allometry are excluded. Depth is arbitrary and so far has been defined by the dataset in use. The dataset should represent samples of as much of the organic carbon as possible, although peatlands present a unique problem. A common sampling depth is 1 m, although this is not an IPCC standard. Adequate sampling depth may be ascertained through local knowledge; 3.9 to 7.9 inches (10 to 20 cm) may be adequate for some forest ecosystems, while others require greater depths. Datasets of soil maps from surveys are another source of data (in addition to P3 plots). SOC variability extends to relatively large-scale maps such as locations surrounding P2/P3 plots. That is, soils maps are based on data with the same variability as seen in the P3 subplot-to-subplot precision.

Note that the pool definitions used by FVS do not match definitions used by FIA in all cases. While the main categories of live and dead biomass will include the same elements, the FIA definition of forest floor includes fine woody debris, while the FVS-FFE definition places fine woody debris in the DDW category. FIA considers trees under 1 inch (2.5 cm) DBH to be part of the understory pool, while FVS tracks these as trees regardless of size. Future work is likely to include the capability of FVS-FFE to generate a carbon report with pools corresponding to the definitions used by FIA in national accounting.

6.2.3 Estimation Methods

The flexibility in using the best obtainable data balanced with the needs and resources of each individual forest owner can provide good/valid forest carbon estimates if some basic guidelines are followed:

- Carbon pools should be explicitly identified to make it possible to identify possible gaps or overlaps between pools. Identifying and recognizing that a gap exists (for example, there are no seedling data, or standing dead trees were not measured) is more useful than fuzzy boundaries between pools.
- Consistent pool definitions and methods for carbon estimation within those pools are required for valid estimates of change. That is, change should be based on the same pools and methods at both time 1 and time 2.

6.2.3.1 Live Trees

Various approaches are used for estimates of tree biomass or carbon content; ultimately, each relies on allometric relationships developed from a characteristic subset of trees. Here, live trees include stems with DBH of at least 1 inch (2.5 cm). Allometry can incorporate whole trees or components such as coarse roots (greater than 0.08 to 0.20 inches (0.2 to 0.5 cm); published distinctions between fine and coarse roots are not always clear), stems, branches, and foliage. Live tree belowground carbon estimates can be troublesome, but overall accuracy is best if the boundary is set to conform to available data rather than a predefined threshold.

Recommended options for obtaining estimates of carbon stock of live trees are:

- Small landowners (as defined in Section 6.2.1): Values obtained from lookup tables (e.g., either those in Smith et al., 2006, or as otherwise provided) categorized by geographic region, forest type, and age class.
- Large landowners (as defined in Section 6.2.1): Standard forest inventory, estimates calculated using individual tree measurement (diameter) and the FVS-FFE module with the Jenkins biomass equations (Jenkins et al., 2003a).

Biomass equations must be applied appropriately; using equations outside the diameter or geographic ranges for which they were developed will introduce additional error to the estimates. Given the hundreds of different tree species growing in diverse habitats across the United States, it is beyond the scope of this document to suggest the magnitude of the effect of alternative tree volume models beyond the national-scale models suggested herein. Regardless of the estimation approach selected, it is critical to use that method consistently over time. Estimates produced from different methods will vary; changing estimation methods over time will introduce additional error.

Although we are currently specifying only the use of biomass equations by Jenkins et al. (2003a), it is understood that these equations may not be the most appropriate in all circumstances. For example, using equations outside the diameter or geographic ranges for which they were developed will introduce additional error to the estimates. Some Jenkins equations have limits to the allowable diameters. Specific guidance will be developed in the future to facilitate the use of different biomass equations such as those used by FIA based on the CRM and locally-specific equations. Refer to Figure 6-3 for a decision tree for the forest carbon accounting source category showing which carbon accounting assumptions (e.g., simulation model, allometric equations, and lookup tables) are recommended for an entity depending on the size class and type of activity data available.

Sampling and Allometry. Recommended approaches are based on the application of allometric relationship to sampled inventory data. The FIADB-based estimates of live tree carbon are based on the plot data–P2 data and CRM biomass estimation (Woodall et al., 2011). In addition, a large number of other allometric relationships have been developed for tree biomass (biomass regression equations). Many biomass equations are available for a variety of forest types; for example, possible older citations are Ter-Mikaelin and Korzukhin (1997); see also citations in Jenkins et al. (2003b). The equations recommended in this report are the Jenkins et al. (2003a) equations, which are nationally consistent and straightforward to apply. Future development or integration of this method into a software tool should consider implementation of the CRM biomass estimation method in order to better align with the methods used for U.S. GHG inventory reporting. The CRM approach is computationally complex, and is not included at this time.

Inventory designs and protocols are well documented by a variety of authors and will not be discussed further here. A good example is Pearson et al. (2007), which is written specifically for carbon inventories.

Lookup Tables. Published summary values of similar or representative forests provide quick and inexpensive means of roughly assessing likely forest carbon. A good example of such lookup values are the past revised 1605(b) guidelines, with the forest tables published as Smith et al. (2006). Alternative versions of representative values include FIA online applications such as FIDO or EVALIDator, FIA-related applications such as COLE, or models from spatial data such as the FIA biomass map or the National Land Cover Dataset layers.

Simulations/Modeling. Not only do forest biometrical models provide a platform for estimating future scenarios of forest carbon stocks, but they can also be a rapid methodology for entity-level calculation of current forest carbon stocks. The FVS is one such simulation tool that can provide estimates of current forest carbon stocks given an elementary forest inventory was conducted (e.g., number of trees, size, and species). In addition, and perhaps a more powerful aspect of such a tool, is that projections of future stand attributes can be acquired (e.g., forest carbon stocks 50 years from present) as described in Dixon (2002) and Hoover and Rebain (2008; 2011).

6.2.3.2 Understory

Estimation procedures and data sources are limited for this pool. Unless an entity has the capability to develop localized understory models and allometric relationships, the development of carbon estimates for these pools will be limited to lookup tables and simulations/modeling. Values are provided in the Smith et al. (2006) lookup tables, which are based on Birdsey (1996) and modified to apply to FIA data; see U.S. EPA Annex 3.12 (2010) for additional details. The FIADB condition table includes estimates based on this model, so estimates based on similar stands can be obtained from the FIADB. Understory values are provided in the carbon reports in FVS and are regional default values set within the model.

6.2.3.3 Standing Dead

The prevailing difference in volume/biomass/carbon estimation of standing dead trees from live trees is the incorporation of decay reduction factors and rotting/missing/cull components (Domke et al., 2011; Harmon et al., 2011).

Sampling and Allometry. FIA inventory-based estimation for standing dead trees is from P2 plot, condition, and tree records. Tree mass in the FIADB is calculated according to CRM methods (Woodall et al., 2011) with refinements to the CRM approach specific to standing dead trees proposed by Domke et al. (2011). During a standard forest inventory, standing dead trees are measured and tallied, and large landowners can use this information with FVS to produce estimates of the biomass and carbon in this pool.

Lookup Tables. Published summary values of similar or representative forests provide quick and inexpensive means of roughly assessing likely forest carbon. A good example of such lookup values are the past revised 1605(b) guidelines, with the forest tables published as Smith et al. (2006). Alternative versions of representative values include FIA online applications such as FIDO or EVALIDATOR, and FIA-related applications such as COLE. Note that some differences may appear among pool estimates compared to the sample estimates, because some or all are based on empirical models (regressions) and not the direct plot-level measurements that are now available within the FIADB. Small landowners can obtain estimates of the standing dead pool using the Smith et al. (2006) lookup tables.

6.2.3.4 Down Dead Wood

The recommended method for obtaining estimates of carbon stock of DDW for large landowners is estimation from transect data collected during the inventory. Care should be taken to adhere to the bounds between the DDW and forest floor pools (noting that fine woody debris is considered part of the forest floor pool in this guidance). Small landowners may refer to the lookup tables for pool estimates.

Sampling and Allometry. A variety of sampling and estimation protocols is available for the DDW pool; a straightforward and commonly used approach can be found in Pearson et al. (2007).

Lookup Tables. Regional averages by forest type are as described in Smith et al. (2006), or estimates can be summarized and extracted from the FIADB condition table to correspond to the entity's forest. However, note that the current FIADB's DDW from the condition table is a model independent of P3 sampling. See Smith et al. (2006), U.S. EPA Annex 3.12 (2010), Woodall et al. (2013), and Domke et al. (2013) for details.

Simulations/Modeling. DDW carbon values are provided in the carbon reports in FVS. Values may be supplied by the landowner; if these data are not available, regional default values based on P3 data or available data for the region and forest type are automatically input by the model.

6.2.3.5 Forest Floor or Litter

Recommended options for obtaining estimates of carbon stock of forest floor for all landowners is the use of lookup tables based on forest type, region, and stand age. Large landowners who are changing land uses from non-forest to forest may wish to collect data for this pool.

Sampling and Allometry. Landowners wishing to estimate these pools from field data can use fine woody debris sampling and carbon conversion according to Woodall and Monleon (2008), and forest floor using the approach described by Pearson et al. (2007). Note that while Pearson et al. (2007) apply a mass to carbon conversion factor of 0.5 (Smith et al., 2006)), others use a conversion factor of 0.37. Landowners who are estimating the forest floor pool using field data should apply the 0.37 conversion factor.

Lookup Tables. Regional averages by forest type are as described in Smith et al. (2006); estimates can also be summarized and extracted from the FIADB condition table to correspond to the entity's forest. These estimates are based on simulations described in Smith and Heath (2002). Note that the current FIADB condition table estimates of forest floor are these modeled values independent of the P3 sampling.

Simulations/Modeling. Forest floor carbon values are provided in the carbon reports in FVS. Values may be supplied by the landowner; if these data are not available, regional default values based on P3 data or available data for the region and forest type are automatically input by the model (FVS employs the 0.37 mass to carbon conversion factor when estimating this pool).

6.2.3.6 Soil Organic Carbon

Possible options for obtaining estimates of SOC stocks are:

- Sampling, following standard field methods;
- Datasets such as the Soil Survey Geographic (SSURGO) Database, State Soil Geographic (STATSGO) Database, or the Digital General Soil Map of the United States (STATSGO2); and
- Stand/forest classification: extract range of modeled estimates from FIADB condition table.

Sampling and Allometry. Soil sampling and carbon estimation according to FIA P3 plot protocols can be found at the USDA Forest Service FIA Library: Field Guides for Standards (Phase 3) Measurements;[7] methods are also available in Pearson et al. (2007), Hoover (2008), and others.

Soils data are generally considered difficult to measure and spatially quite variable. The consequence is that the costs are high and the payoff is likely low. Our recommendation is that sampling is only useful if there is an important reason to do so, such as a change from non-forest to forest or vice versa. If a wildfire occurs and there is significant consumption of peatlands, sampling should be conducted and emissions calculated using FOFEM and/or CONSUME models. This situation is most likely to be found in the Southeast or North Central States.

Lookup Tables. Forest soil organic carbon estimates—representative values or lookup tables. Data sets such as STATSGO or SSURGO are possible sources. Estimates can be summarized and extracted from the FIADB condition table to correspond to the entity's forest; these are based on a STATSGO/P2 overlay (Smith et al., 2006; U.S. EPA, 2010).

6.2.4 Limitations, Uncertainty, and Research Gaps

There is often tremendous uncertainty associated with estimates of forest carbon baselines, such that even at large scales (e.g., state-level) the power to detect statistically significant changes in forest carbon stocks is limited to major disturbances (Westfall et al., 2013). Compounding the sampling error often associated with forest inventories, there is measurement and model error that may not be acknowledged. Users of any inventories, lookup tables, or models should remain aware of these potential errors during their application of information.

There is a level of uncertainty associated with not only tree volume/biomass equations, but also with the various forest carbon pools (e.g., belowground to forest floor) found across a diversity of forest ecosystems (e.g., tropical to boreal) in the United States. Research to refine approaches to forest carbon accounting and refinements of associated models is currently in progress. Perhaps

[7] http://fia.fs.fed.us/library/field-guides-methods-proc/

some of the most needed improvements are for individual tree volume/biomass equations, especially for traditionally non-commercial species. Another forest carbon pool that is being investigated is soil organic carbon. Although the soil carbon pool is not expected to change quickly in comparison to live tree pools, in many areas of the United States it is the largest carbon stock (e.g., northern Minnesota). Beyond reducing the uncertainty associated with estimates of carbon pools, research is being conducted to refine understanding of the effects of disturbance and climate change on carbon pools.

6.3 Establishing, Re-establishing, and Clearing Forests

Methods for Establishing, Re-establishing, and Clearing Forest

- IPCC algorithms developed by Aalde et al. (2006).
- These options use:
  - Allometric equations from Jenkins et al. (2003a), or FVS with the Jenkins et al. equations where applicable; and
  - Default lookup tables from Smith et al. (2006; GTR NE-343)—default regional values based on forest type and age class developed from FIA data.
- These methods were selected because they provide a range of options dependent on the size of an entity's forest land.

6.3.1 Description

Conventional parlance attributes changes of carbon on a site undergoing land-use change into three directional processes: establishing (i.e., afforestation), re-establishing (i.e., reforestation), and clearing forest (i.e., deforestation). In recent years, the term forest degradation has been used to acknowledge that an existing forest can be significantly reduced in carbon stocks and can be considered a source of emissions, as long as the reduction in carbon stocks is not an aspect of normal forest management. However, this is not a form of land-use change because the land remains in forests. This is an important consideration under forest management, but may also be important when human use and removals of forest stocks take place even when not prescribed by a management regime. The most important source of GHG emissions from forests is associated with forest clearing (IPCC, 2007). The conversion of forests to other land uses immediately reduces the stock of carbon in aboveground biomass and soil organic matter, and is likely to reduce the long-term carbon storage potential of the land. The carbon that was once stored in forest biomass and soil is reduced through rapid oxidation by fire or slowly over time by microbial decomposition. Some of the biomass can also be removed from the site and converted to forest products such as lumber, paper, pulp, and other products that have longer term but variable decomposition rates—and hence longer term and variable emissions over time. All of these components of land-use change need to be accounted for when determining the changes in site carbon stocks due to land-use change.

A parcel of land can be converted to forest, plantation, or other treed landscape either through intentional planting or the natural process of secondary succession. Land that had once been in forest is returned to forest through re-establishment. Note that this applies to land that is not currently in forest, not to forest land that is regenerated as part of forest management. Land that had not been in forest, such as grasslands, can be converted to forests through establishment. In either case, generally speaking, the stock of carbon in biomass and soil organic matter will increase over time as a result of this type of land-use change. Biomass increases predictably as trees and other vegetation are established on the site. Soil organic matter also changes, but in less predictable ways. For instance, the establishment of a forest plantation on grassland in cool temperate regions may result in a temporary loss of carbon in soil organic matter before it builds up again after the plantation is fully established. For both accounting and planning purposes, these changes in stocks of carbon must be estimated and accounted for when assessing the effects of land-use change.

Current international definitions are presented below and draw a distinction between lands that have never been under forest cover and those which were in forest cover in the past but have not been forested recently (e.g., for the last 50 years). These definitions are presented here because they are commonly used in the literature; however, in terms of carbon accounting for live biomass, there is no practical difference between the two categories. The greatest impact is on the soil carbon pool. Where the aim is to estimate entity-level GHG fluxes, these two categories will be treated together and termed “establishing forest” in this guidance.

6.3.1.1 Establishing Forest

Establishment is the conversion of a non-forest site that is not naturally a forested or treed ecosystem or had never been in forest to a forest or similar tree-dominated land cover. Examples of establishment include the conversion of bare land to a forest and conversion of grasslands to forests or plantation. In practical terms, and for the sake of this guidance, land that had been in agriculture or other non-forest land cover for a long time (e.g., more than 50 years) that is converted to tree cover can also be viewed as establishment. Hence, established forest land is that which has not been dominated by trees for more than 50 years.

6.3.1.2 Re-establishing Forest

Re-establishment is the reversion of forests or tree cover on sites that had formerly and recently been (e.g., less than 50 years) in forest or dominated by tree cover. Examples of re-establishment include natural regeneration of a disturbed or cleared parcel of forest to a secondary forest, conversion of agricultural land to a forest, and establishment of a plantation on a site that had once been forest but is now in another land use (such as cropland). It is important to distinguish between re-establishment as a land-use change and forest regrowth as part of forest management or the result of a natural disturbance. For example, a land-use change from agriculture to forest is considered here as re-establishment, where forest regeneration following a wind throw or clear-cutting is not considered a land-use change resulting in re-establishment.

In the international conventions, the IPCC Special Report on Land Use, Land-Use Change, and Forestry (IPCC, 2000), which was developed explicitly for carbon inventory, defines re-establishment as "the establishment of trees on land that has been cleared of forest within the relatively recent past; the planting of forests on lands which have, historically, previously contained forests but which have been converted to some other use." Establishment and re-establishment both refer to establishment of trees on non-treed land. Re-establishment refers to creation of forest on land that had recent tree cover, whereas establishment refers to land that has been without forest for much longer. A variety of definitions differentiate between these two processes. Some definitions of establishment are based on phrases such as "has not supported forest in historical time;" others refer to a specific period of years, and some make reference to other processes, such as "under current climate conditions." The IPCC Guidelines define establishment as the "planting of new forests on lands which, historically, have not contained forests" (IPCC, 2000).

As noted above, for the practical purposes of reporting under these methods, a change from non-forest to forest cover will be termed establishing forest, and the 50 year time horizon will not apply.

6.3.1.3 Clearing Forest

Clearing is the conversion of a forest or tree-dominated site to another land use other than forest or a tree-dominated site. Often clearing results in the complete removal of aboveground live biomass. Examples of clearing include the conversion of a forest woodlot to cropland or pasture, conversion of a forest woodlot to commercial or residential use, and conversion of a natural forest to agriculture.

6.3.1.4 Other Important Considerations

Distinction between Land-Use Change and Land-Cover Change. It is very important to understand and delineate the difference between land-*cover* change and land-*use* change. Because the terms "land use" and "land management" are often confused or used interchangeably the distinction is defined here. A basic definition of land cover is "the observed physical and biological cover of the Earth's land as vegetation or human-made features." A basic definition of land use is "the total of arrangements, activities, and inputs undertaken in a certain land-cover type (a set of human actions). The social and economic purposes for which land is managed (e.g., grazing, timber extraction, conservation)." The conventions found in the literature—Turner et al. (1994), Skole (1994), and Lambin et al. (2006)—are followed and were adopted by the IPCC in 2000. It is recognized that in adoption of the terminology of land use, land-use change, and forestry, the IPCC Good Practice Guidance document (IPCC, 2006) generalized the use of terms to include the six broad land-use categories defined in IPCC (2003) Chapter 2 and recognized that these land-use categories are a mixture of land cover (e.g., forest, grassland, wetlands) and land use (e.g., cropland, settlements) classes. For convenience, they are here referred to as *land-use* categories.

We recognize here that the term land-use change can be adopted to include land-cover changes, as well as land-use changes. Thus, for this guidance, as with IPCC, land-use change will be the conversion of the "type of vegetation" from one cover type, such as a forest dominated by trees, to a completely different cover type, such as cropland dominated by non-woody food crops. The direction of cover change determines the nature of the change in carbon stocks (e.g., forest clearing versus establishment). Generally speaking, land-use change is the most important consideration for a landowner, since this process usually results in the largest change in onsite carbon.

However, we also recognize that landowners will have important changes to their lands through the management activities that they deploy, and these activities can have important implications for carbon stocks and GHG emissions and removals. Thus, we also recognize the concept and terminology of land-management change, which is a change in the type of activity being carried out on a unit of land, and thus how it is managed or used, such as changing the management practices within a forest from selective harvest to protection. Land-management change may or may not have a significant impact on carbon and other GHGs.

Land management explicitly refers to how the land is being managed or used, while land use refers to what is on the land. An example of land management is a tree-dominated site that is used as a working forest or woodlot. As such, a landowner can change the management plan for the site—for instance, changing its use to a forest reserve —without radically changing its cover. Nonetheless, even such change in use can affect the amount of carbon stored on the site and in the soils. Typically, when a forest stand land management is changed without affecting its cover type it is considered a managed forest, and its accounting protocols follow those for forest management rather than for establishing forests. Thus it is important to determine and document both the land-use and land-management changes that occur on the site, and explicitly associate the carbon estimation approach to either establishing/clearing forests (Section 6.3) or forest management (Section 6.4), but not both.

Establishing and Clearing Forest versus Forest Management. For reasons of order and consistency, establishing and clearing forest is distinguished from management, which is addressed in Section 6.4. Forestry operations such as thinning, artificial regeneration, and harvesting are associated with managed forest systems. Unless forestry activities lead to a change from one land use to another land-use, these activities are not treated using establishing and clearing forest accounting principles. The initial conversion from forest to agriculture, for example, would use the establishing and clearing forest rules, followed by the application of rules for agriculture. Similarly, when a non-forest land cover is converted to a managed forest the initial conversion would be treated as establishing forest and use these methods, but subsequent management of the stand would follow forest management (e.g., forest carbon accounting and forest management) methods.

Types of Forest. From a strict carbon accounting point of view, the land-cover designation does not matter, nor does its change in cover type as long as one has good estimates of carbon stocks, and can measure or estimate their changes. However, data used to estimate changes in carbon are often reported and organized by forest type, so the composition and structure of the forest often comes into the computation methods. Moreover, to avoid double counting, it is important to define what type of landscapes can be considered as a forest for establishing and clearing forest. There are two elements of a definition of forests that are warranted. The first is a basic definition of a forest. There are a range of conditions of treed landscapes where establishing and clearing forest activities can take place, from preserved forests to woodlots to open and widely spaced tree landscapes and urban treed landscapes. There are hundreds of variations of definitions of forest (Lund, 1999) and for each of these there are subtypes. Examining the implications of each variant would not be fruitful; the result would be greater confusion, rather than the clarity sought. In a strict sense, a forest is defined here using the U.S.-specific definition of forest land (Smith et al., 2009). These are lands with tree crown cover (or equivalent stocking level) of more than 10 percent, width of at least 120 feet (36.6 m), and area of 1 acre (0.4 ha). Trees should be able to reach a minimum height of 6.6–16.4 feet (2–5 m) at maturity *in situ*. A forest-land unit may consist of closed forest formations where trees of various stories and undergrowth cover a high proportion of ground, or open forest formations with a continuous vegetation cover in which tree crown cover exceeds 10 percent.

Second, landowners may have a diverse land base that is affected by different forestry activities, managed at different intensities, or that has a variety of existing data. One of the first steps in preparing entity-wide or sub-entity estimates of carbon fluxes from forests is to organize the underlying data on land conditions into manageable units, referred to here as forest *strata*. Land should be grouped into forest strata using a logical framework that aggregates similar land units. For example, land could be partitioned by average tree age, forest type, productivity class, and management intensity. In many cases forest strata will be contiguous, although this is not a necessary condition. The landowner can select the type of stratification scheme to employ; and there are several guides available to do this. The better the stratification, the more accurate and precise are the carbon estimations with the minimal amount of data collection.

The definition of a forest is useful for consistency in reporting and covers a wide range of conditions. However, note that the technical methods can apply to any treed landscape. The adoption of the international nomenclature for forests allows the consideration of a range of site conditions and situations. Forests in the United States are varied, from scrub woodlands in semi-arid zones to mature deciduous and coniferous complexes in the humid zones. In addition, human managed systems, such as woodlots and plantations, are considered as forests.

Similar Modalities and Variants of Establishing, Re-establishing, and Clearing Forest. This section recognizes that establishing and clearing forest are similar to and indeed conceptually related to several other land-cover change modalities, which are treated in other protocols. These include but are not limited to agro-forestry, which involves the use of trees on farms; urban forests and widely spaced tree complexes; trees on landscapes outside of forests; woodlands and savanna systems; orchards; and palm and horticulture complexes. Although the measurement and estimation methods described here may be easily adapted to these land covers and land uses, they are not treated in this section.

6.3.2 Activity Data Collection

Activity data are measurements or estimations of magnitude of human activity resulting in emissions or removals taking place during a given period of time. Most often the area of land that is converted from one land use to another is the most important type of activity data. Data on area burned, management practices, and lime and fertilizer use are other examples of activity data. For establishing and clearing forest, activity data consists mostly of information, preferably in map form with delineated boundaries. For small landowners, it is possible to delineate an area of land-cover change by foot using simple distance measurements or with the aid of a GPS. A landowner may have different activities occurring on a single property, and thus each of the forest strata should be mapped and have separately delineated activities. Remote sensing or aerial photography can be useful for any landowner with access to these data, but are especially useful for larger land units. Historical information on changes in the areas of land uses on a property is also important, and these data are frequently found in air photo archives or other map records. In addition to the areas and rates of clearing and/or establishment, it is necessary to collect data on specific aspects and details of these activities. This may include data on tree types, biomass, clearing intensity, wood removals, tree planting densities, and other factors that described the modality of the establishing and clearing forest activities.

6.3.2.1 Establishing Forest

For an establishment activity, it is important to gather basic information on the area and location of each stratum of land use that is being established. For the most part an establishment activity will be a plantation or similar type of establishment/forestation activity. Thus, basic information on site preparation, species selection, and densities of plantings can be used with a projection of the long-term plan for the site to make a reasonable *ex-ante* calculation. If natural regeneration is the primary means of establishment, estimates of seedling counts can be used to develop a growth projection. Alternatively, regional yield tables may be used to estimate projected stocks. The prior use and management of the stratum or land use should also be documented, since the historical use of the land influences carbon stock and stock change estimates. For instance establishment of a forest stand on grassland will have a different result in terms of carbon than establishment on a row crop agricultural field. Once a forest is well established, for all practical purposes it becomes a managed forest and should be treated using the methods in the next section on forest management. We consider the land-use stratum to be a forest when the characteristics of the stand meet the definition of a forest. Most often this will be when the site is well stocked to the definitional crown cover and height of trees.

6.3.2.2 Clearing Forest

The most important activity data to collect are the area and rates of forest clearing for each stratum or parcel in the project area. It is also important to know the intensity of clearing and if there are remaining trees or other vegetation left on site after clearing. To estimate emissions, it is necessary to know also the characteristics of the stratum that is to be cleared, including the biomass and soil organic matter of the site. The process of clearing a site is an activity that can also be characterized. Information needed includes the fraction of the aboveground biomass that would be burned, the fraction that is left behind onsite as slash and debris, the fraction that would be removed in the form of wood products, and the fraction that is removed in the form of other products.

6.3.3 Estimation Methods

This section lays out the minimum necessary parts of a computation scheme for estimating carbon stocks and carbon emissions in biomass and soil associated with establishing and clearing forest. The descriptions laid out here are generalized. The basic concept behind them is simple: the stock, or mass, of carbon on a site changes, and the task of estimation is to compute the difference in stocks between the land use before and after the intervention or disturbance. When a site is cleared, stocks go down and this results in emissions to the atmosphere. When a site is established, stocks go up and this results in removals from the atmosphere.

6.3.3.1 Units of Measurement

All stock computations are performed in terms of mass of carbon in kilograms or metric tons per unit area in metric system units (carbon per hectare or C ha^{-1}). Rate data are reported in terms of change in carbon per ha over time, as in carbon per hectare per year (C ha^{-1} $year^{-1}$). All carbon biomass is referenced to its dry weight basis and the fraction of biomass in carbon. For the purpose of this guidance, the fraction of dry biomass that is carbon is 0.5. An example stock is 100 metric tons C ha^{-1}, and an example stock change is 100 metric tons C ha^{-1} $year^{-1}$. It is important to differentiate between units of carbon and CO_2 equivalents (CO_2-eq) and report the appropriate units to the reporting entity. For example, some reporting programs (e.g., carbon markets) require the conversion of metric tons of carbon to metric tons CO_2-eq. This convention places all carbon mass estimates into units of CO_2, which can be derived by multiplying the carbon mass by 44/12.

6.3.3.2 Stocks and Fluxes

The stock of carbon is the amount of carbon in biomass and soil on a site. The stock change is the difference in the stocks from one time period to the next. This change can be positive or negative, depending on whether the site is experiencing clearing, degradation, restoration, or establishment. Declining stocks over time from clearing or degradation result in emissions, while accumulating stocks over time from establishment or restoration are referred to as sequestration.

6.3.3.3 Delineating and Characterizing the Site Used in Computation

To estimate carbon stocks and fluxes, it is necessary to define the mapped extent and the features of the site. For small areas, such as a farm woodlot or forest stand, the boundaries are defined geographically using a GPS device. If surveyors' reports or other forms of maps and photos such as aerial imagery are available, they can be used. There are a growing number of online tools that are available (e.g., Google Maps) that provide detailed imagery of land that can be used to draw boundaries of the proposed sites. After defining the precise boundaries, a land-cover classification should be performed to define the various vegetation, cover, or soil strata within the site. For instance, a re-establishment project with two zones within the boundaries, one for a commercial plantation and the other for natural regeneration, would be stratified into two stands. If the project or property is to be a single cover, such as a natural regeneration forest or a plantation forest, the project site can be a single stratum; but other factors may be important, such as land slope or soil conditions. If there will be a future management activity associated with the project, this stratum should also be delineated. In short, any area within the project boundary that would have different cover or carbon characteristics should be separately delineated. Standard mapping coordinates, projections, and geodetic datums should be used.

6.3.3.4 Carbon Pools under Consideration

Generally, IPCC and other sources reference five pools of carbon to measure—aboveground live biomass, belowground live biomass, standing dead and downed debris, litter, and soil organic carbon. The landowner or project developer should identify from the beginning the pools that will be accounted. All pools should be included, unless one can show that a pool's stock changes are small and unimportant—the *de minimis* assumption (less than 10 percent of the total baseline stock, see more below)—or can show that a pool would not have stock losses or emissions (e.g., forest clearing). In these cases, the landowner is choosing to be conservative in estimation of the impact of the establishing and clearing forest on the atmosphere for that pool. For instance, in an establishment project where the estimation of soil carbon change may be difficult, time consuming, or costly, and the soil carbon change is assumed to be *de minimis* in magnitude, it may be eliminated. Or, if it can be demonstrated that the soil pool will be accumulating carbon, the landowner may select to not count that pool and thus be conservative in the sequestration potential of the project. Wood products that are removed from the site through harvest are not by themselves considered a separate pool, but the landowner is advised to document this amount and its fate, whereby fate can be, for example hardwood products, paper products, or firewood (see Section 6.5).

6.3.3.5 Initial Carbon Stock Measurement

The carbon stocks in the measured pools that are to be reported need to be determined at the beginning of the project in order to define a reference carbon amount to which future changes will be compared. Whether the site is a forest before its conversion or agricultural land before re-establishment of tree cover, the initial conditions in terms of carbon must be reported. The initial carbon stocks in all strata are individually determined from lookup tables, satellite imagery, or FIA database, or are measured and reported according to the detailed measurement methods given below. The reporting of the baseline can get complicated in some cases. Typically the baseline is the current carbon stocks. However, in situations where the carbon stocks are changing, the baseline is computed over time as the forward looking carbon stocks that would occur in the absence of the project or intervention.

6.3.3.6 The Ex-Ante Computation

Once initial carbon stocks are determined (the Type I estimate), the project developer needs to make a forward projection of the expected carbon stock changes, and its deviation from what would have occurred on the site without the intervention of a project or land-cover change (Type II and III estimates). This is somewhat problematic since it is not possible to predict the future with certainty. However, a number of tools and methods are available to make these projections with reasonable certainty (see Table 6-3). An important reason for making this computation is that the carbon stock would change over time in the absence of the project's intervention. For example, an abandoned farm field could be expected to naturally go through old-field succession even without a reestablishment project. Hence, the project-related carbon changes need to be compared with the no intervention/no action estimate over time, not just from the start of the project, to get a true accounting of net carbon benefits. Landowners would want to make the *ex-ante* computation so that they can evaluate a range of future establishment, clearing, or management options to select the one that best suits their carbon and other outcome needs.

6.3.3.7 Measurement and Monitoring

After the initiation of the project intervention (e.g., tree planting), ongoing measurements of actual carbon stock changes need to occur. This is often referred to as the monitoring phase of the project. Methods for ongoing measurement are described below. The project developer should keep organized records of the measurements made over a routine and standard time frame. Annual measurements are usually either not logistically possible or too time-consuming and expensive.

Thus, it is recommended that after the initial measurement, these measurements are repeated every 5 years.

6.3.3.8 Permanent Sample Plots

For small projects such as farm woodlots, or tree and forest stands, a complete inventory of carbon in the reporting pools, strata, and project land can be performed. However, for large areas, installing and delineating a number of sample plots is required. These sample plots are established in the project area on a stratified basis, laid out randomly or systematically—i.e., each land cover stratum has an established number of systematically or randomly placed plots. Methods for forest inventory are well described and available from a variety of sources and will not be further described here (e.g., Pearson et al., 2007). Both the number and location of the plots need to be considered. It is important to remember that the plots are established for the purpose of sampling a forest stand or project stratum. The sample estimate will be as accurate as the number and location of the sample plots permit. The number of plots will relate to the accuracy of the estimates; in simple strata such as plantations, the number of sample plots can be extremely low, but in complex natural stands the number will have to be greater. A good stratification will reduce the necessary number of plots. The location of the plots is important to capture the spatial heterogeneity of the stand. The plots are to be well marked and made permanent for repeat measurements over many years. For forest clearing computations, it is not necessary to make permanent plots unless the process of clearing is selective degradation over a long period of time. For forest clearing, lots only need to be measured once before the intervention and once after the intervention has been completed.

6.3.3.9 Measurement versus Estimation

In some cases, it will not be possible to measure the initial carbon stocks or post-intervention carbon directly. For instance, a forest clearing event may occur without the opportunity to establish plots in the forest, or it may not be possible to measure a large-area establishment event. In these cases, regional summary values of the forest carbon stocks may be of use (Smith et al., 2006).

6.3.3.10 Allometry, Biomass Expansion Factors, and Standard Values

The conventional approach to biomass estimation is to use allometric equations based on species-specific information (Jenkins et al., 2003b; 2003a). An allometric approach can be based on DBH or a combination of DBH, canopy height (H), and wood density on an individual tree basis for the entire stand or for trees in the permanent plots. The allometric equation predicts either volume of wood in the main stem or whole tree biomass or carbon. In the former case, it is then necessary to estimate a whole tree biomass expansion factor (Smith et al., 2003). Alternatively, the entity can use standard values for stocks and growth rates based on lookup tables (DOE, 1992; Smith et al., 2006). For large areas of forests converted through clearing, it may be acceptable to use standard values for stocks per unit area, such as those published by IPCC (2003; 2006).

6.3.3.11 Stocks versus Change in Stocks over Time

For estimation of forest establishment it is necessary to compute the change in stocks over time, which will be a measurement of net sinks of carbon through sequestration. Forest clearing computation is essentially the same but with the opposite sign to indicate emissions. The subtle difference is that establishment requires some means to estimate the accumulation of carbon on the project site over time. This is accomplished using either direct measures or yield models. For forest clearing, it is necessary to know the initial stock of carbon in the forest stand, and how it then changes with disturbance. The latter requires data on the partitioning of post-disturbance carbon components, as removals, and slash and debris left on site.

6.3.3.12 Forest Clearing Removals and Dead Material on Site

The difference of carbon stocks before and after forest clearing is the carbon that has been removed by harvest as wood products or other products (e.g., energy feedstocks), and that left behind on the site as slash and debris (Skog, 2008). If these mass amounts are known, they can be included directly into the computations. If they are not known, they can be estimated and represented as fractions of the original standing stocks prior to disturbance. All removals such as these constitute immediate and future emission sources, as they decay over different time scales. Therefore, it is necessary to assign mass amounts to four long-term decay pools with turnover times of 1, 10, 100, and 1,000 years. The emissions are computed along an exponential decay function related to the turnover time of the pool. For example, carbon lost due to immediate oxidation by fire is placed into the 1-year pool, and the charcoal component is placed into the 1,000-year pool. Other removals are placed into the 10- and 100-year pools.

6.3.4 Specific Protocol for Computation

6.3.4.1 Actual Carbon Removals by Sinks in Establishing Forest

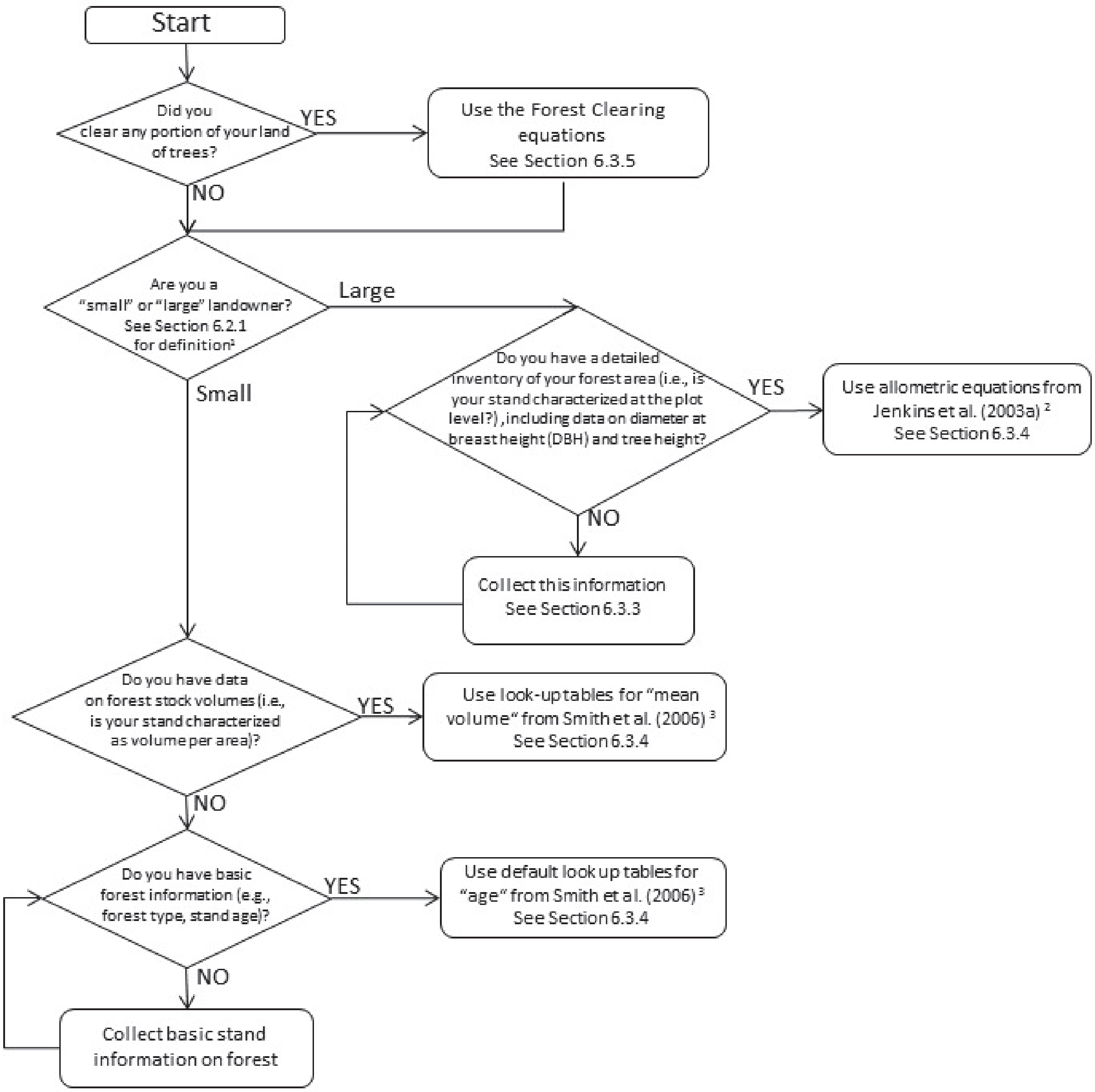
The basic approach to estimation of emissions to, or removals from, the atmosphere is to multiply the activity data by emission factors or, in this case, multiply the land-use change area by site biomass carbon and soil organic matter carbon. These procedures describe the recommended method of estimating carbon—using allometric equations to estimate biomass directly from DBH using the equations of Jenkins et al. (2003a).

Stratification of the project area may be carried out to improve the accuracy and the precision of the carbon estimates. Where required, stratification could be made according to tree species, age classes, or forest management practices. Figure 6-5 shows a decision tree indicating which method is more applicable for a particular landowner.

This protocol will follow the two-tier approach described earlier in the document. Small landowners can use default tables (i.e., Smith et al., 2006) and equations for the appropriate region and forest type group to estimate biomass of their forest systems. Large landowners should use basic forest data collected in the field on sample plots with allometric equations (Jenkins et al., 2003a) to estimate the biomass of individual trees and entire stands. If small landowners want to use sample plots and allometric equations, they are free to do so. Small landowners should contact a consulting forester or perhaps a university extension person to best understand requirements for field sampling.

While most of the fluxes from an establishment project are removals from the atmosphere, there may be some emissions associated with some aspects of the project. The actual net CO_2 removals by sinks can be estimated using the equations in this section. When applying these equations for *ex-ante* calculations of net anthropogenic CO_2 removals by sinks, landowners will provide estimates of the values of those parameters that are not available before the start of the project period and commencement of the monitoring activities. Participants should retain a conservative approach in applying these estimates.

Figure 6-5: Decision Tree for Establishing, Re-establishing, and Clearing Forests Showing Methods Appropriate for Estimating Forest Carbon Stocks



[1] Small landowners (see Section 6.2 for definition) may use generalized lookup tables based on region, forest type, and age class to estimate carbon stocks. Large landowners (see Section 6.2 for definition) should collect standard forest inventory data and use allometric equations to estimate live tree biomass carbon (other carbon pools may be obtained from lookup tables). However, large landowners who do not engage in any management activities or plan to manage their holdings may use lookup tables for all pools; but if active management occurs, the inventory approach should be used.

[2] Jenkins et al. (2003a).

[3] Smith et al. (2006).

The actual net CO_2 removals by sinks in year t are equal to:

Equation 6-1: The Actual Net CO_2 Removals by Sinks in Year t

$$\Delta C_{ACTUAL,t} = \Delta C_{PJ,t}$$

Where:

$\Delta C_{ACTUAL,t}$	= Actual net CO_2 removals by sinks in year t (metric tons CO_2 eq year^{-1})
$\Delta C_{PJ,t}$	= Project CO_2 removals by sinks in year t (metric tons CO_2 eq year^{-1})

Equation 6-2: Project CO_2 Removals by Sinks are Calculated as Follows (between two dates for a time period of t)

$$\Delta C_{PJ,t} = \sum_{i=1}^{t} \Delta C_{project,i,t} \times 44/12$$

$$\Delta C_{project,i,t} = [(C_{trees,i,t2} - C_{trees,i,t1}) / T] + \Delta C_{soil,i,t}$$

Where:

$\Delta C_{PJ,t}$	= Project CO_2 removals by sinks in year t (metric tons CO_2 eq year^{-1})
$\Delta C_{project,i,t}$	= Average CO_2 removals by living biomass of trees and soil for stratum i, for year t (metric tons carbon year^{-1})
$C_{trees,i,t}$	= Carbon stock in living biomass of trees for stratum i, in year t (metric tons carbon)
$\Delta C_{soil,t}$	= Average annual change in carbon stock in soil organic matter for stratum i, for year t (metric tons carbon year^{-1})
T	= Number of years between years t_2 and t_1 (years)

Estimation of Carbon Stock in Living Biomass of Trees at the Stratum Level. The carbon stock in living biomass of trees for stratum i ($C_{trees,i,t}$) is estimated using the following approach: The mean carbon stock in aboveground biomass per unit area is estimated based on field measurements in permanent sample plots.

Step 1: Determine based on measurements (*ex post*), the DBH at typically 4.3 feet (1.3 m) above ground level, and also preferably height (H), of all the trees above some minimum DBH in the permanent sample plots.

Step 2: Calculate the aboveground biomass for each individual tree of a species, using allometric equations appropriate to the tree species (or groups of them if several tree species have similar growth habits) in the stratum.

Step 3: Estimate carbon stock in aboveground biomass for each individual tree l of species j in the sample plot located in stratum i using the selected or developed allometric equation applied to the tree dimensions resulting from **Step 1**, or multiply the result of **Step 2** by 0.5 (i.e., the fraction of dry biomass to carbon conversion factor), and sum the carbon stocks in the sample plot.

Step 4: Convert the carbon stock in aboveground biomass to the carbon stock in belowground biomass using the equations provided in Jenkins et al. (2003a) or by multiplying the result of **Step 3** by 0.26 (i.e., the root-to-shoot ratio). Sum the aboveground carbon stock and belowground carbon stocks.

Step 5: Calculate total carbon stock in the living biomass of all trees present in the sample plot *sp* in stratum *i* at time *t*.

Step 6: Calculate the mean carbon stock in living biomass of trees for each stratum, as per Equation 6-6.

Equation 6-3: Estimate Carbon Stock in Aboveground Biomass for Each Individual Tree

$$C_{AB,\,i,\,sp,\,j,\,t} = \sum_{t=1}^{N\,j,\,sp} CF_j \times f_j\,(DBH, H)$$

Where:

$C_{AB,i,\,sp,\,j,\,t}$ = Carbon stock in aboveground biomass of trees of species *j*, on sample plot *sp*, for stratum *i* (metric tons carbon)

CF_j = Carbon fraction of dry matter (dm) for species or group of species type *j* (metric tons carbon (metric ton dm)$^{-1}$)

$f\,j\,(DBH,H)$ = An allometric equation linking aboveground biomass of a living tree (metric tons dm) to DBH and possibly tree height (H) for species *j*, in year *t* (metric tons dm)

Note: For *ex-ante* estimations, mean DBH and H values should be estimated for stratum *i,* in year *t* using a growth model or yield table that gives the expected tree dimensions as a function of tree age*.* The allometric relationship between aboveground biomass and DBH and possibly H is a function of the species considered. Alternatively there are estimators and tools that project carbon growth rates directly without input of DBH.

i = 1, 2, 3, … M_{PS} strata in the project scenario

j = 1, 2, 3, … S_{PS} tree species in the project scenario

l = 1, 2, 3, … $N_{j,sp}$ sequence number of individual trees of species *j*, in sample plot *sp*

t = 1, 2, 3, … *t** years elapsed since the start of the project activity

Equation 6-4: Convert the Carbon Stock in Aboveground Biomass to the Carbon Stock in Belowground Biomass

$$C_{BB,\,i,\,sp,\,j,\,t} = C_{AB,\,i,\,sp,\,j,\,t} \times R_j$$

Where:

$C_{BB,\,i,\,sp,\,j,\,t}$	= Carbon stock in belowground biomass (BB) of trees of species *j*, in plot *sp*, in stratum *i*, for year *t* (metric tons carbon)
$C_{AB,\,i,\,sp,\,j,\,t}$	= Carbon stock in aboveground biomass (AB) of trees of species *j*, in plot *sp*, in stratum *i*, for year *t* (metric tons carbon)
R_j	= Root:shoot ratio appropriate for biomass stock, for species *j* (dimensionless)

Equation 6-5: Calculate Total Carbon Stock in the Living Biomass of All Trees Present in the Sample Plot

$$C_{tree,\,i,\,sp,\,t} = \sum_{j=1}^{Sps} \left(C_{AB,i,sp,j,t} + C_{BB,i,sp,j,t}\right)$$

Where:

$C_{tree,\,i,\,sp,\,t}$	= Carbon stock in living biomass of trees on plot *sp* of stratum *i*, for year *t* (metric tons carbon)
$C_{AB,\,i,\,sp,\,j,\,t}$	= Carbon stock in aboveground biomass (AB) of trees of species *j*, in plot *sp*, in stratum *i*, for year *t* (metric tons carbon tree^{-1})
$C_{BB,\,i,\,sp,\,j,\,t}$	= Carbon stock in belowground biomass (BB) of trees of species *j*, in plot *sp*, in stratum *i*, for year *t* (metric tons carbon tree^{-1})
i	= 1, 2, 3, … M_{PS} strata in the project scenario (PS)
j	= 1, 2, 3, … S_{PS} tree species in the project scenario (PS)
t	= 1, 2, 3, … *t** years elapsed since the start of the project activity

Equation 6-6: Calculate Mean Carbon Stock in Tree Biomass for Each Stratum

$$C_{tree,\,i,\,t} = (A_i / Asp_i) \sum_{sp=1}^{Pi} C_{tree,\,i,\,sp,\,t}$$

Where:

$C_{tree,i,t}$	= Carbon stock in living biomass of trees in stratum *i*, for year *t* (metric tons carbon)
$C_{tree,\,i,\,sp,\,t}$	= Carbon stock in living biomass of trees on plot *sp*, of stratum *i*, for year *t* (metric tons carbon)
Asp_i	= Total area of all sample plots in stratum *i* (ha)
A_i	= Area of stratum *i* (ha)
sp = 1, 2, 3, …	= Pi sample plots in stratum i in the project scenario
i = 1, 2, 3, …	= M_{PS} strata in the project scenario (PS)
t = 1, 2, 3, …	= t^* years elapsed since the start of the project activity

Soil Organic Carbon. For strata that contain only mineral soils, *ex-ante* and *ex-post* $\Delta C_{soil,\,i,\,t}$ change is estimated from Equation 6-7.

Equation 6-7: Estimating Change in Carbon Stocks for Strata That Contain Only Mineral Soils

$$\Delta C_{soil,\,i,\,t} = A_i * \Delta C_{forest,\,i} \text{ for } t \leq t_{equilibrium,\,i}$$

$$\Delta C_{soil,\,i,\,t} = 0 \text{ for } t > t_{equilibrium,\,i}$$

Where:

$\Delta C_{soil,\,i,\,t}$	= Average annual change in carbon stock in soil organic matter for stratum *i*, for year *t* (metric tons C year^{-1})
A_i	= Area of stratum *i*; hectare (ha)
$\Delta C_{forest,\,i}$	= Average annual increase in carbon stock in soil organic carbon pool for forest system in stratum *i* (metric tons C ha^{-1} year^{-1})
$t_{equilibrium,i}$	= Time from start of the project activity until a new equilibrium in carbon stock in soil organic matter is reached for forest system in stratum *i* (years)

The default value of $\Delta C_{forest\ i}$ = 0.5 metric tons C ha^{-1} year^{-1}, and a $t_{equilibrium}$ of 20 years,$_i$ shall be used.

Changes in carbon stock in soil organic matter are not monitored *ex-post* (i.e., measured before and after the equilibrium period), but are instead estimated *ex-ante* (i.e., predicted based on the specified default value and equilibrium period).

Other Pools. Sample plots need to be set up in such a ways that the small herbs and bushes, as well as forest floor litter is also measured. To do this, establish several small collection plots measuring 3.3 feet by 3.3 feet (1 m by 1 m) on the forest floor. Collect all liter, herbs, and small debris in the subplot and weigh it using a field scale, and dry small sample to get the dry weight fraction.

Multiply the average dry weight of litter by 0.37 to compute the plot litter carbon, and by 0.5 to compute the plot herbs and seedling carbon. For small trees and bushes establish a few small plots measuring 16.4 feet by 16.4 feet (5 m by 5 m) in the sample plot. Cut and weigh all small trees and bushes. Establish a dry weight basis and multiply the dry weight by 0.5 to compute a subsample carbon value. Standing dead wood also needs to be estimated. Most published studies suggest this pool is small and can be ignored.

Non-CO_2 GHGs. Non-CO_2 GHGs, including CH_4 and N_2O are calculated based on emission factors applied to the parcel biomass. Thus, the parcel biomass is multiplied by a factor from default values for that time of stand or planting activity. These emissions and removals will vary depending on the management practice, e.g., natural succession, plantations, fertilization.

6.3.5 Actual GHG Removals and Emissions by Sources and Sinks from Forest Clearing

The above suite of equations can be used to estimate the sources and sinks of carbon from forest clearing, with the results having a different sign than establishment and re-establishment. The fundamental computation is in Equation 6-8.

Equation 6-8: Computing Emissions of Carbon from a Forest Clearing

$$E_d = f(D \times C/ha)$$

Where:

E_d	= Emissions of carbon from forest clearing, D (metric tons carbon year^{-1})
D	= The rate of forest clearing (ha year^{-1})
C/ha	= The stock of carbon in the forest system prior to clearing (metric tons carbon ha^{-1})

The precise computation in Equation 6-9 requires the measurement or estimation of the differences in carbon stocks in the forest system and the land-cover system that it is converted to. It also requires an understanding a computation of the partitioning of the products that were removed from the site or left as slash and debris. For material left onsite and burned, GHG emissions should be calculated using the CONSUME model. Hence, C_f is estimated from standard per-area forest type carbon stocks or from plot data. The fractions *fy* and *dy* are estimated or directly measured (for simplicity it is possible to assume that *dy* is the fraction of the turnover time, as in 1/1, 1/10, 1/100 or 1/1,000). E_s is the soil flux that is represented in lookup tables, and based on the time-varying rate of carbon loss as a percentage of the original forest soil carbon.

Equation 6-9: Computing the Partitioning of the Products That Were Removed from the Site or Left as Slash or Debris in 1 Year

$$E_d = D\,[\,(C_f - C_c) \times \sum_{i=1}^{y} (fy \times dy)\,] + E_s$$

Where:

E_d = Emissions of carbon from forest clearing, D (metric tons carbon year^{-1})

D = The rate of forest clearing (ha year^{-1})

C_f = The carbon stock prior to forest clearing (metric tons carbon ha^{-1})

C_c = The carbon stock after forest clearing (metric tons carbon ha^{-1})

fy = The fraction of original carbon stock in long-term decay pool *y*

dy = The decay function for the mass quantities in decay pool y (long-term decay pools are 1-, 10-, 100- and 1,000-year turnover times)

Es = Emissions from soil (metric tons carbon year^{-1})

6.3.6 Limitations and Uncertainty

There are published methods for formally estimating uncertainty of the estimation, generally based on the number and distribution of the permanent plots, and how they are applied to the whole stratum. These uncertainty estimates can be used *a priori* to establish the number of plots needed to achieve a level of accuracy. They can also be used to attach an uncertainty value to the final estimate. But perhaps the most challenging component of uncertainty lies in the use of various expansion factors where precise field estimates are not known. In particular, the estimation of non-CO_2 GHG fluxes is very uncertain, and must be used with some degree of caution. This is especially true for N_2O in all activities and CH_4 in cases of forest establishment. Considerably more research is necessary to make these estimates.

Another uncertainty in most estimates is the fraction of standing dead biomass. Based on some work (Woodall and Monleon, 2008), it is believed to be small, but the variation with forest types, stand age, conditions, and activities is large. When using default values this may be a challenge to the final estimation. In the case where direct measurements are to be made onsite, the standing dead can be measured along with standing live biomass. This may be an approach that has special benefit if the site being cleared has been intensely damaged by pests or disease.

Perhaps the most problematic area is the computation of whole tree biomass from allometry. There is a very good North American literature on allometry for stem volumes and biomass but less on whole tree volume and biomass. Most allometry is based on volumes rather than whole tree biomass or carbon. Frequently a limited number of simple expansion factors are deployed to expand the volume of the main stem to the biomass of the whole tree including its branches. These models need to be refined to better make the estimation. This may be important since most landowners will not have the ability or interest to conduct their own destructive tree sampling to extract local whole tree biomass allometry (i.e., a Tier 3 approach).

Table 6-3: Examples of Forest Carbon Calculators

Developer	Website
USDA Forest Service tools for carbon inventory, management, and reporting	http://www.nrs.fs.fed.us/carbon/tools/
FAO ExACT	http://www.fao.org/tc/exact/en/
TARAM (BioCF and CATIE)	http://wbcarbonfinance.org/Router.cfm?Page=DocLib&CatalogID=31252
CO_2Fix	http://www.efi.int/projects/casfor/models.htm
GORCAM	http://www.joanneum.at/gorcam.htm
CASS	http://www.steverox.info/software_downloads.htm
FullCam	http://www.ieabioenergy-task38.org/workshops/canberra01/cansession1.pdf
COLE	http://www.ncasi2.org/COLE/
Reforestation/Afforestation Project Carbon Online Estimator	http://ecoserver.env.duke.edu/RAPCOEv1/
Winrock AFOLU Calculator	http://winrock.stage.datarg.net/CarbonReporting/Welcome

6.4 Forest Management

Methods for Forest Management

- Range of options dependent on the size/management intensity/data availability of the entity's forest land including:
  - FVS-FFE with Jenkins (2003a) allometric equations;
  - Default lookup tables of management practice scenarios; and
  - FVS may be used to develop a supporting product providing default lookup tables of carbon stocks over time by region; forest type categories, including species group (e.g., hardwood, softwood, mixed); regeneration (e.g., planted, naturally regenerated); management intensity (e.g., low, moderate, high, very high); and site productivity (e.g., low, high).
- The methods were selected because they provide a consistent and comparable set of carbon stocks over time under management scenarios common to the forest types and management intensities.

6.4.1 Description

Forest management is concerned with meeting landowner objectives for a forest while satisfying biological, economic, and social constraints. Forest managers use a wide variety of silvicultural techniques to achieve management objectives, most of which will have impacts on the carbon dynamics (see Table 6-4). The primary impacts of silvicultural practices on forest carbon include enhancement of forest growth (which increases the rate of carbon sequestration) and forest harvesting practices (which transfers carbon from standing trees into wood products and residues, which eventually decay). Some forest management activities will result in accelerated loss of forest carbon, such as when soil disturbance increases the oxidation of soil organic matter, or when prescribed burning releases CO_2. Furthermore, some forest management activities result in fossil fuel emissions (e.g., from the utilization of mechanized equipment, transportation). However, recent evidence suggests these emissions are fairly minor. Markewitz (2006) estimated that fossil emissions from silvicultural activities in intensively managed pine plantations were about 3 Mg C ha^{-1} over a 25-year rotation. These emissions were very low relative to the subsequent sequestration of carbon in the forest and in wood products. Côté et al. (2002) report emissions from silvicultural activities totaled about 9 percent of total emissions from a pulp and paper operation and about 4 percent of gross forest sequestration. In a life-cycle analysis from the Pacific Northwest, Johnson et al. (2005) reported fossil emissions of CO_2 from forestry operations amounted to 8.02 to 8.12 kg CO_2-eq m^{-3} of harvested logs, or less than 1 percent of the 935 kg CO_2-eq contained in a cubic meter of a Douglas-fir log. In the dry Ponderosa pine forests of Arizona, a thinning treatment resulted in CO_2 emissions from fossil fuels of 334 kg CO_2-eq ha^{-1}, about 1.1 percent of the 30,213 kg CO_2-eq ha^{-1} of firewood removed in the thinning operation (Finkral and Evans, 2008).

This section describes general categories of forest management activities and their impacts on carbon storage. The details vary widely across the United States with different forest types, ownership objectives, and forest stand conditions. It is important to engage professional foresters when considering harvests or other silvicultural practices. An important distinction to be made at the outset is between planted forests, or plantations, and forests that have been naturally regenerated. Productivity rates, silvicultural practices, and management objectives may be markedly different for planted versus natural forests. In planted forests, conditions are typically optimized for increased growth, which increases carbon sequestration over slower growing, naturally regenerated forests. However, methods for inventorying, monitoring, and assessing carbon storage in both planted and natural forests are the same; variability may be less in single-species plantations, but approaches are identical. Small landowners will use the regional default tables to estimate the potential changes in GHG fluxes from changes in forest management, while large landowners will use standard forest inventory data in combination with the simulation feature of the FVS-FFE to assess changes in sequestration and emissions from changes in practice.

Table 6-4: Common Forest Management Practices

Practice	Description	Benefits
Stand density management	Controlling the numbers of trees per unit area in a stand through a variety of techniques, such as underplanting, precommercial thinning, and commercial thinning	▪ Maintains stand at a tree density that provides optimal growing space per tree for best utilization of site resources ▪ Allows concentration of site resources on "crop" trees
Site preparation	Preparing an area of land for forest establishment by removing debris, removing competing vegetation, and/or scarifying soil when needed	▪ Improves survival and initial growth of planted or naturally regenerated seedlings or sprouts ▪ Enhances regeneration of desired species ▪ Provides conditions favorable for planting of seedlings
Vegetation control	Removing, through chemical or mechanical means, undesirable vegetation that would compete with the desired species being regenerated	▪ Improves survival and growth of desired trees/species
Planting	Planting of seedlings by hand or machine to establish a new forest stand	▪ Controls species composition and genetics of newly established stand ▪ Controls stocking (density) of trees per unit area for optimal growth/survival

Practice	Description	Benefits
Natural regeneration	Establishing a new forest stand by allowing/enhancing natural seeding or sprouting	▪ Results in mix of species ▪ Species that sprout from stumps and roots will rapidly recapture the site ▪ Low cost relative to planting ▪ May involve less soil disturbance thereby reducing erosion
Fertilization	Augmenting site nutrients through the application of nitrogen, phosphorous, or other elements essential to tree growth	▪ Enhances growth of trees ▪ Reduces the time for trees to reach merchantable size ▪ Eliminates or reduces nutrient deficiencies that would impair forest growth/survival
Selection of rotation length	Choosing the timing of final harvest so as to optimize the mix of forest products that can be obtained from the stand	▪ Controls the relative amounts of pulpwood and sawtimber products ▪ Allows landowner to respond to wood products markets by optimizing product mix
Harvesting and utilization	Removal of trees from the forest, and cutting and separating logs for forest products markets	▪ Selection of appropriate harvesting systems can provide logs for markets while minimizing damage to residual trees or disturbance of soil ▪ Choice of harvesting and silvicultural cutting system will impact subsequent regeneration of the stand; systems can be chosen to influence the species composition of the regenerated stand
Fire and fuel load management	Reducing the risk of loss to wildfire by controlling the quantity of fuels in a forest stand by controlled fire or mechanical treatments	▪ Reduces the damage caused by severe wildfires by eliminating excessively high fuel loads ▪ May influence the species composition of the understory
Reducing risk of emissions from pests and disease	Recovering value of timber after damaging events and/or preventing further damage by interrupting spread of pests/diseases	▪ Salvage harvests recovers value in damaged timber by removing it before it is unusable ▪ Sanitation harvests prevent spread of pests/diseases
Short-rotation woody crops	Producing merchantable trees in very short time periods through intensive management (genetics, herbicide, fertilization)	▪ Reduces the time for trees to reach merchantable size

The remainder of this section describes these forest management practices and their impact on carbon stocks.

6.4.1.1 Stand Density Management

Management of forest stand density (number of trees per unit area) is important to achieve optimal growth. Overstocked stands (too many trees) or understocked stands (too few trees) will grow less fiber, and therefore store less carbon, than might be desirable. In overstocked stands, trees compete with each other for scarce resources (nutrients, water, and light), and such stands may have high numbers of trees of poor size and quality and are highly susceptible to wildfire or other reversal disturbances. Reducing the stocking in overstocked stands will concentrate growth in trees of more desirable species and quality. Understocked stands do not fully utilize the resources of the site and therefore do not achieve the growth potential of a fully stocked stand. Stand density management seeks to maintain a fully stocked stand.

Density of an existing forest stand may be increased by underplanting, which involves planting additional trees (possibly of different species) beneath an existing tree canopy. This treatment may be desirable for stands in which adequate advanced regeneration of desired species is lacking. Underplanting is designed to increase the likelihood of successful regeneration following the eventual harvest of the overstory. Thus, while the immediate carbon impact of this treatment is low, there may be substantial eventual improvement in carbon stocks compared with a stand without underplanting.

Decreasing the density of a forest stand is accomplished through thinning, or cutting some proportion of the trees in a stand. This may be done as precommercial thinning, in which case most of the trees to be cut are too small to economically justify their removal from the forest, and they are left in the stand to decay naturally. While precommercial thinning provides no immediate economic benefits, it may be used to improve the stocking level, species composition, and overall health of a stand; it represents an investment in creating a more valuable, productive forest. Precommercial thinning and stand density management also can reduce the risk of reversal from drought, insects, disease, and possibly fire. From a carbon standpoint, precommercial thinning will remove carbon from the live tree pool and increase the carbon in the dead wood pool. If the slash is burned, the GHG emissions should be accounted for using the CONSUME model when the burn occurred.

If trees to be thinned are of proper species, size, and quality, commercial thinning may be performed. In commercial thinning, trees are targeted for removal based on their species, size, and the management objectives. Thinned trees are removed from the stand and sold to appropriate forest products markets. Thus, commercial thinning will shift carbon from the live tree pool and into dead wood and litter (branches, foliage, and stumps remaining in the stand after harvest), and HWP pools.

6.4.1.2 Site Preparation Techniques

Regenerating a forest stand after harvest may require treatments to create the most desirable conditions for development of the new stand. This may involve removing debris from the prior stand, removing undesirable competing vegetation, scarifying or disturbing the soil for enhanced regeneration of species that require such conditions, and creating space or proper conditions for planting trees.

A wide variety of techniques are available to meet the specific regeneration objectives; they vary considerably across geographic regions, topography, site conditions, and forest species under management. General categories of site preparation techniques include mechanical methods, chemical applications, and prescribed fire.

Mechanical methods displace unwanted vegetation, move or break down logging residues, and/or cultivate the soil (Nyland, 2002). Mechanical site preparation uses a variety of machines and equipment, and may be limited by site factors such as terrain and soil conditions. Because mechanical site preparation involves soil disturbance, there is increased oxidation and emission of CO_2 from the soil organic matter for a period of time after site preparation.

Chemical applications involve the use of herbicides targeted at controlling undesirable vegetation so that the preferred species of trees have improved survival. Chemicals may be applied through ground or air spraying or injection into individual trees. Chemical site preparation involves little to no soil disturbance and has minimal effect on soil carbon emissions.

Prescribed burning may be used to reduce the amount of debris (limbs, tops, and foliage) from prior harvests, kill advanced regeneration of trees of undesirable species, and control pests that inhabit decaying wood left from the prior stand. Some fire-adapted species require burning to open cones and disperse seed for the new stand. Clearly, prescribed fire for site preparation will result in combustion and emission of CO_2 from woody materials left on the site, but will avoid the soil disturbance of mechanical techniques. The FOFEM model for natural fuels and the COMSUME model for activity generated fuels can be used to address this type of burning and allows estimation of GHG emissions and consumption.

6.4.1.3 Vegetation Control

Control of competing vegetation is one means of enhancing the growth of desirable trees in a forest. For example, in a pine plantation, where pine trees are the species of primary interest, growth of pines is increased when hardwood competition is removed. Vegetation control may be accomplished mechanically (such as girdling undesirable trees) or chemically. Vegetation control is especially important at two stages in the life of a stand: at establishment (planting or regeneration) and later in the rotation but before trees are past the sapling stage.

At establishment (e.g., of a plantation), the primary competition may come from herbaceous vegetation that can quickly outgrow the planted trees and suppress their growth or increase mortality. Herbicides may be effective at controlling herbaceous competition and providing the newly planted trees a chance to grow sufficiently to capture the site. Mid-rotation release of trees may require an additional application of chemical control to reduce competition and focus growth on desirable trees.

Vegetation control has been estimated to have contributed 35 percent of the substantial gain in plantation productivity relative to unimproved plantations (Stanturf et al., 2003). The primary carbon stock impact of vegetation control is a transfer of carbon stock from the live tree to standing dead biomass pool. Trees released from competition will usually exhibit a growth response to balance the loss of growth on the vegetation removed (i.e., overall forest productivity and sequestration will remain unchanged).

6.4.1.4 Planting

One popular form of regenerating a forest stand following clearcutting is to establish a plantation by planting trees of a desirable, fast-growing species, potentially utilizing an improved genetic source, at a consistent spacing selected to optimize growth. Plantation management practices include combinations of treatments to control competing vegetation and manage tree nutrition through fertilization, thinning, and use of genetically improved stock (Vance et al., 2010). Because of these efforts, plantations may be up to six times more productive than naturally regenerated stands of the same species (Carter and Foster, 2006). Successful plantation establishment entails careful selection of species, genetics, and spacing (planting density).

Species used in planted stands typically are selected for high growth rates, low susceptibility to damage from insects and disease, and quality and value. For example, in the U.S. South, loblolly pine is the most widely planted tree species because it is native to the area, fast-growing relative to other pines, and resistant to disease (Schultz, 1997). Longstanding genetic improvement programs have led to the production of improved genetic sources for forest plantation species. Genetically improved seedlings are available from commercial and state tree nurseries; essentially all of the 1.2 billion loblolly pine seedlings planted annually in the U.S. South are the result of tree improvement programs (McKeand et al., 2003). In the Pacific Northwest, genetic improvement in Douglas fir trees has led to increases in productivity (volume production) in excess of 25 percent (St. Clair et al., 2004). Finally, selection of planting density (trees per unit area) can affect overall stand productivity, necessity for thinning, ability to access the stand with equipment to conduct silvicultural operations, and time required until trees reach merchantable diameters. All of these factors combine to determine the likely survival and growth rates of a forest plantation. Plantation productivity is directly related to rate of forest sequestration. Any activity increasing productivity will improve sequestration rates.

6.4.1.5 Natural Regeneration

Certain forest types are regenerated most efficiently using natural regeneration, in which seedlings and sprouts from a recently harvested or disturbed forest will grow quickly after removal of a portion or all of the forest overstory. In this case, the species will be predictable based on the species composition of advanced regeneration from the previous stand, or if species present in the previous stand are prolific in sprouting. The species can also be predicted based on post-harvest regeneration of seedlings from residual overstory trees or from surrounding stands. Density will not be controlled during the regeneration process; frequently natural regeneration results in very dense vegetation that then goes through a natural process of competition.

Because neither the genetic source nor density are controlled during natural regeneration, these stands are frequently less productive than plantations but may be more desirable based on the objectives of the landowner (e.g., for recreation, wildlife, or different products than plantations would provide). The process of natural regeneration may entail minimal (if any) site preparation and less soil disturbance and cost than would plantations. Depending on the level of soil disturbance from the harvest of the previous stand, early soil CO_2 emissions may be lower than in planted stands.

6.4.1.6 Fertilization

Fertilization has been shown to dramatically improve the productivity of forest stands in which nutrients are limiting plant growth. For example, in the U.S. South, nitrogen and phosphorus are commonly deficient in pine plantations (Fox et al., 2007). In these areas, phosphorus fertilization may increase volume production by more than 100 percent (Jokela et al., 1991). Nitrogen and phosphorus fertilization has been shown to increase growth by 1.6 tons acre^{-1} year^{-1} (Fox et al., 2007).

The two primary types of forest fertilization currently practiced in the South are phosphorus-fertilization on deficient sites (usually at or near time of planting), and nitrogen and phosphorus fertilization in mid-rotation stands (e.g., ages 8 to 12). Volume gains vary, with highest gains where stands are most nutrient-limited.

The direct carbon impact of fertilization of forests is the observable increase in growth and therefore sequestration. Other impacts have been noted in agricultural settings, including increased emissions of other GHGs such as NO_x and N_2O. Results from agricultural fertilizer applications may not be directly applicable to forestry operations. Recent research in western Canadian forests showed soil GHG fluxes were neutral following fertilization (Basiliko et al., 2009). In an analysis of fertilization of pine plantations in the southeastern United States, Albaugh et al. (2012) found that carbon sequestration in forest growth far exceeded the emissions associated with fertilizer production, transport, and application (8.70 Tg year^{-1} CO_2 sequestration versus 0.36 Tg year^{-1} emissions). Thus, forest fertilization when applied appropriately can dramatically increase carbon sequestration when compared to unfertilized stands.

6.4.1.7 Selection of Rotation Length

One significant decision that forest managers make is the selection of the rotation length, or target age at which a regeneration harvest (final harvest; often but not necessarily a clearcut) will occur.

The decision affects the timing of other stand treatments. For example, thinnings and some fertilization treatments are targeted for a certain time before final harvest. It also affects the mix of forest products that might be expected from the harvested stand. Stands harvested at relatively young ages will yield primarily trees suitable for pulpwood markets, while longer rotations may involve more thinnings and will increase the proportion of sawtimber-sized trees in the stand. Because these different products have different longevities (see Section 6.5), the rotation length will have a significant impact on the overall carbon dynamics of a forest (and its subsequent pool of carbon in HWPs). Furthermore, longer rotations result in greater average carbon storage in the forest, with resulting higher levels of sequestration (Stainback and Alavalapati, 2002). It is widely recognized that increasing rotations from harvesting at financial maturity to harvesting closer to ages at which stands reach a steady state between growth and mortality can be beneficial for carbon storage (van Kooten et al., 1995).

A variety of decision criteria are available for identifying the optimal rotation length for different sets of objectives. If carbon storage is one of the important objectives, longer rotations will be beneficial (Liski et al., 2001).

6.4.1.8 Harvesting and Utilization Techniques

Regeneration harvests (also called rotation harvests or final harvests) are conducted to harvest trees for forest products markets and to promote the regeneration of desirable species for the next stand. To meet the twin objectives of regeneration and production of merchantable timber, forest managers may choose from a wide array of techniques and operational approaches. The silvicultural system will be chosen to determine which trees are to be removed from the stand, and a harvesting system will be chosen to determine the best logging approach to do so.

The silvicultural system determines what proportion of the forest stand is to be removed in the harvest, and will dictate whether the resulting stand will be even-aged (a stand of trees of a single age class) or uneven-aged (a stand of trees with three or more age classes) (Helms, 1998). Harvests range from clearcuts, in which most or all of the overstory is removed, to a variety of partial harvests. Partial harvests include systems such as seed-tree, shelterwood, group selection, individual tree selection, diameter-limit, and others. Harvest techniques that open most or all of the canopy (such as clearcutting or seed-tree harvests) will promote the regeneration of species that thrive in sunlight and do not tolerate shade. Clearcutting is also the preferred technique when the next stand is to be established by planting rather than natural regeneration.

After selection of a silvicultural system for regeneration, forest managers will select a harvesting system for the felling and extraction of trees from the site. Again a wide variety of systems are available, from individual tree-felling by chain saw with extraction by horse teams, to highly mechanized systems involving skidders, feller-bunchers, forwarders, and other types of equipment. When terrain conditions prevent ground-based vehicular extraction of felled trees, it may be done using cable yarding systems or helicopters. Logging systems that minimize soil disturbance and impacts on unharvested trees and understory may reduce these harvest-associated emissions.

When trees are harvested from a forest, they may produce a variety of products for specific markets. For example, large-diameter trees of certain species are preferred for sawtimber markets, while pulpwood markets accept roundwood with smaller diameters or even chips. Thus, a harvesting operation will often involve merchandising—the sorting, cutting, and separating of logs for delivery to different markets. Depending on the silvicultural system chosen, trees without market value (e.g., too small, poor form, or undesirable species) may be cut and left onsite to decay. In addition, a great deal of logging “slash” may be produced; this material may consist of branches, portions of trees beyond merchantability limits (tops), roots, and foliage. Where biomass energy markets exist, some of this material may be removed and used to replace fossil energy GHG sources;

otherwise it may be left onsite to decay or be burned during site preparation with associated GHG emissions. The proportion of woody material removed from a harvesting operation is termed utilization; high levels of utilization mean more woody biomass is removed and less remains on site.

There are many carbon consequences to the selection of a silvicultural and harvest system. Partial harvests will leave substantial carbon in live trees on the site, whereas clearcut harvest will leave very little. On certain soils, mechanized systems for felling and extracting trees will result in more soil disturbance and subsequent CO_2 emissions than low-impact systems (Nave et al., 2010). The harvesting impact on soil carbon is greater for the forest floor than for carbon in the mineral soil, but these effects are shorter lived and may be modest over longer time intervals (Nave et al., 2010). The availability of markets for smaller-diameter material or trees of nonmerchantable species will affect how much residue (slash) is left on the site. Availability of strong markets will generally lead to higher utilization and less residue. It is important to keep accounting boundaries in mind to ensure that there is no omission or double counting of emissions or removals. The IPCC methodologies have adopted the convention that emissions from burning biomass for energy should not be accounted in the energy sector, but should be accounted in the land-use sector. We conform to this convention. If, for example, forest residues are burned for energy, the CO_2 emissions are not counted in the energy sector, and there should be a reduction in the amount of fossil fuel burned. But the CO_2 emissions from the burned residue will be accounted as a decrease in carbon stocks in the land-use sector, and emissions will be no different than if the residues had been piled and burned in the forest. That is, a complete accounting of emissions when residues are burned for energy will show emissions saved in the energy sector but no change in the land-use sector.

6.4.1.9 Fire and Fuel Load Management

Many forest types have a natural dependence on disturbance from fire. As mentioned previously, it may play a role in natural regeneration, but it has many other functions including nutrient release, natural thinning and pruning, as well as modifying fuel structure and loading. Without prescribed fire, many forest types may be at a much higher risk of reversal of growing carbon stock. In regions of the country where wildfire is a concern, forest managers may take a more active role in managing the levels of potential fuels in a forest. Fuel management cannot prevent ignitions of wildfires, but can decrease levels of intensity, severity, and spread. Two common approaches to fuel load management are prescribed burning and mechanical fuel treatments.

Prescribed fire is any fire intentionally ignited by management under an approved plan to meet specific objectives. When forest fuels are burned under carefully selected conditions (weather, fuel, moisture, etc.), fuels can be reduced to levels that decrease the risk of damaging wildfires. Other objectives for use of fire and controlled burn may be to reduce threat from non-native invasive species and maintenance of many endangered species throughout the United States.

Mechanical fuel treatments are similar to harvesting operations, in that specific classes of trees are cut and removed. For example, all trees below a threshold diameter may be removed in a thinning (Johnson et al., 2007). The result should be decreased availability of fuels that would increase wildfire severity.

The carbon impact of fuel treatments is two-fold. First, it inevitably results in emissions of CO_2 from the material removed or burned. However, second, its goal is to reduce the potential for much larger future emissions (and increased environmental damage) from wildfires in areas where they are a threat. A wildfire could result in a reversal of the previous gains in carbon on the site. Wildfire intensity and resultant loss of carbon is highly variable and depends upon site specific conditions and effects. Wildfire can occur at low to moderate intensity, which like a prescribed fire may result in a more resilient and productive site over the long term. The challenge is that the immediate CO_2

emissions from a wildfire or prescribed fire/control burn are readily quantifiable, whereas the avoided emissions from potential wildfires are not and, because treatments may not take place in the areas where wildfire occurs, they could create extra emissions that would not otherwise have happened. Recent research indicates that prescribed burning has a minimal impact on forest carbon budgets, especially in the eastern United States. Impacts observed from mechanical and fire treatments were also fairly short-lived (Boerner et al., 2008). Disposition of removed materials is a key factor to consider when assessing the GHG implications of fuel management treatments. Prescribed fire can have significant effects on reducing the risk of reversal that could result from a wildfire.

6.4.1.10 Reducing Risk of Emissions from Pests and Disease

Silvicultural intervention may also be called for when forests are damaged by weather, insects, or disease. For example, when insect outbreaks such as pine beetle infestations kill patches of trees, removal of trees at or near the infestation site may prevent populations of harmful insects from spreading further. When harvests are designed to respond to pest and disease problems, they may be called sanitation harvests.

When weather events such as ice storms, hurricanes, or severe winds (or a wildfire) cause extensive damage to forest stands, quick removal of the downed timber may provide an opportunity to recover some of the financial value of the timber and may prevent the buildup of very large fuel loads. When economic value is captured from a harvest of damaged timber, it is termed a salvage harvest.

Both salvage and sanitation harvests remove trees, sometimes with market value and sometimes without. The carbon impacts are reflected in the amount of woody material removed from the forest and whether the material removed enters markets for wood products or for energy. Similar to wildfire treatments, in both sanitation and salvage harvests, however, the removal of biomass may be compared with the alternative of leaving the material in the forest to decay or burn, resulting in CO_2 emissions. For some carbon accounting systems, this difference is crucial; the assumption that emissions would have occurred without the activity affects baseline assumptions against which carbon sequestration is measured.

6.4.1.11 Short-Rotation Woody Crops

Short-rotation woody crops, also called biomass plantations or biomass energy plantations, are tree plantations managed with a very high intensity to produce fiber crops in a relatively short time frame (e.g., 5–10 years). These plantations are more like agricultural crops in the level of intensity of treatments (e.g., fertilization, weed control, and sometimes irrigation). Wood grown in this manner is usually suitable for use by biomass energy facilities or possibly pulp mills, but the cost to produce this wood is very high compared with traditional plantations. For some species, it is possible to regenerate these stands by coppicing, or cutting to promote sprouting from intact root systems, which avoids the cost of planting new trees. Regeneration by sprouts can result in dense stands exhibiting very fast growth.

The carbon dynamics in a short-rotation woody crop system are similar to conventional plantations, except for the accelerated growth and reduced rotation length. In some short-rotation woody crop systems, cover crops may be grown to prevent erosion and maintain soil fertility. Cover crops would also serve to increase carbon storage on site.

6.4.2 Activity Data

Carbon storage from forest management activities is estimated applying three different types of estimates. Estimate Type I focuses on the effects of management activities on carbon stocks for a given year. Estimate Type II focuses on the effects of management activities on carbon stocks over a period of years in the future and must be based on projections. Estimate Type III examines the difference in projected carbon stocks between sets of alternative scenarios of potential management. This section will discuss the activity data needs for each of the types of estimates for the various forest management activities. In general, however, the estimation approaches and data needs will be of two types: (1) forest inventory data; and (2) stand projection models.

For Type I, in cases where a management activity has altered the carbon stock in specific pools, the best estimates may be obtained by having forest inventory data before and after the treatment, such that the difference can be attributed to the management activity. Forest inventory data should include measurements obtained in the forest at a series of plots, with lists of the trees in each plot. Usually for each tree it is necessary to know the species, diameter, and sometimes height. From these measurements, stand-level estimates of tree density (trees per unit area), basal area (cross-sectional bole area at 4.5 feet (1.4 m) from the ground), species composition, and tree volume and biomass can be computed.

Another approach, used for Type II and Type III estimates, requires the use of stand projection models to estimate the responses of the forest to management activities. Such models have been created for a wide variety of forest types and treatments; an example is the FVS family of models discussed earlier. Projection models for forecasting forest conditions (and carbon stocks) typically require measures or indices of forest productivity. A commonly used measure of forest productivity is site index, which represents the height that trees on a site will reach by a certain base age. For example, on land with a site index of 65 (base age 25), the average height of dominant and co-dominant trees in a stand will be 65 feet (19.8 m) when the trees reach age 25.

The most accurate Type II and Type III estimates are from models developed specifically for a given plantation species or narrowly defined forest type. For example, there are many models available to estimate effects of management on commonly planted and highly researched species such as Douglas fir or loblolly pine (e.g., Amateis and Burkhart, 2005; Burkhart, 2008; Carlson et al., 2008; Li et al., 2007; Sucre et al., 2008). At this time, the FVS family of models is the recommended method for estimating forest carbon stocks. In incorporating this method into any software tool, a data portal that allows the user to load their existing stand data and management activity data for translation into the FVS format is recommended and would prove useful. Future development may also permit custom models to interface with an estimation tool. At this time, however, such capability is not available. In cases where such models are not available, it may be necessary to generalize by aggregating forest types and management activities and perform projections based on categories of management intensity for general forest types. Management intensity categories are defined in Section 6.4.3.

The remainder of this section is organized as follows:

- Stand Density Management
- Site Preparation Techniques
- Vegetation Control
- Planting
- Natural Regeneration
- Fertilization
- Selection of Rotation Length
- Harvesting and Utilization Techniques
- Fire and Fuel Load Management

- Reducing Risk of Emissions from Pests and Disease
- Short-Rotation Woody Crops

6.4.2.1 Stand Density Management

Stand density management activities include underplanting, precommercial thinning, and commercial thinning. In each case, the primary data requirements for Type I estimates are tree inventories before and after the treatment, which can indicate the change in stocking levels and the quantity of biomass removed during thinnings. In the case of thinnings, it is important to know the volume or biomass directed to different wood products markets (e.g., pulpwood, sawtimber, or energy) to properly account for the carbon in HWPs.

For Type II and III estimates of the future carbon dynamics of the stand after these treatments, stand projection models will require a measure of site index in addition to the inventory information collected for Type I estimates.

6.4.2.2 Site Preparation

The primary information requirement for estimates of stock changes due to site preparation is whether soil disturbance has occurred during site preparation. Mechanical site preparation techniques that involve soil disturbance will be assumed to lead to a short-term loss of soil carbon storage followed by a recovery. Chemical or other treatments that don't involve soil disturbance will not result in soil CO_2 emissions beyond what may have occurred during harvesting. For Type II and III estimates, the site preparation technique should be recorded in the event that models may differentiate between growth rates corresponding to various site preparation techniques.

6.4.2.3 Vegetation Control

For Type I estimates, it is necessary to have inventory information before and after vegetation control treatments if the vegetation control involves woody material. (Carbon stocks are not expected to be substantially different for herbaceous control treatments near time of planting.) When vegetation is killed but not removed, the carbon stock impacts involve primarily the redirection of stock from one pool (live trees) to another (standing dead trees).

For Type II and III estimates, some models may project stand growth differently if competing vegetation is removed. In such cases, similar inventory information before and after treatment will be necessary.

6.4.2.4 Planting

The act of planting itself involves a negligible carbon stock change for the year of planting. Thus, a Type I estimate would show no carbon stock change following a planting.

For all subsequent years, however, critical parameters are the species planted, the original planting density (trees per acre), and the survival rate (in percent) after one growing season. Because most early mortality occurs within one year of planting, the percentage of trees surviving at year one provides a robust estimate of stand density for growth projections. It will also be important for Type II and III estimates to record the genetic stock used (e.g., first generation, open-pollinated, mass-controlled pollinated, clonal) in the event that projection models are developed for specific genetic sources. Some measure of site productivity (e.g., site index) will be needed as well.

6.4.2.5 Natural Regeneration

As in the case of plantation establishment, carbon stock changes at the time of natural regeneration are negligible.

Type II and III estimates will require information on species mix, stand density, and some information on stand productivity. In cases in which stand productivity cannot be measured directly (by measuring existing trees for site index), some estimates can be derived from soils databases such as SSURGO, or from field characterization of soil series and reference to soil maps and manuals.

6.4.2.6 Fertilization

Type I estimates will show no immediate carbon stock changes relative to fertilization for the year in which the activity occurred. N_2O emissions will occur at time of fertilization; activity data should include number of acres fertilized, application rate, and type of nitrogen applied.

Type II and III estimates involving stand projection may make use of models which incorporate information about the fertilization treatment. Application rates (pounds per acre) and elemental composition (nitrogen, phosphorus, potassium) should be recorded.

6.4.2.7 Selection of Rotation Length

Type I estimates are not applicable to selection of rotation length. Type II and III estimates may entail experimentation with rotation lengths in modeling exercises to test the carbon stock implications of different rotation length strategies. Such experimentation will simply involve the comparison of models run with all parameters held constant except for rotation length.

6.4.2.8 Harvesting and Utilization Techniques

Harvesting has the largest immediate impact on forest carbon stocks. Consequently, for Type I estimates, the landowner needs to collect accurate and sufficiently detailed forest inventory information before harvest and after harvest in the case of partial cutting. Because ongoing sequestration of carbon stocks follows different pathways for different forest products, the disposition of the harvested material into different product pools (e.g., pulpwood, sawtimber) needs to be recorded. This information should be readily available as part of sales records. Default factors are available to estimate carbon in harvesting residues (slash).

In the case of partial harvests (where there is a residual stand to project), or projections of impacts of different harvesting or silvicultural systems, complete inventory data and productivity estimates (e.g., site index) for the stand are needed.

6.4.2.9 Fire and Fuel Load Management

For Type I estimates, pre-treatment data on fuel loading with focus on the material to be removed in the treatment needs to be collected. An example of data collection protocols for fuel data can be found in Brown (1974). Post-treatment assessment of residual material will indicate the amount removed in the treatment. The type of treatment (burn or mechanical) and the disposition of fuel (consumed, left onsite, removed) should be recorded. If consumed, FOFEM or CONSUME can be used to calculate the GHG emissions from a prescribed burn.

Type II and III estimates of the carbon stock impacts of fuel treatments will require specialized fire models that could indicate likely outcomes of the fuel treatment relative to no treatment and a subsequent wildfire; available tools include models such as CONSUME (Joint Fire Science Program, 2009) and the FVS-FFE module (Reinhardt and Crookston, 2003). See Table 6-13 where a low-severity fire could be compared to the crown fire effect based on FOFEM outputs.

6.4.2.10 Reducing Risk of Emissions from Pests and Disease

For estimates of carbon stock impacts of sanitation and salvage harvests, pretreatment and post-treatment inventories are required. In the pretreatment inventory, the extent and nature of damage are needed to estimate the carbon stock that has shifted from live to dead biomass prior to treatment.

Modeling for Type II and III estimates may entail simply projecting the residual (post-treatment) stand. To fully evaluate the carbon stock impacts of the treatment, models or assumptions are needed for estimating the spread of the insect or disease absent the treatment. Tools for such modeling or assumptions may be hard to obtain.

6.4.2.11 Short-Rotation Woody Crops

Estimation of carbon stock impacts from plantations of short-rotation woody crops would follow the same general procedure as other plantation estimates. No stock changes would be expected at time of planting (carbon in seedlings or planting stock is negligible). Projections for Type II and III estimates require the availability of models to project growth and yield of the species planted under the management scenarios envisioned.

6.4.3 Management Intensity Categories

In the previous section, the use of models to predict forest responses to management activities was discussed. Many such models are available for specific management practices in plantations of certain species or in specific forest types. These models are varied in their input requirements and their applications. To develop a nationally consistent approach, the infinite combinations of sequences of specific management activities and forest types need to be generalized. Using a single modeling framework, such as FVS (Dixon, 2002) and categories of management intensities, allows for the simulation of suites of management activities in a wide variety of forest types and conditions with a single set of inputs. This approach to defining management intensity categories is similar to that used by Siry (2002).

Therefore, in this section categories of forest types and management intensities that represent broad combinations of commonly applied activities in the forest types of the United States are defined. Default tables of carbon stocks for these categories could then be developed to provide consistent and useful information about likely carbon stock implications of forest management activities across the country.

6.4.3.1 Defining Forest Type Categories

The first distinction in defining management intensity categories is the identification of the broad species grouping: hardwood, softwood, or mixed. Hardwood forest types are dominated by hardwood tree species such as oak, maple, cottonwood, birch. Softwood types are dominated by softwood tree species such as pine, spruce, or Douglas fir. Mixed types exhibit no clear dominance of one species group. The second major distinction is whether the stand was planted or naturally regenerated. Certain management activities are far more likely to be applied to plantations than natural stands. Most plantations are softwoods, with the exception of some short-rotation woody crops of hardwood types such as cottonwood, willow, hybrid poplar, or aspen.

6.4.3.2 Defining Categories of Management Intensity

Four categories of management intensity are defined based on commonly encountered practices. For example, almost all forest fertilization is applied to plantations rather than naturally regenerated stands, so fertilization will be considered part of management intensities related only to plantations. Similarly, stands that are fertilized are usually also treated with herbicide to control competing vegetation so that the fertilization benefit accrues to the desired crop species.

The four categories of management intensity are low, moderate, high, and very high. Low intensity generally refers to minimal management intervention (e.g., natural regeneration or older softwood plantations without genetically improved stock). Moderate intensity incorporates some level of active management such as intermediate harvests (e.g., thinnings). High intensity applies only to plantations and incorporates the use of superior genetic stock and vegetation control. Very high intensity management applies to aggressively managed softwood or hardwood plantations in which substantial effort is made to maximize growth using genetics, vegetation control, and fertilization. The resulting combinations of forest types, intensities, and management practices are summarized in Table 6-5.

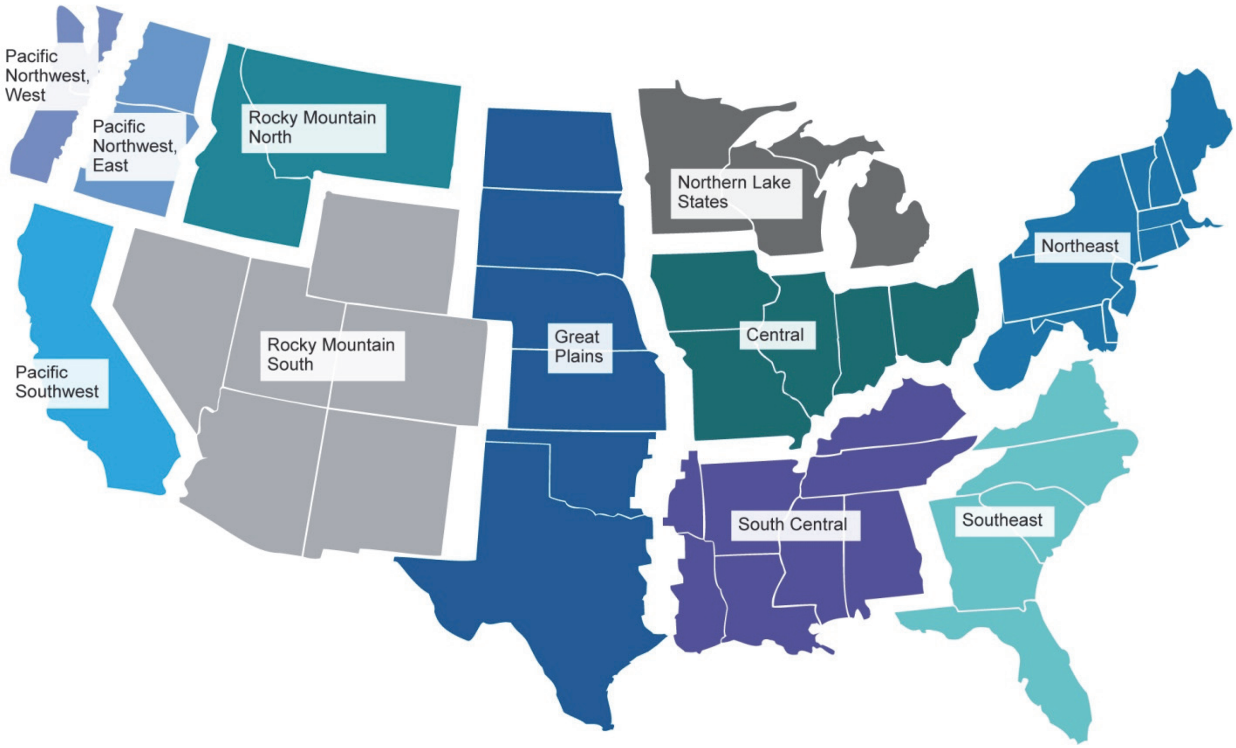
Table 6-5: Management Intensity Categories

Forest Type[a]/Management Intensity[b]	Stand Density Mgmt	Planting	Superior Genetics	Vegetation Control	Fertilization
Hardwood/low					
Hardwood/moderate	X				
Mixed/low					
Mixed/moderate	X				
Softwood (Nat)/low					
Softwood (Nat)/moderate	X				
Softwood (Plt)/low		X			
Softwood (Plt)/moderate	X	X	X		
Softwood (Plt)/high	X	X	X	X	
Softwood (Plt)/very high	X	X	X	X	X
Hardwood (Plt)/very high[c]	X	X	X	X	X

[a] Forest type refers to the combination of species group and regeneration (Nat = naturally regenerated; Plt = Planted).
[b] An X indicates that the practice indicated is applied for the management intensity category.
[c] Very high intensity hardwood plantations are usually encountered in the context of short-rotation wood crops or biomass plantations.

Figure 6-6 shows the specific regions (e.g., Pacific Northwest, West; Pacific Northwest, East; Pacific Southwest; Rocky Mountain North; Rocky Mountain South; Great Plains; Northern Lake States; Central; South Central; Northeast; and Southeast) for which silvicultural options by the most commonly managed forest type were developed.

Figure 6-6: Map of Specific Regions of Forest Management



For the management intensity categories illustrated in Table 6-5, common silvicultural options by the most commonly managed forest types for specific regions of forest management (see Table 6-6) are described. This list is not exhaustive, since silvicultural prescriptions may often be tailored to site specific conditions; however, the list provides the practices frequently applied in commonly managed forest types. The management objective may not necessarily be timber production; in some regions and types habitat restoration, rangelands, or forest health may be the primary management objectives. Table 6-6 provides a list of commonly used silvicultural prescriptions for common forest types in each region.

Table 6-6: Common Silvicultural Options by Most Commonly Managed Forest Type

Region	Forest Type	Generalized Practice
Northeast[a]	Northern hardwoods: beech, sugar maple, yellow birch, and associates	Single tree selection: harvest 40–50 ft^2 per acre every 20 years across a range of size classes in stands with 120–130 ft^2 basal area (BA)
		Clearcut: when 120–130 ft^2, then commercial thinning
		Commercial thin: At age 90–100 (120ft^2) thin to 70–80 ft^2
		Standard shelterwood: Harvest 40–50 ft^2 from below, leaving 80 ft^2 in overstory; remove overstory in 10–15 years
	Spruce–fir: red/white spruce, balsam fir	Shelterwood: Harvest 60 ft^2 from below (leave 100 ft^2); harvest remainder in 10–15 years
		Single tree selection: At 160 ft^2, remove 50 ft^2 in all sizes, every 20 years

Region	Forest Type	Generalized Practice
		Commercial thinning: At age 50–60, thin from 150 down to 100 ft^2
Central[b]	Oak–hickory	Clearcut
		Shelterwood: following local guidelines
		Group selection with commercial thinning to B-level stocking on Gingrich Guide (Gingrich, 1967)
		Precommercial/commercial thinning to B-level stocking on Gingrich Guide
		Diameter limit cut: To 12 inches DBH
		Prescribed fire: to promote oak regeneration or woodland restoration
	Elm–ash–cottonwood	Clearcut
		Individual tree selection: following local guidance
		Diameter limit cut: To 12 inches DBH
	Maple–beech–birch	Clearcut
		Shelterwood: following local guidance
		Group selection with commercial thinning to B-level stocking on Gingrich Guide
		Individual tree selection:
		Commercial thinning to B-level stocking on Gingrich Guide
		Diameter limit cut: To 12 inches DBH
	Oak–pine	Clearcut:
		Shelterwood:
		Group selection with commercial thinning to B-level stocking on Gingrich Guide
		Diameter limit cut: To 12 inches DBH
		Prescribed fire: To promote woodland restoration
Rocky Mountain South[c]	Dry montane: ponderosa pine, Douglas fir	Selection cutting: Harvest 20–30 ft^2 per acre every 20–30 years across size classes in stands to 40–80 ft^2 BA
		Commercial thinning: At age 60–80 thin to 50–60 ft^2 BA
		Shelterwood: Harvest 60–80 ft^2 BA from below; leave 30 ft^2 in overstory; remove overstory in 5–10 years
	Aspen	Coppice: At age 100
	Lodgepole pine	Clearcut: At age 120–150
	Spruce–fir	Single tree selection: Harvest 20–30ft^2 per acre every 20–30 years across size classes in stands to 80–120 ft^2 BA
	Woodland types: pinyon–juniper, Gambrel oak	Selection cutting: Harvest to 40–60 ft^2 BA
Southeast[d]	Upland hardwood	Clearcut: At age 35–50
		Single tree selection: Harvest 40–60 ft^2 per acre in stands with 100–140 ft^2 per acre
	Bottomland hardwood	Single tree selection: Harvest 40–60 ft^2 per acre in stands with 100–140 ft^2 per acre
	Pine plantation – low intensity	Plant with non-improved seedlings 600–700 per acre, thin to 60–70 ft^2 per acre at age 18–24, clearcut at age 25–35
	Pine plantation – medium intensity	Plant with improved seedlings 600–700 per acre, thin to 60–70 ft^2 per acre at age 18–22, fertilize after thinning with nitrogen and phosphorus (if needed), clearcut 5–7 years after thinning

Region	Forest Type	Generalized Practice
	Pine plantation – high intensity	Plant with improved seedlings 600–700 per acre, herbaceous weed control age 2–4, thin to 60–70 ft^2 per acre at age 16–20, fertilize after thinning with nitrogen and phosphorus (if needed), clearcut 5–7 years after thinning
South Central[d]	Upland hardwood	Clearcut: At age 35–50
		Single tree selection: Harvest 40–60 ft^2 per acre in stands with 100–140 ft^2 per acre
	Bottomland hardwood	Single tree selection: Harvest 40–60 ft^2 per acre in stands with 100–140 ft^2 per acre
	Pine plantation – low intensity	Plant with non-improved seedlings 450–700 per acre, on lower quality sites, thin to 60–70 ft^2 per acre at age 18–20; on higher quality sites, thin to 60–70 ft^2 per acre at age 12–16, on higher quality sites, thin again at age 20–24, clearcut 5–7 years after thinning
	Pine plantation – medium intensity	Plant with improved seedlings 600–700 per acre, on lower quality sites, thin to 60–70 ft^2 per acre at age 18–20; on higher quality sites, thin to 60–70 ft^2 per acre at age 12–16, fertilize after thinning with nitrogen and phosphorus (if needed), on higher quality sites, thin again age 20–24, clearcut 5–7 years after thinning
	Pine plantation – high intensity	Plant with improved seedlings 600–700 per acre, herbaceous weed control age 2–4, on lower quality sites, thin to 60–70 ft^2 per acre at age 18–20; on higher quality sites, thin to 60–70 ft^2 per acre at age 12–16, fertilize after thinning with nitrogen and phosphorus (if needed), on higher quality sites, thin again at age 20–24, clearcut 5–7 years after thinning
Northern Lake States[e]	Aspen–birch	Clearcut: 50–60 year rotation
		Shelterwood: When birch is main component: two cut system, commercial thinning at age 40–50 on high quality sites
	Northern hardwoods	Shelterwood: two stage; first cut 20 years prior to rotation age; commercial thinning as required
		Single tree/group selection with 10–20 year cutting cycle
	Oak	Clearcut: On lower quality sites, and on high quality sites where adequate advanced regeneration is present; commercial thinning as required
		Shelterwood: On high quality sites when adequate advanced regeneration is not present; commercial thinning as required
	Jack pine	Clearcut: 50–60 year rotation (note that jack pine managed for Kirtland's warbler habitat will have additional management requirements)
	Red pine	Clearcut: Commonly followed by site preparation and planting 900 per acre, commercial thinning beginning at age 25–40
		Shelterwood: Where disease risk is low; often used with prescribed fire; commercial thinning beginning at age 25–40
	White pine	Shelterwood: Two stage system; commercial thinning beginning at age 40

Region	Forest Type	Generalized Practice
	White spruce/balsam fir	Clearcut: When adequate regeneration is present
		Shelterwood: Two stage system, when adequate regeneration is not present
	Lowland conifer	Clearcut: When adequate regeneration is present; patch and strip clearcuts may be used in some cases
		Shelterwood: Two stage system, when adequate regeneration is not present
Great Plains[f]	Ponderosa pine	Two-cut Shelterwood: reduce basal area to below 60 ft^2 per acre, then remove remaining overstory after adequate regeneration is present
		Precommercial thinning as necessary to maintain desired densities
		Artificial regeneration may be required after catastrophic disturbances or to establish forests on previously unforested land; this may be done through broadcast seeding or planting
Rocky Mountain North[g]	Ponderosa pine	Plant 400–500 trees per acre, precommercial thin to 200–300 trees per acre, commercial thin to 150–200 trees per acre at age 30–40; clearcut harvest at age 60–80
	Lodgepole pine	Site prepare to expose mineral soil seedbed, natural regeneration by seeding, precommercial thin to 200–400 trees per acre, patch clearcut harvest at age 80–100
Pacific Southwest[h]	Mixed conifer: ponderosa pine, sugar pine, Douglas fir, incense cedar, white fir, Jeffrey pine, and California black oak	Commercial thin: Starting at ages near 40 and continuing at various periodic cycles until regeneration; post-thinning stocking generally ranges between 150–250 ft^2; variable rotation length, depending on objectives
		Commercial thinning with both patch regeneration and reserved areas: Similar to above, but with higher levels of variation in post-thinning stocking levels, small patches of regeneration, primarily to increase pine species, and small areas reserved from harvest, maintaining larger/older trees providing relatively unique wildlife habitats; variable rotation length, depending on objectives
Pacific Northwest, East[i]	Douglas fir/Ponderosa pine – low intensity	Site preparation by site scarification in small spots, natural regeneration, precommercial thin at age 20–25 years to 100–250 trees per acre, patch clearcut or seed-tree harvest at age 50–70
	Douglas fir/Ponderosa pine – medium intensity (on more productive sites)	Mechanical site preparation to scarify soil and remove competing vegetation, plant with improved seedlings at approx. 400–500 per acre, precommercial thin at age 15–20, commercial thin at age 30–40, patch clearcut or seed-tree harvest at age 50–70
Pacific Northwest, West[j]	Douglas fir	Site prepare stand with pre-emergent herbicides, plant with improved seedlings at approx. 450 per acre, commercial thinning as needed at age 20–30, fertilize as needed at age 30–40, clearcut harvest at age 40–50

DBH = Diameter at breast height

[a] Personal communication: Bill Leak.

[b] Personal communication: Steve Shifley.

[c] Personal communication: James Youtz, Jim Thinnes.

[d] Personal communication: Steve Prisley.

[e] Planning documents and silviculture guides, and personal communication with staff on the Huron-Manistee, Ottawa, and

Hiawatha National Forests.
[f] See Shepperd and Battaglia (2002).
[g] See Youngblood (2005).
[h] Personal communication: Joe Sherlock.
[i] See Briggs (2007).
[j] See Hanley and Baumgartner (2005).

6.4.3.3 Applying Default Tables of Management Practice Scenarios

Once the general categories of forest types and management intensities are defined, a modeling framework such as FVS could be used to develop sets of default tables of carbon stocks in various pools over time under management scenarios common to the forest types and management intensities. Note that at this time, these lookup tables are not available; developing default carbon stock values for forest management practices is a task requiring a significant level of time and effort. In the absence of such tables, small landowners wishing to estimate the effects of changing management practices (a Type III estimate) will need to use the methods described for large landowners.

Table 6-7 shows an unpopulated example for the default lookup tables of management practice scenarios. The default tables would provide regional estimates of timber volume and carbon stocks for a specific forest type group (e.g., loblolly-shortleaf pine stands) under a specific (e.g., Softwood (planted)/very high) management intensity on forest land after clearcut harvest in a specific region (e.g., the Southeast) for low productivity and high productivity sites.

Table 6-7: Regional Estimates of Timber Volume and Carbon Stocks for a Specific Forest Type Group (e.g., Loblolly-Shortleaf Pine Stands) Under a Specific (e.g., Softwood (Planted)/Very High) Management Intensity on Forest Land after Clearcut Harvest in a Specific Region (e.g., the Southeast) for Low Productivity and High Productivity Sites

Note: At this time, populated tables are not available; development of such tables is not certain.

Age	Mean Volume	Mean Carbon Density				
		Live Tree	Standing Dead Tree	Down Dead Wood	Forest Floor or Litter	Total Nonsoil
Years	$m^3\ ha^{-1}$	---------------------Metric Tons C ha^{-1} (Low Productivity)------------				
0	-	-	-	-	-	-
5	-	-	-	-	-	-
10	-	-	-	-	-	-
15	-	-	-	-	-	-
20	-	-	-	-	-	-
25	-	-	-	-	-	-
30	-	-	-	-	-	-
35	-	-	-	-	-	-
40	-	-	-	-	-	-
45	-	-	-	-	-	-
50	-	-	-	-	-	-
55	-	-	-	-	-	-
60	-	-	-	-	-	-
65	-	-	-	-	-	-
70	-	-	-	-	-	-
75	-	-	-	-	-	-
80	-	-	-	-	-	-
85	-	-	-	-	-	-
90	-	-	-	-	-	-

Age	Mean Volume	Mean Carbon Density				
		Live Tree	Standing Dead Tree	Down Dead Wood	Forest Floor or Litter	Total Nonsoil
Years	m^3 ha^{-1}	--------------------Metric tons C ha^{-1} (high productivity)-----------				
0	-	-	-	-	-	-
5	-	-	-	-	-	-
10	-	-	-	-	-	-
15	-	-	-	-	-	-
20	-	-	-	-	-	-
25	-	-	-	-	-	-
30	-	-	-	-	-	-
35	-	-	-	-	-	-
40	-	-	-	-	-	-
45	-	-	-	-	-	-
50	-	-	-	-	-	-
55	-	-	-	-	-	-
60	-	-	-	-	-	-
65	-	-	-	-	-	-
70	-	-	-	-	-	-
75	-	-	-	-	-	-
80	-	-	-	-	-	-
85	-	-	-	-	-	-
90	-	-	-	-	-	-

6.4.4 Estimation Methods

6.4.4.1 Stand Density Management

Type I estimates may be developed for stand density management. For underplanting, carbon stocks are essentially unchanged immediately after the treatment. For precommercial thinnings, carbon is moved from the live tree pool to the standing dead pool and/or forest floor pool; quantities will be low and essentially just accelerate the natural mortality of these smaller trees, thus accounting for this activity may be unnecessary. For commercial thinning, the live tree carbon stock is reduced and carbon is moved into HWPs, so these pools need to be estimated using procedures outlined in Section 6.2 and Section 6.5.

Type II and III estimates may be developed using forest growth models (i.e., FVS) specific to the forest type and practices used.

6.4.4.2 Site Preparation Techniques

Carbon stock changes that are due to mechanical site preparation techniques will consist of some oxidation of soil organic carbon that will be replaced over time by forest growth. For long-term monitoring, it may be assumed that soil carbon stocks will be stable under sustainable forest management (Smith et al., 2006). Thus, Type I estimates could reflect short-term losses of soil carbon stocks based on assumptions appropriate to the forest type and region.

6.4.4.3 Vegetation Control

Control of woody vegetation will exhibit patterns similar to precommercial thinning: transfer of carbon stocks from live tree to dead tree pools. Quantities will likely be small and the effect of short duration; hence accounting for these impacts using Type I estimates may be unnecessary.

For Type II and III estimates, vegetation control may be expected to have a beneficial impact on the growth of the residual stand that should be modeled accordingly.

6.4.4.4 Planting

Negligible carbon stock changes are expected at the time of establishment of a new plantation, so Type I estimates will show no stock changes. For Type II and III estimates, the plantation activity establishes a new stand that can then be modeled based on species, site index, and initial stocking (planting density times year 1 survival percent).

6.4.4.5 Natural Regeneration

As in the case of plantation establishment, carbon stock changes at the time of natural regeneration are negligible. For Type II and III estimates involving projections of stand growth over time, initial stocking, species mix, and site productivity will define the stand parameters for growth projections.

6.4.4.6 Harvesting and Utilization

Depending on the harvesting and silvicultural system used, multiple stock changes occur with a rotation harvest. Live tree biomass stocks are reduced by the amount of harvested wood (up to 100 percent of the live tree biomass pool). These removals should be balanced by additions to HWP pools and slash/residue in the forest floor and dead wood pools. Because losses to soil organic carbon pools from disturbance by mechanized harvesting systems are of relatively short duration, it is common to consider the loss and recapture as a steady state (e.g., Smith et al., 2006), though this may differ depending on soil characteristics.

In the case of partial harvests, there is a residual stand for which carbon stocks remain to be projected over time. Post-harvest inventory information provides the critical stand parameters to be input into growth models. In the absence of a post-harvest inventory, pre-harvest inventory data can be adjusted to reflect the loss of trees removed by the harvest (e.g., by decreasing the numbers of trees by species and diameter class based on harvest records).

6.4.4.7 Fire and Fuel Load Management

Type I estimates of carbon stock changes due to fuel treatments or prescribed fire should reflect losses to live tree biomass according to the material burned, killed, or removed (from pre and post-treatment inventory data). For a prescribed fire, emissions can be calculated using FOFEM. If slash is left from the fuel treatment, CONSUME may also be used.

Type II and III estimates simply involve projecting the stand based on information from the post-treatment inventory.

6.4.4.8 Reducing Risk of Emissions from Pests and Disease

Type I carbon stock estimates will involve computation of losses to live tree biomass from the sanitation or salvage harvest, with additions to HWP pools as appropriate.

Type II and III estimates simply involve projecting the stand based on information from the post-treatment inventory.

6.4.4.9 Short-Rotation Woody Crops

Negligible carbon stock changes are expected at the time of establishment of a new plantation, so Type I estimates will show no stock changes. For Type II and III estimates, the plantation activity establishes a new stand that can then be modeled based on species, site index, and initial stocking (planting density times year 1 survival percent).

6.4.5 Limitations and Uncertainty

6.4.5.1 Measurement Uncertainties

Forest inventory data, from which most estimates in this section are derived, contain uncertainty as a result of sampling and measurement error. Furthermore, equations are used to estimate biomass from tree measurements (species, diameters, heights), and these equations introduce additional errors. These uncertainties, however, are well documented and can be quantified.

6.4.5.2 Model Uncertainties

For the development of Type II and Type III estimates, models are used to project current conditions into the future. These types of estimates are based initially on inventory data and are subject to the measurement uncertainties discussed above, but are also subject to modeling error. Modeling error can be documented in part based on the diagnostics reported (if any) from the model development process. Greater uncertainties are introduced when models are applied beyond the conditions for which they were developed (e.g., biomass equations applied to species for which no biomass data were collected, forest growth models applied to stands receiving different management than the stands used for model development, etc.). Model uncertainties also increase with the projection period (the distance into the future for which estimates are obtained). Some of the model uncertainties are cancelled out when results from two similar model runs are compared (i.e., a Type III estimate). For example, if a model slightly overestimates carbon stock in a forest with and without some treatment, the difference between the two model estimates may be accurate even if the individual estimates are not.

6.4.5.3 Generalization Uncertainties

For the purpose of applying nationally consistent estimation methods to Type II and III estimates, it is necessary to generalize situations into broad forest types and management intensities. Thus, some precision is lost in applying a generalized, aggregated estimate to a particular set of management activities.

6.5 Harvested Wood Products

Method for Harvested Wood Products

- Method uses U.S.-specific HWPs tables.
- The HWPs tables are based on WOODCARB II model used to estimate annual change in carbon stored in products and landfills (Skog, 2008).
- The entity uses these tables to estimate the average amount of HWP carbon from the current year's harvest that remains stored in end uses and landfills over the next 100 years.
- This method was selected because it is suitable to represent the amount of carbon stored in products in use and in landfills.

6.5.1 General Accounting Issues

When forest landowners harvest wood for products, a portion of the wood carbon ends up in solidwood or paper products in end uses, and eventually in landfills, and can remain stored for years or decades. This report suggests a specific measure, along with estimation methods, that forest landowners can use to report carbon additions to the stock of HWPs from wood they harvest. The accounting framework used to track HWP carbon is similar to the framework that the United States uses to report national-level annual changes in HWP carbon stocks under UNFCCC.

The national accounting framework and these methods adopt the production approach, which entails the following: (1) estimating the annual carbon additions to and removals from the stock of carbon held in wood products in use and in landfills, (2) tracking only carbon in wood that was harvested in the United States (U.S. EPA, 2011), and (3) providing estimates that track wood carbon held in products, even if is the products are exported to other countries.

Estimates of the annual contribution of HWPs to carbon stocks may be made for Type I, Type II, and Type III estimates of forest carbon change as outlined in Section 6.2:

- For Type I estimates, the focus is on estimating the annual contribution of HWPs to carbon stocks for a given current year or recent past years.
- For Type II estimates, the focus is on estimating the annual contribution of HWPs to carbon stocks for a projected period of years in the future.
- For Type III estimates, the focus is on estimating the *change* in the annual contribution of HWPs to carbon stocks between: (1) a base case with one scenario for forest management (and harvest); and (2) a second scenario for forest management (and harvest) that is intended to change carbon flux.

For each of the Type I, II, or III estimates, these methods recommend that forest landowners report the annual contribution of HWPs to carbon stocks using a specific measure intended to approximate the climate mitigation benefit associated with storing carbon in HWPs over time. The recommended measure is the estimated *average* amount of HWP carbon from the current year's harvest that remains stored in end uses and landfills *over* the subsequent 100 years.

The intent of this measure is to approximate the average annual climate benefit of withholding carbon from the atmosphere by a certain amount each year for 100 years as described by a "decay" curve. This average benefit is one that can be credited in the year of harvest. This estimate of average effect is conceptually similar to the measure of the radiative forcing impact of a current year emission of CO_2, CH_4, or other GHG. One ton of CO_2 emissions—in GHG accounting—is equated to the radiative forcing it causes over the 100 years following the emission. The radiative forcing caused in each year is weighted the same over each of the 100 years. We are suggesting the same convention in weighting the carbon storage in wood products equally for each of 100 years.

An estimate of average fraction of HWP carbon stored over 100 years (average amount stored over 100 years divided by the original product carbon produced) is not exactly the same as the fraction of radiative forcing avoided by storing wood products carbon (and emitting carbon slowly) over 100 years. For decay curves where a constant fraction of remaining HWP carbon is emitted each year the fraction of radiative forcing avoided over 100 years can be 0 to14 percent less than the average fraction of HWP carbon stored over 100 years depending on the decay rate.[8] Estimates of the fraction of radiative forcing avoided over 100 years could be used in place of the average carbon storage. Given the uncertainty in decay rates as an influence on estimates and the greater complexity of the radiative forcing measure, we recommend the measure of average carbon stored as an adequate proxy for the effect of wood products produced in the current year and stored over

[8] The fraction of radiative forcing avoided over 100 years was estimated (and compared to average carbon stored over 100 years) assuming a range of decay rates for first order decay curves for wood products and using the CO_2 radiative forcing response curve from the IPCC Working I 4th Assessment Report (footnote a, p. 213) (IPCC, 2007).

100 years.

The measure—average carbon stored in HWP over 100 years (with variations on how landfill carbon is included)—is used in the Climate Action Reserve (2010) Forest Project Protocols adopted by the California Air Resources Board. The protocols indicate how to calculate the level of annual carbon credits that may be sold by forest landowners who enter carbon contracts.

Note that use of the production approach to accounting is not a life-cycle assessment accounting approach that could take into account how carbon emissions from increased wood burning or increased use of wood products might offset fossil fuel emissions or emissions from making non-wood products over time. The estimates of annual change in carbon in HWPs are not intended to indicate the total impact on GHG levels in the atmosphere of using HWPs (including use of wood for energy), nor are they intended to indicate that the emission to the atmosphere took place in the United States versus other countries where products were exported. Estimation of Type III secondary GHG reduction effects of substitution of wood for fossil fuels or non-wood construction products are complex and would require specification of a baseline from which change is measured and other assumptions that are beyond the scope of these methods.

The production approach is used to acknowledge that harvesting of forests does not immediately release all of the contained carbon to the atmosphere; the accounting counts only the carbon change in HWPs in order to allow annual carbon changes in HWPs to be deducted or added to annual emissions in the energy and manufacturing sectors and carbon changes in forests, so there will be no omission or double counting of sequestration or emissions to the atmosphere. In the national accounting framework, the annual emissions from wood energy are accounted for as part of the aggregated annual change in forest plus HWP carbon.

6.5.2 Estimation Methods

6.5.2.1 Wood Products Fate/Longevity

To allow forest landowners to estimate carbon additions to HWP stocks—using average carbon stored in HWP over 100 years—lookup tables are provided that give estimates of carbon remaining stored after harvest out to 100 years.

There are two types of lookup tables: a “roundwood” type and a “primary product” type.

For the roundwood type, the landowner needs estimates of the carbon in harvested amounts of industrial roundwood: hardwood (HW) or softwood (SW), sawlogs (SL), or pulpwood (PW). Industrial roundwood is wood used for solidwood or paper products and excludes bark and fuelwood. The landowner can begin with estimates in cubic units and convert them to carbon weight or wood weight units then convert them to carbon weight (assuming 0.5 metric tons carbon per metric ton dry wood). Separate lookup “decay” tables are provided by major U.S. region and roundwood type (HW or SW, SL, or PW) that show the fraction of carbon in wood typically stored in wood products in use and in landfills, out to 100 years after the year of harvest, and the average fraction stored over 100 years.

For the primary product type of lookup tables, the landowner needs estimates of the primary wood products made from the wood harvested; i.e., SW or HW lumber, SW or HW plywood, oriented strandboard, or paper (in conventional product units). The landowner then converts these amounts to carbon weight. For each primary product, the lookup “decay” tables show the fraction of wood carbon that is typically stored in wood products in use and in landfills, from the year of harvest out to 100 years, and the average fraction of carbon stored over 100 years.

6.5.3 Activity Data Collection

6.5.3.1 Primary Product Decay Tables

In order to construct the primary product type decay tables, data are used for each U.S. region on:

- The disposition of each primary product (e.g., lumber, structural panels) to major end uses (e.g., percentage of product going to residential housing, non-residential housing, manufacturing (furniture)), and percentage going to exports;
- The decay functions indicating how quickly products go out of use for each end use;
- The fraction of material going out of use that goes to landfills; and
- The fraction of material in landfills that does not decay, and the decay rate for material in landfills that does decay.

It is assumed that there is a national market for primary products and the percentage of primary products going to each end use will be the same for each U.S. region. It is also assumed that primary products exported from the United States are used in the same way as domestic products. That is, there is a national market for each of the primary wood and paper products. Data for items (1) through (4) come from the WOODCARB II model used to estimate annual change in carbon stored in products and landfills for the U.S. Inventory of GHG Emissions and Sinks report (Skog, 2008; U.S. EPA, 2010).

If a landowner knows the traditional number of units of primary products (e.g., thousand board feet of lumber) that were made from the timber harvested from their land in a given year, they can use Tables 6-A-1, 6-A-2, and 6-A-3 to estimate the carbon contents in these products (Table 6-A-1) and estimate the amount of carbon stored in these products (in use and in landfills) out to 100 years and the average amount of carbon stored over 100 years (Table 6-A-2 [in use] and Table 6-A-3 [in landfills]).

The average amount of carbon stored over 100 years for a particular primary product is the total of the averages for products in use and products in landfills shown in Tables 6-A-2 (in use) and Table 6-A-3 (in landfills).

6.5.3.2 Roundwood Decay Tables

In order to construct the roundwood type of decay tables, data are needed for each region on the percentage of HW or SW, SL, or PW that goes to various primary wood products; for example, the fraction of SW SLs in the South that goes to lumber, panels, and paper. After the amounts of primary wood products are estimated, the primary products type decay tables can be used to construct roundwood decay tables. Data needed to divide roundwood into primary products for each region include Forest Service FIA timber product output data and national data on primary wood products production (Howard, 2012; Smith et al., 2007).

If a landowner knows the cubic feet of roundwood, in the form of HW or SW SLs or PW that is harvested from their land in a given year, they can use Table 6-A-4 and 6-A-5 to (1) estimate the weight of wood harvested; (2) convert weight of wood to carbon by multiplying by 0.5 (i.e., the fraction of dry biomass to carbon conversion factor); and (3) estimate the total amount of carbon stored in the products (the sum of amounts in use and in landfills) each year out to 100 years, and the average stored over 100 years.

If the landowner knows the weight of roundwood harvested rather than cubic feet, it would use steps 2 and 3 above.

Annual HWP carbon (average stored over 100 years) is given for each region and roundwood type in Table 6-A-5.

If the landowner is making forest growth and harvest projections (Type II and Type III estimates) and only knows the cubic feet (or weight) of growing stock of HW and SW SLs and PW that will be harvested in given future years, then Table 6-A-6 can be used to estimate the total amount of roundwood that can expected to be harvested (growing stock and non-growing stock). These total amounts of roundwood (HW and SW SLs and PW may then converted to carbon and to carbon stored (and average carbon stored over 100 years) using Table 6-A-4 and Table 6-A-5, as discussed above. To convert 1 cubic foot of dry wood to pounds multiply density by 62.4 lbs ft^{-3}. To convert 1 cubic foot to kilograms multiply density by 28.3 kg ft^{-3}.

A spreadsheet is available showing all the parameters and calculations that produce the carbon storage tables that start with primary products or roundwood harvest (Skog, 2013).

6.5.4 Limitations, Uncertainty, and Research Gaps

6.5.4.1 Uncertainty in C change estimate

General estimates of uncertainty, given as the 95 percent confidence intervals, can be made for HWP measure used in Type I carbon change estimates (current year or recent past years). These estimates of uncertainty could be provided with each of the two types of lookup tables, and can be made using Monte Carlo simulations and assumptions about HWP uncertainty that are used for the Inventory of U.S. Greenhouse Gas Emissions and Sinks report (U.S. EPA, 2011). Uncertainty could be specified for key variables including: (1) fractions of SLs PW going to various primary products; (2) fractions of primary products going to various end uses; (3) rate at which products are discarded from each end use; (4) fraction of discarded wood or paper that goes to landfills; (5) fraction of wood or paper set to landfills that is subject to decay; and (6) rate of decay in landfills of degradable wood/paper carbon.

A spreadsheet is available the could be used as a basis for Monte Carlo simulations to estimate overall uncertainty for estimates of average carbon stored over 100 years (Skog, 2013).

It would be possible but more complex to make uncertainty estimates for Type II and Type III carbon change estimates by adding estimates of uncertainty in parameters used to make projections of harvest.

Additional research is needed to improve differentiation of the various rates at which solidwood products are discarded from uses such as pallets, railroad, railcars, and furniture that are currently grouped into one category. This further differentiation would refine estimates of average carbon stored when the landowner knows which primary wood products are made from the wood that is harvested on their land. Alternate curves for discard rates from end uses, particularly discards from housing, if empirically verified, could improve estimates of average carbon stored. Estimates of uncertainty in parameters over 100 year projections are needed to give a sound estimate of the uncertainty in average carbon stored over 100 years.

6.6 Urban Forests

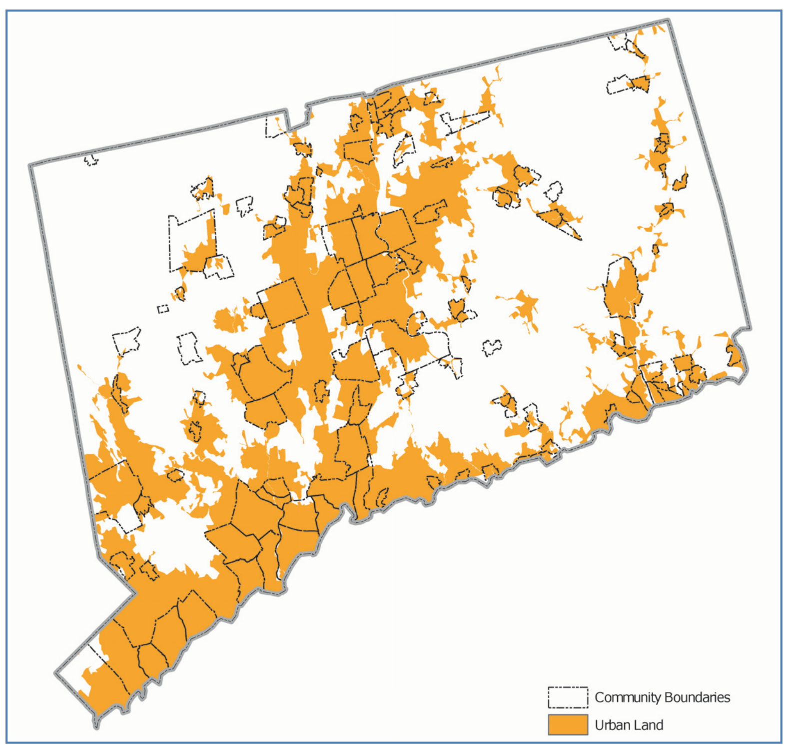
Methods for Urban Forests

- Range of options depends on data availability of the entity's urban forest land.
- These options use:
  - i-Tree Eco model (http://www.itreetools.org) to assess carbon from field data on tree populations; and
  - i-Tree Canopy model (http://www.itreetools.org/canopy/index.php) to assess tree cover from aerial images and lookup tables to assess carbon.
- Quantitative methods are also described for maintenance emissions and altered building energy use and included for information purposes only.
- The methods were selected because they provide a range of options dependent on the data availability for the entity's urban forest land.

6.6.1 Description

6.6.1.1 Defining Urban Areas and Forests

Figure 6-7: Urban and Community Land in Connecticut



Source: U.S. Census Bureau (2007).

Urban forests are composed of a population of all trees within an urban area. To delimit the extent of an urban forest, the boundaries of the urban area must be drawn. This boundary issue can be problematic, as people may conceive or define "urban" differently. To delimit urban areas in the United States, U.S. Census bureau definitions are used. These definitions differ from those used in the National Resources Inventory, which aims to identify areas that are removed from the rural land base and includes land uses such as transportation corridors.

The U.S. Census Bureau (2007) defines urban as all territory, population, and housing units located within urbanized areas or urban clusters. Urbanized area and urban cluster boundaries encompass densely settled territories, which are described by one of the following: (1) one or more block groups or census blocks with a population density of at least 386.1 people km^{-2}

(1,000 people $mile^{-2}$); (2) surrounding block groups and census blocks with a population density of 193.1 people km^{-2} (500 people $mile^{-2}$); and (3) less densely settled blocks that form enclaves or indentations or are used to connect discontinuous areas. More specifically, urbanized areas consist of territories of 50,000 or more people. Urban clusters, a concept new to the 2000 Census, consist of territories with at least 2,500 people but fewer than 50,000 people.

In addition to urban land, the Census Bureau designates places that delimit population concentrations based on incorporated or unincorporated places, such as a city, town, village, and census-designated place. These places, or "communities," also define areas where people reside, but often with a lower population density. The geographic areas of urban and communities overlap (see Figure 6-7), and either or both could be used to define urban forests. The urban land designation delimits higher population densities, but does not follow the boundaries of cities or towns that most people can relate to. The place or community boundaries follow these political boundaries, but often include both rural and urban land.

Urban land is defined based on population density, and community land is often based on political boundaries. Thus, urban forest land overlaps with forest lands. That is, forested stands that are measured as part of other programs can exist within urban or community boundaries. Assessments of urban forest effects thus have the potential to double-count effects found in forests within regional or national scale assessments. The amount of this overlap is estimated as 13.8 percent of urban area or 1.5 percent of forest area in the conterminous United States (Nowak et al., 2013) and is an important consideration for larger scale assessments. This section focuses on assessing the carbon effects of urban or community trees and forests in the United States.

Urban or community forests (hereafter referred to as urban forests) affect the carbon cycle by directly storing atmospheric carbon within the woody vegetation, but also by affecting the local climate and thereby altering carbon emissions affected by local climatic conditions. Urban tree maintenance activities also affect carbon emissions in urban areas. For a true accounting of carbon effects, all of these factors need to be considered. This report focuses on trees (defined as woody vegetation with a diameter of at least 1 inch (2.5 cm) DBH), but similar accounting could be conducted for all urban vegetation.

6.6.1.2 Accounting for Primary Urban Forest Carbon Effects

Trees sequester and store carbon in their tissue at differing rates and amounts, based on such factors as tree size, life span, and growth rate. After a tree is removed, the tree can decompose with the carbon stored in that tree emitted back to the atmosphere, or the carbon may be stored in wood products or the soil. Thus, in order to account for the total carbon in the system at one time, one needs to understand how many trees there are in the system along with information such as species and size (e.g., Nowak and Crane, 2002). To account for how the carbon stock will change through time, one must also account for growth rates, tree mortality and removals, and the disposition of the wood after removal (e.g., chipping, burning, products), which affect decomposition rates and carbon emissions. In addition, the number of new trees entering the system through tree planting and natural regeneration must be considered.

6.6.1.3 Accounting for Secondary Effects

In addition to the carbon stored in trees, the urban forest has secondary impacts on atmospheric carbon by affecting carbon emissions from urban areas. Tree care and maintenance practices often release carbon back to the atmosphere via fossil-fuel emissions from maintenance equipment (e.g., chain saws, trucks, chippers). Thus, some of the carbon gains from tree growth are offset by carbon losses to the atmosphere via fossil fuels used in maintenance activities (Nowak et al., 2002). Trees strategically located around buildings can reduce building energy use (e.g., Heisler, 1986), and consequently reduce carbon emissions from fossil-fuel-burning power plants. These energy effects are caused primarily by tree transpiration (lowering of air temperatures), blocking of winds, and shading of buildings and other surfaces. Trees typically lower building energy use in summer, but can either lower or increase building energy use in the winter depending upon the tree's location relative to a building.

"Altered building energy use" and "maintenance emissions" for urban trees are described in Section 6.6.3.1. However, while quantitative methods are described for estimating altered building energy use and maintenance emissions for urban forestry, they are included for information purposes only, since they have already been developed as part of the i-Tree software suite. However, as previously mentioned in Chapter 1, the scope of this guidance does not include other energy-related source categories that are associated with management activities related to certain agriculture and forestry activities (e.g., transportation, fuel use, heating fuel use).

6.6.2 Activity Data Collection

To estimate carbon storage, annual sequestration, and long-term carbon changes, two general approaches could be used. The first method is based on collecting data on trees in the urban area of interest; the second method involves collecting aerial data on tree cover in the area, and using tables to estimate effects based on field data from other areas. The first method, using local field data, will produce the most accurate estimates for the local area, but at increased costs and time spent by the landowner. The second method is more cost-effective and more straightforward, but its accuracy is more limited (see Table 6-8).

Table 6-8: Comparison of the "Field Data" and "Aerial" Methods for Estimating the Changes in Carbon Stocks for Urban Forests

Field Data Method	Aerial Method
Requires significant time commitment to take field measurements	Requires less time to extract necessary aerial data from an existing database
Requires access to several sample plots across an area	Does not require field measurements, only a computer with internet access
Increases specificity and accuracy	Returns a more approximate estimate
Provides a variety of output data including current carbon stock, annual carbon sequestration, and long term effects	Provides only information on total carbon stored and annual carbon sequestration

The output data from the field data method includes current carbon stock (existing carbon storage), annual carbon sequestration by trees, and long term effects of the forest (accounting for changes in tree population and disposition of carbon from trees). For the field data method (or for producing the default tables that are used in the aerial approach) the following items need to be measured and input by the landowner:

- Current Stock:
  - Number of trees by species and size class (species, DBH, height, condition, competition factor)
- Annual Sequestration:
  - Number of trees by species and size class (species, DBH, height, condition, competition factor)
  - Annual growth rates for each tree based on tree and site conditions (inches per year)
- Long Term Effects:

  - Number of trees by species and size class (species, DBH, height, condition, competition factor)
  - Annual growth rates for each tree based on tree and site conditions (inches per year)
  - Changes in tree population due to tree death and removals, and new trees planted or naturally regenerated (numbers of trees dying by species and size class, number of new trees by species and size class) (number per year)
- Proportion of removed tree biomass that is:
  - Chipped/mulched
  - Burned
  - Burned to produce energy (e.g., heat buildings)
  - Below the ground in roots
  - Used for long-term wood products
  - Left on the ground to decompose naturally
  - Put in landfills
- Decomposed; decomposition rates for wood from removed trees:
  - Percentage of biomass per year decomposed per removal class above
- Maintenance Emissions:
  - Amount (number and hours per year) of maintenance equipment used (e.g., vehicles, chippers, chain saws) for vegetation maintenance (e.g., planting, maintenance, tree removal)
  - Emission factors (g C hr^{-1}) for each maintenance equipment used
- Altered Building Energy Use:
  - Number of trees by species and size class within 60 feet (18.3 m) of residential building by cardinal and ordinal direction

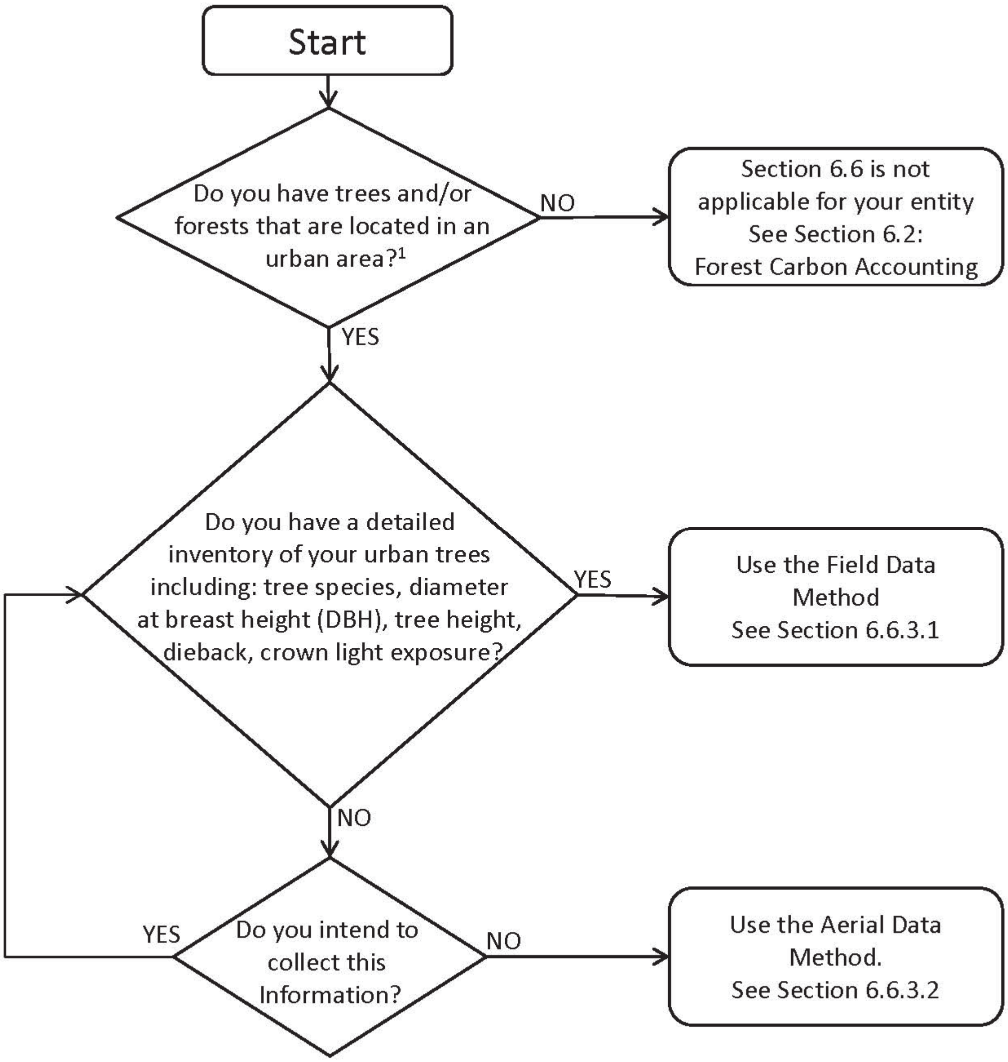
For estimating tree cover using the aerial approach, one would need to know the extent (ha) of the urban area and the percentage of tree cover within the area, and use a default table of values to convert ha of tree cover to primary and secondary tree effects in a city. To estimate change in the population, the tree cover would need to be re-measured through time.

As previously mentioned, altered building energy use and maintenance emissions for urban trees are described in Section 6.6.3.1. However, while quantitative methods are described for estimating altered building energy use and maintenance emissions for urban forestry, they are included for information purposes only, as they are part of the i-Tree software suite or can be calculated from i-Tree data.

6.6.3 Estimation Methods

The methods for estimating carbon effects in an urban forest will be detailed for the field data and aerial approaches separately. The field data method and aerial method for urban forests are described in Section 6.6.3.1 and Section 6.6.3.2. Figure 6-8 shows a decision tree indicating which method is more applicable for each type of activity data.

Figure 6-8: Decision Tree for Urban Forests Showing Methods Appropriate for Estimating Urban Forest Carbon Stocks



[1] The U.S. Census Bureau (2007) defines urban as all territory, population, and housing units located within urbanized areas or urban clusters. Urbanized area and urban cluster boundaries encompass densely settled territories, which are described by one of the following: (1) one or more block groups or census blocks with a population density of at least 386.1 people km^{-1} (1,000 people $mile^{-2}$); (2) surrounding block groups and census blocks with a population density of 193.1 people km^{-2} (500 people $mile^{-2}$); and (3) less densely settled blocks that form enclaves or indentations, or are used to connect discontinuous areas. More specifically, urbanized areas consist of territories of 50,000 or more people. Urban clusters, a concept new to the 2000 Census, consist of territory with at least 2,500 people but fewer than 50,000 people.

6.6.3.1 Field Data Method for Estimating Carbon Storage and Annual Sequestration

The field data method involves using field measurements of urban trees (i.e., a "tree list") to build a tailored, accurate estimate of carbon storage and sequestration in an urban forest. The various steps for estimating carbon (and altered building energy use) effects from an urban forest are:

(1) *Delimit boundary of urban area to be analyzed*. This information is essential to set the boundary of the analysis. U.S. Census boundary files of urban areas or places can be used to delimit the boundaries (U.S. Census Bureau, 2011). Information on these boundaries can be used to determine areas of potential overlap with other carbon estimates (e.g., non-urban forests), and to help set up a

sampling design to collect necessary field data as desired by the landowner.

(2) *Measure all trees within the urban area or sample the tree population*. Within the defined geography, all trees can be measured, or a random distribution of field plots can be measured to quantify the urban tree population as desired by the landowner. To conduct this field sampling and analysis, the i-Tree Eco model (formerly UFORE model) is available free of charge at www.itreetools.org. Field manuals exist on how to randomly select plots locations and collect the needed field data (http://www.itreetools.org/resources/manuals.php). Details on model methods also exist (e.g., Nowak and Crane, 2002; Nowak et al., 2008).

The basic field data procedure is to record information on all trees within the field plots. This information includes:

- Tree species
- DBH
- Tree height
- Dieback
- Crown light exposure
- Distance and direction to buildings

These variables are needed to assess carbon effects, but other tree variables (e.g., crown width, percentage of crown missing) can also be collected to assess other ecosystem services (e.g., air pollution removal, volatile organic compound emissions, effects on building energy use, rainfall interception, and runoff).

(3) *Enter data into i-Tree and run analyses*. After field data are collected (via paper forms or on a mobile device), data are entered into i-Tree, and the program produces standard tables, graphs, and reports that detail carbon and other ecosystem service information. In relation to carbon, results along with sampling standard errors are specifically produced by species and land use regarding:

- Carbon storage: amount of carbon currently in the existing tree stock;
- Gross annual carbon sequestration: one-year estimate as sequestration based on estimated annual tree growth, which varies by location, tree condition, and crown competition; and
- Net annual carbon sequestration: gross sequestration minus estimated carbon lost from dead or removed trees due to decomposition.

Altered Building Energy Use. In addition to the carbon effects estimated by the field data method, the i-Tree program can estimate tree effects on residential building energy use and consequent carbon emissions using methods detailed in McPherson and Simpson (1999).

Maintenance Emissions. For estimating maintenance emission effects, the following steps are suggested:

(1) *Determine vehicle use related to tree maintenance*. Determine the number of miles driven by various vehicle types.

(2) *Calculate carbon emissions from vehicles*. To estimate carbon emissions from vehicles, the latest fuel efficiency information (mpg) will be needed for each vehicle class. Divide the miles driven by the vehicle class mpg to determine the total gallons of gasoline (or other fuel) used. Multiply total gallons (or other units) used by the emissions factor in Table 6-9 to estimate carbon emissions from vehicle use (Nowak et al., 2002).

Table 6-9: Emission Factors for Common Transportation Fuels

Fuel	Emissions (lbs CO_2 per unit volume)
B20 biodiesel	17.71 per gallon
B10 biodiesel	19.93 per gallon
Diesel fuel (No.1 and No. 2)	22.15 per gallon
E85 ethanol	2.9 per gallon
E10 ethanol	17.41 per gallon
Gasoline	19.36 per gallon
Natural gas	119.90 per Mcf
Propane	5.74 per gallon

Source: Table 1.D.1, U.S. DOE (2007).

(3) *Determine maintenance equipment use*. Estimate the number of run hours used for all fossil-fuel-based maintenance equipment used on trees (e.g., chainsaws, chippers, aerial lifts, backhoes, and stump grinders). Estimates of run time for various pruning and removal equipment are given in Table 6-10.

Table 6-10: Total Hours of Equipment Run-Time by DBH Class for Tree Pruning and Removal

DBH	Pruning				Removal					
	2.3 hp Saw	3.7 hp Saw	Bucket Truck[a]	Chipper[b]	2.3 hp Saw	3.7 hp Saw	7.5 hp Saw	Bucket Truck[a]	Chipper[b]	Stump Grinder[b]
1–6	0.05	NA	NA	0.05	0.3	NA	NA	0.2	0.1	0.25
7–12	0.1	NA	0.2	0.1	0.3	0.2	NA	0.4	0.25	0.33
13–18	0.2	NA	0.5	0.2	0.5	0.5	0.1	0.75	0.4	0.5
19–24	0.5	NA	1.0	0.3	1.5	1.0	0.5	2.2	0.75	0.7
25–30	1.0	NA	2.0	0.35	1.8	1.5	0.8	3.0	1.0	1.0
31–36	1.5	0.2	3.0	0.4	2.2	1.8	1.0	5.5	2.0	1.5
+36+	1.5	0.2	4.0	0.4	2.2	2.3	1.5	7.5	2.5	2.0

Note: Table is based on ACRT data (Wade and Dubish, 1995) and assumes that crews work efficiently and equipment is not run idle (Nowak et al., 2002).
hp = Horsepower
DBH = Diameter at breast height
[a] Mean hp = 43 (U.S. EPA, 1991)
[b] Mean hp = 99 (U.S. EPA, 1991)

(4) *Calculate carbon emissions from maintenance equipment*. Calculations for emissions from equipment are based on the formula:

Equation 6-10: Calculate Carbon Emissions from Maintenance Equipment

$$C = N \times HRS \times HP \times LF \times E$$

Where:

C = Carbon emissions (g)

N = Number of units (dimensionless)

HRS = Hours used (hr)

HP = Average rated horsepower (hp)

LF = Typical load factor (dimensionless)

E = Average carbon emissions per unit of use (g hp^{-1} hr^{-1}) (U.S. EPA 1991)

Typical load factors and average carbon emissions for equipment are given in Table 6-11.

Table 6-11: Typical Load Factors (U.S. EPA, 1991), Average Carbon Emissions, and Total Carbon Emissions for Various Maintenance Equipment (from Nowak et al., 2002)

Equipment	Typical Load Factor[a]	Average Carbon Emission (g hp^{-1} hr^{-1})[b]	Total Carbon Emission (kg hr^{-1})[c]
Aerial lift	0.505	147.2	3.2[d]
Backhoe	0.465	147.3	5.3[e]
Chain saw <4 hp	0.500	1,264.4	1.5[f]
Chain saw >4 hp	0.500	847.5	3.2[g]
Chipper/stump grinder	0.370	146.4	5.4[h]

[a] Average value from two inventories (conservative load factor of 0.5 from inventory B was used for chain saws >4 hp due to disparate inventory estimates; inventory average for this chain saw type was 0.71).
[b] Calculated from estimates of carbon monoxide (U.S. EPA, 1991), hydrocarbon crankcase and exhaust (U.S. EPA, 1991), and carbon dioxide emissions (Charmley, 1995), adjusted for in-use effects. Total carbon emissions were calculated based on the proportion of carbon of the total atomic weight of the chemical emission. Multiply by 0.0022 to convert to lbs hp^{-1} hr^{-1}.
[c] Multiply by 2.2 to convert to lbs hr^{-1}.
[d] Mean hp = 43 (U.S. EPA, 1991).
[e] Mean hp = 77 (U.S. EPA, 1991).
[f] hp = 2.3.
[g] hp = 7.5.
[h] Mean hp = 99 (U.S. EPA, 1991).

(5) *Calculate total maintenance carbon emissions*. Add results of carbon emissions from vehicles and maintenance equipment.

Combined Carbon Sequestration, Altered Building Energy Use, and Maintenance Emissions. To determine current net annual urban forest effect on carbon, the carbon emissions from tree maintenance should be contrasted to net carbon sequestration from trees and altered carbon emissions from altered building energy use effects.

Changes in Carbon Sequestration, Altered Building Energy Use, and Maintenance Emissions. To determine how tree and maintenance effects on carbon change through time, the field plots or trees inventoried can be re-measured, and results between the years contrasted to estimate changes in carbon stock, net annual carbon effects, and altered building energy use effects. In addition, maintenance activity estimates should be updated when the re-measurement occurs.

6.6.3.2 Aerial Data Method

The aerial data method uses aerial tree cover estimates and lookup tables to provide a more approximate (i.e., higher degree of uncertainty), but less resource intensive estimate of annual carbon sequestration in an urban forest compared to the field data method. The various steps for estimating carbon effects from an urban forest are:

(1) *Delimit boundary of urban area to be analyzed*. This information is essential to set the boundary of the analysis. U.S. Census boundary files of urban or places can be used to delimit the boundaries (U.S. Census Bureau, 2011). Information on these boundaries can be used to determine areas of potential overlap with other carbon estimates (e.g., non-urban forests).

(2) *Conduct photo interpretation of tree cover in urban area*. To determine percentage of tree cover, the urban area can be photo interpreted using i-Tree Canopy (http://www.itreetools.org/canopy/index.php). This web tool allows users to import a shape file of, or manually delimit their area, and then randomly locate points within the area on Google® aerial imagery. The user then classifies each point according to its cover class (e.g., tree or non-tree). The program produces estimates of percentage cover and associated standard error for the cover classes. This same type of analysis could also be performed with digital aerial images using a Geographic Information System.

(3) *Estimate total tree cover in urban area*. Multiply the percentage of tree cover and standard error by urban area (ha) to produce an estimate of total tree cover and standard error (ha). Note that i-Tree Canopy will make these calculations.

(4) *Estimate carbon effects*. Multiply total tree cover (ha) by average carbon storage or annual sequestration per ha of tree cover in places or urban areas (Table 6-12). i-Tree Canopy will make these calculations based on average state or national data.

Note that to estimate effects for maintenance emissions and altered building energy use based on total tree cover, a table similar to Table 6-12 would need to be developed containing emission rates for these source categories.

Table 6-12: Metric Tons Carbon Storage and Annual Sequestration per Hectare of Tree Cover in Selected Cities and Urban Areas of Selected States (from Nowak et al., 2013)

City, State	Storage		Sequestration	
	Metric tons C ha^{-2}	Standard Error	Metric tons C ha^{-2} $year^{-1}$	Standard Error
Arlington, TX[a]	63.7	7.3	2.9	0.28
Atlanta, GA[a]	66.3	5.4	2.3	0.17
Baltimore, MD[a]	87.6	10.9	2.8	0.36
Boston, MA[a]	70.2	9.6	2.3	0.25
Casper, WY[b]	69.7	15.0	2.2	0.39
Chicago, IL[c]	60.3	6.4	2.1	0.21
Freehold, NJ[a]	115.0	17.8	3.1	0.45
Gainesville, FL[d]	63.3	9.9	2.2	0.32
Golden, CO[a]	58.8	13.3	2.3	0.45
Hartford, CT[a]	108.9	16.2	3.3	0.46
Jersey City, NJ[a]	43.7	8.8	1.8	0.34
Lincoln, NE[a]	106.4	17.4	4.1	0.63
Los Angeles, CA[e]	45.9	5.1	1.8	0.17
Milwaukee, WI[a]	72.6	11.8	2.6	0.33

City, State	Storage		Sequestration	
	Metric tons C ha^{-2}	Standard Error	Metric tons C ha^{-2} $year^{-1}$	Standard Error
Minneapolis, MN[f]	44.1	7.4	1.6	0.23
Moorestown, NJ[a]	99.5	9.3	3.2	0.30
Morgantown, WV[g]	95.2	11.6	3.0	0.37
New York, NY[h]	73.3	10.1	2.3	0.29
Oakland, CA[i]	52.4	1.9	na	na
Omaha, NE[a]	141.4	22.9	5.1	0.81
Philadelphia, PA[j]	67.7	9.0	2.1	0.27
Roanoke, VI[a]	92.0	13.3	4.0	0.58
Sacramento, CA[k]	78.2	15.7	3.8	0.64
San Francisco, CA[l]	91.8	22.5	2.4	0.50
Scranton, PA[m]	92.4	12.8	4.0	0.52
Syracuse, NY[a]	85.9	10.4	2.9	0.30
Washington, DC[n]	85.2	10.4	2.6	0.30
Woodbridge, NJ[a]	81.9	8.2	2.9	0.28
Indiana[o]	88.0	26.8	2.9	0.77
Kansas[p]	74.2	13.0	2.8	0.48
Nebraska[p]	66.7	18.6	2.7	0.74
North Dakota[p]	77.8	24.7	2.8	0.79
South Dakota[p]	30.6	6.6	1.3	0.26
Tennessee[q]	64.7	5.0	3.4	0.21
Average	**76.9**	**13.6**	**2.8**	**0.45**

[a] Unpublished data analyzed using the UFORE model.
[b] Nowak et al. (2006a).
[c] Nowak et al. (2011).
[d] Escobedo et al. (2009).
[e] Nowak et al. (2011).
[f] Nowak et al. (2006c).
[g] Nowak et al. (2012c).
[h] Nowak et al. (2007d).
[i] Nowak (1991).
[j] Nowak et al. (2007c).
[k] Nowak et al. (In review).
[l] Nowak et al. (2007b).
[m] Nowak et al. (2010).
[n] Nowak et al. (2006b).
[o] Nowak et al. (2007a).
[p] Nowak et al. (2012b).
[q] Nowak et al (2012a).

Combined Carbon Sequestration, Altered Building Energy Use, and Maintenance Emissions. To determine current net annual urban forest effect on carbon, the carbon emissions from tree maintenance, if available, should be contrasted to the net carbon sequestration from trees and altered carbon emissions from altered building energy use effects.

Changes in Carbon Sequestration, Altered Building Energy Use, and Maintenance Emissions. To determine tree effects on carbon change through time, the photo-interpretation points can be re-measured when newer photos become available to assess change in tree cover (e.g., Nowak and Greenfield, 2012). The i-Tree Canopy program saves the geographic coordinates of each point so the points can be re-measured in the future. Changes in tree cover and associated carbon effects between the years can be contrasted to estimate changes in carbon stock and net annual carbon effects. Changes in altered building energy use effects and maintenance effects could also be estimated if the appropriate tables are developed.

6.6.4 Limitations and Uncertainty

Field data collection estimates have fewer limitations than the aerial approach, but some limitations exist (Nowak et al., 2008). The main advantage of carbon estimation using the field data approach and i-Tree is having accurate estimates of the tree population (e.g., species, size, distribution) with a calculated level of precision. The modeled carbon values are estimates based on forest-derived allometric equations (Nowak, 1994; Nowak and Crane, 2002). The carbon estimates yield a standard error of the estimate based on sampling error, rather than error of estimation. Estimation error is unknown, and likely larger than the reported sampling error. Estimation error includes the uncertainty of using biomass equations and conversion factors, which may be large, as well as measurement error, which is typically small. The standardized carbon values (e.g., kg C ha^{-1} or lbs C (acre of tree cover)$^{-1}$) fall in line with values for forests (Birdsey and Heath, 1995), but values for cities (places) can be higher (Table 6-12), likely due to a larger proportion of large trees in city environments and relatively fast growth rates due to a more open urban forest structure (Nowak and Crane, 2002).

There are various means to help improve the carbon storage and sequestration estimates for urban trees. Carbon estimates for open-grown urban trees are adjusted downward based on field measurements of trees in the Chicago area (Nowak, 1994). This adjustment may lead to conservative estimates of carbon. More research is needed on the applicability of forest-derived equations to urban trees. In addition, more urban tree growth data are needed to better understand regional variability of urban tree growth under differing site conditions (e.g., tree competition) for better annual sequestration estimates. Average regional growth estimates are used based on limited measured urban tree growth data standardized to length of growing season and crown competition.

There are currently a very limited number of biomass equations for tropical trees in i-Tree. The model needs to be updated with tropical tree biomass equations for more accurate estimates in tropical cities. Future research is needed to obtain biomass equations for urban or ornamental tree species. Estimates of tree decay and net annual sequestration in i-Tree are quite rudimentary (Nowak et al., 2010), and can be improved with future research. The degree of uncertainty of the net carbon sequestration estimates is unknown.

Estimates of maintenance emissions and altered building energy use effects are also rather crude. Accurate maintenance emissions estimates require good estimates of vehicle and maintenance equipment use; then they rely on an average multiplier for emissions from the literature. Energy effects estimates are based on sampling proximity of trees near buildings within various tree size, distance, and direction classes from a building. Energy factors, converted to carbon emission factors based on state average energy distribution (e.g., electricity, oil) are applied to trees in each building location class based on U.S. climate zone and average building types in a state to estimate energy effects (see McPherson and Simpson, 1999). Though these estimates are crude, with an unknown certainty, they are based on reasonable approaches that provide first-order estimates of effects. It should be noted that emission reductions from altered building energy use effects might also be implicitly included in any emission estimation an entity might perform based on actual energy use data (e.g., meter readings) for the building in question.

Estimates based on aerial tree canopy effects have the same limitations as field data approaches, plus some additional limitations and advantages. The advantages include a simple, quick, and accurate means to assess the amount of canopy cover in an area, with measures that are repeatable through time. The disadvantages are that the user must use a lookup value from a table (e.g., mean value per unit of canopy cover) to estimate carbon effects. Though the tree cover estimate will be accurate with known uncertainty (i.e., standard error), the carbon multipliers may be off depending upon the urban forest characteristics. If average multipliers are used, the accuracy of those estimates will decline as the difference increases between the local urban characteristics and the values of the average multipliers. If local field data are not collected, then the discrepancy between the urban forest's characteristics and those of average values is unknown. If the average values in

Table 6-12 truly represent averages, the estimates over a large population of urban areas should be reliable. However, local estimates may be inaccurate depending upon the extent to which characteristics of the local urban forest diverge from the average values.

Both approaches can provide carbon estimates for urban areas, with differing degrees of uncertainty and work required. Both approaches can also be improved with more field data collection in urban areas, and with model and method improvements related to carbon estimation.

6.7 Natural Disturbance – Wildfire and Prescribed Fire

Methods for Emissions from Natural Disturbances

- Range of options depends on the data availability of the entity's forest land including:
  - FOFEM model entering measured biomass; and
  - FOFEM model using default values generated by vegetation type.
- These options use Reinhardt et al. (1997).
- The methods were selected because they provide a range of options dependent on the data availability of the entity's disturbed forest land.

6.7.1 Description

Fire produces GHG emissions directly through fuel consumption. Emissions produced are directly proportional to fuel consumed. Fuel consumption is in turn influenced by fuel quantity and fuel characteristics such as size, moisture content, fire weather, and fire severity. Algorithms exist for estimating fuel consumption for a variety of fuel types and conditions. Fire and other disturbances also convert live vegetation to dead, altering subsequent carbon dynamics on the site by reducing carbon captured by photosynthesis in the short run due to reduced vegetative cover, and increasing emissions from decomposition of dead vegetation. Fire severity, which is driven by the onsite factors that drive consumption as well as other physical factors, will drive the subsequent carbon dynamics and area where reversal of carbon retention may occur.

6.7.2 Activity Data Collection

For all disturbances, key activity data is the area affected. A simple descriptor is used to characterize the severity of the event. For both wildfire and prescribed fire/control burns, descriptors of severity include crown fire, stand-replacement underburn, mixed-severity underburn (some tree mortality), and low-severity underburn. Typically wildfire will be more weighted towards crown fire and higher severity versus lower severity from prescribed fire. For other disturbances, the percentage of live trees killed (or percentage basal area mortality) and the percentage of killed trees that are still standing as was covered previously in Section 6.4.2.10 and Section 6.4.4.8 are used.

6.7.3 Estimation Methods

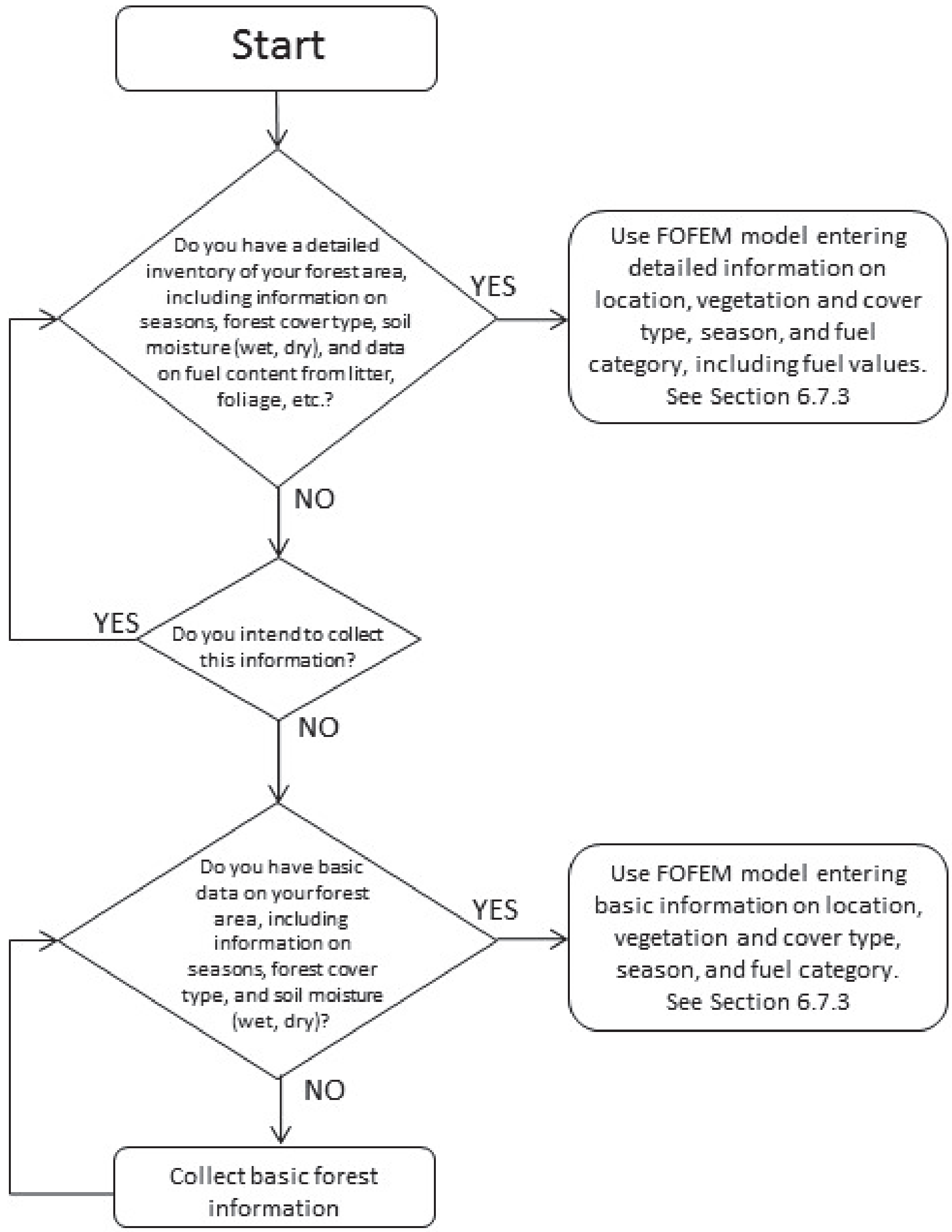
FOFEM[9] (Reinhardt et al., 1997) is recommended for estimating GHG emissions, because it is applicable nationally, computer code is available that can be linked to or incorporated into other

[9] http://www.firelab.org/science-applications/fire-fuel/111-fofem

code, and inputs are defined so that measured biomass can be entered or default values generated by vegetation type. FOFEM produces direct estimates of total CO_2, CO, CH_4, and NO_x emitted, as well as estimates of fuel consumption by component, which can be used to determine residual fuel quantities for estimating subsequent decomposition. FOFEM and/or CONSUME (Joint Fire Science Program, 2009) can also be used directly to compute emissions and consumption from fire. FOFEM algorithms can also be used to compute tree mortality in order to update estimates of live and dead biomass. Although another option is to use FVS-FFE[10] (Rebain, 2010; Reinhardt and Crookston, 2003), it is not the recommended approach for wildfire GHG calculation. FVS-FFE uses many of the same internal algorithms for estimating tree mortality, fuel consumption, and emissions as FOFEM, but also simulates stand, fuel, and carbon dynamics over time. It is a more powerful predictive tool, but substantially more work is involved in understanding the modeling framework, setting up model runs and data preparation. Alternatively, lookup tables can be built using these tools for a range of vegetation types, fuel loadings from natural and/or management processes, and fire severities, or a simplified algorithm can be developed as in the 2006 IPCC Guidelines for National GHG Inventories (IPCC, 2006).

[10] http://www.fs.fed.us/fmsc/fvs/whatis/index.shtml

Figure 6-9: Decision Tree for Natural Disturbances Showing Methods Appropriate for Estimating Emissions from Forest Fires Depending on the Data Available



6.7.3.1 Estimation of Greenhouse Gas Emissions from Fire

The calculation of GHG emissions from fires can be seen in Equation 6-11 below.

Equation 6-11: Calculate GHG Emissions from Fire

$$L_{fire} = A \times MB \times C_f \times G_{ef} \times 10^{-3}$$

Where:

L_{fire} = Amount of greenhouse gas emissions from fire (metric tons of each GHG, i.e., CH_4, N_2O, etc.)

A = Area burned (ha)

MB = Mass of fuel available for combustion (metric tons ha^{-1})

C_f = Combustion factor (dimensionless)

G_{ef} = Emission factor (g (kg dry matter burnt)$^{-1}$)

In order to use this algorithm, an estimate of A by fire severity is used. For MB, the understory, DDW, and forest floor are assumed to be available for combustion. In addition, an estimate of what portion of the live tree biomass is available for combustion (typically only foliage and fine branchwood) is used. For C_f, IPCC (2006) protocols use 0.45 for temperate forests. Separate values for C_f for biomass pools for crown fire, stand-replacement underburn, mixed-severity underburn, and low-severity underburn, by forest type, using FOFEM are provided (see Table 6-13). For G_{ef} emission factors from Urbanski et al. (2009) are recommended: 1619 g (kg dry matter burnt for CO_2)$^{-1}$, 89.6 g (kg dry matter burnt for CO)$^{-1}$, 3.4 g (kg dry matter burnt for CH_4)$^{-1}$, and from Akagi et al. (2011), 2.5 g (kg dry matter burnt for NO_x)$^{-1}$. Note that not all biomass is available for combustion; in particular, standing live tree boles are not available.

For subsequent effects, the GHG estimation methods adopted should match as closely as possible those used in other sections (e.g., HWPs). Decomposition of dead material over time will be projected using a fixed annual loss rate. The conversion of standing dead to dead-and-down should also be projected using a fixed rate and approximating the methods in FVS-FFE.

GHG emissions from natural disturbance wildfires and prescribed fires used for site maintenance and restoration should be reported separately from emissions resulting from management (sites with thinning slash, machine or hand piles, or logging slash) to facilitate the use of the estimates in decision making regarding management practices.

Table 6-13 shows an example for the default lookup tables for consumption fraction (C_f). Regions are those shown in Table 6-13, with the exception of the West region, which represents an average of all western regions.

Table 6-13: C_f Consumption Fraction

Region	Forest Type	C_f Crown Fire	C_f Stand Replacement Underburn	C_f Mixed Severity	C_f Low Severity Underburn
		%			
Northeast	Aspen–birch	84	69	59	45
	Elm–ash–cottonwood	74	47	35	20
	Maple–beech–birch	77	60	44	35
	Oak–hickory	63	49	41	32
	Oak–pine	80	61	50	38
	Spruce–fir	73	73	69	62
	White–red–jack pine	55	45	37	26

Region	Forest Type	C_f Crown Fire	C_f Stand Replacement Underburn	C_f Mixed Severity	C_f Low Severity Underburn
		%			
Northern Lake States	Aspen–birch	84	69	59	45
	Elm–ash–cottonwood	74	47	35	20
	Maple–beech–birch	77	60	44	35
	Oak–hickory	80	61	50	38
	Spruce–fir	73	73	69	62
	White–red–jack pine	55	45	37	26
Northern Prairie States	Elm–ash–cottonwood	74	47	35	20
	Maple–beech–birch	77	60	44	35
	Oak–hickory	80	61	50	38
	Ponderosa pine	60	53	47	37
Pacific Northwest, East	Douglas fir	85	79	72	60
	Fir–spruce–m.hemlock	67	64	58	44
	Lodgepole pine	77	72	64	52
	Ponderosa pine	78	53	41	27
Pacific Northwest, West	Alder–maple	82	67	48	42
	Douglas fir	71	62	55	43
	Fir–spruce–m.hemlock	67	64	58	44
	Hemlock–Sitka spruce	85	77	69	55
Pacific Southwest	Mixed conifer	79	69	50	46
	Douglas fir	66	42	30	17
	Fir–spruce–m.hemlock	67	64	58	44
	Ponderosa Pine	78	53	41	27
	Redwood	82	76	69	56
Rocky Mountain, North and South	Aspen–birch	80	61	50	35
	Douglas fir	85	79	72	60
	Fir–spruce–m.hemlock	67	64	58	44
	Lodgepole pine	77	72	64	52
	Ponderosa pine	78	53	41	27
	Mixed conifer	79	69	50	46
Southeast	Elm–ash–cottonwood	76	45	29	19
	Loblolly–shortleaf pine	66	52	44	35
	Oak–hickory	61	50	44	36
	Oak–pine	62	55	51	45
South Central	Elm–ash–cottonwood	76	45	29	19
	Loblolly–shortleaf pine	66	52	44	35
	Longleaf–slash pine	69	63	57	47
	Oak–hickory	61	50	44	36
	Oak–pine	62	55	51	45
West[a]	Pinyon–juniper	64	55	49	41
	Tanoak–laurel	70	52	43	32
	Western larch	76	68	60	47
	Western oak	65	62	56	48
	Western white pine	68	56	47	33

[a] Represents an average over all western regions for the specified forest types (PNW-W, PNW-E, PSW, RMN, RMS).

6.7.3.2 Estimation of Greenhouse Gas Emissions from Other Disturbances

For other disturbances, the primary effects are indirect: by converting live biomass to dead—and in some cases standing trees to dead, down trees—decomposition is accelerated. Currently grouping non-fire disturbance into two categories is suggested: disturbances that leave dead trees standing (insect and disease-caused mortality) and disturbance that leaves the trees on the ground (wind or ice storms).The landowner will have to estimate mortality (Section 6.7.2); then as in decomposition of fire-killed trees, a fixed decomposition rate (default value 0.015) will be used to simulate subsequent decomposition.

For insect or pathogen-caused mortality, the trees are assumed to be initially standing after death. Conversion of standing dead to dead-and-down will be projected using a fixed rate and approximating the methods in FVS-FEE. Once down, the default decomposition rate from FVS-FFE of 0.015 for dead and down wood will be used to simulate decomposition. For blowdowns or ice storms, the impacted trees are assumed to be dead and down. In this case decomposition begins immediately.

6.7.4 Limitations and Uncertainty

A major source of uncertainty in predicting fire emissions is the preburn fuel quantities. If landowners are doing some kind of inventory of live and dead biomass (see Section 6.7.2) they will have relatively robust estimates of available fuel. If they are using lookup table values by forest type, there will be more uncertainty associated with the estimates since fuel quantities vary greatly within forest type.

A related challenge is determining the appropriate degree of specificity for tracking biomass by pools (e.g., live, dead). Any kind of management or disturbance changes biomass at the time of occurrence, and also the subsequent trajectory. Subsequent management or disturbance should be applied to the changed and changing values, not the original values. This can result in a complicated simulation model like FVS, rather than a calculator. Since prefire fuel quantity is the strongest predictor of fuel consumption, determining the appropriate degree of specificity for tracking biomass by pools is not a completely academic question.

Appendix 6-A: Harvested Wood Products Lookup Tables

Table 6-A-1: Factors to Convert Primary Wood Products to Carbon Mass from the Units Characteristic of Each Product

Solidwood Product or Paper	Unit	Factor to Convert Units to Tons (2,000 lbs) C	Factor to Convert Units to Metric Tons C
Softwood lumber/laminated veneer lumber/glulam lumber/I-joists	Thousand board feet	0.488	0.443
Hardwood lumber	Thousand board feet	0.844	0.765
Softwood plywood	Thousand square feet, 3/8-inch basis	0.260	0.236
Oriented strandboard	Thousand square feet, 3/8-inch basis	0.303	0.275
Non-structural panels (average)	Thousand square feet, 3/8-inch basis	0.319	0.289
Hardwood veneer/plywood	Thousand square feet, 3/8-inch basis	0.315	0.286
Particleboard/medium density fiberboard	Thousand square feet, 3/4-inch basis	0.647	0.587
Hardboard	Thousand square feet,1/8-inch basis	0.152	0.138
Insulation board	Thousand square feet, 1/2-inch basis	0.242	0.220
Other industrial products	Thousand cubic feet	8.250	7.484
Paper	Tons, air dry	0.450	0.496

Table 6-A-2: Fraction of Carbon in Primary Wood Products Remaining in End Uses up to 100 Years After Production (year 0 indicates fraction at time of production)

Year after Production	Softwood Lumber	Hardwood Lumber	Softwood Plywood	Oriented Strandboard	Non-Structural Panels	Misc. Products	Paper
0	1.000	1.000	1.000	1.000	1.000	1.000	1.000
1	0.908	0.909	0.908	0.908	0.908	0.903	0.880
2	0.892	0.893	0.893	0.896	0.892	0.887	0.775
3	0.877	0.877	0.878	0.884	0.876	0.871	0.682
4	0.863	0.861	0.863	0.872	0.861	0.855	0.600
5	0.848	0.845	0.848	0.860	0.845	0.840	0.528
6	0.834	0.830	0.834	0.848	0.830	0.825	0.465
7	0.820	0.815	0.820	0.837	0.816	0.810	0.354
8	0.806	0.801	0.807	0.826	0.801	0.795	0.269
9	0.793	0.786	0.794	0.815	0.787	0.781	0.205
10	0.780	0.772	0.781	0.804	0.774	0.767	0.156
15	0.718	0.705	0.719	0.753	0.708	0.700	0.040
20	0.662	0.644	0.663	0.706	0.649	0.639	0.010
25	0.611	0.589	0.613	0.662	0.595	0.583	0.003
30	0.565	0.538	0.567	0.622	0.546	0.532	0.001
35	0.523	0.492	0.525	0.585	0.501	0.486	0.000
40	0.485	0.450	0.487	0.551	0.460	0.444	0.000
45	0.450	0.411	0.452	0.519	0.423	0.405	0.000
50	0.418	0.376	0.420	0.490	0.389	0.370	0.000
55	0.389	0.344	0.391	0.462	0.358	0.338	0.000
60	0.362	0.315	0.364	0.437	0.329	0.308	0.000
65	0.338	0.288	0.340	0.413	0.303	0.281	0.000
70	0.315	0.264	0.317	0.391	0.280	0.257	0.000
75	0.294	0.242	0.296	0.370	0.258	0.234	0.000
80	0.276	0.221	0.277	0.351	0.238	0.214	0.000
85	0.258	0.203	0.260	0.333	0.220	0.195	0.000
90	0.242	0.186	0.244	0.316	0.203	0.178	0.000
95	0.227	0.170	0.229	0.300	0.188	0.163	0.000
100	0.213	0.156	0.215	0.285	0.174	0.149	0.000
Average	**0.466**	**0.430**	**0.468**	**0.526**	**0.441**	**0.424**	**0.059**

Table 6-A-3: Fraction of Carbon in Primary Wood Products Remaining in Landfills up to 100 Years after Production (year 0 indicates fraction at time of production)

Year after Production	Softwood Lumber	Hardwood Lumber	Softwood Plywood	Oriented Strandboard	Non-Structural Panels	Misc. Products	Paper
0	0.000	0.000	0.000	0.000	0.000	0.000	0.000
1	0.061	0.060	0.061	0.061	0.061	0.064	0.040
2	0.071	0.070	0.071	0.068	0.071	0.074	0.073
3	0.080	0.080	0.080	0.076	0.081	0.084	0.102
4	0.089	0.090	0.089	0.083	0.090	0.094	0.127
5	0.098	0.099	0.097	0.090	0.099	0.103	0.147
6	0.106	0.109	0.106	0.097	0.108	0.112	0.164
7	0.114	0.117	0.114	0.103	0.117	0.121	0.197
8	0.122	0.126	0.122	0.110	0.125	0.129	0.220
9	0.130	0.134	0.130	0.116	0.134	0.138	0.236
10	0.138	0.143	0.137	0.122	0.142	0.146	0.247
15	0.173	0.181	0.172	0.151	0.179	0.184	0.256
20	0.203	0.214	0.202	0.176	0.211	0.217	0.241
25	0.230	0.243	0.229	0.199	0.239	0.246	0.223
30	0.253	0.269	0.252	0.220	0.265	0.272	0.207
35	0.274	0.292	0.273	0.238	0.287	0.296	0.195
40	0.293	0.313	0.292	0.255	0.307	0.316	0.185
45	0.310	0.332	0.308	0.271	0.325	0.335	0.177
50	0.325	0.348	0.324	0.285	0.341	0.352	0.171
55	0.338	0.363	0.337	0.298	0.356	0.367	0.166
60	0.351	0.377	0.349	0.310	0.369	0.380	0.163
65	0.362	0.389	0.361	0.321	0.381	0.393	0.160
70	0.372	0.400	0.371	0.331	0.391	0.404	0.158
75	0.381	0.410	0.380	0.341	0.401	0.414	0.156
80	0.390	0.419	0.389	0.350	0.410	0.423	0.154
85	0.398	0.427	0.397	0.359	0.418	0.431	0.153
90	0.405	0.435	0.404	0.366	0.426	0.439	0.153
95	0.412	0.442	0.411	0.374	0.432	0.446	0.152
100	0.418	0.448	0.417	0.381	0.438	0.452	0.151
Average	**0.297**	**0.317**	**0.296**	**0.264**	**0.311**	**0.321**	**0.178**

Table 6-A-4: Density of Softwood and Hardwood Sawlogs/Veneer Logs and Pulpwood by Region and Forest Type Group[a]

Region	Forest type	Specific Gravity[d] of Softwoods	Specific Gravity[d] of Hardwoods
Northeast	Aspen–birch	0.353	0.428
	Elm–ash–cottonwood	0.358	0.470
	Maple–beech–birch	0.369	0.518
	Oak–hickory	0.388	0.534
	Oak–pine	0.371	0.516
	Spruce–fir	0.353	0.481
	White–red–jack pine	0.361	0.510
Northern Lake States	Aspen–birch	0.351	0.397
	Elm–ash–cottonwood	0.335	0.460
	Maple–beech–birch	0.356	0.496
	Oak–hickory	0.369	0.534
	Spruce–fir	0.344	0.444
	White–red–jack pine	0.389	0.473
Northern Prairie States	Elm–ash–cottonwood	0.424	0.453
	Loblolly–shortleaf pine	0.468	0.544
	Maple–beech–birch	0.437	0.508
	Oak–hickory	0.448	0.565
	Oak–pine	0.451	0.566
	Ponderosa pine	0.381	0.473
Pacific Northwest, East	Douglas fir	0.429	0.391
	Fir–spruce–m.hemlock	0.370	0.361
	Lodgepole pine	0.380	0.345
	Ponderosa pine	0.385	0.513
Pacific Northwest, West	Alder–maple	0.402	0.385
	Douglas fir	0.440	0.426
	Fir–spruce–m.hemlock	0.399	0.417
	Hemlock–Sitka spruce	0.405	0.380
Pacific Southwest	Mixed conifer	0.394	0.521
	Douglas fir	0.429	0.483
	Fir–spruce–m.hemlock	0.372	0.510
	Ponderosa Pine	0.380	0.510
	Redwood	0.376	0.449
Rocky Mountain, North	Douglas fir	0.428	0.370
	Fir–spruce–m.hemlock	0.355	0.457
	Hemlock–sitka spruce	0.375	0.441
	Lodgepole pine	0.383	0.391
	Ponderosa pine	0.391	0.374
Rocky Mountain, South	Aspen–birch	0.355	0.350
	Douglas fir	0.431	0.350
	Fir–spruce–m.hemlock	0.342	0.350
	Lodgepole pine	0.377	0.350
	Ponderosa pine	0.383	0.386

Region	Forest type	Specific Gravity[d] of Softwoods	Specific Gravity[d] of Hardwoods
Southeast	Elm–ash–cottonwood	0.433	0.499
	Loblolly–shortleaf pine	0.469	0.494
	Longleaf–slash pine	0.536	0.503
	Oak–gum–cypress	0.441	0.484
	Oak–hickory	0.438	0.524
	Oak–pine	0.462	0.516
South Central	Elm–ash–cottonwood	0.427	0.494
	Loblolly–shortleaf pine	0.470	0.516
	Longleaf–slash pine	0.531	0.504
	Oak–gum–cypress	0.440	0.513
	Oak–hickory	0.451	0.544
	Oak–pine	0.467	0.537
West[e]	Pinyon–juniper	0.422	0.620
	Tanoak–laurel	0.430	0.459
	Western larch	0.433	0.430
	Western oak	0.416	0.590
	Western white pine	0.376	--

-- = No hardwood trees in this type in this region.
[a] Estimates based on survey data for the conterminous United States from USDA Forest Service, FIA Program's database of forest surveys (FIADB) (USDA Forest Service, 2005) and include growing stock on timberland stands classified as medium- or large-diameter stands. Proportions are based on volume of growing stock trees.
[d] Average wood specific gravity is the density of wood divided by the density of water based on wood dry mass associated with green tree volume.
[e] West represents an average over all western regions for these forest types.

Table 6-A-5: Average Disposition Patterns of Carbon as Fractions in Roundwood by Region and Roundwood Category; Factors Assume No Bark on Roundwood and Exclude Fuelwood

Year after Production	Northeast, Softwood								
	Sawlog					Pulpwood			
	In Use	In Landfills	Total Stored	Total Emissions		In Use	In Landfills	Total Stored	Total Emissions
0	0.569	0.000	0.569	0.431		0.513	0.000	0.513	0.487
1	0.521	0.029	0.550	0.450		0.452	0.021	0.473	0.527
2	0.505	0.037	0.542	0.458		0.400	0.038	0.438	0.562
3	0.491	0.044	0.535	0.465		0.355	0.052	0.407	0.593
4	0.478	0.050	0.528	0.472		0.315	0.064	0.379	0.621
5	0.465	0.056	0.522	0.478		0.279	0.074	0.354	0.646
6	0.453	0.062	0.516	0.484		0.248	0.083	0.331	0.669
7	0.438	0.069	0.507	0.493		0.193	0.099	0.293	0.707
8	0.425	0.075	0.500	0.500		0.152	0.111	0.263	0.737
9	0.414	0.080	0.494	0.506		0.120	0.119	0.239	0.761
10	0.403	0.085	0.489	0.511		0.096	0.124	0.220	0.780
15	0.363	0.105	0.468	0.532		0.038	0.130	0.168	0.832
20	0.332	0.121	0.453	0.547		0.022	0.124	0.146	0.854
25	0.306	0.134	0.440	0.560		0.017	0.116	0.133	0.867
30	0.282	0.146	0.428	0.572		0.015	0.109	0.124	0.876
35	0.260	0.156	0.417	0.583		0.014	0.103	0.117	0.883
40	0.240	0.166	0.406	0.594		0.013	0.099	0.111	0.889
45	0.222	0.174	0.397	0.603		0.012	0.095	0.107	0.893
50	0.206	0.182	0.388	0.612		0.011	0.093	0.104	0.896
55	0.191	0.189	0.380	0.620		0.010	0.091	0.101	0.899
60	0.177	0.195	0.372	0.628		0.009	0.089	0.099	0.901
65	0.165	0.201	0.365	0.635		0.009	0.088	0.097	0.903
70	0.153	0.206	0.359	0.641		0.008	0.087	0.095	0.905
75	0.143	0.210	0.353	0.647		0.008	0.086	0.094	0.906
80	0.133	0.214	0.347	0.653		0.007	0.086	0.093	0.907
85	0.124	0.218	0.342	0.658		0.007	0.085	0.092	0.908
90	0.116	0.221	0.337	0.663		0.006	0.085	0.091	0.909
95	0.108	0.224	0.332	0.668		0.006	0.085	0.091	0.909
100	0.101	0.227	0.328	0.672		0.006	0.085	0.090	0.910
Average	**0.235**	**0.166**	**0.402**			**0.041**	**0.095**	**0.136**	

Table 6-A-5—continued

Year after Production	Northeast, Hardwood								
	Sawlog					Pulpwood			
	In Use	In Landfills	Total Stored	Total Emissions		In Use	In Landfills	Total Stored	Total Emissions
0	0.614	0.000	0.614	0.386		0.650	0.000	0.650	0.350
1	0.559	0.034	0.594	0.406		0.580	0.032	0.613	0.387
2	0.544	0.042	0.586	0.414		0.540	0.046	0.586	0.414
3	0.530	0.049	0.579	0.421		0.503	0.059	0.562	0.438
4	0.516	0.056	0.573	0.427		0.471	0.070	0.541	0.459
5	0.504	0.063	0.567	0.433		0.443	0.079	0.522	0.478
6	0.491	0.069	0.561	0.439		0.417	0.087	0.504	0.496
7	0.477	0.076	0.553	0.447		0.374	0.101	0.475	0.525
8	0.463	0.083	0.546	0.454		0.341	0.111	0.453	0.547
9	0.452	0.089	0.540	0.460		0.316	0.119	0.434	0.566
10	0.441	0.094	0.535	0.465		0.295	0.125	0.420	0.580
15	0.397	0.117	0.514	0.486		0.239	0.137	0.376	0.624
20	0.361	0.136	0.497	0.503		0.215	0.140	0.355	0.645
25	0.330	0.152	0.482	0.518		0.199	0.141	0.340	0.660
30	0.301	0.167	0.468	0.532		0.186	0.142	0.328	0.672
35	0.275	0.180	0.455	0.545		0.175	0.144	0.319	0.681
40	0.252	0.192	0.444	0.556		0.164	0.146	0.310	0.690
45	0.230	0.202	0.432	0.568		0.155	0.148	0.302	0.698
50	0.211	0.211	0.422	0.578		0.146	0.150	0.296	0.704
55	0.193	0.220	0.412	0.588		0.138	0.152	0.290	0.710
60	0.176	0.227	0.403	0.597		0.130	0.154	0.285	0.715
65	0.162	0.234	0.395	0.605		0.123	0.157	0.280	0.720
70	0.148	0.240	0.388	0.612		0.116	0.159	0.275	0.725
75	0.136	0.245	0.380	0.620		0.110	0.161	0.271	0.729
80	0.124	0.250	0.374	0.626		0.104	0.163	0.268	0.732
85	0.114	0.254	0.368	0.632		0.099	0.165	0.264	0.736
90	0.104	0.258	0.362	0.638		0.094	0.167	0.261	0.739
95	0.096	0.261	0.357	0.643		0.089	0.169	0.258	0.742
100	0.088	0.264	0.352	0.648		0.085	0.171	0.255	0.745
Average	**0.244**	**0.192**	**0.437**			**0.178**	**0.145**	**0.323**	

Table 6-A-5—continued

Year after Production	North Central, Softwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.630	0.000	0.630	0.370		0.514	0.000	0.514	0.486
1	0.579	0.031	0.610	0.390		0.454	0.021	0.475	0.525
2	0.561	0.039	0.601	0.399		0.402	0.038	0.440	0.560
3	0.545	0.047	0.592	0.408		0.357	0.052	0.409	0.591
4	0.530	0.055	0.585	0.415		0.317	0.064	0.381	0.619
5	0.516	0.062	0.577	0.423		0.281	0.074	0.356	0.644
6	0.502	0.068	0.570	0.430		0.250	0.083	0.333	0.667
7	0.485	0.076	0.561	0.439		0.196	0.099	0.295	0.705
8	0.470	0.083	0.553	0.447		0.154	0.111	0.265	0.735
9	0.457	0.089	0.546	0.454		0.123	0.119	0.241	0.759
10	0.446	0.094	0.540	0.460		0.098	0.124	0.223	0.777
15	0.401	0.116	0.517	0.483		0.041	0.130	0.171	0.829
20	0.366	0.133	0.500	0.500		0.025	0.124	0.148	0.852
25	0.336	0.148	0.485	0.515		0.020	0.116	0.135	0.865
30	0.310	0.162	0.471	0.529		0.018	0.109	0.126	0.874
35	0.286	0.173	0.459	0.541		0.016	0.103	0.120	0.880
40	0.264	0.184	0.447	0.553		0.015	0.099	0.114	0.886
45	0.243	0.193	0.437	0.563		0.014	0.096	0.110	0.890
50	0.225	0.202	0.427	0.573		0.013	0.093	0.106	0.894
55	0.208	0.209	0.418	0.582		0.012	0.091	0.103	0.897
60	0.193	0.216	0.409	0.591		0.012	0.089	0.101	0.899
65	0.179	0.222	0.401	0.599		0.011	0.088	0.099	0.901
70	0.166	0.228	0.394	0.606		0.010	0.087	0.098	0.902
75	0.154	0.233	0.387	0.613		0.010	0.087	0.097	0.903
80	0.144	0.237	0.381	0.619		0.009	0.086	0.095	0.905
85	0.134	0.242	0.375	0.625		0.009	0.086	0.095	0.905
90	0.125	0.245	0.370	0.630		0.008	0.086	0.094	0.906
95	0.116	0.249	0.365	0.635		0.008	0.086	0.093	0.907
100	0.108	0.252	0.360	0.640		0.007	0.086	0.093	0.907
Average	**0.258**	**0.184**	**0.442**			**0.043**	**0.095**	**0.138**	

Table 6-A-5—continued

Year after Production	North Central, Hardwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.585	0.000	0.585	0.415		0.685	0.000	0.685	0.315
1	0.533	0.032	0.565	0.435		0.613	0.035	0.648	0.352
2	0.518	0.040	0.558	0.442		0.575	0.049	0.624	0.376
3	0.504	0.047	0.550	0.450		0.541	0.061	0.602	0.398
4	0.490	0.054	0.544	0.456		0.511	0.071	0.582	0.418
5	0.477	0.060	0.537	0.463		0.484	0.080	0.565	0.435
6	0.465	0.066	0.531	0.469		0.460	0.089	0.548	0.452
7	0.450	0.073	0.523	0.477		0.421	0.101	0.522	0.478
8	0.437	0.080	0.517	0.483		0.390	0.111	0.501	0.499
9	0.425	0.085	0.511	0.489		0.365	0.119	0.484	0.516
10	0.415	0.090	0.505	0.495		0.346	0.125	0.471	0.529
15	0.372	0.112	0.484	0.516		0.290	0.139	0.429	0.571
20	0.339	0.130	0.468	0.532		0.263	0.144	0.408	0.592
25	0.309	0.145	0.454	0.546		0.245	0.148	0.393	0.607
30	0.282	0.158	0.441	0.559		0.229	0.151	0.380	0.620
35	0.258	0.170	0.428	0.572		0.216	0.154	0.370	0.630
40	0.236	0.181	0.417	0.583		0.203	0.158	0.360	0.640
45	0.216	0.191	0.407	0.593		0.191	0.161	0.352	0.648
50	0.197	0.199	0.397	0.603		0.180	0.165	0.345	0.655
55	0.181	0.207	0.388	0.612		0.170	0.168	0.338	0.662
60	0.165	0.214	0.379	0.621		0.160	0.171	0.332	0.668
65	0.151	0.220	0.372	0.628		0.152	0.174	0.326	0.674
70	0.138	0.226	0.364	0.636		0.143	0.178	0.321	0.679
75	0.127	0.231	0.358	0.642		0.136	0.180	0.316	0.684
80	0.116	0.235	0.351	0.649		0.129	0.183	0.312	0.688
85	0.106	0.239	0.346	0.654		0.122	0.186	0.308	0.692
90	0.098	0.243	0.340	0.660		0.116	0.188	0.304	0.696
95	0.089	0.246	0.336	0.664		0.110	0.191	0.300	0.700
100	0.082	0.249	0.331	0.669		0.104	0.193	0.297	0.703
Average	**0.229**	**0.182**	**0.411**			**0.212**	**0.158**	**0.370**	

Table 6-A-5—continued

Year after Production	Pacific Northwest, East, Softwood			
	In Use	All		Total Emissions
		In Landfills	Total Stored	
0	0.637	0.000	0.637	0.363
1	0.574	0.036	0.610	0.390
2	0.551	0.046	0.597	0.403
3	0.530	0.055	0.585	0.415
4	0.511	0.063	0.574	0.426
5	0.494	0.070	0.564	0.436
6	0.478	0.077	0.555	0.445
7	0.455	0.086	0.541	0.459
8	0.436	0.093	0.529	0.471
9	0.420	0.100	0.520	0.480
10	0.406	0.105	0.512	0.488
15	0.359	0.125	0.484	0.516
20	0.327	0.139	0.466	0.534
25	0.301	0.150	0.451	0.549
30	0.278	0.160	0.438	0.562
35	0.258	0.169	0.427	0.573
40	0.239	0.177	0.416	0.584
45	0.222	0.185	0.406	0.594
50	0.206	0.191	0.397	0.603
55	0.191	0.198	0.389	0.611
60	0.178	0.203	0.381	0.619
65	0.166	0.208	0.374	0.626
70	0.155	0.213	0.368	0.632
75	0.145	0.217	0.362	0.638
80	0.136	0.221	0.356	0.644
85	0.127	0.224	0.351	0.649
90	0.119	0.227	0.347	0.653
95	0.112	0.230	0.342	0.658
100	0.105	0.233	0.338	0.662
Average	**0.238**	**0.177**	**0.415**	

Table 6-A-5—continued

Year after Production	Pacific Northwest, West, Softwoods								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.740	0.000	0.740	0.260		0.500	0.000	0.500	0.500
1	0.674	0.039	0.713	0.287		0.440	0.020	0.460	0.540
2	0.652	0.049	0.702	0.298		0.387	0.037	0.424	0.576
3	0.632	0.059	0.691	0.309		0.341	0.051	0.392	0.608
4	0.613	0.068	0.681	0.319		0.300	0.063	0.364	0.636
5	0.596	0.076	0.672	0.328		0.264	0.074	0.338	0.662
6	0.579	0.083	0.663	0.337		0.233	0.082	0.315	0.685
7	0.558	0.093	0.651	0.349		0.177	0.099	0.276	0.724
8	0.539	0.101	0.640	0.360		0.134	0.111	0.245	0.755
9	0.524	0.108	0.631	0.369		0.102	0.119	0.221	0.779
10	0.510	0.114	0.624	0.376		0.078	0.124	0.202	0.798
15	0.457	0.139	0.596	0.404		0.020	0.129	0.149	0.851
20	0.418	0.158	0.576	0.424		0.005	0.122	0.127	0.873
25	0.384	0.174	0.558	0.442		0.001	0.113	0.114	0.886
30	0.355	0.188	0.543	0.457		0	0.105	0.105	0.895
35	0.328	0.201	0.529	0.471		0	0.098	0.099	0.901
40	0.303	0.213	0.516	0.484		0	0.093	0.093	0.907
45	0.281	0.223	0.504	0.496		0	0.090	0.090	0.910
50	0.260	0.232	0.493	0.507		0	0.086	0.086	0.914
55	0.242	0.241	0.482	0.518		0	0.084	0.084	0.916
60	0.224	0.248	0.473	0.527		0	0.082	0.082	0.918
65	0.209	0.255	0.464	0.536		0	0.080	0.080	0.920
70	0.194	0.261	0.456	0.544		0	0.079	0.079	0.921
75	0.181	0.267	0.448	0.552		0	0.078	0.078	0.922
80	0.169	0.272	0.441	0.559		0	0.078	0.078	0.922
85	0.158	0.276	0.434	0.566		0	0.077	0.077	0.923
90	0.148	0.281	0.428	0.572		0	0.077	0.077	0.923
95	0.138	0.285	0.423	0.577		0	0.076	0.076	0.924
100	0.129	0.288	0.417	0.583		0	0.076	0.076	0.924
Average	**0.298**	**0.213**	**0.511**			**0.030**	**0.090**	**0.119**	

Table 6-A-5—continued

Year after Production	Pacific Northwest, West, Hardwood				Pacific Southwest, Softwood			
	In Use	All		Total Emissions	In Use	All		Total Emissions
		In Landfills	Total Stored			In Landfills	Total Stored	
0	0.531	0.000	0.531	0.469	0.675	0.000	0.675	0.325
1	0.476	0.027	0.503	0.497	0.611	0.036	0.647	0.353
2	0.447	0.038	0.485	0.515	0.587	0.047	0.634	0.366
3	0.421	0.048	0.469	0.531	0.566	0.056	0.622	0.378
4	0.397	0.057	0.454	0.546	0.546	0.065	0.611	0.389
5	0.376	0.064	0.440	0.560	0.528	0.072	0.600	0.400
6	0.357	0.071	0.428	0.572	0.511	0.080	0.591	0.409
7	0.327	0.081	0.408	0.592	0.488	0.089	0.577	0.423
8	0.303	0.089	0.393	0.607	0.468	0.097	0.565	0.435
9	0.284	0.096	0.380	0.620	0.451	0.104	0.555	0.445
10	0.269	0.101	0.369	0.631	0.437	0.110	0.547	0.453
15	0.222	0.115	0.337	0.663	0.387	0.131	0.518	0.482
20	0.197	0.122	0.319	0.681	0.353	0.146	0.499	0.501
25	0.179	0.127	0.306	0.694	0.324	0.159	0.483	0.517
30	0.164	0.132	0.295	0.705	0.299	0.170	0.469	0.531
35	0.150	0.136	0.286	0.714	0.276	0.180	0.457	0.543
40	0.137	0.140	0.278	0.722	0.256	0.190	0.445	0.555
45	0.126	0.144	0.270	0.730	0.237	0.198	0.435	0.565
50	0.115	0.148	0.263	0.737	0.220	0.205	0.425	0.575
55	0.106	0.151	0.257	0.743	0.204	0.212	0.416	0.584
60	0.097	0.155	0.252	0.748	0.189	0.218	0.408	0.592
65	0.089	0.157	0.247	0.753	0.176	0.224	0.400	0.600
70	0.082	0.160	0.242	0.758	0.164	0.229	0.393	0.607
75	0.075	0.163	0.238	0.762	0.153	0.233	0.387	0.613
80	0.069	0.165	0.234	0.766	0.143	0.238	0.381	0.619
85	0.064	0.167	0.231	0.769	0.133	0.241	0.375	0.625
90	0.059	0.169	0.227	0.773	0.125	0.245	0.370	0.630
95	0.054	0.171	0.224	0.776	0.117	0.248	0.365	0.635
100	0.050	0.172	0.222	0.778	0.109	0.251	0.361	0.639
Average	**0.145**	**0.139**	**0.284**		**0.254**	**0.190**	**0.444**	

Table 6-A-5—continued

Year after Production	Rocky Mountain, Softwood			
	In Use	All		Total Emissions
		In Landfills	Total Stored	
0	0.704	0.000	0.704	0.296
1	0.640	0.037	0.677	0.323
2	0.615	0.048	0.663	0.337
3	0.592	0.057	0.650	0.350
4	0.572	0.066	0.638	0.362
5	0.552	0.075	0.627	0.373
6	0.535	0.082	0.617	0.383
7	0.510	0.092	0.602	0.398
8	0.489	0.101	0.590	0.410
9	0.472	0.108	0.579	0.421
10	0.457	0.114	0.571	0.429
15	0.404	0.136	0.540	0.460
20	0.368	0.152	0.520	0.480
25	0.338	0.166	0.504	0.496
30	0.312	0.177	0.489	0.511
35	0.288	0.188	0.476	0.524
40	0.266	0.198	0.464	0.536
45	0.247	0.206	0.453	0.547
50	0.229	0.214	0.443	0.557
55	0.212	0.221	0.433	0.567
60	0.197	0.228	0.425	0.575
65	0.183	0.234	0.417	0.583
70	0.170	0.239	0.409	0.591
75	0.159	0.244	0.403	0.597
80	0.148	0.248	0.396	0.604
85	0.138	0.252	0.390	0.610
90	0.129	0.256	0.385	0.615
95	0.121	0.259	0.380	0.620
100	0.113	0.262	0.375	0.625
Average	**0.265**	**0.198**	**0.463**	

Table 6-A-5—continued

Year after Production	Southeast, Softwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.636	0.000	0.636	0.364		0.553	0.000	0.553	0.447
1	0.578	0.034	0.612	0.388		0.490	0.024	0.514	0.486
2	0.557	0.043	0.600	0.400		0.442	0.040	0.482	0.518
3	0.537	0.052	0.589	0.411		0.399	0.054	0.453	0.547
4	0.519	0.060	0.578	0.422		0.361	0.066	0.427	0.573
5	0.502	0.067	0.569	0.431		0.328	0.076	0.403	0.597
6	0.486	0.074	0.560	0.440		0.298	0.084	0.382	0.618
7	0.465	0.083	0.547	0.453		0.247	0.100	0.347	0.653
8	0.447	0.090	0.537	0.463		0.208	0.111	0.319	0.681
9	0.432	0.096	0.528	0.472		0.178	0.119	0.297	0.703
10	0.418	0.102	0.520	0.480		0.155	0.124	0.279	0.721
15	0.371	0.122	0.494	0.506		0.098	0.132	0.230	0.770
20	0.339	0.137	0.476	0.524		0.079	0.128	0.208	0.792
25	0.311	0.150	0.461	0.539		0.071	0.123	0.194	0.806
30	0.287	0.161	0.448	0.552		0.066	0.118	0.184	0.816
35	0.265	0.171	0.436	0.564		0.062	0.115	0.177	0.823
40	0.245	0.180	0.425	0.575		0.058	0.112	0.170	0.830
45	0.227	0.188	0.415	0.585		0.055	0.110	0.165	0.835
50	0.210	0.195	0.405	0.595		0.052	0.109	0.161	0.839
55	0.195	0.202	0.397	0.603		0.049	0.108	0.157	0.843
60	0.181	0.208	0.389	0.611		0.046	0.108	0.154	0.846
65	0.169	0.213	0.382	0.618		0.044	0.108	0.151	0.849
70	0.157	0.218	0.375	0.625		0.041	0.108	0.149	0.851
75	0.146	0.222	0.369	0.631		0.039	0.108	0.147	0.853
80	0.137	0.226	0.363	0.637		0.037	0.108	0.145	0.855
85	0.127	0.230	0.358	0.642		0.035	0.108	0.143	0.857
90	0.119	0.233	0.353	0.647		0.033	0.109	0.142	0.858
95	0.111	0.236	0.348	0.652		0.031	0.109	0.141	0.859
100	0.104	0.239	0.344	0.656		0.030	0.110	0.140	0.860
Average	**0.243**	**0.180**	**0.423**			**0.082**	**0.109**	**0.191**	

Table 6-A-5—continued

Year after Production	Southeast, Hardwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.609	0.000	0.609	0.391		0.591	0.000	0.591	0.409
1	0.552	0.035	0.587	0.413		0.525	0.027	0.552	0.448
2	0.534	0.043	0.577	0.423		0.480	0.043	0.522	0.478
3	0.518	0.051	0.569	0.431		0.439	0.056	0.495	0.505
4	0.503	0.058	0.561	0.439		0.404	0.067	0.471	0.529
5	0.488	0.065	0.553	0.447		0.372	0.077	0.449	0.551
6	0.475	0.071	0.546	0.454		0.344	0.085	0.430	0.570
7	0.457	0.079	0.537	0.463		0.296	0.100	0.397	0.603
8	0.442	0.086	0.528	0.472		0.260	0.111	0.371	0.629
9	0.429	0.092	0.521	0.479		0.231	0.119	0.350	0.650
10	0.418	0.097	0.515	0.485		0.209	0.124	0.334	0.666
15	0.373	0.119	0.492	0.508		0.153	0.134	0.287	0.713
20	0.338	0.136	0.475	0.525		0.132	0.133	0.265	0.735
25	0.309	0.151	0.460	0.540		0.121	0.130	0.251	0.749
30	0.282	0.164	0.446	0.554		0.113	0.127	0.240	0.760
35	0.258	0.176	0.434	0.566		0.106	0.126	0.232	0.768
40	0.236	0.186	0.422	0.578		0.100	0.125	0.225	0.775
45	0.216	0.196	0.412	0.588		0.094	0.125	0.218	0.782
50	0.198	0.204	0.402	0.598		0.089	0.125	0.213	0.787
55	0.181	0.212	0.393	0.607		0.084	0.125	0.209	0.791
60	0.166	0.218	0.384	0.616		0.079	0.126	0.205	0.795
65	0.152	0.224	0.376	0.624		0.075	0.126	0.201	0.799
70	0.139	0.230	0.369	0.631		0.071	0.127	0.198	0.802
75	0.127	0.235	0.362	0.638		0.067	0.128	0.195	0.805
80	0.117	0.239	0.356	0.644		0.063	0.129	0.193	0.807
85	0.107	0.243	0.350	0.650		0.060	0.130	0.190	0.810
90	0.098	0.247	0.345	0.655		0.057	0.131	0.188	0.812
95	0.090	0.250	0.340	0.660		0.054	0.132	0.186	0.814
100	0.083	0.253	0.336	0.664		0.051	0.133	0.185	0.815
Average	**0.231**	**0.187**	**0.417**			**0.119**	**0.123**	**0.242**	

Table 6-A-5—continued

Year after Production	South Central, Softwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.629	0.000	0.629	0.371		0.570	0.000	0.570	0.430
1	0.569	0.035	0.603	0.397		0.506	0.026	0.532	0.468
2	0.547	0.044	0.591	0.409		0.459	0.041	0.500	0.500
3	0.527	0.053	0.580	0.420		0.417	0.055	0.472	0.528
4	0.509	0.061	0.569	0.431		0.380	0.066	0.447	0.553
5	0.492	0.068	0.560	0.440		0.348	0.076	0.424	0.576
6	0.477	0.075	0.551	0.449		0.319	0.085	0.404	0.596
7	0.455	0.083	0.538	0.462		0.270	0.100	0.370	0.630
8	0.437	0.091	0.527	0.473		0.232	0.111	0.343	0.657
9	0.421	0.097	0.518	0.482		0.202	0.119	0.321	0.679
10	0.408	0.102	0.510	0.490		0.180	0.124	0.304	0.696
15	0.362	0.122	0.484	0.516		0.123	0.133	0.256	0.744
20	0.330	0.136	0.466	0.534		0.103	0.130	0.234	0.766
25	0.303	0.148	0.451	0.549		0.094	0.126	0.220	0.780
30	0.280	0.158	0.439	0.561		0.087	0.122	0.210	0.790
35	0.259	0.168	0.427	0.573		0.082	0.120	0.202	0.798
40	0.240	0.176	0.416	0.584		0.077	0.118	0.195	0.805
45	0.222	0.184	0.406	0.594		0.072	0.117	0.189	0.811
50	0.206	0.191	0.397	0.603		0.068	0.116	0.185	0.815
55	0.192	0.197	0.389	0.611		0.064	0.116	0.181	0.819
60	0.178	0.203	0.381	0.619		0.061	0.116	0.177	0.823
65	0.166	0.208	0.374	0.626		0.058	0.116	0.174	0.826
70	0.155	0.213	0.368	0.632		0.054	0.117	0.171	0.829
75	0.145	0.217	0.362	0.638		0.051	0.117	0.169	0.831
80	0.135	0.221	0.356	0.644		0.049	0.118	0.167	0.833
85	0.126	0.225	0.351	0.649		0.046	0.119	0.165	0.835
90	0.118	0.228	0.346	0.654		0.044	0.119	0.163	0.837
95	0.111	0.231	0.342	0.658		0.042	0.120	0.161	0.839
100	0.104	0.234	0.338	0.662		0.039	0.121	0.160	0.840
Average	**0.239**	**0.176**	**0.415**			**0.099**	**0.116**	**0.215**	

Table 6-A-5—continued

Year after Production	South Central, Hardwood								
	In Use	Sawlog		Total Emissions		In Use	Pulpwood		Total Emissions
		In Landfills	Total Stored				In Landfills	Total Stored	
0	0.587	0.000	0.587	0.413		0.581	0.000	0.581	0.419
1	0.531	0.033	0.564	0.436		0.516	0.027	0.542	0.458
2	0.512	0.042	0.554	0.446		0.470	0.042	0.512	0.488
3	0.495	0.050	0.545	0.455		0.429	0.055	0.484	0.516
4	0.479	0.057	0.536	0.464		0.392	0.067	0.459	0.541
5	0.464	0.064	0.528	0.472		0.360	0.077	0.437	0.563
6	0.450	0.070	0.521	0.479		0.332	0.085	0.417	0.583
7	0.432	0.078	0.510	0.490		0.283	0.100	0.383	0.617
8	0.416	0.085	0.501	0.499		0.246	0.111	0.357	0.643
9	0.403	0.091	0.493	0.507		0.217	0.119	0.336	0.664
10	0.391	0.096	0.487	0.513		0.195	0.124	0.319	0.681
15	0.347	0.116	0.463	0.537		0.138	0.133	0.272	0.728
20	0.314	0.132	0.446	0.554		0.118	0.131	0.250	0.750
25	0.286	0.145	0.432	0.568		0.108	0.128	0.236	0.764
30	0.262	0.157	0.419	0.581		0.101	0.125	0.226	0.774
35	0.239	0.168	0.407	0.593		0.095	0.123	0.217	0.783
40	0.219	0.177	0.396	0.604		0.089	0.121	0.210	0.790
45	0.200	0.186	0.386	0.614		0.084	0.121	0.204	0.796
50	0.183	0.193	0.377	0.623		0.079	0.120	0.199	0.801
55	0.168	0.200	0.368	0.632		0.075	0.121	0.195	0.805
60	0.154	0.206	0.360	0.640		0.070	0.121	0.191	0.809
65	0.141	0.212	0.353	0.647		0.067	0.121	0.188	0.812
70	0.129	0.217	0.346	0.654		0.063	0.122	0.185	0.815
75	0.118	0.222	0.340	0.660		0.060	0.123	0.182	0.818
80	0.108	0.226	0.334	0.666		0.057	0.124	0.180	0.820
85	0.099	0.229	0.329	0.671		0.054	0.124	0.178	0.822
90	0.091	0.233	0.324	0.676		0.051	0.125	0.176	0.824
95	0.084	0.236	0.319	0.681		0.048	0.126	0.174	0.826
100	0.077	0.238	0.315	0.685		0.046	0.127	0.173	0.827
Average	**0.215**	**0.177**	**0.393**			**0.110**	**0.119**	**0.229**	

Table 6-A-5—continued

Year after Production	Other West, Hardwood			
	In Use	All		Total Emissions
		In Landfills	Total Stored	
0	0.568	0.000	0.568	0.432
1	0.516	0.028	0.544	0.456
2	0.494	0.038	0.532	0.468
3	0.473	0.046	0.520	0.480
4	0.455	0.054	0.509	0.491
5	0.438	0.061	0.499	0.501
6	0.422	0.068	0.490	0.510
7	0.399	0.077	0.476	0.524
8	0.381	0.084	0.465	0.535
9	0.365	0.090	0.455	0.545
10	0.352	0.095	0.447	0.553
15	0.307	0.113	0.421	0.579
20	0.277	0.126	0.403	0.597
25	0.253	0.136	0.389	0.611
30	0.232	0.146	0.377	0.623
35	0.212	0.154	0.366	0.634
40	0.195	0.162	0.356	0.644
45	0.179	0.169	0.347	0.653
50	0.164	0.175	0.339	0.661
55	0.151	0.181	0.331	0.669
60	0.138	0.186	0.324	0.676
65	0.127	0.190	0.318	0.682
70	0.117	0.195	0.312	0.688
75	0.108	0.198	0.306	0.694
80	0.099	0.202	0.301	0.699
85	0.091	0.205	0.296	0.704
90	0.084	0.208	0.292	0.708
95	0.078	0.210	0.288	0.712
100	0.072	0.213	0.284	0.716
Average	**0.195**	**0.161**	**0.357**	

Table 6-A-6: Regional Factors to Estimate Carbon in Roundwood Logs, Bark on Logs, and Fuelwood

Region[a]	Timber Type	Roundwood Category	Ratio of Roundwood to Growing-Stock Volume that is Roundwood[b]	Ratio of Carbon in Bark to Carbon in Wood[c]	Fraction of Growing-Stock Volume that is Roundwood[d]	Ratio of Fuelwood to Growing-Stock Volume that is Roundwood[b]
Northeast	SW	Sawlog	0.991	0.182	0.948	0.136
		Pulpwood	3.079	0.185		
	HW	Sawlog	0.927	0.199	0.879	0.547
		Pulpwood	2.177	0.218		
North Central	SW	Sawlog	0.985	0.182	0.931	0.066
		Pulpwood	1.285	0.185		
	HW	Sawlog	0.960	0.199	0.831	0.348
		Pulpwood	1.387	0.218		
Pacific Coast	SW	Sawlog	0.965	0.181	0.929	0.096
		Pulpwood	1.099	0.185		
	HW	Sawlog	0.721	0.197	0.947	0.957
		Pulpwood	0.324	0.219		
Rocky Mountain	SW	Sawlog	0.994	0.181	0.907	0.217
		Pulpwood	2.413	0.185		
	HW	Sawlog	0.832	0.201	0.755	3.165
		Pulpwood	1.336	0.219		
South	SW	Sawlog	0.990	0.182	0.891	0.019
		Pulpwood	1.246	0.185		
	HW	Sawlog	0.832	0.198	0.752	0.301
		Pulpwood	1.191	0.218		

SW=Softwood, HW=Hardwood.
[a] North Central includes the Northern Prairie States and the Northern Lake States; Pacific Coast includes the Pacific Northwest (West and East) and the Pacific Southwest; Rocky Mountain includes Rocky Mountain, North and South; and South includes the Southeast and South Central.
[b] Values and classifications are based on data in Tables 2.2, 3.2, 4.2, 5.2, and 6.2 of Johnson (2001).
[c] Ratios are calculated from carbon mass based on biomass component equations in Jenkins et al. (2003a), applied to all live trees identified as growing stock on timberland stands classified as medium- or large-diameter stands in the survey data for the conterminous United States from USDA Forest Service, FIA Program's database of forest surveys (FIADB)(Alerich et al., 2005; USDA Forest Service, 2005). Carbon mass is calculated for boles from stump to 4-inch (10.2 cm) top, outside diameter.
[d] Values and classifications are based on data in Tables 2.9, 3.9, 4.9, 5.9, and 6.9 of Johnson (2001).

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Chapter 7 Quantifying Greenhouse Gas Sources and Sinks from Land-Use Change

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Acronyms, Chemical Formulae, and Units

C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
DOM	Dead organic matter
EPA	Environmental Protection Agency
FIA	Forest Inventory and Analysis
GHG	Greenhouse gas
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
N_2O	Nitrous Oxide
NRI	Natural Resources Inventory
PRISM	Parameter-Elevation Regressions on Independent Slopes Model
SOC	Soil organic carbon
SSURGO	Soil Survey Geographic Database
USDA	U.S. Department of Agriculture

7 Quantifying Greenhouse Gas Sources and Sinks from Land-Use Change

This chapter provides guidance on estimating the net greenhouse gas (GHG) flux resulting from changes between land use types—i.e., conversions into and out of cropland, wetland, grazing land, or forestland—at the entity scale. In some cases, it is sufficient to estimate the net GHG flux associated with the new land use. If changing from one land use to another has a significant effect on carbon stocks (e.g., changes in forest carbon stocks, changes in soil carbon), it will be necessary to represent that influence associated with a specific land-use change (e.g., wetland to cropland, grazing land to cropland, forestland to cropland). Table 7-1 provides a summary and description of the sources covered in this chapter.

Table 7-1: Overview of Land-Use Change Sources and Associated GHGs

Source	Method for GHG Estimation			Description
	CO_2	N_2O	CH_4	
Annual change in carbon stocks in dead wood and litter due to land conversion	✓			Live and dead biomass carbon stocks and soil organic carbon constitutes a significant carbon sink in many forest and agricultural lands. Following land-use conversion, the estimation of dead biomass carbon stock changes during transition periods requires that the area subject to land-use change on the entity's operation be tracked for the duration of the 20-year transition period.
Change in soil organic carbon stocks for mineral soils	✓			Soil organic carbon stocks are influenced by land-use change (Aalde et al., 2006) due to changes in productivity that influence carbon inputs, and to changes in soil management that influence carbon outputs (Davidson and Ackerman, 1993; Ogle et al., 2005; Post and Kwon, 2000). The most significant changes in soil organic carbon occur with land-use change, particularly conversions to croplands, due to changes in the disturbance regimes and associated effects on soil aggregate dynamics (Six et al., 2000).

7.1 Overview

In many cases, the methods proposed to estimate contributions to the GHG flux resulting from land-use change are the same as those used to estimate carbon stock changes in the individual chapters on Cropland and Grazing Land, Forestry, and Wetlands; although, in specific cases guidance is also provided on reconciling carbon-stock estimates between discrete data sets and estimation methods (e.g., reconciling forest soil carbon estimates and cropland soil carbon estimates for land-use change from forest land to cropland). Table 7-2 presents the methodologies for each source and indicates their section.

Table 7-2: Overview of Cropland and Grazing Land Systems Sources, Method and Section

Section	Source	Proposed Method
7.4.1	Annual change in carbon stocks in dead wood and litter due to land conversion	The change in carbon stocks in dead wood and litter due to land conversion is estimated as the difference in carbon stocks in the old and new land-use categories applied in the year of the conversion (carbon losses), or distributed uniformly over the length of the transition period (carbon gains) (Aalde et al., 2006).
7.4.2	Change in soil organic carbon stocks for mineral soils	The methodologies to estimate soil carbon stock changes for organic soils and mineral soils are adopted from IPCC (Aalde et al., 2006).

The remainder of this chapter is organized as follows:

- Definitions
- Caveats
- Steps for estimating GHG flux from land-use change
- Overlaps, issues, and assembly instructions for GHG flux estimation from land-use change

7.2 Definitions of Land Use

A land-use categorization system that is consistent and complete (both temporally and spatially) is needed to assess land use and land-use change status within an entity's boundaries. Each entity should ensure that it characterizes all of the land within its boundary according to the following land-use types: cropland, grazing land, forest land, wetland, settlements (e.g., residential, farm, and commercial buildings), and other land (e.g., bare soil, rock). The land-use definitions provided below are expected to be adopted by entities using this report. It is critical that individual parcel areas are estimated accurately and when combined add up to the total land area reported by the entity before and after the land-use change.

Current definitions for land use that are consistent with other policy programs related to GHG estimation (e.g., Intergovernmental Panel on Climate Change (IPCC), Natural Resources Inventory (NRI)) are provided below. These definitions are specific to the United States and are based predominantly on criteria used in the land-use surveys for the United States. Specifically, the definition of forest land is based on the Forest Inventory and Analysis (FIA) definition of forest,[1] while definitions of cropland, grazing land, and settlements are based on the NRI.[2] The definitions for other land and wetlands are based on the IPCC (2006) definitions for these categories.

- *Forest Land*: A land-use category that includes areas at least 36.6 meters wide and 0.4 hectares in size with at least 10 percent cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated. Forest land includes transition zones, such as areas between forest and non-forest lands that have at least 10 percent cover (or equivalent stocking) with live trees and forest areas adjacent to urban and built-up lands. Roadside, streamside, and shelterbelt strips of trees must have a crown width of at least 36.6 meters and continuous length of at least 110.6 meters to qualify as forest land. Unimproved roads and trails,

[1] See FIA Glossary http://socrates.lv-hrc.nevada.edu/fia/ab/issues/pending/glossary/Glossary_5_30_06.pdf
[2] See National Resource Inventory Glossary of Selected Terms (p. 9) http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1041379.pdf

streams, and clearings in forest areas are classified as forests if they are less than 36.6 meters wide or 0.4 hectares in size; otherwise they are excluded from forest land and classified as settlements. Tree-covered areas in agricultural production settings, such as fruit orchards, or tree-covered areas in urban settings, such as city parks, are not considered forest land (Smith et al., 2009).

- *Cropland*: A land-use category that includes areas used for the production of adapted crops for harvest; this category includes both cultivated and non-cultivated lands. Cultivated crops include row crops or close grown crops and also hay or pasture in rotation with cultivated crops. Non-cultivated cropland includes continuous hay, perennial crops (e.g., orchards), and horticultural cropland. Cropland also includes land with alley cropping and windbreaks, as well as lands in temporary fallow or enrolled in conservation reserve programs (i.e., set-asides[3]). Roads through cropland, including interstate highways, state highways, other paved roads, gravel roads, dirt roads, and railroads are excluded from cropland area estimates and are, instead, classified as settlements.
- *Grazing Land*:[4] A land-use category under which the plant cover is composed principally of grasses, grass-like plants, forbs, or shrubs suitable for grazing and browsing. This category includes both pastures and native rangelands and areas where practices such as clearing, burning, chaining, and/or chemicals are applied to maintain the grass vegetation. Savannas, some wetlands and deserts, and tundra are considered grazing land. Woody plant communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also classified as grazing land if they do not meet the criteria for forest land. Grazing land includes land managed with agroforestry practices such as silvopasture and windbreaks, assuming the stand or woodlot does not meet the criteria for forest land. Roads through grazing land, including interstate highways, state highways, other paved roads, gravel roads, dirt roads, and railroads are excluded from grazing land area estimates and are, instead, classified as Settlements.
- *Wetlands:*[5] A land-use category that includes land with hydric soils, native or adapted hydrophytic vegetation, and a hydrologic regime were the soil is saturated during the growing season in most years. Wetland vegetation types may include marshes, grasslands, or forests. Wetlands may have water levels that are artificially changed, or where the vegetation composition or productivity is manipulated. These lands include undrained forested wetlands, grazed woodlands and grasslands, impoundments managed for wildlife, and lands that are being restored following conversion to a non-wetland condition (typically as a result of agricultural drainage). Provisions for engineered wetlands including

[3] A set-aside is cropland that has been taken out of active cropping and converted to some type of vegetative cover, including, for example, native grasses or trees.

[4] Note that this definition is the "grassland" definition from the NIR with "grassland" replaced with "grazing land."

[5] The jurisdictional definition of a wetland is "those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas" (EPA, 1980). The 1987 Corps of Engineers Wetland Delimitation Manual & Regional Supplements (U.S. Army Corps of Engineers, 1987) is used to identify wetlands in the field.

storm water detention ponds, constructed wetlands for water treatment, and farm ponds or reservoirs are not included. Natural lakes and streams are also not included.

- *Settlements:* A land-use category representing developed areas consisting of units of 0.25 acres (0.1 ha) or more that includes residential, industrial, commercial, and institutional land; construction sites; public administrative sites; railroad yards; cemeteries; airports; golf courses; sanitary landfills; sewage treatment plants; water control structures and spillways; parks within urban and built-up areas; and highways, railroads, and other transportation facilities. Also included are tracts of less than 10 acres (4.05 ha) that may meet the definitions for Forest Land, Cropland, Grassland, or Other Land but are completely surrounded by urban or built-up land, and so are included in the settlement category. Rural transportation corridors located within other land uses (e.g., Forest Land, Cropland, and Grassland) are also included in Settlements.
- *Other Land*: A land-use category that includes bare soil, rock, ice, and all land areas that do not fall into any of the other five land-use categories, which allows the total of identified land areas to match the managed land base.

7.3 Caveats

The methods presented here for quantifying GHG flux from land-use change are intended for use at the entity scale on lands managed to enhance the production of food, feed, fiber, and renewable energy. Methods are currently not provided for estimating emissions from energy used when converting land use from one category to another. Methods are also not provided for land-use change from settlements or the "other land" category to forest land, cropland, grazing land, or wetlands. These methods have been developed for U.S. conditions and are considered applicable to agricultural and forestry production systems in the United States.

7.4 Estimating GHG Flux from Land-Use Change

Method for Estimating GHG Flux from Land-Use Change

- The GHG flux associated with land-use change is estimated based on the balance of carbon losses from the previous land use following conversion and the carbon gains with the current land use.
- This section only covers methodologies for dead organic matter carbon and soil organic matter carbon. Guidance on biomass carbon methods are provided in the land-use-specific sections (Cropland, Grazing land, Forest Land, and Wetlands).
- The change in carbon stocks in dead wood and litter due to land conversion is estimated as the difference in carbon stocks in the old and new land-use categories applied in the year of the conversion (carbon losses), or distributed uniformly over the length of the transition period (carbon gains).
- The methodologies to estimate soil carbon stock changes for organic soils and mineral soils are adopted from IPCC (Aalde et al., 2006).

Rationale for Selected Method

This method is based on the IPCC 2006 Guidelines (IPCC, 2006)and represents the most consistent method for estimating emissions from land-use change. Other methods are provided for land parcels that are not undergoing land use change, and arguably those methods are more comprehensive for estimating emissions for the specific land use. However, it is critical that an individual land parcel has a consistent, seamless method for estimating carbon stock changes throughout the time series. Otherwise artificial changes in stocks can be estimated due to a change in the method. The methods based on the IPCC 2006 Guidelines (IPCC, 2006) provide this consistency and seamless integration. Further testing and development will be needed before the more comprehensive methods provided in each land use section can be integrated into a seamless approach for estimating the carbon stock changes.

Description of Method

For inventory purposes, changes in carbon stock in biomass should be estimated for: (1) land remaining in the same land-use category; and (2) land converted to a new land-use category. The methods provided in this section are strictly for portions of an entity's operation that have undergone a land use change. The soil carbon changes must be addressed over a 20-year period. Aboveground and below ground biomass are estimated on an annual basis. Note that this section only addresses dead organic matter carbon and soil organic matter carbon. Biomass carbon methods should follow the guidance provided in the land-use-specific sections (Cropland, Grazing land, Forest Land, and Wetlands).

The reporting convention is that all carbon stock changes and non-CO_2 GHG emissions associated with a land-use change are reported in the new land-use category. For example, in the case of conversion of forest land to cropland, both the carbon stock changes associated with the clearing of the forest as well as any subsequent carbon stock changes that result from the conversion, are reported under cropland (IPCC, 2006).

The GHG flux associated with land-use change is essentially the sum of the GHG fluxes associated with previous (i.e., old) land-use categories plus the sum of the GHG fluxes associated with the current (i.e., new) land-use categories for a specified area undergoing conversion from the old to new land-use category. GHG emissions and stock changes not resulting from a land-use change are estimated with methods in the land-use-specific sections.

Equation 7-1: Annual Carbon Stock Changes for a Land-Use Change Estimated as the Sum of Changes in All Land-Use Categories

$$\Delta C_{LUC} = \Delta C_{LUC\ o} + \Delta C_{LUC\ n}$$

and

$$\Delta C_{LUC} = \Delta C_{LUC\ FL} + \Delta C_{LUC\ CL} + \Delta C_{LUC\ GL} + \Delta C_{LUC\ WL}$$

Where:

ΔC = carbon stock change (metric tons CO_2-eq ha^{-1} $year^{-1}$)

Indices denote the following land-use categories:

LUC = land-use change
o = old land-use category
n = new land-use category
FL = forest land
CL = cropland
GL = grazing land
WL = wetlands

For each land-use category undergoing a land-use change, it is important to estimate the annual carbon stock change occurring within each stratum or subdivision (e.g., carbon pool, management regime) for that land-use category.

Equation 7-2: Annual Carbon Stock Changes for a Land-Use Change as a Sum of Changes in Each Stratum Within a Land-Use Change

$$\Delta C_{LUC} = \Sigma_i \Delta C_{LUC\ I}$$

Where:

ΔC_{LUC} = carbon stock changes for a land-use change as defined in Equation 7.1 (metric tons CO_2-eq ha^{-1} $year^{-1}$)

i = denotes a specific stratum or subdivision within the land uses undergoing land-use conversion (by any combination of species, climatic zone, ecotype, management regime, etc.), i = 1 to n

For example, in the case of conversion of forest land to cropland, the carbon stock changes associated with each of the forest carbon pools plus harvested wood products should be assessed, as well as any subsequent carbon stock changes that result from the conversion (specific annualized changes in dead organic matter, soil carbon, etc.).

7.4.1 Carbon Pools in Live Biomass, Dead Biomass, and Soil Organic Carbon

Live and dead biomass carbon stocks and soil organic carbon constitute a significant carbon sink in many forest and agricultural lands. Sector-specific methods for estimating changes in biomass carbon stocks are detailed in the individual sector chapters and should be used when estimating the effect of land-use change. In addition to estimating the changes in biomass carbon stocks before and after the land-use change using the sector-specific methods, it is also important to estimate any increase in the harvested wood pool resulting from clearing/harvest of the forest following the methods outlined in Chapter 6, Forestry. Any biomass that is retained on the land during the land-use conversion will need to be included in the estimation, such as conversion of forest to grasslands, where some trees are left to provide shade for grazing livestock.

Following land-use conversion, the estimation of dead biomass carbon stock changes during transition periods requires that the area subject to land-use change on the entity's operation be tracked for the duration of the 20-year transition period. For example, dead organic matter (DOM) stocks are assumed to increase for 20 years after conversion to forest land. After 20 years, the area converted becomes forest and remaining forest land, and no further DOM changes are assumed. The conceptual approach to estimating changes in carbon stocks in dead wood and litter pools is to estimate the difference in carbon stocks in the old and new land-use categories and to apply this change in the year of the conversion (carbon losses), or to distribute it uniformly over the length of the transition period (carbon gains).

7.4.2 Changes in Soil Carbon

Soil organic carbon stocks are influenced by land-use change (Aalde et al., 2006) due to changes in productivity that influence carbon inputs, and to changes in soil management that influence carbon outputs (Davidson and Ackerman, 1993; Ogle et al., 2005; Post and Kwon, 2000). The most significant changes in soil organic carbon occur with land-use change, particularly conversions to croplands, due to changes in the disturbance regimes and associated effects on soil aggregate dynamics (Six et al., 2000). While there is considerable evidence and mechanistic understanding about the influence of land-use change on soil organic carbon, there is less known about the effect on soil inorganic carbon. Consequently, current methods do not include impacts on inorganic carbon uncertainty associated with estimates of land use and management impacts on soil carbon stocks.

Equation 7-3: Annual Change in Carbon Stocks in Dead Wood and Litter Due to Land Conversion

$$\Delta C_{DOM} = (C_n - C_o) \times A_{on} \div T_{on}$$

Where:

ΔC_{DOM} = annual change in carbon stocks in dead wood or litter (metric tons C year^{-1})

C_o = dead wood/litter stock, under the old land-use category (metric tons C ha^{-1})

C_n = dead wood/litter stock, under the new land-use category metric tons C ha^{-1})

A_{on} = area undergoing conversion from old to new land-use category (ha)

T_{on} = time period of the transition from old to new land-use category (year) (The default is 20 years for carbon stock increases and 1 year for carbon losses.)

Estimating changes in GHG emissions, including carbon stocks, require consistency in the methods that are applied across a time series. Applying different methods to account for changes in carbon stocks as the land shifts from one land use to another will lead to artificial changes in the stocks beyond the actual change occurring on the land. Thus, in order to ensure consistency, changes in soil organic carbon stocks will be estimated for the entire time series being reported, using the method described in this section. As noted earlier, estimates should be made separately for each parcel of land that undergoes a change in land use. However, the stock changes will only be reported as a land-use change effect for a 20-year transition period. Applying the same method across the entire time series will limit errors in the estimation of mineral soil organic carbon stock changes that would result from changing methods after the 20-year transition period.

7.4.2.1 Description of Method

Models have been adopted from the IPCC methods to estimate soil organic carbon stock change (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. 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). Emission estimation methods from organic soils should be consistent with their appropriate sector methodologies (i.e., forestry, croplands, grazing lands, or wetlands).

Mineral Soils: The model to estimate changes in soil organic carbon stocks for mineral soils has been adopted from the method developed by IPCC (Aalde et al., 2006). The change would need to be estimated separately for each area in the entity's operation that is converted from one land use to another. The change in stocks for each area is estimated over five year intervals for the entire reporting time series, using the following equation:

Equation 7-4: Change in Soil Organic Carbon Stocks for Mineral Soils

$$\Delta C_{Mineral} = [(SOC_f - SOC_i) \times CO_2MW] \div D$$

Where:

$\Delta C_{Mineral}$	= annual change in mineral soil organic carbon stock (metric tons CO_2-eq year^{-1})
SOC_f	= soil organic carbon stock at the end of year 5 (metric tons C)
SOC_i	= soil organic carbon stock at the beginning of year 1 (metric tons C)
CO_2MW	= ratio of molecular weight of CO_2 to C = 44/12 (dimensionless)
D	= time dependence of stock change factors (20 years)

Carbon stocks are estimated using the following equation adapted from the IPCC (Aalde et al., 2006):

Equation 7-5: Soil Organic Carbon Stock for Mineral Soils

$$SOC = SOC_{REF} \times F_{LU} \times F_{MG} \times F_I \times A$$

Where:

SOC	= soil organic carbon stock at the beginning (SOC_i) and end of the five years (SOC_f) (metric tons C)
SOC_{REF}	= reference soil organic carbon stock (metric tons C ha^{-1})
F_{LU}	= stock change factor for land use (dimensionless)
F_{MG}	= stock change factor for management (dimensionless)
F_I	= stock change factor for input (dimensionless)
A	= area of land-use change (ha)

The stock change factors (F_{LU}, F_{MG}, F_I) and reference carbon stocks (SOC_{REF}) are country-specific values developed for the United States (EPA, 2011; Ogle et al., 2003; Ogle et al., 2006). The reference stocks are based on the SOC stocks in croplands (Table 7-3), while the land-use factors represent the relative change in SOC between cropland and grazing lands, forest land, and set-aside cropland (Table 7-4). The management factors represent the influence of tillage in croplands and grassland condition in grazing lands. The input factors represent influences of changing plant productivity on carbon input to soils. Management and input factors are not needed for forest lands (Factors are set to a value of 1).

Organic Soils: The methodology for estimating soil carbon stock changes in organic soils has been adopted from IPCC (Aalde et al., 2006), and is described accordingly in Chapter 4, Wetlands, and Chapter 3, Croplands, and Grazing Lands. Chapter 6, Forestry, recommends soil sampling in cases where there have been significant changes in soil carbon (e.g., land conversion).

Table 7-3: Reference Carbon Stocks (Mg C ha-1) (±1 SE) To Estimate Soil Organic Carbon Stock Changes for Mineral Soils

Soil Type	USDA Taxonomy	Cool Temperate Dry	Cool Temperate Moist	Warm Temperate Dry	Warm Temperate Moist	Sub-Tropical Dry	Sub-Tropical Moist
High activity clay soils	Vertisols, Mollisols, Inceptisols, Aridisols, and high base status Alfisols	42±1.4	65±1.1	37±1.1	51±1.0	42±2.6	57±13.0
Low activity clay soils	Ultisols, Oxisols, acidic Alfisols, and many entisols	45±3.0	52±2.3	25±1.4	40±1.2	39±4.8	47±13.9
Sandy soils	Any soils with greater than 70% sand and less than 8% clay (often Entisols)	24±4.8	40±3.7	16±2.4	30±2.0	33±1.9	50±7.9
Volcanic soils	Andisols	124±11.4	114±16.7	124±11.4	124±11.4	124±11.4	128±15.0
Spodic soils	Spodosols	86±6.5	74±6.8	86±6.5	107±8.3	86±6.5	86±6.5
Wetland soils	Soils with Aquic suborder	86±	89±	48±	51±	63±	48±

Source: EPA (2011) and Ogle et al. (2003) Ogle et al. (2006).

Table 7-4: Carbon Stock Change Factors (±1 SE) to Estimate Soil Organic Carbon Stock Changes for Mineral Soils

Factor	Warm Temperate Moist/Subtropical Moist	Warm Temperate Dry/Subtropical Dry	Cool Temperate Moist	Cool Temperate Dry
Land-Use Factor				
Long-term cultivated	1	1	1	1
Forest/grassland	1.42±0.06	1.37±0.05	1.24±0.06	1.20±0.06
Set-aside	1.31±0.06	1.26±0.04	1.14±0.06	1.10±0.05
Cropland Management				
Full till	1	1	1	1
Reduced till	1.08±0.03	1.01±0.03	1.08±0.03	1.01±0.03
No-till	1.13±0.02	1.05±0.03	1.13±0.02	1.05±0.03
Grassland Management[a]				

Factor	Warm Temperate Moist/Subtropical Moist	Warm Temperate Dry/Subtropical Dry	Cool Temperate Moist	Cool Temperate Dry
Non-degraded	1	1	1	1
Moderately degraded	0.95±0.06	0.95±0.06	0.95±0.06	0.95±0.06
Severely degraded	0.7±0.14	0.7±0.14	0.7±0.14	0.7±0.14
Improved	1.14±0.06	1.14±0.06	1.14±0.06	1.14±0.06
Cropland input				
Low	0.94±0.01	0.94±0.01	0.94±0.01	0.94±0.01
Medium	1	1	1	1
High	1.07±0.02	1.07±0.02	1.07±0.02	1.07±0.02
High with amendment[a]	1.38±0.06	1.34±0.08	1.38±0.06	1.34±0.08
Grassland input[a]				
Medium	1	1	1	1
High	1.11±0.04	1.11±0.04	1.11±0.04	1.11±0.04

[a] Grassland management and input factors are from the 2006 IPCC Guidelines (Verchot et al., 2006) as well as the high input systems with manure in croplands (Lasco et al., 2006).

7.4.2.2 Activity Data

Mineral soils require the following activity for croplands:

- Crop selection and rotation sequence;
- Residue management, including harvested, burned, grazed, or left in the field;
- Irrigation, yes or no;
- Mineral fertilization, yes or no;
- Lime amendments, yes or no;
- Organic amendments, yes or no;
- Tillage implements, which can be used to determine tillage classification (i.e., full tillage, reduced tillage, and no-till); and
- Cover crops, yes or no.

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

- Degradation status, non-degraded, moderately degraded, severely degraded;
- Irrigation, yes or no;
- Mineral fertilization, yes or no;
- Seeding legumes, yes or no;
- Lime amendments, yes or no; and
- Organic amendments, yes or no.

The method for forest land does not require any management activity data because the method provided here assumes limited influence on soil organic carbon stock changes associated with forest management after a land-use change (i.e., the land-use change has the largest impact).

The activity data are used to classify land-use, management, and input classes. The classifications can be found in Lasco et al. (2006) for cropland (Figure 5.1), and Verchot et al. (2006) for grassland (Figure 6.1).

7.4.2.3 Ancillary Data

Ancillary data include climate regions and soil types, consistent with the method developed by the IPCC (Bickel et al., 2006). Weather data may be based on national datasets such as the Parameter-Elevation Regressions on Independent Slopes Model (PRISM) data (Daly et al., 2008) and are classified according to the IPCC classification as refined for the United States (Table 7-5). Soils data may also be based on national datasets such as the Soil Survey Geographic Database (SSURGO) (Soil Survey Staff, 2011), and are classified according to the IPCC classification (Bickel et al., 2006; Figure 3A.5.3). However, entities may also substitute field-specific soils data, as long as entities characterize the soil pedons necessary for use of the IPCC classifications. These characteristics include sand and clay content, soil order, and suborder (See Table 7-3).

Table 7-5: Climate Classification for the Soil Organic Carbon Methods Associated with Land-Use Change

Climate Type	Mean Annual Temperature (°C)	Mean Annual Precipitation (mm)
Cool temperate dry	<10	<Potential evapotranspiration
Cool temperate moist	<10	≥Potential evapotranspiration
Warm temperate dry	10-20	<Potential evapotranspiration
Warm temperate moist	10-20	≥Potential evapotranspiration
Subtropical dry	>20	<1000
Subtropical moist	>20	1000-2000

Source: Bickel et al. (2006).

7.4.2.4 Model Output

Model output is generated as an absolute quantity of emissions. The change in mineral soil organic carbon stocks is estimated based on stock changes over five-year time periods (Equation 7.4). In addition, trends in soil organic carbon will be estimated for the entire time series associated with the parcel of land, including 20 previous years of history, in order to present the longer term trends and provide an adequate baseline of data and consistency in the time series for reporting purposes.

7.4.2.5 Limitations and Uncertainty

The limitations of the mineral soil organic carbon method include no assessment of the effect of land-use change at deeper depths in the profile (IPCC method only addresses changes in top 30 cm of soil profile; Aalde et al., 2006), and no assessment of erosion, transport, and deposition of carbon. Uncertainties in the mineral soil methods include imprecision in the emission factors, in addition to uncertainties in the activity and ancillary data. Uncertainty in the emission factors is provided in this guidance (Ogle et al., 2003; Ogle et al., 2006). Uncertainty in the activity data is based on the entity input, as well as the ancillary data to the extent that this information is provided by the entity. Uncertainties can be combined using a Monte Carlo simulation approach.

7.4.3 Changes in other GHG emissions

As previously mentioned, changes in other GHG emissions—i.e., non-CO_2 emissions— associated with a land-use change should be included in any estimation of the GHG flux strata associated with the outgoing or incoming land-use change. While changes in biomass and soil carbon stocks are likely to dominate the GHG flux, there are a number of activities that may occur during land-use conversion that might result in non-CO_2 emission. For example, if forest harvest residues (and other dead organic matter) are piled and burnt as part of the conversion of forest land to another land use, in addition to the change in carbon stock the residue burning will result in emissions of N_2O and CH_4; and if wetlands are cleared and drained prior to conversion to another land use (e.g., grazing lands, peat extraction), in addition to the change in carbon stock from clearing, the draining will result in a reduction in emissions of CH_4, and a possible increase in emissions of N_2O, depending on the nitrogen content of the wetland soil (i.e., peat).

Sector-specific methods for estimating changes in biomass burning non-CO_2 emissions (e.g., for cropland and grazing land systems) and soil non-CO_2 emissions (e.g., for wetland systems) are detailed in the individual sector chapters.

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Chapter 8 Uncertainty Assessment for Quantifying Greenhouse Gas Sources and Sinks

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[1] All authors of Chapters 3, 4, 5, and 6 provided strategic input in the parameters in the uncertainty chapter.

Acronyms, Chemical Formulae, and Units

CONUS STATSGO	Continental United States Soil Geographic Database
DBH	Diameter at breast height
DNDC	DeNitrification-DeComposition
EPA	U.S. Environmental Protection Agency
ERS	USDA Economic Research Service
FOFEM	First Order Fire Effects Model
FVS	Forest Vegetation Simulator
GHG	Greenhouse gas
IPCC	Intergovernmental Panel on Climate Change
NADP	National Atmospheric Deposition Program
NARR	North American Regional Reanalysis
NASS	USDA National Agricultural Statistics Service
NCEP	National Centers for Environmental Prediction
NLCD	National Land Cover Database
NOAA	National Oceanic and Atmospheric Administration
NRCS	USDA Natural Resources Conservation Service
NRI	National Resource Inventory
PDF	Probability density function
PRISM	Parameter-elevation Regressions on Independent Slopes Model
SOC	Soil organic carbon
SSURGO	Soil Survey Geographic
USDA	U.S. Department of Agriculture

8 Uncertainty Assessment for Quantifying Greenhouse Gas Sources and Sinks

Quantifying the uncertainty of greenhouse gas (GHG) emissions and reductions from agriculture and forestry practices is an important aspect of decision-making for farmers, ranchers and forest landowners as the uncertainty range for each GHG estimate communicates our level of confidence that the estimate reflects the actual balance of GHG exchange between the biosphere and the atmosphere. In particular, a farm, ranch, or forest landowner may be more inclined to invest in management practices that reduce net GHG emissions if the uncertainty range for an estimate is low, meaning that higher confidence in the estimates exists. This chapter presents the approach for accounting for the uncertainty in the estimated net emissions based on the methods presented in this report.[2] A Monte Carlo approach was selected as the method for estimating the uncertainty around the outputs from the methodologies in this report as it is currently the most comprehensive, sound method available to assess the uncertainty at the entity scale. Limitations and data gaps exist; however, as new data become available the method can be improved over time. Implementation of a Monte Carlo analysis is complicated and requires the use of a statistical tool to produce a probability density function (PDF)[3] around the GHG emissions estimate.[4] From the PDF, the uncertainty estimate can be derived and reported.

[2] The IPCC Good Practice Guidance (IPCC, 2000) recommends two approaches—Tier 1 and Tier 2—for developing quantitative estimates of uncertainty for emissions estimates for source categories. The Tier 1 method uses error propagation equations. These equations combine the uncertainty associated with the activity data and the uncertainty associated with the emission (or other) factors. This approach is appropriate where emissions (or removals) are estimated as the product of activity data and an emission factor or as the sum of individual sub-source category values. The Tier 2 method utilizes the Monte Carlo Stochastic Simulation technique. Using this technique, an estimate of emission (or removal) for a particular source category is generated many times via an uncertainty model, resulting in an approximate PDF for the estimate. Where sufficient and reliable uncertainty data for the input variables are available, the Tier 2 method is the preferred option.

[3] The integral of a PDF over a given interval of values is the probability for a random variable to take on some value in the interval. That is, the PDF is a function giving probability "densities" and its integral gives probabilities. A narrower PDF for an estimate indicates smaller variance around the central/most likely value, i.e., a higher probability of the value to be closer to the central/most likely value. The uncertainty for such an estimate is lower.

[4] Given the complexity of Monte Carlo analysis and the necessity for a tool, the approach presented here is not intended for development by a landowner, rather it is intended for use in developing a tool that a landowner would use to assess uncertainty estimates.

Monte Carlo Analysis for Assessing Uncertainty

- In the Monte Carlo method, uncertain inputs (parameters and other data) and uncertain model structure are described via PDFs. By randomly selecting from each of these PDFs, and running the selected inputs through the selected model, an uncertainty model output is obtained. Combining these model outputs across many random selections leads to an approximate PDF describing uncertainty in the model output, reflecting known sources of uncertainty in the inputs and model structure.
- A tool is needed to run a Monte Carlo analysis to assess the uncertainty for model outputs. Farmers and landowners are not expected to perform a Monte Carlo analysis on their own.
- A centralized database is needed to store information on the known uncertainties associated with the activity and emission factor data for each emissions source. This report presents readily available data that can form the initial foundation for such a database.

Uncertainty in GHG emissions estimation arises because of unknown or incompletely known factors associated with:

- Parameters – Due to limitations associated with available input data (e.g., activity data and emission factors).
- Sampling methods – Due to either measurement errors during sample collection or potential variations in values obtained from sampling (i.e., when the chosen sample is not fully representative of the entire population).
- Large datasets – Due to measurements errors during data collection, and variations in dataset values for a given set of conditions.
- Models – Due to approximation errors and estimation errors. Approximation error arises because the model is a simplification of the real system, while estimation error arises because the theoretical model is fitted using limited data.
- Concepts – This is closely related to model approximation error and occurs because the conceptual scope does not capture the actual/real scope thus creating a bias. For an entity, this conceptualization uncertainty may be relatively small.

The approach to address uncertainty does not address conceptual uncertainty because it is expected to be small and difficult to quantify. This chapter addresses parameter uncertainty, sampling uncertainty, large dataset uncertainty, and model approximation uncertainty. Where data are currently unavailable or incomplete for establishing PDF's and estimating uncertainty, the authors provide expert judgment and/or a qualitative description of uncertainty in the interests of making the GHG management methods as transparent and complete as possible. In the future, new data can be used to refine and improve the estimation of uncertainty.

In this chapter, Section 8.1 includes the components and inputs to an entity-scale Monte Carlo uncertainty assessment, and Section 8.2 highlights research gaps.

8.1 Components and Inputs to an Entity-Scale Monte Carlo Uncertainty Assessment

To conduct a Monte Carlo uncertainty analysis for each of the GHG quantification methods and resulting net GHG emissions, information is required about the uncertainty associated with: (1) the input variables (i.e., parameters); (2) sampling methods used to obtain data; (3) existing large datasets used as data sources; and (4) external models used. Ideally, this information would consist of specific PDFs (e.g., normal, triangular, uniform, beta). Alternatively, the uncertainty might be described with summary statistics, such as lower and upper bounds for intervals with specified confidence, minimum, maximum, mean, and standard deviation. This summary information forms the basis for constructing approximate PDFs for the Monte Carlo method. Repeated selections are made from these PDFs. These selections represent the range of possible outcomes from each PDF. Random sampling from the PDFs will ensure such representativeness.[5] By randomly selecting from each of these PDFs and running the selected inputs through the model, a range of outputs is obtained. Combining these model outputs across many random selections leads to an output PDF that can be used to describe uncertainty in the estimate, accounting for known sources of uncertainty in the inputs and model structure.

This section presents readily available information on each of the key components of uncertainty. In summary, although information on all the components are described here, the Monte Carlo method for assessing net GHG emissions uncertainty relies most heavily on parameter uncertainty, for which the best PDF data and information are available. Other components of uncertainty are discussed, including limitations such as characterizing the uncertainty associated with other components. These components can be readily improved or refined in the uncertainty analysis as additional information becomes available. Overall uncertainty is typically greater than any particular uncertainty component (e.g., sampling, large data sets, models) and can be readily improved or refined as additional information becomes available. As the uncertainty associated with the other components is addressed, the uncertainty will increase (i.e., addressing only parameter uncertainty sets a lower bound for overall uncertainty). Therefore, the quantification of parameter uncertainty sets a lower bound for overall uncertainty.

8.1.1 Parameter Uncertainty

Parameter uncertainty is the primary source of uncertainty in the net GHG estimates. This section presents readily available information on parameters used to estimate net GHG emissions from animal production systems, croplands and grazing lands, and forestry GHG estimation methods. For each input variable, readily available information was collected on the probability distribution; variance; standard deviation; expected mean, median, and mode; most likely value; minimum; maximum; relative uncertainty absolute values; confidence interval; and data sources. The information was collected primarily from published literature, such as the Intergovernmental Panel on Climate Change (IPCC) Guidelines (2006), the U.S. National GHG Inventory Report (U.S. EPA, 2012), and peer-reviewed journals. In the absence of published data, default factors are indicated based on expert judgment obtained from the Working Groups. The information obtained to date is presented in Appendix 8-B.[6, 7, 8]

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[5] An alternative approach to selecting from the PDFs is *Latin hypercube sampling* (McKay et al., 1979; Helton and Davis, 2003).

[6] Uncertainty for the forestry sector is mainly driven by modeling and sampling uncertainty; consequently, only a few parameters have been listed in Appendix 8-B.

[7] The Wetlands Chapter methods suggest use of the FVS and DNDC models in combination with the lookup tables for dominant shrub and grassland vegetation types found in Chapter 3, for estimating biomass carbon, soil carbon, N_2O, and CH_4 emissions and removals in wetlands. Descriptions of these models and the uncertainty associated with the look-up tables are included in the Uncertainty Assessment (Chapter 8).

[8] An uncertainty assessment was not completed for the Land-use Change Chapter methods (i.e., annual change in carbon stocks in dead wood and litter due to land conversion, change in soil organic carbon stocks for mineral soils) as they are based upon IPCC 2006 Guidance and no U.S. specific customizations were made to these methods. Uncertainty assessments for each land-use and transition into or out of a land-use category

In order to make the uncertainty estimation process feasible and consistent at the entity scale for use by an entity or landowner, a tool will be needed that provides the following uncertainty information for input variables:

- PDFs or distributions – Default
- Emission factors – Default
- Activity data – Default, but customizable

With default uncertainty information available, it is feasible to quantify parameter uncertainty via PDFs and to combine the uncertainty via Monte Carlo methods. These PDFs are often relatively crude, relying on default values and conservative expert judgment. Options to improve the PDFs (i.e., improve parameter uncertainty quantification) are to: (1) develop a method to help elicit and refine these uncertainty distributions at an entity scale; and (2) conduct new research to better understand the key parameters identified in this report and to quantify their uncertainties.[9]

The uncertainty associated with the various inputs to the GHG estimation equation or models are combined to estimate overall uncertainty at the entity level for: (1) each source category emission estimate; and (2) total emission estimate arrived at by aggregating each source category's estimate. Although most inputs within a category and across categories are independent, certain variables might be the same, similar, or highly co-related, and will need to be accounted for appropriately in the uncertainty analysis.

8.1.2 Sampling Method Uncertainty

Some sampling methods (i.e., field measurements) will be conducted to support the estimation of emissions using the GHG quantification methods. For example, for the forestry sector, conducting field measurements on sampling plots for large forest and on urban forests is used to determine aggregate forest characteristics (e.g., tree cover). Additionally, some large datasets and external models that the methods use also utilize data that were obtained from a variety of outside sampling methods. For example, forest inventory data used in the forest vegetation simulator (FVS) model and the average carbon sequestration rates used in the i-Tree model use data obtained through sampling methods. In addition, there are instances in these external models and large datasets where the variation in measurements obtained from the sampling methods is not taken into account, but they can impact uncertainty.

If forest stand sampling is conducted at an entity level using a formal probability sampling design, then unbiased estimates of sampling error variance can be computed via standard techniques from the field of survey statistics. The exact form of the variance estimate depends on the particular design used for the stand sampling. Though additional uncertainties arise from the actual measurement protocols used in the field, the sampling error variance is a major part of the sampling uncertainty. A currently feasible approach to incorporating information on sampling error variances into the uncertainty analysis is to model the sampling error PDF as a normal distribution with zero as its mean and the estimated sampling error variance as its variance.[10] This section

are contained in the associated land-use category chapter of the 2006 IPCC Guidelines. http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html

[9] A tool could provide an option to use pre-defined values such as those provided in Appendix 8-B or user (i.e., landowner) supplied values to define PDFs.

[10] Similarly, estimates derived from existing surveys and tools such as Forest Inventory Data Online in the Forest Inventory and Analysis National Program or the USDA Natural Resources Inventory (NRI) Summary Reports have associated sampling error variances from well-established statistical procedures. In these cases,

provides the sampling methods and their potential sources of uncertainty. However, given the complexity in incorporating information on uncertainty for sampling data in the Monte Carlo uncertainty assessment, a tool will be needed to quantify the impact of sampling uncertainty on the estimate of net GHG emissions.[11]

8.1.2.1 Forest Stand Sampling with Plots for Use with the Forest Vegetation Simulator Model

The FVS model is a family of forest growth simulation model variants. FVS estimates forest carbon stocks based on sample data parameters (e.g., the diameter, height, species, and canopy density of trees from representative sample plots established across the forest). For sampling purposes, a number of plots are established within a forest that can serve as a representative sample of the entire forest. As the variance in forest types increases, the number of plots will increase. The size and number of plots should be determined based on the variance in carbon stocks between plots. Complete forest estimates have sampling uncertainty, but larger and more numerous plots help to create a more representative sample and lower the uncertainty associated with carbon stock estimates produced by FVS. Both permanent and temporary plots can be used in sampling; however, a larger uncertainty is associated with temporary plots. Note that the use of permanent plots is recommended in this report. This type of sampling methodology is commonly referred to as a forest inventory.

Once plots have been defined, all trees above a certain diameter at breast height (DBH) (commonly 2.5 cm. or 1 in.) are measured and recorded.[12] DBH, height, and a variety of other measurements are recorded, but DBH alone is sufficient for use with FVS. FVS uses DBH and available information to develop carbon density estimates for the entire forest. If provided, FVS models growth estimates for future years based on average growth rates and variables such as thinning.[13] Selecting plots to represent entire forests means FVS outputs are subject to the sampling method uncertainty. However, uncertainty in the FVS outputs can be lowered (i.e., more representative carbon stock estimates can be obtained) by collecting more detailed tree data beyond DBH as well as ensuring that sample plots are large and numerous enough to cover the variety of tree growth settings in a forest.

The forest inventory data recommended for use with FVS in this report is based on the sampling methods described in the U.S. Department of Agriculture (USDA) Measurement Guidelines for the Sequestration of Carbon (Pearson et al., 2007). These guidelines also describe the potential uncertainty associated with such sampling methods. According to these measurement guidelines, a reasonable estimate of the net change in carbon stocks would be within 10 percent of the true value of the mean at the 95 percent confidence level that can be achieved by having a sufficiently large sample size (Pearson et al., 2007). Different carbon pools in a forest can have different variances; however, focusing on the standing live tree component for forestry activities can capture most of the total variance.

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it may be feasible to use these estimated variances fairly directly. In other cases, it might be necessary to consider small area estimation techniques to describe uncertainty as large-scale survey data are downscaled to entity levels. Describing this type of uncertainty would require building statistical models for complex survey data and, hence, is not addressed in this report as additional research is required.

[11] For example, in addressing uncertainty in growth of forest biomass, algorithms to account for nonlinear growth patterns will be needed.

[12] Under common stand exams, even trees less than 2.5 cm or 1 in can be measured, but these trees are often considered to be part of the understory (e.g., by FIA).

[13] Other variables, such as fertilization, only apply to a few FVS variants.

The forest inventory data that is used for modeling the changes in forest stands is likely the largest source of parameter uncertainty for the inputs and assumptions used in the FVS model; in addition, there are many components in the FVS model where variation in measurements for this data is not taken into account. For example, the potential error distribution from month to month in carbon storage associated with leaves and foliage is not accounted for in the FVS model.

In a research paper on obtaining sampling uncertainty in FVS, experts note the challenge of reporting distributions of model inputs that include sampling uncertainty in FVS projections (Gregg and Hummel, 2002). They state in their introduction that, "it hasn't been possible to compute the effects of sampling uncertainty because classical statistical methods are not available to make inferences about FVS projections. A variance estimator is not available for the results of simulation."

As provided in Appendix 8-A, the FVS model provides quantitative information on the range and variability of sampling data. This FVS application, called FVSBoot, uses "bootstraps" to determine fluctuation in estimate outcomes (Gregg and Hummel, 2002) (i.e., allows modeler to empirically approximate the sampling distribution of any statistic/FVS attribute for which the modeler wants to make inferences).

Bootstrap sampling[14] using the FVSBoot program can be used to empirically approximate the sampling distribution of statistics for which inferences are to be drawn. New samples of stand conditions can be generated by sampling the original plots with replacement to create a bootstrap sample. A bootstrap mean can be generated from the bootstrap sample. Repeating this process multiple times will generate a Monte Carlo approximation of the distribution of bootstrap means. The standard deviation of this approximation will be an estimation of the true standard deviation for the entire population. FVSBoot does not cover all potential sources of variation but it can give a measure of important components of uncertainty in FVS model projections. While the FVSBoot program can be used to determine the sampling uncertainty in FVS, it was not developed originally to produce an overall uncertainty estimate for FVS outputs. However, FVSBoot has been used for sensitivity analysis of some FVS outputs (Hummel et al., 2013). A tool would be needed to facilitate developing an estimate of the uncertainty based on a combination of the results from the FVSBoot program and underlying equations used in the FVS model.

8.1.2.2 Urban Tree Population Sampling for Use with the Field Data Method Using the i-Tree Eco Model

The i-Tree Eco model estimates urban forest carbon stocks and gross and net annual carbon sequestration based on sample data parameters (e.g., the tree species, diameter, height, dieback, and crown light exposure) with a calculated level of precision. As desired by the landowner, all trees can be measured or a random distribution of field plots can be measured to quantify the urban tree population. Larger and more numerous plots help to create a more representative sample and lower the uncertainty associated with the sampling. The i-Tree Eco model uses the sample data parameters and forest-derived allometric equations to estimate carbon values. The model also estimates the standard error of the estimated carbon value, which is based on the sampling uncertainty, rather than the error of estimation from applying the allometric equations. Estimation error is unknown and likely larger than the reported sampling error. A Monte Carlo

[14] Bootstrapping is the process of estimating variance by repeated random sampling with replacement of an existing data set. For example, to determine the probability distribution of average DBH for a sample of 100 trees, resamples of the data set of 100 trees can be taken to approximate the variance.

analysis tool could use the standard error of the estimated carbon value to evaluate the uncertainty associated with an entity's total net GHG emissions.

8.1.2.3 Sampling and DAYCENT for Estimating Biomass Carbon in Grazing Land and Agroforestry Systems

Sampling uncertainty will exist when estimating biomass carbon using the method provided in Chapter 3. For example, peak forage estimates for grazing lands can be sampled using the biomass clipping method.[15] This method is destructive with the removal of forage samples from the field. This method has been shown to produce estimates with low uncertainty (Lauenroth et al., 2006; Byrne et al., 2011). Non-destructive methods can also be used including the comparative yield method for rangelands,[16] or the robel pole method on rangelands or pastures (Harmoney et al., 1997; Vermeire et al., 2002). The biomass clipping method and comparative yield methods have less uncertainty than the robel pole or visual obstruction method. Destructive sampling methods, however, are more time and labor intensive. Uncertainty associated with the robel pole method was assessed in the Black Hills of South Dakota in a study by Uresk and Benzon (2007). The authors compared destructive (clipping) methods for estimating biomass and the robel pole method. They found there was a linear relationship between the two methods and the standard error of the robel pole estimate for a single mean was 373 kg ha^{-1}. The study further recommends that a minimum of three transects be sampled for monitoring areas less than 259 ha to be within 20 percent of the mean and 80 percent confident (Uresk and Benzon, 2007). In a similar study by Vermeire et al., (2002), a single visual obstruction model (i.e., robel pole method) effectively estimated herbage standing crop across range types and produced a coefficient of determination of 0.93. 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 USDA Natural Resources Conservation Service (USDA-NRCS) in Ecological Site Descriptions.[17] The sampling uncertainty will depend on the method used to collect the sample and should be provided by the farmer or landowner.

8.1.3 Large Dataset Uncertainty

Information from several large datasets will be used with the GHG quantification methods to estimate emissions from the animal agriculture, cropland and grazing land, and forestry sectors. Large datasets can be considered any grouping of data points that cover a wide time-series and/or level of reported variables. These data sets include multiple data layers, GIS data, databases, and other such reporting catalogues.

The large datasets to be used include the Smith et al. (2006, also known as GTR-NE-343), Forest Inventory and Analysis, FVS, Daymet, and the dataset from the i-Tree model. These datasets provide values for estimation equation and model inputs. These inputs include region- and species-specific tree growth rates, land and tree cover, inferred and observed meteorological data, soil type and distribution, ammonia content, and historical climate data for the North American continent. These data will be used to inform the carbon densities of small forest holdings, coverage of urban trees, direct and indirect N_2O emissions, soil pH, organic matter values, ambient air ammonia concentrations, and daily air temperature and velocity.

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[15] See Section 15, "Standing Biomass" (USDA NRCS, 2011b).
[16] See Section 13, "Dry Weight Rank" (USDA NRCS, 2011b).
[17] See USDA NRCS (2011a).

This section includes a description of the large datasets used for estimating forestry and agroforestry sector carbon stocks and stock changes, GHG emissions and removals from wetlands, soil carbon stocks, and ammonia emissions. The section also provides uncertainty information obtained from the dataset developers and an approach to incorporating uncertainty associated with these datasets into the overall uncertainty analysis.

Many of the large datasets are complex and cover multiple parameters. In some instances uncertainty information is available for some variables but not for others, making it difficult to assess the uncertainty of the entire dataset. Table 8-1 below summarizes the uncertainty documentation available for each large dataset used. The majority of datasets did not have publically available documentation characterizing the associated uncertainty.

Table 8-1: Availability of Uncertainty Information for Large Datasets

Dataset Name	Dataset Abbreviation	Availability of Uncertainty Documentation
Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States (Smith et al., 2006)	GTR-NE-343	No published quantification of uncertainty. Standard errors available for carbon density for live and standing dead trees at the 50th and 99th percentile of volume.
National Land Cover Database	NLCD	No published quantitative uncertainty information found. Authors only provide information on contributing factors.
Daily Surface Weather and Climatological Summaries	Daymet	No published quantitative uncertainty information found.
Contiguous United States Soil Geographic Database	CONUS STATSGO	No published quantitative uncertainty information found.
Soil Survey Geographic Database	SSURGO	No published quantitative uncertainty information found.
Ammonia Monitoring Network	AMoN	No published quantitative uncertainty information found.
Parameter-Elevation Regressions on Independent Slopes Model	PRISM	No published quantitative uncertainty information found.
North American Regional Reanalysis	NARR	Regional-scale accuracy and bias reported by Mesinger (2006).
Natural Resources Inventory	NRI	Data are collected using a two-stage sampling process. Statistically valid uncertainties in management practices are computable at Major Land Resource Areas or State level.
National Agricultural Statistics Service Agricultural Census	NASS-agricultural census	No published quantitative uncertainty information found.
National Agricultural Statistics Service – Cropland Data Layer	NASS – Cropland Data Layer	NASS provides accuracy information and error matrices (total accuracy, errors of omission and co-mission), but not on an annual basis for crops and States.
Economic Research Service Cropping Practices Survey	ERS-CPS	No published quantitative uncertainty information found.
Economic Resource Service Agricultural Resource Management Survey	ERS-ARMS	No published quantitative uncertainty information found.

Dataset Name	Dataset Abbreviation	Availability of Uncertainty Documentation
National Climatic Data Center of the National Oceanic and Atmospheric Administration	NCDC (NOAA)	NCDC provides values that describe the range of the uncertainty, or simply "range," of each month's, season's, or year's global temperature anomaly. These values are provided as plus/minus values.
Modern-Era Retrospective Analysis for Research and Applications	MERRA (NASA)	No published quantitative uncertainty information found.
National Centers for Environmental Prediction	NCEP (NOAA)	No published quantitative uncertainty information found.
National Atmospheric Deposition Program	NADP	Regional-scale uncertainty was assessed in Dennis et al. (2011).

8.1.3.1 GTR-NE-343 Carbon Density Values

Estimates of carbon stocks and stock changes from the report, "Methods for Calculating Forest Ecosystem and Harvested Carbon with Standard Estimates for Forest Types of the United States" (Smith et al., 2006) (USDA Forest Service, General Technical Report NE-343), are based on regional averages and reflect the current best available data. However, according to GTR-NE-343, "quantitative expressions of uncertainty are not available for most data summaries, coefficients, or model results presented in the [GTR-NE-343] tables." GTR-NE-343 lookup tables include some information about the confidence intervals for live and standing dead tree carbon densities at two different average volumes (see Table 20 of GTR-NE-343), but it does not prescribe a method for applying these summary uncertainty statistics to stand level carbon stock estimates.

The uncertainty associated with these reported regional average carbon stock values is likely higher as these values are applied to smaller-scale projects rather than regions. Sampling uncertainty associated with the regional averages, that are based on data summaries or models, can influence estimates for specific projects. These projects are generally small subsets of a region. Yet, variability within a region for values in a dataset will likely have a much greater influence on uncertainty than the actual sampling uncertainty associated with collecting regional values (Smith et al., 2006).

Once the user finds the table in GTR-NE-343 that describes the forest's species mix and region, the user can use the age (or volume) of the forest stand (which is also collected with a high level of uncertainty) to find out the metric tons of carbon per acre density value for live tree carbon, down deadwood, organic soil carbon, and other categories. The uncertainty information is given as 95 percent confidence intervals for the carbon density of live and standing dead trees, at two different growing stock volumes—the 50^{th} percentile and the 99^{th} percentile. These confidence intervals are given for each forest type and region. To use this information in an uncertainty analysis requires extrapolation to other growing stock volumes, which requires modeling the relationship between growing stock volume and variation in carbon density. While these tables are simple and easy to use, the uncertainty of results obtained by using representative average values may be high relative to other techniques that use site- or project-specific data. Additional research is needed to include this uncertainty into a Monte Carlo analysis framework.

8.1.3.2 National Land Cover Database Map

The National Land Cover Database (NLCD) Map is the product of the Multi-Resolution Land Characterization partnership, a consortium of Federal agencies including the U.S. Geological Survey, Environmental Protection Agency (EPA), National Oceanic and Atmospheric Administration (NOAA), and the USDA Forest Service that are continuously developing digital land cover data. This association has successfully provided land cover data for the lower 48 States, Hawaii, Alaska, and Puerto Rico from decadal Landsat satellite imagery and other associated imaging datasets. The database provides Landsat-based, 30-meter resolution, land coverage characteristics including thematic class (e.g., urban, agriculture, and forest), percent impervious surface, and percent tree canopy cover.

Regarding uncertainty, the NLCD map documentation indicates, “Unfortunately, there is no readily available reference dataset with which to compare the inventory to generate accuracy statistics. Reference data have to be specifically generated through manual interpretation of remote sensing data for a sample of locations, as has been done for accuracy assessment of land cover maps. In lieu of such an approach, which is outside the scope of this study, the best that can be done currently to describe the uncertainty of the inventory data is to identify the known conditions that contribute to it” (National Land Cover Database, 2008).

8.1.3.3 Continental United States Soil Geographic Database

The Continental United States Soil Geographic Database (CONUS STATSGO) is a digital general soil association map that has been developed by the National Cooperative Soil Survey and distributed by the USDA NRCS. It consists of broad based inventory of soils and non-soil areas that occur in a repeatable pattern on the landscape and that can be cartographically shown at scale and mapped. No information is readily available on the uncertainty associated with this dataset.

8.1.3.4 Soil Survey Geographic Database

The Soil Survey Geographic (SSURGO) database has been developed by the National Geospatial Management Center, formerly the National Cartography and Geospatial Center. The SSURGO database depicts information about the kinds of soils and distribution of soils on the landscape. This dataset is a digital soil survey and generally is the most detailed level of soil geographic data available. Uncertainty information was not readily available for this database beyond the disclaimer that the accuracy of data points ‘met national map accuracy standards.’

8.1.3.5 Ammonia Monitoring Network

The Ammonia Monitoring Network is part of the National Atmospheric Deposition Program (NADP), and was originally initiated by the U.S. State Agricultural Experiment Stations. The dataset provides consistent, long term record of ammonia gas concentrations in the United States, drawing from 50 monitoring sites across 37 states in total. Uncertainty was not directly addressed in the dataset materials, aside from the disclaimer that the NADP’s Central Analytical Laboratory (CAL) analyzes, quality assures, and provides the analytical data to the NADP (2011).

8.1.3.6 Parameter-elevation Regressions on Independent Slopes Model

The Parameter-elevation Regressions on Independent Slopes Model (PRISM) is a climate mapping system developed by the PRISM Climate Group. PRISM is a knowledge-based system that uses point measurements of precipitation, temperature, and other climatic factors to produce continuous, digital grid estimates of monthly, yearly, and event-based climatic parameters. No information is readily available on the uncertainty associated with this dataset.

8.1.3.7 Daymet Weather Dataset

Daymet is a weather model developed by Oak Ridge National Laboratory that provides interpolations extracted from daily meteorological observations onto a gridded dataset where no such observations are present. Daymet provides output parameters including temperature, precipitation, humidity, solar radiation, and snow water equivalent. The Daymet dataset is based on the spatial convolution of a truncated Gaussian weighting filter with the set of station locations. Sensitivity to the typical heterogeneous distribution of stations in complex terrain is accomplished with an iterative station density algorithm. The weather datasets are produced as outputs from the Daymet model run. This dataset is used as an input for estimating GHG emissions from croplands and grazing lands, and ammonia emissions from manure management. No information is readily available on uncertainty associated with this dataset.

8.1.3.8 North American Regional Reanalysis Weather Dataset

The DAYCENT model simulations use the North American Regional Reanalysis (NARR) data product for daily temperature and precipitation. The NARR dataset was chosen because it provides full, gap-filled coverage for the conterminous U.S. and is maintained and updated regularly. As described by Mesinger (2006), "The National Centers for Environmental Prediction (NCEP) North American Regional Reanalysis (NARR) is a long-term, dynamically consistent, high-resolution, high-frequency, atmospheric and land surface hydrology dataset for the North American domain. It covers the 25-year period 1979–2003, and is being continued in near-real time as the Regional Climate Data Assimilation System, R-CDAS. Essential components of the system used to generate NARR are the lateral boundaries from and the data used for the NCEP/DOE Global Reanalysis, the NCEP Eta Model and its Data Assimilation System, a recent version of the NOAA land surface model, and the use of numerous data sets additional to or improved compared to those of the Global Reanalyses. In particular, NARR has successfully assimilated high quality and detailed precipitation observations into the atmospheric analysis. Consequently, the forcing to the land surface model component of the system is more accurate than in previous reanalyses, so that NARR provides a much improved analysis of land hydrology and land-atmosphere interaction." No quantitative information is readily available on uncertainty associated with this dataset.

8.1.3.9 DAYCENT Land Management Data Sets

Data on past land use and management (prior to the year 2000) are the basis for representative cropland management systems, selected by the entity landowner, that are used to initialize ("spinup") the DAYCENT model for computing soil organic carbon stock changes. The attributes of the management systems are based primarily on three large datasets for the US: the National Resources Inventory (NRI), the National Agricultural Statistics Service (NASS) cropland surveys, and USDA Economic Research Service Cropping Practices Survey. The use of representative crop management systems for the DAYCENT initialization process introduces some uncertainty when applied to a specific farm or ranch entity (which has a unique management history that may be different from the regionally-based representative management histories specified by Major Land Resource Areas. However, the major uncertainty for the model initialization is driven by the timing of major land-cover change (e.g., conversion of grassland to cropland) which can be user-specified for the particular entity and land parcel.

National Resources Inventory. The NRI is an inventory of land cover and use, soil erosion, prime farmland, wetlands, and other natural resource characteristics. NRI was designed as a tool to assess conditions and trends for soil, water, and related natural resources primarily on non-Federal lands of the United States (Nusser and Goebel, 1997). The NRI is a stratified two-stage area sample of over several hundred thousand points distributed across the United States and Caribbean. Each point in the survey is assigned an area weight (i.e., expansion factor) based on other known areas and land-use information so that each point has a statistically assigned area that it represents (Nusser and Goebel, 1997). It should be noted that there is some uncertainty associated with scaling the point data to a region or the country using the expansion factors. In general, those uncertainties decline at larger scales, such as States compared to smaller county units, because of a larger sample size.

National Agricultural Statistics Service Crop Surveys. Data from the NASS county agricultural production surveys were used to construct representative crop rotations for the period prior to (i.e., before 1979) the data record in the NRI. NASS conducts thousands of surveys each year covering many facets of U.S. agriculture. Estimates include crop acreage, yield, production, irrigation, and livestock numbers. State-level crop estimates are available from as early as 1866 depending on the State and variable of interest. Some county-level crop data is available from as early as 1915, with most crops available for most States by about 1960. Data aggregated to the county level are subject to a high level of quality control, including data screening for outliers, double checking with primary data collectors and comparisons with other aggregate data sets such as from the USDA Farm Services Agency.

USDA Economic Research Service (ERS) Cropping Practices Survey. Ancillary data on historical management practices used in the DAYCENT model initialization include nitrogen fertilizer rates (USDA ERS, 1997; 2011). Mean fertilizer rates since 1990 were estimated for all major crops, summarized by ERS at the State-level. If a State was not surveyed for a particular crop or if there were not enough data to produce a State level estimate, then data were aggregated to USDA Farm Production Regions in order to estimate a mean and standard deviation for fertilization rates (Farm Production Regions are groups of States with similar agricultural commodities). Crop-specific regional fertilizer rates prior to 1990 were based largely on extrapolation or interpolation of fertilizer rates from the years with available data. For crops in some agricultural regions, little or no data were available, and, therefore, a geographic regional mean was used to simulate nitrogen fertilization rates (e.g., no data are available for the State of Alabama during the 1970s and 1980s for corn fertilization rates; therefore, mean values from the southeastern United States were used to simulate fertilization to corn fields). No uncertainty data are available for this dataset.

8.1.3.10 DNDC Input Datasets

The DeNitrification-DeComposition (DNDC) model is proposed to estimate GHG emissions and removals from wetlands systems. DNDC is a soil biochemistry model that simulates thermodynamic and reaction kinetic processes of carbon, nitrogen, and water driven by the plant and microbial activities in ecosystems (Olander and Haugen-Kozyra, 2011). The DNDC model relies on specific input datasets that can be categorized into five sources: (1) cropland/land-use data; (2) crop management data; (3) soils data; (4) weather data; and (5) atmospheric deposition data (Salas et al., 2012). These primary sources of data and uncertainty associated with the dataset are provided below.

National Agricultural Statistics Service Cropland Data Layer dataset. The DNDC model uses the NASS Cropland Data Layer as a source of cropland/land-use data. The NASS Cropland Data Layer is an online geospatial exploring tool generated from satellite image observations at a 30 meter resolution. NASS provides accuracy information and error matrices (total accuracy, errors of omission and co-mission), but not on an annual basis for crops and States.

NASS Agricultural Census. The census is available every five years, and used at the county scale. It provides information on U.S. farms and ranches and is the only source of uniform, comprehensive agricultural data at the county level. Farmers and ranchers are asked to respond to the census by mail or online. Information including production expenses, market value of products, and operation characteristics are a few of the categories of data. Uncertainty is not assessed for these data.

Remote Sensing. DNDC uses remote sensing to build regional databases on cropland on a project and as needed basis. The range of sensors used includes RapidEye, Landsat, MODIS, and SAR (PALSAR, Radarsat, ENVISAT, etc.). Remote sensing is used for estimating hydroperiods (i.e., where the water table is at any given time). As DNDC does not have a groundwater modeling component, remote sensing is used to identify when wetlands are flooded. Uncertainty is not assessed for these data.

USDA, ERS Agricultural Resource Management Survey (ARMS). ARMS data are used to populate the crop management component of the DNDC module. USDA ERS ARMS provide data on the financial condition, production practices, and resource use of farmers at the field level within the United States. ARMS data are released and/or revised twice a year. Uncertainty is not assessed for these data.

CONUS STATSGO (See description above). These data are used to associate soil types and uncertainty of soils data within the model.

SSURGO (See description above). SSURGO data are retrieved by DNDC via an automated retrieval script and extract four key soil attributes: clay content (texture), bulk density, organic matter (soil organic carbon), and pH.

NOAA National Climatic Data Center. DNDC uses station data from the NOAA National Climatic Data Center (NCDC) to input temperature, dew point, relative humidity, precipitation, wind speed and direction, visibility, and atmospheric pressure. Data are provided at the subhourly, hourly, daily, monthly, annual, and multiyear timescale. NCDC provides values that describe the range of the uncertainty, or simply "range," of each month, season, or year global temperature anomaly. These values can be used as plus/minus values within an overall Monte Carlo framework; however, a tool is needed to utilize this information.

Daymet (See description above). These weather data are used by DNDC and have been available for much of North America from 1980 to 2012. Uncertainty information is not available for this dataset.

National Aeronautics and Space Administration Modern-Era Retrospective Analysis for Research and Applications (MERRA). The DNDC model relies on MERRA satellite data as input for the hydrological cycle. MERRA provides global data on various aspects of moisture distribution and variability. Nearly 30 years of data are available and has undergone an online bias correction for satellite radiance observations. This was done to calibrate observations from different satellites. Uncertainty data are not available for MERRA output.

National Oceanic and Atmospheric Administration National Center for Environmental Prediction (NCEP). DNDC inputs NCEP national weather, water, and climate data into the NCEP model. NCEP creates climate, water, ocean, space, and environmental hazard outputs. Uncertainty data are not available for NCEP output.

National Atmospheric Deposition Program National Trends Network (NTN) Stations. DNDC requires total nitrogen deposition and estimates of average concentration. DNDC relies on the NADP NTN stations to input total nitrogen deposition (NO_3 and NH_4) into the model. NADP NTN stations collect precipitation and chemistry samples away from urban area and point sources of pollution. The station's Central Analytical Laboratory reviews data for completeness and accuracy and flags samples that were mishandled or compromised. Sample data are further reviewed by the NADP program office to do a final check to resolve discrepancies. Once data are made available online, DNDC calculates mean nitrogen deposition for the simulation time period and incorporates the data into the project database. NADP NTN station data do not have associated uncertainty data available, however regional uncertainty was analyzed in a presentation by Dennis et al. (2011).

8.1.3.11 Approach for Incorporating Large Dataset Uncertainty

Among the large datasets to be used for the GHG quantification methods, only GTR-NE-343 has some quantified uncertainty information for use in a Monte Carlo assessment of net GHG emissions. Because confidence intervals for only two stock volumes are available, only a linear relationship can be modeled with GTR-NE-343 information, and no departures from linearity can be assessed. Further analysis of carbon density at other growing stock volumes requires computation of additional confidence intervals.

Given the lack of uncertainty information for most of the relevant large datasets, estimating this source of uncertainty is not feasible. Instead, reliance of the methods on the large datasets is explicitly acknowledged and readily available information on uncertainty is summarized as provided above.

Some large "wall-to-wall" datasets are formed via interpolation of existing data from a fixed set of measurement locations. For such datasets, a potential near-term next step might be to incorporate uncertainty by imputing measurements from randomly-selected measurement locations. This random selection could use probabilities inversely proportional to the distance between the measurement locations and the entity. If most locations are far from the entity, then the imputations are increasingly uncertain.

In the longer term, both new research and synthesis of existing research will be required to quantify large dataset uncertainty. Methods from geostatistics, for example, might be used to describe an uncertain large dataset obtained by interpolation.

8.1.4 Model Uncertainty

In the case of the external models, it is hard to appropriately account for approximation error and often only one model exists to represent or estimate emissions (or removals) from a specific activity or process. Since comparable models do not exist, it is almost impossible to estimate the uncertainty associated with using one particular model versus another. While this report specifies the use of several external models—DAYCENT, DNDC, FVS, i-Tree Canopy, i-Tree Eco, First Order Fire Effects Model (FOFEM)—given the above considerations, limited published data was found on external model uncertainty inherent with these models.

This section includes a description of the external models used for estimating carbon stocks and stock changes from the croplands and grazing lands, wetlands, and forestry sectors, uncertainty information obtained from the model developers. These models help provide a quantitative and geographical view into the emissions associated with a variety of factors from agricultural and forestry systems. For example, given inputs such as area, tree diameter, tree height, species, soil type, and geography, the suite of forestry models can provide emission estimates from fire disturbances, approximate changes in forest carbon stocks, or provide urban forest carbon stock data. Table 8-2 below summarizes the uncertainty information obtained from the model developers for each of the models used to estimate net GHG emissions. Given the lack of quantitative information on model uncertainty, this component of uncertainty will not be part of the Monte Carlo uncertainty assessment.

Table 8-2: Uncertainty Information for Process-based Models

Model	Availability of Uncertainty Documentation	Occurrence of Uncertainty Biases
DAYCENT	Ogle et al., 2010	Biases by practice are quantified in Ogle et al. (2010).
DNDC[a]	Input uncertainty: Li et al. (2002) and Zhang et al. (2009). There have been	A Monte Carlo approach or Most Sensitive Factor analysis can be run on certain input parameters

Model	Availability of Uncertainty Documentation	Occurrence of Uncertainty Biases
	no papers focused on quantification of DNDC model structural uncertainty.	(i.e., soil measurements) to assess the variability of the parameters (includes excerpts from C_AGG whitepaper by Salas et al., 2012).
Forest Vegetation Simulator	No published quantification of model uncertainty was found.	Exists but not quantifiable according to experts.
i-Tree Canopy (Aerial Data Method)	No published quantification of model uncertainty found.	Model bias is likely low, according to model developer.
i-Tree Eco (Field Data Method)	No published quantification of model uncertainty found.	Values are standardized, bias is minimized. Unknown bias for national density estimates.
First Order Fire Effects Model	No published quantification of model uncertainty found.	Regional biases (North Rocky Mountains, Pacific Northwest regions).

[a] DNDC does not provide uncertainty parameterization of outputs at the site level, however, the regional model provides an option for assessing uncertainty due to input uncertainty.

8.1.4.1 DAYCENT Model

The DAYCENT model has inherent uncertainty associated with predicting soil organic carbon (SOC) stock changes (Ogle et al., 2010; U.S. EPA, 2013). The uncertainty is associated with imperfect simulation of the plant and soil processes associated with the algorithms and parameters. To address this uncertainty, the simulated model predictions of SOC stocks need to be compared to measurements. The comparison leverages the scalability of the process-based model to the wide range of conditions that exist in agricultural lands, while having an underlying measurement basis to support the reporting (Conant et al., 2011).

The differences between measurements and simulated SOC stocks and stock changes have been analyzed using an empirically based approach in which a statistical model was developed that quantifies the accuracy and precision in the simulated predictions (Ogle et al., 2007). The linear mixed-effect modeling approach was used for this analysis, and various environmental conditions (e.g., climate and soil characteristics) and management practices were evaluated to determine if the model is more accurate or precise for particular conditions or management systems. The approach relied on measurements of SOC stocks from a network of sites across the U.S. agricultural lands. A network is currently being expanded by the USDA NRCS that is expected to provide additional measurements supporting the entity-scale methods for estimating SOC stock changes. This uncertainty analysis will be updated as new measurements become available from the network and will be incorporated into a Monte Carlo assessment.

8.1.4.2 DNDC Model

Structural uncertainty is related to the inherent uncertainty of a model that remains even if none of the input data had any variability. Estimating model structural uncertainty requires the use of independent validation data (i.e., field measurement data that were not used to develop the model algorithms). This approach requires not only access to sufficient independent field data, but also that the data include all the input data that DNDC requires. A number of validation tests with independent field data have been published although summary studies are currently not available to quantify DNDC structural uncertainty.

8.1.4.3 Forest Vegetation Simulator

As previously described, a source of uncertainty for the FVS model is sampling uncertainty associated with the tree list (the main user input). The additional uncertainty associated with the model uncertainty is difficult to quantify.

In the FVS model, diameter growth is the only variable that is considered stochastic. For the diameter growth module, a random seed is used for projections of changes in forest stands rather than using the mean diameter value to avoid underestimating growth. This process increases error propagation because the results of the diameter growth module are used to make further estimates in the model, e.g., using growth and yield equations (i.e., Jenkins equations). However, the stochasticity of diameter growth is not the main driver of model uncertainty. Uncertainty associated with the FVS model is complex because it is derived from 20 different regionally specific model variants that were developed independently. Each model run or analysis has to be calibrated to account for local tree variety and growth rates, introducing another level of complexity (Van Dyck, 2012). Additionally, errors may propagate from the bias in regional factors, adjusting to local geographies, climates, the use of field data, and sampling uncertainty. Given the overall complexity inherent in the model, FVS does not incorporate uncertainty in the output or post-analysis of model runs and additional research is required to quantify model uncertainty.

8.1.4.4 i-Tree Model

i-Tree (formerly the Urban Forest Effects model) is an urban forestry analysis model developed by David J. Nowak (USDA FS), Daniel E. Crane (NRS), and Patrick McHale (SUNY College of Environmental Science and Forestry). The i-Tree model helps quantify the structure of community trees and the environmental services that they provide. It provides six analytical tools including:

- **i-Tree Eco:** Provides a full picture of the entire urban forest (used in the Field Data Method)
- **i-Tree Streets:** Quantifies benefits from a municipalities street level trees
- **i-Tree Hydro:** Models the effects of trees on watershed stream flow and water quality
- **i-Tree Vue:** Uses NLCD satellite imagery to assess tree canopy
- **i-Tree Design:** Assesses multiple trees at parcel level
- **i-Tree Canopy:** Provides a quantifiable estimate of tree cover and other land cover types (using in the Aerial Data Method)

i-Tree Eco and i-Tree Canopy are recommended in this report for use by an entity to estimate the change in carbon stocks in their urban forests.

i-Tree Eco Uncertainty Information: The i-Tree Eco model produces uncertainty estimates based on sampling error, but it does not calculate a model estimation error. According to i-Tree developers, estimation error is based on the uncertainty inherent in the biomass conversion equations and emission factors. The developers also note that model bias is likely low given that the input assumes a given random sample of trees, and tree species equations are selected based on stand height. If a particular species equation is not available the model uses the average of available equations from the closest genera (Nowak, 2012). A Monte Carlo analysis tool could use the standard error of the estimated carbon value to evaluate the uncertainty associated with an entity's total net GHG emissions.

i-Tree Canopy Uncertainty Information: The i-Tree Canopy model produces a statistical estimate of the standard error of the percent tree cover estimate based on the ratio of sample points classified as trees to total sample points. In i-Tree Canopy the user imports a shape file, samples points, and classifies them as either trees or non-trees. An analysis of the tree point to total point ratio is used to estimate the standard error associated with the percent tree cover estimate, as described in the i-Tree Canopy technical notes,[18] and shown in Equation 8-1 below.

Equation 8-1: Estimating Standard Error of Percent Tree Cover from i-Tree Canopy

$$SE = \sqrt{pq/N} \text{ (e.g., } \sqrt{0.33 \times 0.67/1000} = 0.0149\text{)}$$

Where:

N= Total number of sampled points (e.g., 1,000)

n = Total number of points classified as a tree (e.g., 330)

p = n/N (e.g., 330/1,000 = 0.33)

q = 1 − p (e.g., 1 − 0.33 = 0.67)

Table 8-3 shows estimates of the standard error as related to the ratio of tree points to total sample points (p value), where the total number of sampled points (N) equals 1,000.

Based on the standard error formula, standard error is greatest when p equals 0.5, and is least when p is very small or very large (see Table 8-3). A Monte Carlo analysis tool could use the standard error of the estimated percent tree cover value to evaluate the uncertainty associated with an entity's total net GHG emissions.

Table 8-3: Estimates of Standard Error (SE) (N = 1,000) of Percent Tree Cover from i-Tree Canopy with Varying p Values

p	SE
0.01	0.0031
0.1	0.0095
0.3	0.0145
0.5	0.0158
0.7	0.0145
0.9	0.0095
0.99	0.0031

8.1.4.5 First Order Fire Effects Model

FOFEM is a computational model for predicting tree mortality, fuel consumption, smoke production, and soil heating caused by either prescribed fire or wildfire. FOFEM was developed by the Intermountain Fire Sciences Laboratory in Missoula, MT, of the USDA Forest Service. First order fire effects are those characterized with the direct immediate consequences of a fire including GHG emission estimates. FOFEM is divided into four national regions: Pacific West, Interior West, North East, and South East. The model includes several forest cover types to provide an additional level of detail resolution. The quantitative output can be used in assessments after fire damage, in analyzing prescribed fire impacts, and modeling vulnerabilities in regional forest groups.

FOFEM has a regional bias given that the empirical relationships and assumptions are based on forested systems in the North Rocky Mountains and the Pacific Northwest. However, these uncertainties are not quantified or adjusted for use in different regions. For instance, Southeast fires burn well at humidity levels that would not support them in the West. This phenomenon is not accounted for in the model and there is no uncertainty quantification around the output. There are also material differences such as litter bulk density that influences consumption and emission which can vary considerably region to region (Lutes, 2012).

[18] I-Tree Canopy Technical Notes: http://www.itreetools.org/canopy/resources/iTree_Canopy_Methodology.pdf

8.1.4.6 Approach for Incorporating Model Uncertainty

Given the lack of uncertainty information for most of the relevant external models, it is not currently feasible for the GHG quantification methods to quantify this source of uncertainty. Instead, reliance of the methods on the models will be explicitly acknowledged. The potential impacts of uncertain models on the accuracy and precision of the resulting estimates is described qualitatively in the previous sections.

It may be possible in the near term to elicit expert judgments on the level of model uncertainty at the entity level. Models used in the GHG quantification methods are typically constructed at scales no smaller than the entity level. It is expected that the model uncertainty at the entity level would be no smaller than the model uncertainty at the model's scale, and possibly larger due to additional error from downscaling to the entity level.

In the longer term, more research is needed to evaluate model predictions with independent data, not used in the development of the model. The differences between model predictions and independent data are the best possible source of information regarding model uncertainty.

8.2 Research Gaps

The readily available information on parameter uncertainty is provided in the tables in Appendix 8-B. As indicated, much of the information to characterize the uncertainty is not available and the data that are provided are mostly default values from the literature and assumed probability density functions. To conduct a Monte Carlo analysis for uncertainty estimation, it is important to obtain probability density functions or summary statistics for all uncertain variables. Significant research is needed to obtain new data and to synthesize existing and new data in order to truly assess uncertainty associated with a range of factors causing uncertainty in the GHG estimates developed using the recommended methods described in this report. In particular, more research is needed to assess parameter, sampling, large data sets, and model uncertainties.

Appendix 8-A: Example Output File from FVS Sampling Uncertainty Bootstrapping Application FVSBoot (as provided in Gregg and Hummel, 2002)

The following table illustrates standard deviation surrounding the sampling error of the Basal Area outputs. FVSBoot can be configured to determine standard deviation of the sampling error for any FVS output.

Table 8-A-1: Example Output File from FVS Sampling Uncertainty Bootstrapping Application FVSBoot (as provided in Gregg and Hummel, 2002)

```
Data from FVS Model:  SUMMARY STATISTICS.

    Stand ID      = 1022144
    Management ID = NONE

 FVS Variable           =  Cycle( 3), BA

  FVS-PI
    Mean                =      129.56
    Number of samples   =         201
    Standard Deviation  =       1.295

  SEPI:
    Number of samples   =         500
    Mean                =      131.87
    Standard Deviation  =       14.00
    Bootstrap Median    =      132.00
    Max outcome         =      175.00
    Min outcome         =       93.00
    Range of outcomes   =       82.00

BOOTSTRAP SAMPLING ERROR PREDICTION INTERVALS

     Variable             Mean    Percent  Lower    Upper
  ----------------      --------   -----  -------- --------
  Cycle( 3), BA          129.56     68    118.00    146.00
                                    80    114.00    150.00
                                    90    109.00    157.00
                                    95    105.00    162.00
                                    99     99.00    173.00
  -----------------------------------------------------------

 Frequency distribution for (   500 ) bootstrap samples for "Cycle( 3), BA" from FVS.

 Interval  Midpoints   Counts
 --------  ---------   ---+------------------------------------------------------------
     1       95.05      2 |II
     2       99.15      3 |III
     3      103.25      9 |IIIIIIIII
     4      107.35     13 |IIIIIIIIIIIII
     5      111.45     20 |IIIIIIIIIIIIIIIIIIII
     6      115.55     31 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
     7      119.65     44 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
     8      123.75     40 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
     9      127.85     50 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
    10      131.95     83 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
    11      136.05     54 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
    12      140.15     43 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
    13      144.25     31 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
    14      148.35     31 |IIIIIIIIIIIIIIIIIIIIIIIIIIIIIII
```

Appendix 8-B: Uncertainty Tables

This section presents readily available data on the uncertainty associated with activity and emission factor data. Table 8-B-1 lists the data elements that are provided in the subsequent tables for each agriculture system. In particular, readily available uncertainty information is provided in the following tables:

- Table 8-B-2: Cropland Uncertainty Template
- Table 8-B-3: Animal Population Uncertainty Template
- Table 8-B-4: Enteric Fermentation and Housing Uncertainty Template
- Table 8-B-5: Manure Management Uncertainty Table
- Table 8-B-6: Forestry Uncertainty Table

Table 8-B-1: Data Elements Provided

Column Label	Description
Data Element Name	The name of the variable
Abbreviation / Symbol	The shorthand representation used in the report
Emission Type	Emissions estimates that depend on the data element (CH_4, N_2O, NH_3, CO_2)
Data Input Unit	Unit associated with the data element
Input Source	Entity entry, default entry, model output, or from a database
Statistic	Available statistic for the parameter
Type of Statistic	Mean, median, or mode
Probability Distribution Type	The probability distribution function of the data element (normal, lognormal, uniform, triangular, beta)
Relative Uncertainty	Range of values around the most likely value, expressed as a percent of the most likely value
Confidence Level	The probability that the confidence range captures the true value of the data element given a distribution of samples.
Effective Lower Limit	Minimum value for data element (excluding outliers)
Effective Upper Limit	Maximum value for data element (excluding outliers)
Data Source	Reference for information related to the data element and associated uncertainty

Table 8-B-2: Cropland Uncertainty Template

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Croplands – Multiple Sub-sources	Area	A	CH_4, N_2O, CO_2	Hectares	Entity Entry									
Croplands – Multiple Sub-sources	Crop Yield	Y	CH_4, N_2O, CO_2	Metric tons dry matter crop yield/ year	Entity Entry									
Croplands – Multiple Sub-sources	Meat yield per parcel of land	Meat	CO_2	kg carcass yield	Entity Entry									
Croplands – Multiple Sub-sources	Milk production per parcel of land	Milk Prod.	CO_2	kg fluid milk yield	Entity Entry									
Biomass Carbon Stock Changes	Mean annual woody biomass (t=current year's stocks)	W_t	CO_2	Metric tons CO_2-eq year^{-1}	Model Output	DAYCENT model simulations and growth functions for agro-forestry								
Biomass Carbon Stock Changes	Mean annual woody biomass (t-1=Previous year's stocks)	W_{t-1}	CO_2	Metric tons CO_2-eq year^{-1}	Model Output	DAYCENT model simulations and growth functions for agro-forestry								
Biomass Carbon Stock Changes	Mean annual herbaceous biomass (t=current year's stocks)	H_t	CO_2	Metric tons CO_2-eq year^{-1}	Entity Entry									
Biomass Carbon Stock Changes	Mean annual herbaceous biomass (t-1=Previous year's stocks)	H_{t-1}	CO_2	Metric tons CO_2-eq year^{-1}	Entity Entry									

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Biomass Carbon Stock Changes	Forage Yield	Forage yield for grazing lands	CO_2	Metric tons dry matter per hectare	Entity Entry									
Biomass Carbon Stock Changes	Number of trees by age of diameter class for each agroforestry practice	Number of Trees	CO_2	Number	Entity Entry									
Biomass Carbon Stock Changes	Diameter at breast height for a subsample of trees	DBH	CO_2	Meters	Entity Entry									
Biomass Carbon Stock Changes	Root to Shoot Ratio	R:S	CO_2	Ratio	Default Entry									West et al. (2010)
Biomass Carbon Stock Changes	Dry matter content of harvested crop biomass or forage	DM	CO_2	Dimensionless	Entity Entry									
Biomass Carbon Stock Changes	Harvest Index	HI	CO_2	Fraction	Default Entry									West et al. (2010)
Biomass Carbon Stock Changes	Crop harvest or forage yield, corrected for moisture content	Y_{dm}	CO_2	Metric tons biomass ha^{-1}	Entity Entry									
Biomass Carbon Stock Changes	Approximate fraction of calendar year representing the growing season	Y_f	CO_2	Fraction	Entity Entry									
Biomass Carbon Stock Changes	Carbon fraction of aboveground biomass	C	CO_2	Fraction	Default Entry	0.45		Normal	11.0	11.0				IPCC (1997)
CO_2 from Liming	Annual application of lime	M	CO_2	Metric tons/ year	Entity Entry									

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
CO_2 from Liming	Metric tons CO_2 emissions per metric tons of lime	EF	CO_2	Metric tons C/metric tons lime	Default Entry	-0.04			46.0	46.0				West and McBride (2005)
CO_2 from Urea Fertilizer Application	Annual amount of urea fertilization	M	CO_2	Metric tons urea/year	Entity Entry									
CO_2 from Urea Fertilizer Application	Proportion of C in urea	EF	CO_2	Metric tons C/metric tons urea	Default Entry	0.2								de Klein et al. (2006)
Direct N_2O Emissions	Area of organic soils (histosols) drained on a parcel of land (ha)	A_{os}	N_2O	ha	Entity Entry									
Direct N_2O Emissions	Prior-year crop type		N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									
Direct N_2O Emissions	Emission rate modeled at 0 level of N input (N_t =0)	ER_0	N_2O	Metric tons N_2O-N $ha^{-1}year^{-1}$	Model Output									
Direct N_2O Emissions	Emission factor for the typical fertilization rate	$EF_{typical}$	N_2O	Metric tons N_2O-N metric $tons^{-1}$ N	Model Output									
Direct N_2O Emissions	Typical N fertilizer rate	N_{tf}	N_2O	Metric tons N ha^{-1} $year^{-1}$	Database									
Direct N_2O Emissions	Emission rate for the typical case modeled	$ER_{typical}$	N_2O	Metric tons N ha^{-1} $year^{-1}$	Model Output									
Direct N_2O Emissions	Actual N fertilizer rate, including synthetic and organic	N_f	N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									

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Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Direct N_2O Emissions	Base EF scalar for ΔN_f > zero and non-grassland crops	S_{EF}	N_2O	(metric tons N_2O-N (metric tons N)$^{-2}$) ha year	Default Entry	0.0274								Appendix 3-A
Direct N_2O Emissions	Base EF scalar for ΔN= > zero and grassland	S_{EF}	N_2O	(metric tons N_2O-N (metric tons N)$^{-2}$) ha year	Default Entry	0.117								Appendix 3-A
Direct N_2O Emissions	Base EF scalar for ΔN_f < zero	S_{EF}	N_2O	(metric tons N_2O-N (metric tons N)$^{-2}$) ha year	Default Entry	0								Appendix 3-A
Direct N_2O Emissions	Dry matter content of harvested biomass	DM	N_2O		Entity Entry									
Direct N_2O Emissions	Residue:yield ratios		N_2O	Ratio	Entity Entry									
Direct N_2O Emissions	Amount of residue harvested, burned or grazed	R_r	N_2O		Entity Entry									
Direct N_2O Emissions	Fraction of live biomass removed by grazing	F_r	N_2O		Entity Entry									
Direct N_2O Emissions	N mineralization from manure	N_{man}	N_2O		Entity Entry and Model Output									
Direct N_2O Emissions	N mineralization from compost	N_{comp}	N_2O		Entity Entry and Model Output									
Direct N_2O Emissions	Total dry matter yield of crop		N_2O	Metric tons dry matter year^{-1}	Entity Entry									
Direct N_2O Emissions	Stocking rates and methods		N_2O	Head/acre	Entity Entry									

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Direct N_2O Emissions	Scaling factor for slow-release fertilizers, 0 where no effect	S_{sr}	N_2O	Dimensionless	Default Entry	-0.21		Normal				-0.3	-0.12	Meta-analysis
Direct N_2O Emissions	Scaling factor nitrification inhibitors – semi-arid/arid climate	S_{inh}	N_2O	Dimensionless	Default Entry	-0.38		Normal				-0.51	-0.21	Meta-analysis
Direct N_2O Emissions	Scaling factor nitrification inhibitors – mesic climate	S_{inh}	N_2O	Dimensionless	Default Entry	-0.4		Normal				-0.52	-0.24	Meta-analysis
Direct N_2O Emissions	Scaling factor for no till, semi-arid/arid climate, <10 years following no-till adoption	S_{till}	N_2O	Dimensionless	Default Entry	0.38						0.04	0.72	van Kessel et al. (2012); Six et al. (2004)
Direct N_2O Emissions	Scaling factor for no till, semi-arid/arid climate, ≥10 years following no-till adoption	S_{till}	N_2O	Dimensionless	Default Entry	-0.33						-0.5	-0.16	van Kessel et al. (2012); Six et al. (2004)
Direct N_2O Emissions	Scaling factor for no till, mesic/wet climate, <10 years following no-till adoption	S_{till}	N_2O	Dimensionless	Default Entry	-0.015						-0.16	0.16	van Kessel et al. (2012); Six et al. (2004)
Direct N_2O Emissions	Scaling factor for no till, mesic/wet climate, ≥10 years following no-till adoption	S_{till}	N_2O	Dimensionless	Default Entry	-0.09						-0.19	0.01	van Kessel et al. (2012); Six et al. (2004)

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Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Direct N_2O Emissions	N in slow-release N fertilizer applied to the parcel of land	N_{sr}	N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									
Direct N_2O Emissions	N in manure amendments (and sewage sludge) added to the parcel	N_{man}	N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									
Direct N_2O Emissions	N excreted by cattle, poultry and swine directly on the parcel of land (metric tons N year-1 ha-1)	N_{prp}	N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									
Direct and Indirect N_2O Emissions	N in synthetic fertilizer applied to a parcel of land	Nsfert	N_2O	Metric tons N $year^{-1}$ ha^{-1}	Entity Entry									
Direct N_2O Emissions	N from a change in soil organic matter mineralization due to LUC or tillage change applied to a parcel of land	N_{min}	N_2O	Metric tons N $year^{-1}$	Entity Entry	DAYCENT model derived								
Direct N_2O Emissions	N fraction of aboveground biomass for the crop or forage	N_a	N_2O	Dimensionless	Entity Entry									
Direct N_2O Emissions	N fraction of belowground biomass for the crop or forage	N_b	N_2O	Dimensionless	Entity Entry									

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Indirect N_2O Emissions	Emission rate for cropped histosols,	ER_{OS}	N_2O	Metric tons N_2O-N ha^{-1} $year^{-1}$	Default Entry	0.008		Uniform				0.002	0.024	IPCC (2006)
Indirect N_2O Emissions	N fertilizer applied of organic origin including manure, sewage sludge, compost and other organic amendments	F_{ON}	N_2O	Metric tons N $year^{-1}$	Entity Entry									
Indirect N_2O Emissions	Fraction of NSN that volatizes as NH3 and NOx	FR_{SN}	N_2O	kg N kg^{-1} Nsfert	Default Entry	0.1		Uniform				0.03	0.3	IPCC (2006)
Indirect N_2O Emissions	Fraction or proportion of F_{ON} that volatizes as NH_3 and NO_x	FR_{ON}	N_2O	kg N kg^{-1} NON	Default Entry	0.2		Uniform				0.05	0.5	IPCC (2006)
Indirect N_2O Emissions	Emission factor for volatilized N or proportion of N volatilized as NH_3 and NO_x that is transformed to N_2O in receiving ecosystem	EF_{vol}	N_2O	kg N_2O-N kg^{-1} N	Default Entry	0.01		Uniform				0.002	0.05	IPCC (2006)
Indirect N_2O Emissions	Fraction or proportion of Nt that leaches or runs off	FR_{Leach}	N_2O	kg N kg^{-1} N	Default Entry	0.3		Uniform				0.1	0.8	IPCC (2006)

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Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Indirect N_2O Emissions	Emission factor for leached and runoff N or proportion of leached and runoff N that is transformed to N_2O in receiving ecosystem	EF_{Leach}	N_2O	kg N_2O-N kg^{-1} N	Default Entry	0.0075		Uniform				0.0005	0.02 5	IPCC (2006)
Methane from Wetland Rice Cultivation	Cultivation period for rice under i, j, and k conditions	t_{ijk}	CH_4	Days	Entity Entry									
Methane from Wetland Rice Cultivation	Annual harvested area of rice for i, j, and k conditions	A_{ijk}	CH_4	Hectares/ year	Entity Entry									
Methane from Wetland Rice Cultivation	Application rate of organic amendment(s)	ROA_i	CH_4	Metric tons/ hectare	Entity Entry									
Methane from Wetland Rice Cultivation	Baseline emission factor for continuously flooded fields without organic amendments	EF_c	CH_4	kg CH_4/ha/ day	Default Entry	1.3		Uniform				0.8	2.2	IPCC (2006)
Methane from Wetland Rice Cultivation	Water regime during the cultivation period – Scaling Factor	SF_w for continuously flooded	CH_4	Scaling Factor from IPCC	Default Entry	1		Uniform				0.79	1.26	IPCC (2006)
Methane from Wetland Rice Cultivation	Water regime during the cultivation period – Scaling Factor	SF_w for single aeration	CH_4	Scaling Factor from IPCC	Default Entry	0.6		Uniform				0.46	0.8	IPCC (2006)

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Methane from Wetland Rice Cultivation	Water regime during the cultivation period – Scaling Factor	SF_w for multiple aerations	CH_4	Scaling Factor from IPCC	Default Entry	0.52		Uniform				0.41	0.66	IPCC (2006)
Methane from Wetland Rice Cultivation	Water regime before the cultivation period – Scaling factor	SF_p for non-flooded pre-season <180 days	CH_4	Scaling Factor from IPCC	Default Entry	1		Uniform				0.88	1.14	IPCC (2006)
Methane from Wetland Rice Cultivation	Water regime before the cultivation period – Scaling factor	SF_p for non-flooded pre-season > 180 days	CH_4	Scaling Factor from IPCC	Default Entry	0.68		Uniform				0.58	0.8	IPCC (2006)
Methane from Wetland Rice Cultivation	Water regime before the cultivation period – Scaling factor	SF_p for flooded pre-season > 30 days	CH_4	Scaling Factor from IPCC	Default Entry	1.9		Uniform				1.65	2.18	IPCC (2006)
Methane from Wetland Rice Cultivation	Organic amendment used – scaling factor	SF_o	CH_4	Scaling Factor from IPCC	Default Entry									
Methane from Wetland Rice Cultivation	Organic amendment conversion factor	$CFOA_i$ for straw incorporation less than 30 days before cultivation	CH_4	Conversion factor from IPCC	Default Entry	1		Uniform				0.97	1.04	IPCC (2006)
Methane from Wetland Rice Cultivation	Organic amendment conversion factor	$CFOA_i$ for straw incorporation more than 30 days before cultivation	CH_4	Conversion factor from IPCC	Default Entry	0.29		Uniform				0.2	0.4	IPCC (2006)

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Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Methane from Wetland Rice Cultivation	Organic amendment conversion factor	$CFOA_i$ for compost	CH_4	Conversion factor from IPCC	Default Entry	0.05		Uniform				0.01	0.08	IPCC (2006)
Methane from Wetland Rice Cultivation	Organic amendment conversion factor	$CFOA_i$ for farm yard manure	CH_4	Conversion factor from IPCC	Default Entry	0.14		Uniform				0.07	0.2	IPCC (2006)
Methane from Wetland Rice Cultivation	Organic amendment conversion factor	$CFOA_i$ for green manure	CH_4	Conversion factor from IPCC	Default Entry	0.5		Uniform				0.3	0.6	IPCC (2006)
Methane Uptake by Soils	Potential CH_4 Oxidation in soils	PCH_4 for grassland	CH_4	kg CH_4 ha^{-1} $year^{-1}$	Default Entry	3.2		Normal				0	6.9	Del Grosso et al. (2000)
Methane Uptake by Soils	Potential CH_4 Oxidation in soils	PCH_4 for coniferous forest	CH_4	kg CH_4 ha^{-1} $year^{-1}$	Default Entry	2.8		Normal				0.1	5.5	Del Grosso et al. (2000)
Methane Uptake by Soils	Potential CH_4 Oxidation in soils	PCH_4 for deciduous forest	CH_4	kg CH_4 ha^{-1} $year^{-1}$	Default Entry	11.8		Normal				1.9	21.6	Del Grosso et al. (2000)
Methane Uptake by Soils	CH_4 oxidation attenuation factor: cropland including set-aside (CRP) grassland, grazing land, and fertilized or recently harvested forests	AF	CH_4	N/A	Default Entry	0.30		Normal				0.07	1	Smith et al. (2000)
Methane Uptake by Soils	CH_4 oxidation attenuation factor: natural vegetation, 0-100 years after abandonment of agricultural production or timber harvest	AF	CH_4	N/A	Default Entry	0.3 + (0.007 × years since abandonment)		Normal				0.07 + (0.007 × years since abandonment)	1	Smith et al. (2000)

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Methane Uptake by Soils	CH_4 oxidation attenuation factor:>100 years post-management or never used for agricultural management or timber harvest	AF	CH_4	N/A	Default Entry	1		Normal				0.07	1	Smith et al. (2000)
N_2O from Wetland Rice	Total N inputs from all agronomic sources: mineral fertilizer, organic amendments, residues, and additional mineralization from LUC or tillage change (metric tons N year^{-1})	N_t	N_2O	Metric tons N year^{-1}	Entity Entry									
N_2O from Wetland Rice	Fertilizer N management		N_2O	Rate	Entity Entry									
N_2O from Wetland Rice	Organic fertilizer N		N_2O	% N	Entity Entry									
N_2O from Wetland Rice	Crop residue N		N_2O	% N	Entity Entry									
N_2O from Wetland Rice	Emission factor or proportion of N_t transformed to N_2O	EF	N_2O	kg N_2O-N (kg N)$^{-1}$	Default Entry	0.0022		Normal	0.2%	0.2%				Akiyama et al. (2005)
N_2O from Wetland Rice	Scaling factor to account for drainage effects	SF_D for continuously flooded systems	N_2O	Dimensionless	Default Entry	0								Akiyama et al. (2005)

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
N_2O from Wetland Rice	Scaling factor to account for drainage effects	SF_D for aerated systems	N_2O	Dimensionless	Default Entry	0.59		Normal	0.4%	0.4%				Akiyama et al. (2005)
Non CO_2 Emissions Biomass Burn	Boreal Forest (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.34		Normal	102%	102%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Wildfire	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.4		Normal	340%	340%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Crown fire	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.43		Normal	104%	104%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Surface fire	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.15		Normal	96%	96%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Post logging slash burn	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.33		Normal	130%	130%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Temperate Forest (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.45		Normal	51%	51%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Post logging slash burn	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.62		Normal	264%	264%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Shrublands (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.72		Normal	147%	147%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	*Calluna* health	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.71		Normal	121%	121%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Fynbos	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.61		Normal	195%	195%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Savanna woodlands (early dry season burns) (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.4		Normal	93%	93%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Savanna woodlands (mid/late dry season burns) (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.74		Normal	99%	99%				IPCC (2006)

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Non CO_2 Emissions Biomass Burn	Savanna woodland (mid/late)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.72		Normal	270%	270%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Tropical savanna	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.73		Normal	598%	598%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Other savanna woodlands	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.68		Normal	931%	931%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Savanna grasslands (early dry season burns) (all)	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.74		Normal	183%	183%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Tropical/sub-tropical grassland	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.74		Normal	270%	270%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Tropical/sub-tropical grassland	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.92		Normal	151%	151%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Tropical pasture	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.35		Normal	427%	427%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Savanna	Combustion Efficiency (C)	CH_4/N_2O		Default Entry	0.86		Normal	85%	85%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Emission factor	EF for grassland	CH_4	g GHG/kg burned biomass	Default Entry	2.3	Mean	Normal	8.0%	8.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Emission factor	EF for crop residue	CH_4	g GHG/kg burned biomass	Default Entry	2.7	Mean	Normal	50.0%	50.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Emission factor	EF for grassland	N_2O	g GHG/kg burned biomass	Default Entry	0.21	Mean	Normal	93.0%	93.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Emission factor	EF for crop residue	N_2O	g GHG/kg burned biomass	Default Entry	0.07	Mean	Normal	50.0%	50.0%				IPCC (2006)

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Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Non CO_2 Emissions Biomass Burn	Combustion efficiency	C for shrublands	CH_4/N_2O	% Burned	Default Entry	0.72	Mean	Normal	68.0%	68.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Combustion efficiency	C for grasslands with early season burns	CH_4/N_2O	% Burned	Default Entry	0.74	Mean	Normal	50.0%	50.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Combustion efficiency	C for grasslands with mid to late season burns	CH_4/N_2O	% Burned	Default Entry	0.77	Mean	Normal	66.0%	66.0%				IPCC (2006)
Non CO_2 Emissions Biomass Burn	Combustion efficiency	C for small grains	CH_4/N_2O	% Burned	Default Entry	0.9	Mean	Normal	50.0%	50.0%				Expert Assessment
Non CO_2 Emissions Biomass Burn	Combustion efficiency	C for large grain and other crop residues	CH_4/N_2O	% Burned	Default Entry	0.8	Mean	Normal	50.0%	50.0%				Expert Assessment
Non CO_2 Emissions Biomass Burn	Moisture content of residues and forage		CH_4/N_2O	% moisture	Default Entry									
Non CO_2 Emissions Biomass Burn	Residue to yield ratio of crop	R:Y	CH_4/N_2O	Metric tons residue / metric tons dry matter yield	Default Entry									
SOC Change Mineral Soils	Soil organic C stock at the end of the year	SOC_t	CO_2	Metric tons C ha^{-1}	Model Output	DAYCENT Model derived								Ogle et al. (2007); EPA (2013)

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
SOC Change Mineral Soils	Soil organic C stock at the beginning of the year	SOC_{t-1}	CO_2	Metric tons C ha^{-1}	Model Output	DAYCENT Model derived								Ogle et al. (2007); EPA (2013)
SOC Change Mineral Soils	Crop selection and Rotation Sequence		CO_2	Management List Developed by Experts	Entity Entry									
SOC Change Mineral Soils	Irrigation application rate		CO_2	Gallons per minute	Entity Entry									
SOC Change Mineral Soils	Mineral Fertilizer application Rate		CO_2	lbs/square foot	Entity Entry									
SOC Change Mineral Soils	Lime Amendment application Rate		CO_2	lbs/square foot	Entity Entry									
SOC Change Mineral Soils	Organic Amendment application Rate		CO_2	lbs/square foot	Entity Entry									
SOC Change Mineral Soils	Number of passes in each operation		CO_2	Number	Entity Entry									
SOC Change Mineral Soils	Depth of drainage		CO_2	Meters	Entity Entry									
SOC Change Mineral Soils	Length of field		CO_2	Meters	Entity Entry									
SOC Change Mineral Soils	Historical Weather Patterns		CO_2	PRISM Weather Data	Model Output									
SOC Change Mineral Soils	Physical and Chemical Properties of Soil		CO_2	NRCS SURRGO database	Model Output									
SOC Change Mineral Soils – Grazing Land	Animal Size used for grazing		CO_2	lbs	Entity Entry									

Cropland Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
SOC Change Mineral Soils – Grazing Land	Stocking Rate		CO_2	Head/acre	Entity Entry									
SOC Change Mineral Soils – Grazing Land	Irrigation application rate		CO_2	Gallons per minute	Entity Entry									
SOC Change Mineral Soils – Grazing Land	Mineral Fertilizer application Rate		CO_2	lbs/square foot	Entity Entry									
SOC Change Mineral Soils – Grazing Land	Depth of drainage		CO_2	Meters	Entity Entry									
SOC Change Organic Soils	Emission factor	EF for cropland in cool temperate regions	CO_2	Metric tons C $ha^{-1} year^{-1}$	Default Entry	11	Mean	Normal	45.0	45.0				Ogle et al. (2003)
SOC Change Organic Soils	Emission factor	EF for cropland in warm temperate regions	CO_2	Metric tons C $ha^{-1} year^{-1}$	Default Entry	14	Mean	Normal	35.0	35.0				Ogle et al. (2003)
SOC Change Organic Soils	Emission factor	EF for cropland in subtropical regions	CO_2	Metric tons C $ha^{-1} year^{-1}$	Default Entry	14	Mean	Normal	46.0	46.0				Ogle et al. (2003)
SOC Change Organic Soils	Emission factor	EF for grazing land in cool temperate regions	CO_2	Metric tons C $ha^{-1} year^{-1}$	Default Entry	2.8	Mean	Normal	45.0	45.0				Ogle et al. (2003)
SOC Change Organic Soils	Emission factor	EF for grazing land in warm temperate regions	CO_2	Metric tons C $ha^{-1} year^{-1}$	Default Entry	3.5	Mean	Normal	35.0	35.0				Ogle et al. (2003)
SOC Change Organic Soils	Emission factor	EF for grazing land in subtropical regions	CO_2	Metric tons C ha-1 year-1	Default Entry	3.5	Mean	Normal	46.0	46.0				Ogle et al. (2003)

Table 8-B-3: Animal Population Uncertainty Template

Animal Population Data Element Name	Abbreviation/ Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative Uncertainty Low (%)	Relative Uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Number of Animals													
Beef replacement heifers	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Dairy replacement heifers	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Mature beef cows	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Steers (>500 lbs)	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Bulls	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Stockers (All)	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Cattle on feed	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Dairy cow	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Cattle	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
American bison	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Sheep NOF	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Feedlot sheep	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Goats	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Horses	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment
Mules/burros/asses	N	CH_4, N_2O		Entity Entry				-1.0%	1.0%				Expert Assessment

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Table 8-B-4: Enteric Fermentation and Housing Uncertainty Template

Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Daily Milk Production	Milk	CH_4	kg milk/animal/day	Entity Entry				3%	5%				Expert Assessment
Days in milk	DIM	CH_4	Days	Entity Entry									
Dry matter intake	DMI	CH_4	kg/animal/day	Entity Entry									
Daily Work Done by Animal	Work	CH_4	Hours/day	Entity Entry									
Average live body weight – lactating beef cows	BW	CH_4	kg	Entity Entry									
Beef Cow Mature Weight	MW	CH_4	lbs	Entity Entry									
Steer Daily Weight Gain to 24 months	WG	CH_4	lbs/day	Entity Entry									
Beef Steer Mature Weight	MW	CH_4	lbs	Entity Entry									
Beef Heifer Mature Weight	MW	CH_4	lbs	Entity Entry									
Net energy required by the animal for maintenance	NE_m	CH_4	MJ day^{-1}	Entity Entry									
Milk Fat Content	Fat	CH_4	Percent	Entity Entry									
Starch Content of Diet (Dairy Cows)	Starch	CH_5	kg/animal/day	Entity Entry									
Acid Detergent Fiber Content of Diet	ADF	CH_4	kg/head/day	Entity Entry									
DE – Each Feed Type	DE	CH_4	Percent of gross energy	Entity Entry									
Neutral Detergent Fiber in Diet (Dairy Cows)	NDF	CH_4	Percent	Entity Entry									
Crude Protein in Diet	CP	CH_4	Percent	Entity Entry									
Acid Detergent Fiber Content of Diet (Dairy Cows)	ADF	CH_4	Percent	Entity Entry									
Neutral Detergent Fiber in Diet	NDF	CH_4	Percent	Entity Entry									
Supplemental Fat (feedlot)	S.Fat	CH_4	Percent	Entity Entry	3%	Mean					2	4	Expert Assessment
Dietary Forage %		CH_4	Percent	Entity Entry									
Total Digestible Nutrients (Dairy Cows)	TDN	CH_4	kg	Entity Entry									
Ym Feedlot – All Regions	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Beef Cattle Not on Feed (stocker)	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Beef Cattle Not on Feed (all foraging animals except dairy)	Ym	CH_4	% GE converted to CH_4	Default Entry									

Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Ym Dairy Repl. Heif. – California	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif. – West	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif. – Northern Great Plains	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif.– Southcentral	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif. – Northeast	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif. – Midwest	Ym	CH_4	% GE converted to CH_4	Default Entry									
Ym Dairy Repl. Heif. – Southeast	Ym	CH_4	% GE converted to CH_4	Default Entry									
Maximum daily emissions for dairy cows	E_{max}	CH_4	MJ/head	Default Entry	45.98								Mills et al. (2003)
Average live body weight for lactating cows	BW	N_2O/NH_3	kg	Entity Entry									
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (cool, humid region)	NH_3 loss	N_2O 3	Percent of N_{ex}	Default Entry							15%	30%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Open dirt lots (hot, arid region)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							30%	45%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							5%	15%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (shallow pit under floor)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Dairy Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							30%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (cool, humid region)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							30%	45%	Koelsh and Stowell (2005)

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Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Typical Ammonia Losses from Beef Housing Facilities – Open dirt lots (hot, arid region)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							40%	60%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (bedded pack)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Beef Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	NH_3 loss	N_2O	Percent of N_{ex}	Default Entry							30%	40%	Koelsh and Stowell (2005)
N_2O Emission Factor for manure in housing (dry lots and pit storage)	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry									IPCC (2006)
Nitrogen Excretion from Beef Cattle—Days on feed for an individual ration	DOF_x	N_2O, NH_3	Days	Entity Entry									
Nitrogen Excretion from Beef Cattle—Live body weight at finish of feeding period	BW_F	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Beef Cattle—Live body weight at the start of feeding period	BW_I	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Beef Cattle—Standard reference weight for expected final body fat	SRW	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Beef Cattle—Concentration of crude protein of total ration	C_{CP-x}	N_2O, NH_3	g crude protein/g dry feed	Entity Entry									
Monthly Beef Feedlot NH_3 Emissions—Dietary crude protein	CP	NH_3	Percent of dry matter	Entity Entry									
Average monthly temperature	T	N_2O, NH_3	Degrees Kelvin	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Average daily feed intake over finishing period	$ADFI_G$	N_2O, NH_3	g/ day	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Concentration of crude protein of total (wet) ration	C_{CP}	N_2O, NH_3	Percent	Entity Entry									

Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Nitrogen Excretion from Grow-Finish Pigs –Days on feed to finish animal (grow-finish phase)	DOF_G	N_2O, NH_3	Days	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Final (market) body weight	BW_F	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Average dressing percent (yield) at final weight	DP_F	N_2O, NH_3	Percent	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Initial body weight	BW_I	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Grow-Finish Pigs –Average fat-free lean percentage at final weight	$FFLP_F$	N_2O, NH_3	Percent	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Average daily feed intake over finishing period	$ADFI_G$	N_2O, NH_3	g/ day	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Concentration of crude protein of total (wet) ration	C_{CP}	N_2O, NH_3	Percent	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Days on feed to finish animal (nursery phase)	DOF_N	N_2O, NH_3	Days	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Average fat-free lean gain from 20 to 120kg	FFLPG	N_2O, NH_3	g/ day	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Final body weight in nursery phase	BW_{F-N}	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Weaning Pigs –Initial body weight in nursery phase	BW_{I-N}	N_2O, NH_3	kg	Entity Entry									
Nitrogen Excretion from Gestating Sows –Average daily feed intake during gestation	$ADFI_S$	N_2O, NH_3	g/ day	Entity Entry									
Nitrogen Excretion from Gestating Sows –Concentration of crude protein	C_{CP}	N_2O, NH_3	Percent	Entity Entry									
Nitrogen Excretion from Gestating Sows –Gestation period length	GL	N_2O, NH_3	Days	Entity Entry									
Nitrogen Excretion from Gestating Sows –Gestation lean tissue gain	GLTG	N_2O, NH_3	Kg	Entity Entry									

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Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Nitrogen Excretion from Gestating Sows –Number of pigs in litter	LITTER	N_2O, NH_3	Head	Entity Entry									
Nitrogen Excretion from Lactating Sows—Average daily feed intake during lactation	$ADFI_{LACT}$	N_2O, NH_3	g/ day	Entity Entry									
Nitrogen Excretion from Lactating Sows—Concentration of crude protein	C_{CP}	N_2O, NH_3	Percent	Entity Entry									
Nitrogen Excretion from Lactating Sows—Lactation length (days to weaning)	LL	N_2O, NH_3	Days	Entity Entry									
Nitrogen Excretion from Lactating Sows—Lactation lean tissue gain	LLTG	N_2O, NH_3	Kg	Entity Entry									
Nitrogen Excretion from Lactating Sows— Litter weight at weaning	L_{WEAN}	N_2O, NH_3	Kg	Entity Entry									
Nitrogen Excretion from Lactating Sows—Litter weight at birth	LW_{BIRTH}	N_2O, NH_3	kg	Entity Entry									
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (flushed or scraped) Roofed facility (daily scrape and haul)	$\%NH_3$ loss	NH_3	Percent of N_{ex}	Default Entry							5%	15%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (shallow pit under floor)	$\%NH_3$ loss	NH_3	Percent of N_{ex}	Default Entry							10%	20%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (bedded pack)	$\%NH_3$ loss	NH_3	Percent of N_{ex}	Default Entry							20%	40%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Swine Housing Facilities –Roofed facility (deep pit under floor, includes storage loss)	$\%NH_3$ loss	NH_3	Percent of N_{ex}	Default Entry							30%	40%	Koelsh and Stowell (2005)
Nitrogen Excretion from Broilers, Turkeys, and Ducks—Feed intake per phase	FI_x	N_2O, NH_3	g feed/ finished animal	Entity Entry									
Nitrogen Excretion from Broilers, Turkeys, and Ducks—Concentration of crude protein of total ration in each phase	$C_{CP\text{-}X}$	N_2O, NH_3	g crude protein/ g (wet) feed	Entity Entry									

Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Nitrogen Excretion from Broilers, Turkeys, and Ducks—Retention factor for nitrogen	N_{RF}	N_2O, NH_3	Fraction	Entity Entry									
Nitrogen Excretion from Laying Hens—Feed intake	FI	N_2O, NH_3	g feed/ finished animal	Entity Entry									
Nitrogen Excretion from Laying Hens—Concentration of crude protein of total ration	C_{CP}	N_2O, NH_3	g crude protein/ g (wet) feed	Entity Entry									
Nitrogen Excretion from Laying Hens—Egg weight	Egg_{wt}	N_2O, NH_3	g	Entity Entry									
Nitrogen Excretion from Laying Hens—Fraction of eggs produced each day	Egg_{pro}	N_2O, NH_3	Eggs/ hen/ day	Entity Entry									
Typical Ammonia Losses from Poultry Housing –Roofed facility (litter) (Meat Producing birds)	%NH_3 loss	NH_3	Percent of N_{ex}	Default Entry							25%	50%	Koelsh and Stowell (2005)
Typical Ammonia Losses from Poultry Housing –Roofed facility (stacked manure under floor - , includes storage loss) (Egg-producing birds)	%NH_3 loss	NH_3	Percent of N_{ex}	Default Entry							25%	50%	Koelsh and Stowell (2005)
Methane Emissions from Goats – Emission factor for goats	EF_G	CH_4	kg CH_4/head/day	Default Entry	0.0137	Mean							IPCC (2006)
Methane Emissions from Bison – Emission factor for bison	EF_{AB}	CH_4	kg CH_4/head/day	Default Entry									

Table 8-B-5: Manure Management Uncertainty Template

Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Beef Finishing Cattle		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	2.4	Mean	-20	20			ASABE (2005)
Total Dry Manure – Beef Cow (confinement)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	6.6	Mean	-20	20			ASABE (2005)
Total Dry Manure – Beef Growing calf (confinement)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	2.7	Mean	-20	20			ASABE (2005)
Total Dry Manure – Dairy Lactating cow		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	8.9	Mean	-20	20	8.7	11.3	ASABE (2005)
Total Dry Manure – Dairy Dry cow		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	4.9	Mean	-20	20	8.8	11.2	ASABE (2005)
Total Dry Manure – Dairy Heifer		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	3.7	Mean	-20	20			ASABE (2005)
Total Dry Manure – Dairy Veal 118 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.12	Mean	-20	20			ASABE (2005)
Total Dry Manure – Horse Sedentary 500 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	3.8	Mean	-20	20			ASABE (2005)
Total Dry Manure – Horse Intense exercise 500 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	3.9	Mean	-20	20			ASABE (2005)
Total Dry Manure – Poultry Broiler		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.03	Mean	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (male)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.07	Mean	-20	20			ASABE (2005)
Total Dry Manure – Poultry Turkey (females)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.04	Mean	-20	20			ASABE (2005)
Total Dry Manure – Poultry Duck		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.04	Mean	-20	20			ASABE (2005)
Total Dry Manure – Layer		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.02	Mean	-20	20			ASABE (2005)
Total Dry Manure – Swine Nursery pig (12.5 kg)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.13	Mean	-20	20			ASABE (2005)
Total Dry Manure – Swine Grow finish (70 kg)		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.47	Mean	-20	20			ASABE (2005)
Total Dry Manure – Swine gestating sow 200 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.5	Mean	-20	20			ASABE (2005)

Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total Dry Manure – Swine Lactating sow 192 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	1.2	Mean	-20	20			ASABE (2005)
Total Dry Manure – Swine Boar 200 kg		CH_4, N_2O, NH_3	kg dry manure/animal/day	Entity Entry	0.38	Mean	-20	20			ASABE (2005)
Volatile solids – Beef Finishing cattle	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.81	Mean	-25	25			ASABE (2005)
Volatile solids – Beef Cow (confinement)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.89	Mean	-25	25			ASABE (2005)
Volatile solids – Beef Growing calf (confinement)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.85	Mean	-25	25			ASABE (2005)
Volatile solids – Dairy Lactating cow	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.84	Mean	-25	25			ASABE (2005)
Volatile solids – Dairy Dry cow	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.85	Mean	-25	25			ASABE (2005)
Volatile solids – Dairy Heifer	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.86	Mean	-25	25			ASABE (2005)
Volatile solids – Dairy Veal 118 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry		Mean	-25	25			ASABE (2005)
Volatile solids – Horse Sedentary 500 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.79	Mean	-25	25			ASABE (2005)
Volatile solids – Horse Intense exercise 500 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.79	Mean	-25	25			ASABE (2005)
Volatile solids – Poultry Broiler	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.73	Mean	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (male)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.8	Mean	-25	25			ASABE (2005)
Volatile solids – Poultry Turkey (females)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.79	Mean	-25	25			ASABE (2005)
Volatile solids – Poultry Duck	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.58	Mean	-25	25			ASABE (2005)
Volatile solids – Layer	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.73	Mean	-25	25			ASABE (2005)
Volatile solids – Swine Nursery pig (12.5 kg)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.83	Mean	-25	25			ASABE (2005)
Volatile solids – Swine Grow finish (70 kg)	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.8	Mean	-25	25			ASABE (2005)
Volatile solids – Swine gestating sow 200 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.9	Mean	-25	25			ASABE (2005)
Volatile solids – Swine Lactating sow 192 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.83	Mean	-25	25			ASABE (2005)
Volatile solids – Swine Boar 200 kg	VS	CH_4, N_2O	kg VS/kg dry manure	Entity Entry	0.89	Mean	-25	25			ASABE (2005)
Storage temperature	T	CH_4	Kelvin	Entity Entry							
Manure temperature	T_{manure}	NH_3	Kelvin	Entity Entry							
Ambient air velocity	V_a	NH_3	m/s	Default Entry							
Height	h	N_2O	m	Entity Entry							
Width	W	NH_3	m	Entity Entry							
Radius	r	NH_3	m	Entity Entry							
pH	pH	NH_3	-	Entity Entry	7.5				6.5	8.5	Expert Assessment

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Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total nitrogen at a given day – beef finishing cattle		N_2O	kg N/kg dry manure	Entity Entry	0.07	Mean					ASABE (2005)
Total nitrogen at a given day – beef cow (confinement)		N_2O	kg N/kg dry manure	Entity Entry	0.03	Mean					ASABE (2005)
Total nitrogen at a given day – beef growing calf (confinement)		N_2O	kg N/kg dry manure	Entity Entry	0.05	Mean					ASABE (2005)
Total nitrogen at a given day – dairy lactating cow		N_2O	kg N/kg dry manure	Entity Entry	0.05	Mean					ASABE (2005)
Total nitrogen at a given day – dairy dry cow		N_2O	kg N/kg dry manure	Entity Entry	0.05	Mean					ASABE (2005)
Total nitrogen at a given day – dairy heifer		N_2O	kg N/kg dry manure	Entity Entry	0.03	Mean					ASABE (2005)
Total nitrogen at a given day – dairy veal 118 kg		N_2O	kg N/kg dry manure	Entity Entry	0.13	Mean					ASABE (2005)
Total nitrogen at a given day – Horse Sedentary 500 kg		N_2O	kg N/kg dry manure	Entity Entry	0.02	Mean					ASABE (2005)
Total nitrogen at a given day – Horse Intense Exercise		N_2O	kg N/kg dry manure	Entity Entry	0.04	Mean					ASABE (2005)
Total nitrogen at a given day – poultry, broiler		N_2O	kg N/kg dry manure	Entity Entry	0.04	Mean					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (male)		N_2O	kg N/kg dry manure	Entity Entry	0.06	Mean					ASABE (2005)
Total nitrogen at a given day – poultry, turkey (females)		N_2O	kg N/kg dry manure	Entity Entry	0.06	Mean					ASABE (2005)
Total nitrogen at a given day – poultry, duck		N_2O	kg N/kg dry manure	Entity Entry	0.04	Mean					ASABE (2005)
Total nitrogen at a given day – layer		N_2O	kg N/kg dry manure	Entity Entry	0.07	Mean					ASABE (2005)
Total nitrogen at a given day – swine nursery pig (12.5 kg)		N_2O	kg N/kg dry manure	Entity Entry	0.09	Mean					ASABE (2005)
Total nitrogen at a given day – swine grow finish (70 kg)		N_2O	kg N/kg dry manure	Entity Entry	0.08	Mean					ASABE (2005)
Total nitrogen at a given day – swine gestating sow 200 kg		N_2O	kg N/kg dry manure	Entity Entry	0.06	Mean					ASABE (2005)
Total nitrogen at a given day – swine lactating sow 192 kg		N_2O	kg N/kg dry manure	Entity Entry	0.07	Mean					ASABE (2005)
Total nitrogen at a given day – swine boar 200 kg		N_2O	kg N/kg dry manure	Entity Entry	0.07	Mean					ASABE (2005)
Total ammonia nitrogen in the manure – beef earthen lot	TAN	NH_3	kg NH_3/m^3	Entity Entry	0.1	Mean			0	0.02	ASABE (2005)

Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Total ammonia nitrogen in the manure – poultry, leghorn pullets	TAN	NH_3	kg NH_3/m^3	Entity Entry	0.85	Mean			0.66	1.04	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, leghorn hen	TAN	NH_3	kg NH_3/m^3	Entity Entry	0.88	Mean			0.54	1.22	ASABE (2005)
Total ammonia nitrogen in the manure – poultry, broiler	TAN	NH_3	kg NH_3/m^3	Entity Entry	0.75	Mean					ASABE (2005)
Ammonia concentration in the liquid – dairy lagoon effluent	NH_3	NH_3	kg NH_3/m^3	Calculated	0.08	Mean					ASABE (2005)
Ammonia concentration in the liquid – dairy slurry (liquid)	NH_3	NH_3	kg NH_3/m^3	Calculated	0.14	Mean					ASABE (2005)
Ammonia concentration in the liquid – Swine Finisher-Slurry wet-dry feeders	NH_3	NH_3	kg NH_3/m^3	Calculated	0.5	Mean					ASABE (2005)
Ammonia concentration in the liquid – Swine Slurry storage-dry feeders	NH_3	NH_3	kg NH_3/m^3	Calculated	0.34	Mean			0.19	0.49	ASABE (2005)
Ammonia concentration in the liquid – Swine flush building	NH_3	NH_3	kg NH_3/m^3	Calculated	0.14	Mean					ASABE (2005)
Ammonia concentration in the liquid – Swine agitated solids and water	NH_3	NH_3	kg NH_3/m^3	Calculated	0.05	Mean					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon surface water	NH_3	NH_3	kg NH_3/m^3	Calculated	0.04	Mean					ASABE (2005)
Ammonia concentration in the liquid – Swine Lagoon sludge	NH_3	NH_3	kg NH_3/m^3	Calculated	0.07	Mean					ASABE (2005)
Methane Conversion Factor (MCF) [a]– Dairy Cow	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Cattle	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Buffalo	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Market Swine	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Breeding Swine	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor [a] – Layer (Dry)	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Broiler	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Turkey	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Duck	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Sheep	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Goat	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Horse	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Mule/Ass	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)

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Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Methane Conversion Factor[a] – Buffalo	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – In vessel manure composting	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Static pile manure composting	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Intensive windrow	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Methane Conversion Factor[a] – Passive windrow	MCF	CH_4	%	Default Entry			-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Beef Replacement Heifers	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.33		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Replacement Heifers	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.17		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Mature Beef Cows	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.33		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Steers (>500 lbs)	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.33		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Stockers (All)	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.17		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle on Feed	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.33		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Dairy Cow	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.24		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Cattle	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.19		-20	20			U.S. EPA (2011)
Maximum Methane Producing Capacities – Buffalo[b]	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.1						IPCC (2006)
Maximum Methane Producing Capacities – Market Swine	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.48		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Breeding Swine	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.48		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (dry)	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.39		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Layer (wet)	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.39		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Broiler	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.36		-30	30			IPCC (2006)

Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Maximum Methane Producing Capacities – Turkey	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.36		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Duck	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.36		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Sheep	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.19		-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Feedlot sheep	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.36		-20	20			IPCC (2006)
Maximum Methane Producing Capacities – Goat	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.17		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Horse	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.3		-30	30			IPCC (2006)
Maximum Methane Producing Capacities – Mule/Ass	B_o	CH_4	m^3 CH_4/kg VS	Default Entry	0.33		-30	30			IPCC (2006)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with steel or lined concrete or fiberglass digesters with a gas holding system (egg shaped digesters) and monolithic construction	$EF_{CH4, leakage}$	CH_4	%	Default Entry	2.8						CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – UASB type digesters with floating gas holders and no external water seal	$EF_{CH4, leakage}$	CH_4	%	Default Entry	5						CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Digesters with unlined concrete/ferrocement/brick masonry arched type gas holding section; monolithic fixed dome digesters	$EF_{CH4, leakage}$	CH_4	%	Default Entry	10						CDM (2012)
Emission factor for the fraction of CH_4 produced that leaks from the anaerobic digester – Other digester configurations	$EF_{CH4, leakage}$	CH_4	%	Default Entry	10						CDM (2012)
Temporary storage of liquid/slurry manure –N_2O emission factor[c]	EF_{N20}	N_2O	kg N_2O-N/kg N	Default Entry	0.005		-50	100			U.S. EPA (2011)
Long-term storage of solid manure –N_2O emission factor[c]	EF_{N20}	N_2O	kg N_2O-N/kg N	Default Entry	0.002		-50	100			U.S. EPA (2011)

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Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Long-term storage of slurry manure – N_2O emission factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.005		-50	100			U.S. EPA (2011)
Cattle and Swine Deep Bedding (Active Mix)- N_2O emission factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.07						IPCC (2006)
Cattle and Swine Deep Bedding (No Mix)- N_2O emission factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.01						IPCC (2006)
Pit Storage Below Animal Confinements- N_2O emission factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.002						IPCC (2006)
Natural aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.01		-50	100			IPCC (2006)
Forced aeration aerobic lagoons – N_2O conversion factor[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.005		-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure with a crust[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0.8		-50	100			IPCC (2006)
N_2O emission factor for liquid storage – uncovered liquid manure without a crust[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0		-50	100			IPCC (2006)
N_2O emission factor for liquid storage – covered liquid manure[c]	EF_{N2O}	N_2O	kg N_2O-N/kg N	Default Entry	0		-50	100			IPCC (2006)
Composting – Ammonia emission (loss) relative to total nitrogen in manure	EF_{NH3}	NH_3	kg NH_3-N/kg N	Default Entry	0.05						Hellebrand and Kalk (2000)
Manure Management – Multiple Sources – collection efficiency, covered storage (with or without crust)	η	CH_4	percentage	Default Entry	1						Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage with crust formation	η	CH_4	percentage	Default Entry	0						Sommer et al. (2004)
Manure Management – Multiple Sources – collection efficiency, uncovered storage without crust formation	η	CH_4	percentage	Default Entry	-0.40						Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_1)	b_1	CH_4	dimensionless	Default Entry	1						Sommer et al. (2004)
Manure Management – Multiple Sources – Rate correcting factors (b_2)	b_2	CH_4	dimensionless	Default Entry	0.01						Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, cattle	A	CH_4	g CH_4 /kg VS/hr	Default Entry	43.33						Sommer et al. (2004)
Manure Management – Multiple Sources – Arrhenius parameter, swine	A	CH_4	g CH_4 /kg VS/hr	Default Entry	43.21						Sommer et al. (2004)

Data Element Name	Data Element Abbreviation/Symbol	Emission Type	Data Input Unit	Input Type	Estimated Value	Type of Estimate	Relative uncertainty Low (%)	Relative uncertainty High (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Potential methane yield of the manure cattle	$E_{CH4, pot}$ -	CH_4	kg CH_4/kg VS	Default Entry	0.48						Sommer et al. (2004)
Potential methane yield of the manure - swine	$E_{CH4, pot}$	CH_4	kg CH_4/kg VS	Default Entry	0.5						Sommer et al. (2004)
Manure Management – Multiple Sources – Kinematic viscosity of air[e]	v	NH_3	m^2/s	Default Entry							White (1999)
Manure Management – Multiple Sources – Mass diffusivity of NH_3[e]	D	NH_3	m^2/s	Default Entry							Watson (1966) and Baker (1969)
Temporary stack and long-term stockpile – Resistance to mass transfer through the manure[e]	R_s	NH_3	s/m	Default Entry							Rotz et al. (2011)
Temporary stack and long-term stockpile – Resistance to mass transfer through the cover[e]	R_c	NH_3	s/m	Default Entry							Rotz et al. (2011)
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - cattle liquid manure	VS_d/VS_T	CH_4	Unitless	Default Entry	0.46						Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio degradable volatile solids to total volatile solids - swine liquid manure	VS_d/VS_T	CH_4	Unitless	Default Entry	0.89						Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio Non-degradable volatile solids to total volatile solids - cattle liquid manure	VS_{nd}/VS_T	CH_4	Unitless	Default Entry	0.54						Møller et al. (2004)
Temporary stack and long-term stockpile – Ratio non-degradable volatile solids to total volatile solids – swine liquid manure	VS_{nd}/VS_T	CH_4	Unitless	Default Entry	0.11						Møller et al. (2004)
Solid-liquid separation – Efficiency of mechanical solid-liquid separation[e]		CH_4, N_2O, NH_3	Percent	Entity Entry							Ford and Fleming (2002)

[a] The values for methane conversion factor (MCF) vary depending on the temperature and the manure management system. IPCC (2006) provides estimated uncertainty ranges for these MCFs.
[b] There are no data for North America region; the data from Western Europe are used to calculate the estimation. There is no reported uncertainty for this adapted value.
[c] IPCC (2006) reports large uncertainties with default N_2O emission factors. The N_2O EF values vary depending on the animal species and temperature of the manure management system.
[d] Values for N_2O conversion factors are available for dairy cow, cattle, swine, and other animals and can be found in the chapter.
[e] Default values are available in the chapter.

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Table 8-B-6: Forestry Uncertainty Table

Forestry Sub-Source Category	Data Element Name	Abbreviation/Symbol	Emission Type	Data Input Unit	Input Source	Statistic	Type of Statistic	Probability Distribution Type	Relative uncertainty Low (%)	Relative uncertainty High (%)	Confidence Level (%)	Effective Lower Limit	Effective Upper Limit	Data Source
Urban Forestry – Aerial Data Method	Tree Cover Percent		CO_2	%	Model Output									i-Tree Canopy
Urban Forestry – Aerial Data Method	Urban Area		CO_2	m^2	Entity Entry									
Urban Forestry – Aerial Data Method	Average Annual Carbon Sequestration		CO_2	kg C/m^2/year	Default Entry	2.8	mean		14.8	14.8				Nowak et al. (2013)
Urban Forestry – Aerial Data Method	Average Carbon Storage		CO_2	kg C/m^2	Default Entry	7.69	mean							Nowak et al. (2013)

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