Chapter 4
Quantifying Greenhouse Gas Sources and Sinks in Managed Wetland Systems

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Contents:

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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.

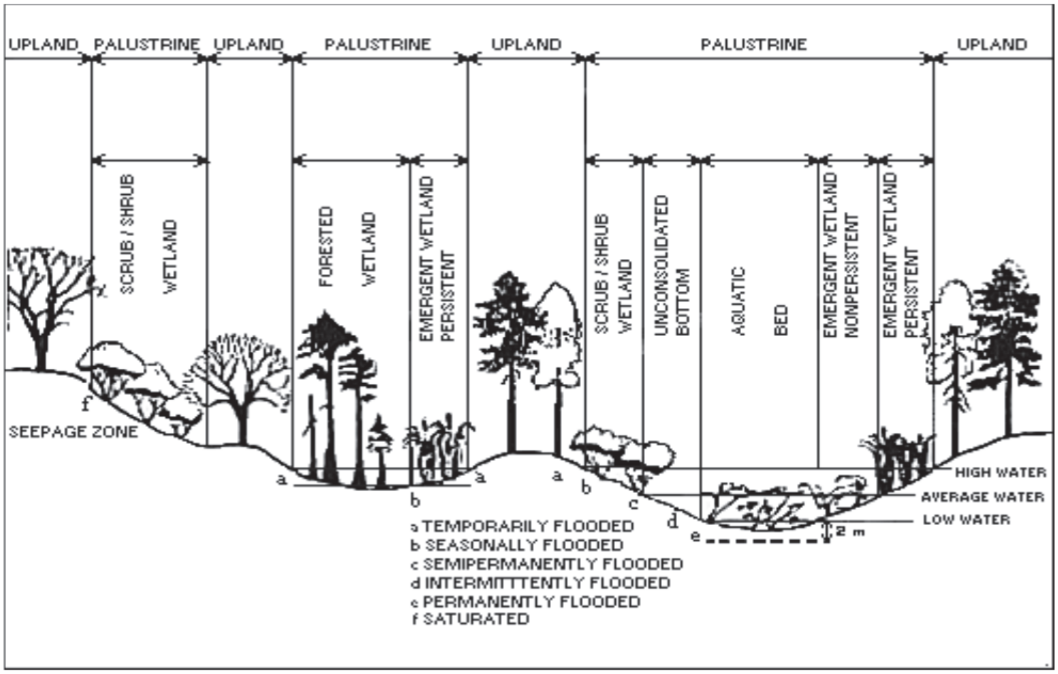
4.1.1.1 Description of Sector

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

[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.

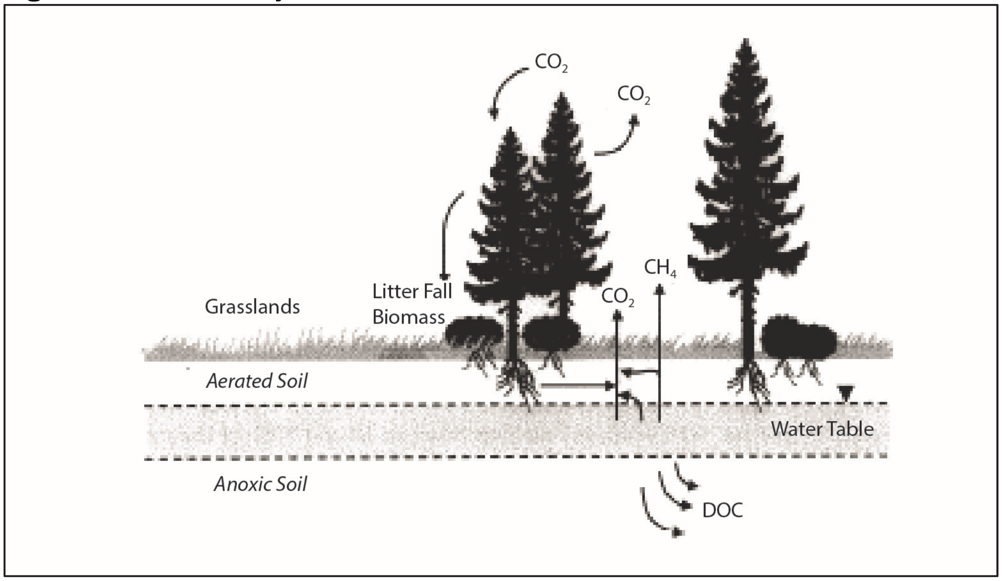
4.1.1.3 Risk of Reversals

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.

[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.

Chapter 4 References

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