CHAPTER 5

INLAND MINERAL SOIL WETLANDS

Contents

5.1 INTRODUCTION

This chapter provides guidance for estimating and reporting greenhouse gas emissions and removals from *Managed Inland Wetlands having Mineral Soils*. This will be referred to as IMS wetlands (inland mineral soil wetlands).

This chapter builds on the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (*2006 IPCC Guidelines*), Volume 4 chapter 2. Following the 2006 Guidelines, greenhouse gas emissions and removals can be estimated in two ways: 1) Net changes in carbon stocks in the five IPCC C pools over time (used for most CO_2 fluxes); and 2) Directly as gas flux rates to/from the atmosphere using emission factors (used for non-CO_2 emissions and some CO_2 emissions and removals).

These wetlands can occur in any of the six IPCC land classes. For example, a riverine wetland with trees may be classified as a forest, while a marsh may be used for grazing and classified as grassland. The precise details of this classification are specific to each country so it is not possible to say exactly how an inland wetland on mineral soil may be classified. This guidance applies to all inland wetlands on mineral soils, however they are classified. The classification is important when reporting these emissions, and there is no intention to change in any way how land is classified, however there may be a need to sub-divide some land types to reflect differing management actions.

Inland Wetlands Having Mineral Soils meet the following two criteria:

1) Have *Mineral Soils* (i.e. not organic soils)
2) Do not meet the definition of *Coastal Wetland*

Mineral soils are those that <u>do not</u> fit the definition of *organic soils*. Organic soils are those that satisfy requirements 1 & 2, or 1 & 3 below (FAO, 1998; *2006 IPCC Guidelines* Vol. 4, Chap. 4 Glossary):

1) O horizon thickness $\geqslant$ 10 cm; a horizon of less than 20 cm must have 12% or more organic carbon when mixed to a depth of 20 cm.
2) Soils that are never saturated for more than a few days must contain more than 20% organic C by weight (i.e. about 35% organic matter).
3) Soils are subject to water saturation episodes and have either:
    a. At least 12% organic carbon by weight (i.e. about 20% organic matter) if the soil has no clay; or
    b. At least 18% organic carbon by weight (i.e. about 30% organic matter) if the soil has 60% or more clay; or
    c. An intermediate, proportional amount of organic carbon for intermediate amounts of clay.

Coastal wetlands are covered in Chapter 4 of this Supplement.

Saline Inland Wetlands are a group of very specific Inland Wetlands. A review of the literature for potential methods to estimate carbon stock changes and greenhouse gas fluxes was conducted for inland saline wetlands (i.e. saline wetlands not covered in Chapter Four). Also known as playas, pans, salt lakes, brackish wetlands, salinas, and sabkhas (generally associated with coasts), saline wetlands are important parts of arid landscapes across the globe (Shaw and Bryant 2011). Carbon stocks and greenhouse gas fluxeshave been little studied in inland saline wetlands. In a recent review of the literature characterizing known information on pans, playas and salt lakes, carbon stocks and carbon dioxide, methane and nitrous oxide fluxes were not discussed (Shaw and Bryant 2011), likely indicating little research carbon and greenhouse gas fluxes has been conducted. Because of the briny nature of inland saline wetlands and their periodic flooding, saline wetlands are generally not vegetated. A review of the broader literature on saline wetlands indicates that only one study has assessed soil C in inland saline wetlands (Bai et al. 2007), and no studies have measured greenhouse gas fluxes or other biomass categories from inland saline wetlands. The Bai et al. (2007) study was conducted in northeast China and found relatively low soil carbon stocks when compared to other wetlands in China. Although only a single study, the soil carbon stocks found in Bai et al. (2007) of 41-47 Mg ha^{-1} to 30 cm are more similar to upland soils (Table 2.3 in the *2006 IPCC Guidelines*) than freshwater mineral wetland soils (Table 5.1). At present the lack of data on inland saline wetlands does not allow for default of carbon stock changes or greenhouse gas emission factors to be given. However the same methods as described here for IMS wetlands can be used with nationally measured factors and applied to inland saline wetlands. Note that it is good practice for inventory compilers to account for all areas of inland saline wetlands as sub-categories of the six IPCC land classes, to ensure that all land is accounted.

Figure 1 in Chapter 1 shows a decision tree to orient the inventory compiler to what types of wetlands are covered in this Chapter.

5.1.1 Inland Mineral Soil Wetlands

Mineral wetland soils are estimated to cover ~5.3% of the world's land surface, or 7.26 x 10^6 km^2 (Batjes, 2010). The most important climate zone is boreal (2%), followed by Tropical Moist (0.67%), Cool Temperate Moist (0.63%), Tropical Wet (0.61%), Polar (0.60%), and Warm Temperate Moist (0.23%) soils (Batjes, 2010). Climate zones with less than 0.20% mineral wetland soils include Cool and Warm Temperate Dry, Tropical Dry, and Tropical Montane.

Numerous inland wetland types have been described based on various criteria; the Ramsar Convention Wetland Classification includes 24 inland wetland types alone. Wetland classification can be simplified by considering broad generalizations of landform and hydroperiod (Semeniuk and Semeniuk, 1995, 1997). Inland mineral soil wetlands (IMS wetlands) are found in a variety of landscape settings, including basins, channels, flats, slopes, and highlands (Semeniuk and Semeniuk, 1995). It is common to find IMS wetlands adjacent to flowing waters (riparian wetland) and lake and pond margins. The hydroperiod, or degree of wetness over time, of IMS wetlands range from inundated (covered by water) to saturated (water is at or just below the surface) to unsaturated, and from permanent to seasonal or intermittent. Hydroperiod is directly related to climate (precipitation and evaporation), mechanisms of water discharge and recharge, and permeability of underlying sediment. Hydroperiod can significantly impact wetland carbon and nitrogen cycling pathways and rates, and is commonly altered by management activities. Therefore knowledge of wetland hydroperiod (inundated vs. un-inundated, permanent vs. seasonal) is useful for inventories of emissions and removals, particularly CH_4. An additional characteristic that is used to classify IMS wetlands is dominant vegetation community, and can include trees (forested wetland), woody shrubs, and/or emergent and non-emergent vascular plants. Vegetation type and productivity is important to carbon cycling, and is commonly impacted by management activities. For instance, emergent vascular plants can significantly enhance CH_4 production and emission from wetland sediments (Whiting et al., 1991; Whiting and Chanton, 1992, 1993).

An important agricultural use of inland wetlands is rice cultivation, which is covered in the *2006 IPCC Guidelines* (Vol. 4, Chapter 5 – Croplands), and not addressed in this supplement. Other potentially important agricultural uses of wetlands on mineral soils include lotus and mat rush cultivation, particularly in Asia (Seo et al., 2010; Maruyama et al., 2004). Currently there is little available information on C stock changes or greenhouse gas emissions for this type of cultivation. Further research will be required to develop methodologies for these types of cultivation practices. Indirect effects of agricultural activities include agricultural runoff to adjacent wetlands. Runoff would be expected to increase sedimentation rates, and may also alter greenhouse gas fluxes, e.g. nitrogen-rich inputs from agricultural runoff may lead to higher N_2O emissions (Bridgham et al., 2006). The impact of nitrogen fertilization on methane emissions is currently unclear as interactions are complex, and can affect both methane production and methane consumption (Bodelier, 2011).

Grazing is an important activity in wetlands within grassland or forest landscapes (Liu et al., 2009; Oates et al., 2008; Wang et al., 2009). The direct effects of livestock grazing on wetlands can be selective removal of plant biomass, trampling of plants, changes in below-ground biomass and soil properties, nutrient inputs, and bacterial contamination from animal waste. Overgrazing can influence biomass and carbon stocks. The intensity of grazing can also affect species composition and diversity. Reduction of aboveground biomass by grazing can affect plant-mediated gas transport between soil and the atmosphere, which may alter greenhouse gas fluxes in wetlands. Grazed wetlands and wetlands receiving run-off from adjacent livestock grazing areas, may also alter N_2O (and NO) emissions from wetlands.

Forest management activities on forested wetlands can vary in management intensity depending on the silvicultural system. The intensity may range from selective cutting treatments to large area clearcuts. It represents a loss in biomass pools and can also alter the ecologic and hydrologic conditions of the site. Possible consequences of harvesting in wetlands can be changes in water table, and changes in microclimatic conditions such as increased solar radiation and evapotranspiration. Some of these changes can substantially affect primary production, respiration, and fluxes of CO_2, CH_4, and N_2O. Many studies have reported temporary increases in water levels after harvesting from increased interception and decreased transpiration. The increase in water level may result in decreasing decomposition rates but acceleration in CH_4 and N_2O emissions. This point is further discussed below in rewetting. Another important point is the wetland ecosystem type where harvesting takes place, for example seasonally flooded riparian ecosystems are known to have more soil and biomass carbon content compared to upland ecosystems. Therefore, harvesting may be expected to cause larger carbon emissions in these areas compared to dryer environments. A specific accounting method for harvesting in riparian ecosystems is a topic for future improvement.

5.1.2 Guidance for Inland Mineral Soil Wetlands

In this Chapter guidance and methodologies mainly follow the *2006 IPCC Guidelines*, in particular the generic guidance given in Volume 4 Chapter 2. This chapter provides additional information to be used in applying the methods in the *2006 IPCC Guidelines* and should be read in conjunction with volume 4 of the 2006 Guidelines.

Management activities that impact CO_2 and non-CO_2 (CH_4 and N_2O) emissions include water level management as well as activities that impact vegetation (such as grazing, vegetation removal, and cultivation, nutrient amendments). Figure 1 shows a decision tree to guide inventory compilers on which Tier approach should be used to report on Inland Wetlands on Mineral Soils. Table 5.1. clarifies the scope of the assessment, and the corresponding sections of this chapter.

TABLE 5.1
SECTIONS ADDRESSING MAJOR GREENHOUSE GAS EMISSIONS FROM INLAND MINERAL SOIL WETLANDS

Land-use category/GHG	CO_2	CH_4	N_2O
IMS wetlands (General guidance)	Section 5.2.1	Section 5.2.2	Included Elsewhere[1]
Drainage, Restoration and Creation of IMS wetlands	Section 5.3	Section 5.3	Section 5.3
Inland Saline Wetlands	No Guidance[2]		

NOTES:

[1] N_2O emissions from FWMS wetlands are included in the estimation of indirect N_2O from agricultural or other run-off, and waste water.

[2] No available information for providing default factors for inland saline wetlands however the same methods as used for IMS wetlands can be used with national factors. Note that it is good practice for inventory compilers to account for all areas of inland saline wetlands as sub-categories of the six IPCC land classes.

5.1.3 Choice of Activity Data

IMS Wetlands may occur in any of the six IPCC land use classes described in volume 4 of the *2006 IPCC Guidelines*. It is good practice to follow the guidance given the 2006 Guidelines in categorising land. While the use of this supplementary guidance does not ask for changes to the land classification required in the 2006 guidelines it may be necessary to sub-divide some categories according to ecosystem and management activities especially where these involve changes in water level.

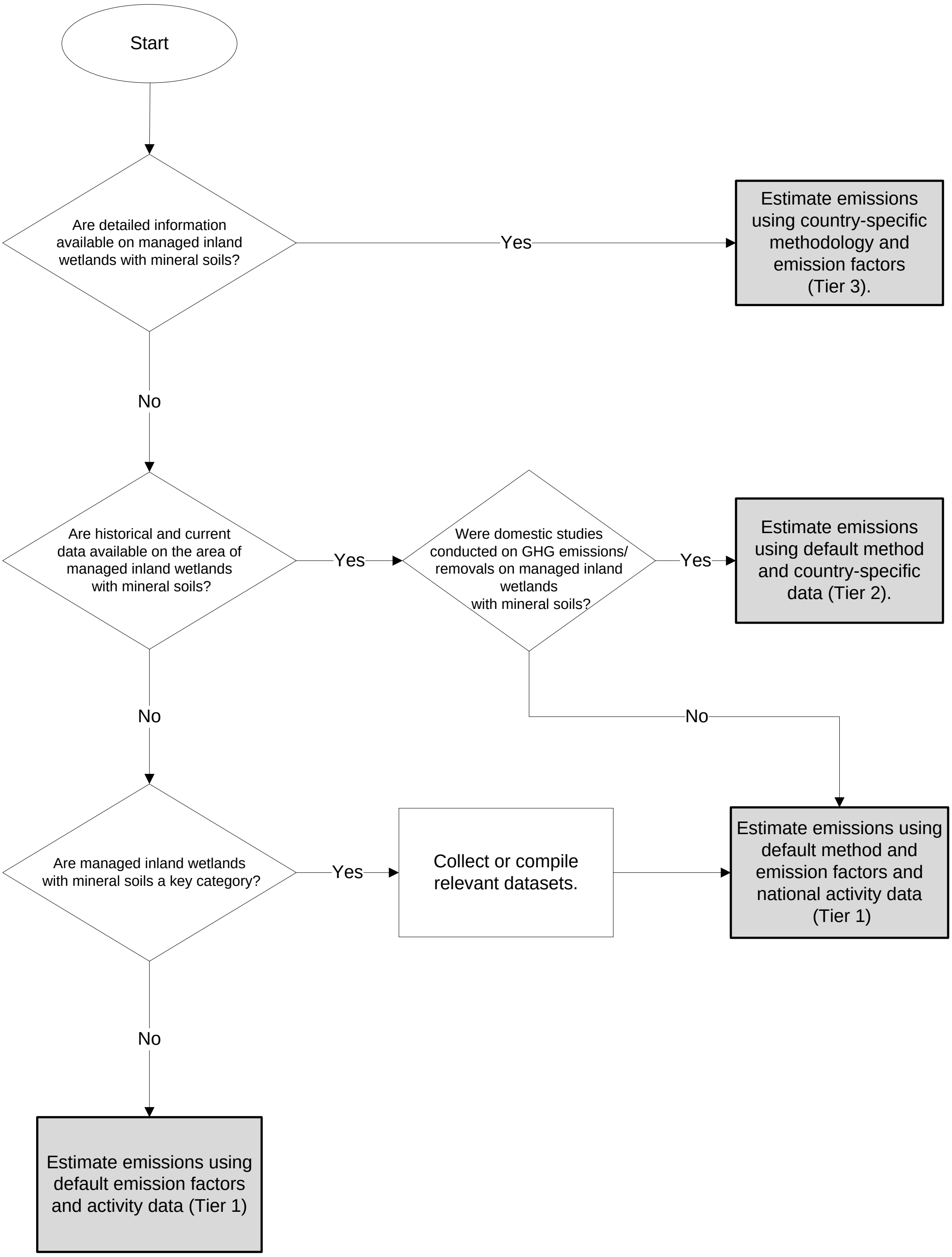
All areas should be estimated as accurately as possible. In a Tier 1 level some degree of aggregation can be performed in the estimation of biomass. For example, initial land use of grassland and some types of croplands can be aggregated and biomass estimations for both done with the same methodology. At higher level Tiers it is good practice to use a matrix of initial land use and final wetlands types to estimate changes in biomass stocks. Tier 1 assumes that transition from any land use to wetland occurs in the year of conversion, whereas in Tiers 2 and 3 biomass stocks are monitored during transitions years and expressed on an annual average basis.

5.1.4 Reporting Inland Mineral Soil Wetlands

As noted above, these inland wetlands may occur in any of the 6 IPCC land classes depending on the precise national definitions used. Emissions should be reported in the land use category under which they are classified. Note that a change in management practice may, or may not, result in the emissions being classified as “land converted to” lands. For example, changes in harvesting trees in a wetland classified as forest would generally not result in the emissions being reported as land use change while there would be a change in the management and the guidance in this chapter.

Inventory compilers should also note that while the emission factors in this chapter are, in part, based on those form pristine wetlands there is no intention for unmanaged wetlands to be reported. Methods applicable for the drainage, creation and restoration of IMS wetlands are covered in section 5.3 while section 5.2 covers other management uses of IMS wetlands. For Tier 1 it is assumed that these IMS wetlands reach an equilibrium state after 10 years (inventory compilers are encouraged to us more appropriate periods for wetlands in their country if such data is available). Thus after 10 years it is assumed that the impact of drainage, creation and restoration ceases and the emissions would then be estimated following section 5.2.

185 **Figure 1 Decision tree to the inventory on Inland Wetlands on Mineral Soils**



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5.2 GENERAL METHODS

5.2.1 CO_2

5.2.1.1 BIOMASS AND DEAD ORGANIC MATTER

The set of general equations to estimate the annual carbon stock changes on Managed Inland Wetlands are given in Volume 4, Chapter 2 of the *2006 IPCC Guidelines.*

Figure 1.2 in Chapter 1 of Volume 4 of the 2006 Guidelines shows a decision tree for the identification of appropriate Tiers for the inventory of land remaining in the same land-use category. Refer also to Figure 4.2 of Chapter 4 in Volume 1 of the 2006 Guidelines for assigning key categories.

CHOICE OF METHOD AND TIER

Where there is no change in management or change in land-use the Tier 1 approach assumes no change in biomass, dead wood or soil carbon in these wetlands. Following a change in management practices the biomass and dead organic matter will not vary significantly after 10 years (Miller and Fujii, 2010), and this land area will not fall into the definition of a key category (see Figure 1.2. in Chapter 1 of Volume 4 in the 2006 Guidelines for guidance on defining key categories). Where there are significant changes in management, biomass stocks can change accordingly. The Tier 1 approach of no change in biomass can be used if the land is not considered to be a key category, but if there is reliable data about rates of biomass change then countries should use a higher Tier to estimate emissions and removals associated with changes in biomass and dead organic matter. For Tier 2 and 3, it is good practice to implement country-specific biomass and carbon stock inventories. If national data are not available for Tier 2, countries can use globally-compiled databases (e.g. FAO) and assume that wetland vegetation does not have substantially different biomass carbon densities than upland vegetation (Bridgham et al., 2006). It is also good practice to use modern satellite imagery and field surveys to estimate sub-types of Wetlands, as outlined in Chapter 3 of the *2006 IPCC Guidelines*. Where resources are not available to obtain country-specific data, wetland cover can also be derived from globally-compiled databases (e.g. WWF Global Lakes and Wetlands Database – GLWD). Since several definitions exist for the classification of land as wetland (Mitsch and Gosselink, 2011), it is good practice that countries explicitly describe criteria of classification.

UNCERTAINTY ASSESSMENT

As stated in the 2006 Guidelines, Volume 1, Chapter 3 provides information on forest biomass uncertainties associated with sample-based studies. FAO (2006) provides uncertainty estimates for forest carbon factors; basic wood density (10 to 40%); annual increment in managed forests of industrialized countries (6 %); growing stock (industrialized countries 8%, nonindustrialized countries 30%); combined natural losses for industrialized countries (15%); wood and fuelwood removals (industrialized countries 20%). The major sources of uncertainty of wood density and biomass expansion factors are stand age, species composition, and structure. To reduce uncertainty, countries are encouraged to develop country- or region specific biomass expansion factors and BCEFs for FWMW. In case country- or regional-specific values are unavailable, the sources of default parameters should be checked and their correspondence with specific conditions of a country should be examined.

Uncertainty in dead organic matter pools is high relative to other carbon pools (Bradford et al. 2010). Dead organic matter pools are dependent on the vegetation type (Currie and Nadelhoffer 2002), management (Scheller et al. 2011), age of stand (if forested) (Sun et al. 2004), disturbance history (Tinker and Knight 2000) and the presence of soil fauna (Hale et al. 2005). Errors as high as 100% are common when measuring DOM (Bradford et al. 2009). For IMS wetlands it is highly unlikely that approaches other than inventory methods (Tier 2) will be available to assess changes in DOM pools. Hence, good practice for inventory methods is critical for an accurate assessment of DOM (see inventory discussion in Uncertainty Assessment in Soil C section – 5.2.3.5). When no data on DOM are available, no change in DOM stocks can be assumed unless changes in DOM are associated with a key category.

5.2.1.2 SOIL CARBON

Management practices can significantly change mineral soil wetland C stocks, especially when flow is regulated and sediment deposition rates change (McCarty and Ritchie 2002; Chmura et al. 2003). Changes in water table affect redox conditions in soils which affect decomposition rates and ultimately C stocks (Zdruli et al. 1995). Restoration and rewetting of mineral soil wetlands can increase C stocks over time (Ballantine and Schneider (2009).

CHOICE OF METHOD

Little information is available to conduct Tier 2 and Tier 3 soil C stock analyses for organic C in IMS wetlands. For example, only two studies have assessed site specific changes in soil C pools following mineral soil wetland drainage (Zdruli et al. 1995; Page and Dalal 2011). An alternative approach for Tier 1 could characterize the direct soil emissions of CO_2 by vegetation type, climate zone, management practices and soil type, however little information is available to use a direct CO_2 emission approach. Only four studies have assessed CO_2 emissions following a change in IMS wetlands (Danevcic et al. 2010; Fromin et al. 2010; Samaratini 2011; Sgouridis 2011). Similarly, only three studies have measured direct emissions of CO_2 following mineral soil wetland restoration (Pfeifer-Meister 2008; Gleason et al. 2009). Because of the paucity of data on both changes in soil C stock and on direct CO_2 emissions, a Tier 1 approach based on annual multiple inventories is described to assess changes in soil C stocks for mineral soil wetlands. If data from multiple inventories is not available, changes to soil C stocks in IMS wetlands is assumed to be zero at Tier 1 level of estimation.

TIER 1

Chapter 2 of Volume 4 of the 2006 Guidelines provides general information about mineral soil classification and soil C stock estimations. The annual change in carbon stocks in mineral soils is calculated using Equation 2.25 (IPCC 2006, Volume 4 chapter 2). The Tier 1 approach is based on changes in soil C stocks over a finite period of time, assuming (i) over time, soil organic C reaches a spatially-averaged, stable value specific to the soil, climate, land-use and management practices and (ii) soil organic C stock changes during the transition to a new equilibrium SOC occurs in a linear fashion (IPCC 2006). To account for changes in SOC, countries need to estimate wetland areas according to climate zones, management practices and soil types. Table 5.1 below gives some updated reference soil organic carbon stocks that should be used in preference to those in the 2006 Guidelines in table 2.3 of volume 4, chapter 2.

TABLE 5.1. DEFAULT REFERENCE SOIL ORGANIC CARBON STOCKS FOR IMS WETLANDS.

		IPCC (2006), Table 2.3[a]		Batjes, 2011[b]	
Region	Depth	Mg C ha^{-1}	Error	Mg C ha^{-1}	Error
Boreal	0-30cm	146*	131	116	94
Cold temperate, dry	0-30cm	87	78		
Cold temperate, moist	0-30cm	87*	78	128	55
Warm temperate, dry	0-30cm	88*	79	74	45
Warm temperate, moist	0-30cm	88*	79	135	101
Tropical, dry	0-30cm	86	77	22	11
Tropical, moist	0-30cm	86	77	68	45
Tropical, wet	0-30cm	86	77	49	27
Tropical, montane	0-30cm	86	77	82	73

[a] SOC stocks for mineral soils under natural vegetation presented as Tier 1 values in Table 2.3 of *the IPCC 2006 Guidelines* for National Greenhouse Gas Inventories, Volume 4. They are derived from soil databases described by Jobbagy and Jackson (2000) and Bernoux et al. (2002).

[b] This study presents revised estimates of the IPCC 2006 SOC stocks for mineral soils under natural vegetation based on an expanded version of the ISRIC-WISE database (Batjes, 2009) which contains 1.6 times the number of soil profiles of the databases used in the IPCC 2006 SOC stocks estimate.

*SOC stock estimates from IPCC 2006 that were not based on expert estimates (Batjes, 2011).

For this assessment only new values of SOC_{ref} were found to support the derivation of general stock change factors for IMS wetlands utilizing the second equation in 2.25 from the *2006 IPCC Guidelines*. Inventory compilers should use the data from the appropriate chapters of Volume 4 of the *2006 IPCC Guidelines* in conjunction with the data in table 5.1, above. If countries have data that can be used to derive stock change factors or suitable literature values for these parameters for wetlands by climate region it is good practice to use them.

TIER 2

For Tier 2, it is good practice to conduct soil inventories for the appropriate classification of soils, but if data are not available, aggregate data (e.g. FAO) can be used for general classification. To conduct a Tier 2 approach soil C stocks need to be known at two periods in time and stock changes are simply a function of the difference in C stock divided by the time period (years) (Equation 5.1).

EQUATION 5.1

ANNUAL CHANGE IN ORGANIC CARBON STOCKS IN MINERAL SOILS

$$\Delta C_{Mineral} = \frac{\left(SOC_0 - SOC_{(0-T)}\right)}{T}$$

Where:

$\Delta C_{Mineral}$ = annual change in carbon stocks in mineral soils, tonnes C yr-1

SOC_0 = soil organic carbon stock in the last year of an inventory time period, tonnes C

$SOC_{(0-T)}$ = soil organic carbon stock at the beginning of the inventory time period, tonnes C

SOC_0 and $SOC_{(0-T)}$ are calculated using the SOC equation in the box where the reference carbon stocks and stock change factors are assigned according to the land-use and management activities and corresponding areas at each of the points in time (time = 0 and time = 0-T)

T = number of years between inventory time periods, yr

UNCERTAINTY ASSESSMENT

Because of lack of data at this time, the only reliable higher Tier approach to assess C stock changes in soils for IMS wetlands is through repeated inventories. The repeated inventory (stock changes) approach works well for any of the disturbances and restoration activities discussed in this chapter. Inventory methods, if conducted with consistent methods for measurement and analysis from year to year tend to be very accurate in assessing change (Gillespie 1999). As stated in the 2006 Guidelines, the precision of an inventory is increased and confidence ranges are smaller with more sampling. If plot locations are not re-locatable, if measurement methods change or if lab analyses protocols are not consistent with time, uncertainty increases for inventories.

As indicated in the 2006 Guidelines, uncertainties in activity statistics may be reduced through a better national system, such as developing or extending a ground-based survey with additional sample locations and/or incorporating remote sensing to provide additional coverage. It is *good practice* to design a classification that captures the majority of land-use and management activities with a sufficient sample size to minimize uncertainty at the national scale.

5.2.2 Non-CO_2 Emissions from IMS Wetlands

5.2.2.1 METHANE EMISSIONS FROM IMS WETLANDS

Methane is produced in soils of IMS during anaerobic decomposition of organic matter, and emitted to the atmosphere after diffusion or ebullition through the water column or through plant-mediated transport. Several factors have been identified as important controls on methane production and emission, including water level, temperature, and vegetation community and productivity (Whiting and Chanton, 1993). Despite current understanding of the processes involved in methane production and emission from wetlands, it remains difficult to accurately predict methane emissions with a high degree of confidence. Studies show high spatial variability in methane emissions across large areas that have similar climate, vegetation, and topography, and within small areas that have microscale variation in topography (Ding et al., 2003; Saarnio et al., 2009). In addition, there are very few studies of methane emissions from IMS wetlands in Europe (Saarnio et al., 2009), tropical regions (Mitsch et al., 2010), and certain regions of North America (Pennock et al., 2010). Therefore, the default emission factors we present necessarily have large uncertainties.

TIER 1

The basic equation to estimate CH_4 emission is shown in Eq. 5.2, where wetland area subject to a particular hydroperiod is multiplied by the default emission factor, and by the annual fractional period that the wetland area is inundated by water, if known. This allows for the incorporation of wetland hydroperiod, which may be determined by natural causes or as a consequence of management activity. If the annual fractional period of inundation is unknown, then a default fraction of 1 is assumed.

EQUATION 5.2
METHANE EMISSIONS FROM IMS WETLANDS

$$\boldsymbol{M_{IMS}} = \sum_{\boldsymbol{all\,IMS\,wetlands}} \{\boldsymbol{EF} \times \boldsymbol{A} \times \boldsymbol{T} \times \boldsymbol{10^{-3}}\}$$

Where:

M_{IMS} = annual mass of methane emitted from IMS wetlands, kg yr^{-1}
EF = annual emission factor, g C-CH_4 m^{-2} yr^{-1}
A = wetland area experiencing a particular hydroperiod, m^2
T = annual fractional period of inundation, (dimensionless)

Table 5.2 gives default CH_4 emission factors and error ranges for IMS wetlands that are permanently inundated, specific for different climate zones.

TABLE 5.2 DEFAULT CH_4 EMISSION FACTORS FOR PERMANENTLY INUNDATED IMS WETLANDS

Region	Emission factor EF_{CH4} (g CH_4-C m^{-2} yr^{-1})	Range (g CH_4-C m^{-2} yr^{-1})	References
Boreal	5.04	2.52 – 7.55	Bridgham et al., 2006
Temperate Cool	33.5	4.21 – 80.0	Bridgham et al., 2006; Altor and Mitsch, 2005; Kim et al., 1998; Badiou et al., 2011
Temperate Warm	43.9	7.69 - 182	Bridgham et al., 2006; Yu et al., 2008; Pulliam, 1993
Tropical	91.7	6.70 - 350	Devol et al., 1990; Smith et al., 2000; Nahlik and Mitsch, 2010

Several studies of CH_4 emissions from seasonally inundated IMS wetlands have been conducted, especially in temperate regions. If a wetland is known to be seasonally inundated and is located in a temperate or tropical region, an alternative is to apply the following default emission factors (Table 5.3) to the total wetland area to calculate an annual emission.

TABLE 5.3 DEFAULT CH_4 EMISSION FACTORS FOR SEASONALLY INUNDATED IMS WETLANDS

Region	Emission factor EF_{CH4} (g CH_4-C m^{-2} yr^{-1})	Range (g CH_4-C m^{-2} yr^{-1})	References
Temperate	16.7	0.13 – 46.8	Bridgham et al., 2006; Altor and Mitsch, 2005; Pennock et al., 2010; Gleason et al., 2009; Morse et al., 2012; Danevcic et al., 2010; Song et al., 2009; Ding and Cai, 2007; Song et al., 2003; Bartlett et al., 1993
Tropical	135	9.5 - 350	Nahlik and Mitsch, 2010; Bartlett et al., 1993

CALCULATING STEPS FOR TIER 1

Step 1: Determine the area of wetland experiencing a particular hydroperiod; if hydroperiod is unknown then assume permanent inundation ($T_{inundation}$=1). Using equation 5.2, the appropriate EF_{CH4} (Table 5.2), and the annual fractional period of inundation, estimate the annual CH_4 emission from that area of wetland. If a wetland is known to be seasonally inundated, an alternative is to apply the appropriate EF_{CH4} (Table 5.3), setting $T_{inundation}$=1.

Step 2: Repeat Step 1 for each wetland area experiencing a particular hydroperiod.

Step 3: Sum the annual CH_4 emissions from each wetland area to calculate a total annual CH_4 emission from IMS wetlands.

TIER 2

Tier 2 calculations use country-specific emission factors and parameters, to reflect regionally important wetland types and hydrologic dynamics. For example if seasonally-inundated wetlands are a dominant IMS wetland type of the country, it is recommended that wetland hydroperiod be determined.

TIER 3

Tier 3 calculations include site-specific determinations of methane emissions from dominant wetland types, or may include approaches such as dynamic modeling of methane fluxes (Saarnio et al., 2009). Models based on simple regressions (Christensen et al., 1996; Saarnio et al., 1997; Juutinen et al., 2003) or sophisticated process-based models (Walter and Heimann, 2000; Kettunen, 2003; Cui et al., 2005) can be applied to specific wetland ecosystems, however these models require input data sets that may be difficult to obtain.

5.2.2.2 NITROUS OXIDE EMISSIONS FROM IMS WETLANDS

Nitrous oxide (N_2O) is an important greenhouse gas.The microbial processes involved in production of N_2O are nitrification (nitrifier denitrification, in particular) and denitrification. Therefore, N_2O emissions vary with climate, soil-water conditions and management. Due to complexity of the interactions between these factors, N_2O shows very high spatial and temporal variations; even diurnal variations.

A wetland system can be a major N_2O source or sink as a result of ecologic conditions and microbial activity. The increase in anthropogenic nitrogen discharge to the natural environment may enhance the N_2O flux to the atmosphere. Generally, wetlands with low N availability are considered relatively minor sources of N_2O (Durand et al., 2010). Relatively high N_2O emissions from IMS wetlands are associated with agricultural runoff (Phillips and Beeri, 2008; DeSimone et al., 2010), and livestock runoff (Chen et al., 2011; Oates et al., 2008; Jackson et al., 2006; Holst et al., 2007; Walker et al., 2002).In order to avoid double-counting N_2O emitted from the use of fertilizers, and urine and dung deposition from grazing animals, it is suggested to follow 2006 Guidelines (Volume 4, Chapter 11) for estimating N_2O emissions from those IMS wetlands receiving agricultural or other runoff.

EQUATION 5.3

N_2O EMISSIONS FROM IMS WETLANDS

$$N_{IMS} = EF \times A$$

Where

N_{IMS} = N_2O emissions form IMS wetlands, kg yr^{-1}

EF = Emission Factors (see Table 5.4 for defaults) kg km^{-2} yr^{-1}

A = Area of IMS wetlands km^{-2}

TABLE 5.4 EMISSION FACTORS FOR ESTIMATING N_2O EMISSIONS FROM IMS WETLANDS.

Region	Emission factor EF_{N2O}	Uncertainty range	Reference/Comments
Polar			
Boreal			
Temperate	155-231 kg km^{-2} yr^{-1}		Wang et al., 2009. In China Eutrophic conditions
	0-1367 kg km^{-2} yr^{-1}		Hasegawa et al., 2000 Paddy field
	0.5 kg km^{-2} yr^{-1}	±0.4	Vilain et al., 2010. Riparian buffer
	-0.2-0.9 kg ha-1 yr−1	± 0.8	Boeckx and Cleemput, 2006 Riparian buffer
	4.07 kg ha-1 yr−1 2.09 kg ha-1 yr−1 4.07 kg ha-1 yr−1 2.09 kg ha-1 yr−1 4.07 kg ha-1 yr−1 4.90 kg ha-1 yr−1_	± 1.72 ± 0.79 ±1.72 ±0.79 ±1.72 ± 1.52_	Jiang et al, 2009 (Marsh) Jiang et al, 2009 (Rice field) Fertilizer application Jiang et al, 2009 (Conversion of Marsh to rice field) Jiang et al, 2009 (Conversion of Marsh to dryland)
Tropical	15.7 µg m-2 h-1 98.9 µg m-2 h-1 138.8 µg m-2 h-1 429.5 µg m-2 h-1	-177.6–163.1 -265.1–2101.4 -278.0–437.0 71.0–1641.1	Wang et al.,2006. In China Eutrophied conditions

5.3 DRAINAGE, CREATION, AND RESTORATION OF INLAND MINERAL SOIL WETLANDS

This section provides some additional methodological guidance for cases when the water table is changed (either raised or lowered) through drainage, restoration and creation of IMS wetlands.

5.3.1 Introduction

Drainage and restoration of wetlands are very common management activities that may result directly from management or may be part of land use conversion from, or to, wetlands. Management activities that commonly occur on wetlands include agriculture, grazing, and forestry.

Draining of mineral wetlands generally leads to other land uses such as agriculture, grazing or forestry, although wetlands may be drained and still meet the criteria for wetlands. Following limited drainage, wetlands can be very productive even if still meeting the wetland criteria. Drainage leads to lower water tables which affect processes like decomposition, above and belowground productivity, and organic matter and nutrient mineralization. Drainage can lead to changes in vegetation, soil carbon stocks, and greenhouse gas fluxes. Although some wetlands have been drained for centuries, drainage increased tremendously approximately 100 years ago with the onset of modern agriculture and the growth of urban environments. Globally, approximately 54% of mineral soil wetlands have been converted to other land uses, mostly by drainage (Bridgham et al., 2006).

Restoration is “the process of assisting the recovery of an ecosystem that had been degraded, damaged, or destroyed” (SER 2004). Wetland restoration is a common activity in response to significant wetland loss and degradation on a global scale. Wetland restoration can take many forms, including: invasive species removal, conversion of agricultural lands back to wetlands, filling or blocking ditches, reducing nutrient and sediment levels, to name a few. There is large potential for increased carbon storage from restoring mineral soil wetlands. For instance, Bridgham et al. (2006) estimated that mineral soil wetlands are currently losing roughly 45 Mt C yr^{-1} of carbon sequestration potential from wetland conversion. This potential to sequester more carbon has been borne out by many restoration studies (Euliss et al. 2006, Gleason et al. 2009, Ballantine and Schneider 2009, Card et al. 2010, Badiou et al. 2011). Badiou et al. (2011) estimated from their study of 22 wetlands that restored wetlands were accumulating 2.7 Mg C ha^{-1} yr^{-1} as soil organic carbon. Ballantine and Schneider (2009) surveyed 35 restored wetlands in New York and found that 17% (6% to 23%) of the reference carbon stock had accumulated over a 55 year chronosequence. Ballantine and Schneider (2009) also summarized the literature and found that wetlands recovered about 50% of their reference soil carbon after 55 years, whereas plant standing biomass reached reference conditions at 55 years. Time to recover soil C after wetland restoration varies greatly depending upon the amount of soil C lost, soil wetness, vegetation, and hydrogeomorphic setting. For instance, Card et al. (2010) estimated that it would only take 7-11 years after restoration in riparian wetlands to come back to reference conditions.

Although restored wetlands tend to increase their ability to sequester carbon, restored wetlands may also increase their emissions of CH_4 and N_2O, and so could increase their climate change impacts despite storing more carbon (Bridgham et al. 2006). This is especially true for mineral wetlands, which can have relatively large CH_4 emissions. For example, peatlands occupy 32% more land then mineral wetlands, but emit 46% less CH_4 (Bridgham et al. 2006). Much less information exists for trace gas emissions compared to soil carbon contents in restored wetlands. In general, emissions of CH_4 and N_2O are affected by hydroperiod, vegetation, and substrate quality and quantity. A few studies in seasonal wetlands have found that restoration had little impact on either CH_4 or N_2O emissions (Gleason et al. 2009, Pfeifer-Meister 2008). Conversely, other studies, especially in restored wetlands with permanent or semi-permanent hydroperiods, were found to have increased CH_4 and/or N_2O (Badiou et al. 2011).

Data is still sparse to document how wetland restoration affects emissions of total greenhouse gas emissions (see summaries by Roulet 2000, Bridgham et al. 2006). Few studies of restoration effects on mineral wetlands have been conducted. Badiou et al. (2011) measured greenhouse gas fluxes across a range of prairie potholes in Canada and calculated that they would sequester approximately 3.25 Mg CO_2 equivalents ha^{-1} yr^{-1}, even after accounting for an increase in CH_4 and N_2O emissions. Gleason et al. (2009) found no significant difference in CO_2, CH_4, or N_2O exchange between cropland and restored prairie pothole wetlands on cropland 16 years after restoration.

5.3.2 Methodological Issues

The primary challenges to wetland restoration and creation are the development of wetland hydrology and the establishment of vegetation (US EPA, 2003). Wetlands are restored or created for a variety of reasons, including water-quality enhancement (treatment of wastewater, stormwater, acid mine drainage, agricultural runoff; Hammer, 1989), flood minimization, and habitat replacement (Mitsch et al., 1998). Wetlands created for the purposes of wastewater treatment are not covered in this Chapter; please refer to Chapter 6 (Constructed Wetlands) in this Supplement for guidance on these types of wetlands. Flooding of land to create reservoirs is covered in Chapter 7 of Volume 4 of 2006 Guidelines.

The inventory of greenhouse gas emissions requires the assessment of all 5 IPCC carbon pools as well as emissions of non-CO_2 gases, stratified by climatic zones and conversion types. For example, the initial raising of the water table might result in death of all or part of the living biomass, with transfers to dead organic matter and litter pools or decomposition and losses to the atmosphere. However, with time, original vegetation might colonize the wetland, increasing living biomass pool again. Thus, changes and transfers in C pools will vary according to the stage of the conversion until a new steady state is achieved.

The appropriate Tier to be used in the inventory can be decided based on Figure 1.3 of Chapter 1 of Volume 4 in the 2006 Guidelines.

When a new wetland is created transfers of C among pools can be abrupt or follow different transitional stages until a new steady state is achieved. For this reason, all carbon pools and exchanges among them have to be accounted for in the first year of conversion and in the subsequent 9 years. The 10 year transitional stage is assumed based on studies of vegetation recovery during wetland restorations but might not apply for every land use change. Countries are encouraged to establish monitoring programs in representative plots of key categories to determine which time frame is more appropriate.

5.3.2.1 BIOMASS

TIER 1.

In a Tier 1 approach changes in biomass carbon stock wetland creation are calculated using Equation 2.15 from Chapter 2 of Volume 4 of the 2006 Guidelines. For simplicity, in Tier 1 it can be assumed that once the land is rewetted all the vegetation will initially die and the resulting plant organic matter will decompose, with the ecosystem reaching a new steady-state immediately after the conversion. Average C stock changes are calculated from the difference between initial and final stocks. Default values for biomass carbon stocks for each type of other land use (Forest, Grassland, Cropland) can be obtained from their respective chapters in Volume 4 of the 2006 Guidelines. After this transition period of 10 years the land is assumed to have reached a new equlibrium state and would be reported in a “land remaining” category (e.g. forest land remaining forest land,or grassland remaining grassland).

Tier 1 requires the estimation of biomass stocks in land before and after the conversion to Wetland. Although rewetting will cause variable death rates depending on the type of living biomass prior to conversion, at this level a simple assumption can be made that all biomass is lost and the ecosystem achieves a new steady-state in the year of conversion. Hence, biomass stock after conversion is zero. Initial biomass is estimated using the methods provided in the 2006 Guidelines for each type of Land cover, that is, Chapter 4 for Forest, Chapter 5 for Cropland and Chapter 6 for Grassland. Annual change in biomass carbon stock is calculated using Equations 2.15 and 2.16 from Chapter 2 of Volume 4 of 2006 Guidelines. Since biomass after conversion is assumed to be zero, ΔCCONVERSION term in equation 2.15 also equals to zero. If data is not available, countries might use global biomass stocks datasets and emission/removal factors suggested in the referred respective chapters.

TIER 2

Tier 2 uses country-specific data for each climatic, ecosystem type and management practice to estimate biomass changes following Land conversion to Wetland. In this case the Tier 1 assumption of a new steady-state condition in the first year is replaced by the monitoring of the transitions in time of average carbon pools. Equations 2.5 and 2.6 from Chapter 2 of Volume 4 of the 2006 Guidelines are used. To estimate biomass pools suggested methods and respective equations are the same as those in Section 5.2.1.above.

It is good practice to implement country-specific biomass and carbon stock inventories. If national data are not available for Tier 2, countries can use globally-compiled databases (e.g. FAO) and assume that wetland vegetation does not have substantially different biomass carbon densities than terrestrial vegetation (Bridgham et al., 2006). It is also good practice to use modern satellite imagery and field surveys to estimate sub-types of Wetlands, as delineated in Chapter 3 of the *2006 IPCC Guidelines*. Where resources are not available to obtain country-specific data, wetland cover can also be derived from globally-compiled databases (e.g. WWF Global Lakes and Wetlands Database – GLWD). Since several definitions exist for the classification of land into the Wetland category (Mitsch and Gosselink, 2011), it is good practice that countries explicitly describe criteria of classification.

Under a Tier 2 approach empirical data is used to evaluate the evolution in time of biomass stocks in Other Land Converted to Wetland immediately after the conversion and in the following 9 years of succession. It is good practice to obtain country-specific data from each previous kind of vegetation (forest, crop and grassland) under each climatic region and to subsequently follow changes in biomass stocks according to management practices under the Land Converted to Wetland. In biomass carbon stock changes accounted for longer periods of time, results are converted to average annual values.

TIER 3

Tier 3 uses country and ecosystem specific data to model the evolution of biomass carbon stocks in time, from the conversion year until a new steady-state is reached

5.3.2.2 DEAD ORGANIC MATTER

Dead Organic Matter and Litter summed constitute the Dead Organic Matter pool. These pools vary greatly according to the type of initial land use (Forest, Crop, Grassland or Other Uses) and the rates of transfers from other pools to the DOM pool will also be quite different depending on the velocity of the conversion and the types of *Other Land Use Converted to Wetland*. For example, after rewetting of a forest leaves might fall and decompose while trunks will remain as DOM, whereas grasses might be all decomposed in the first year of conversion. If anaerobic conditions develop after complete rewetting, decomposition might slow down after the initial phases of transition.

Because average DOM stock changes will depend on the stage of the transition from one type of land use to another it is good practice to follow the transitional process until a new steady-state is reached. Estimates should be expressed on an annual average basis for consistency. Given the variability of the DOM stock, there are no default values for this carbon pool and is good practice that countries strive to obtain country specific data.

For simplicity Tier 1 assumes that a new steady-state is achieved in the first year of conversion while it is more likely that the transition for this new state will take longer and countries are encouraged to use Tiers 2 and 3 to obtain more accurate estimations. If countries choose to use Tier 1 method after the first year this land should be classified as *Wetland Remaining Wetland.*

5.3.2.3 SOIL CARBON

Few studies have assessed the carbon implications of restoring and creating IMS wetlands. In general, the restoration and creation of IMS wetlands leads to increases in soil C stocks over time (Badiou et al. 2011; Wolf et al. 2011; Ballantine and Schneider 2009; Meyer et al. 2008; Wiggington et al. 2000). Research on C dynamics in created wetlands is sparse with only a few a studies addressing C stocks in soils or greenhouse gas fluxes (Sha et al. 2011; Wolf et al. 2011; Ahn and Peralta 2009). Although there are more data on C pools and fluxes from wetland restoration studies, many do not report bulk density so that stocks can be calculated (e.g. Hartman et al. 2008), measurement systems such as eddy covariance include landscape components other than restored wetlands (e.g. Herbst et al. 2011), or difficulty in interpreting numbers to make independent calculations (e.g. Lu et al. 2007). The most detailed analysis of both soil c stocks and greenhouse gas fluxes in restored wetlands has been conducted in the Prairie Pothole Region of Canada and the United States (Badiou et al. 2011; Card et al. 2010; Gleason et al. 2009; Euliss et al. 2006).

TIER 1

Tier 1 characterises the direct soil emissions of CO_2 by climate zone, management practices and soil type, whereas for non-CO2 emissions, Tier 1 estimates include factors of climate zone, management practices and inundation regime Equation 5.4 gives the method that is applicable to CO_2, CH_4 and N_2O. Emissions are estimated by multiplying the emission rate for pristine wetlands by an adjustment factor AF. Values of F are given in table 5.5. Emission rates for pristine wetlands for CH_4 are given in tables 5.2 and 5.3 and for N_2O in table 5.4 with emission rates for CO_2 listed in table 5.6, below.

EQUATION 5.4

SOIL EMISSIONS FROM DRAINAGE, CREATION, AND RESTORATION OF IMS WETLANDS

$$E_{p,l} = EF_p \times AF_{p,l} \times A_l$$

Where:

$E_{p,l}$ = Emissions from Soil from management, restoration or creation of IMS wetlands, (tonnes)

With:

p = gas (CO_2, CH_4 or N_2O)

l is land use type.

$AF_{p,l}$ = Adjustment Factor to account for drainage, creation and restoration for gas p and land type l, (+ve = emission, -ve = removal) Table 5.5 provides default values. (dimensionless)

EF_p = Emission Factor of pristine wetland for gas p

A_l = area of land type l (ha)

Table 5.5 provides emission adjustment factors based on the data reported in the studies above. However, the numbers were obtained from many different approaches, such as modelling by Li et al. (2004), gaseous fluxes measurements by Page and Dalal (2011) and biomass changes measured by Ballantine and Schneider (2009). Countries are encouraged to develop their own studies based on local climate and vegetation types.

TABLE 5.5. SOIL EMISSION ADJUSTMENT FACTORS DUE TO DRAINAGE OR RESTORATION OF IMS WETLANDS TO BE USED IN EQUATION 5.4 (DIMENSIONLESS) (TO BE COMPLETED)

Region		Deforestation/Drainage			Restoration			Source
		CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	
Boreal		+3	-1	+9	-1.2	+7	-7	Li et al., 2004*
					-3	-0.7 to +1	+0.3 to +10	Badiou et al, 2011[+]
Temperate		+1	-0.25 to -1	+0 to +30				Page & Dalal, 2011[++]
					0	0	0	Gleason et al., 2009[#]
	North America	+0.25			-0.4			Euliss et al. 2006[†]
	Canada	+0.2			-0.3			Euliss et al. 2006[†]
	Humid				-0.2			Mckenna, 2003**
Sub-Tropical		+2	-1	+8	-1.5	+47	-1	Li et al., 2004*
	Humid				-8			Craft et al., 2003[††]
	Continental				-0.6 to -0.7			Ballantine & Schneider, 2009[‡]
Global		+1						Bridgham et al., 2006***

Notes:

* Values based on a process-based model, Wetland-DNDC

[+] Praire potholes, long-term restoration (> years)

[++] Melaleuca freshwater forests

\# Wetland Croplands compared to Wetland Grasslands

[†] Aquatic croplands, values computed as OC over a period of 10 years

[††] 1 year old restored marsh

** Woodland converted to upland grass wetland 6 years before the study

[‡] Depressional wetlands with 10 to 50 years of restoration*** Estimates based on losses of wetland area only

TABLE 5.5. SOIL CO_2 EMISSION FACTORS FOR IMS WETLANDS
TO BE USED IN EQUATION 5.4 (DIMENSIONLESS) (TO BE COMPLETED)

Region		CO_2	Source
Boreal			
Temperate			
	North America		
	Canada		
	Humid		
Sub-Tropical			
	Humid		
	Continental		
Global			
Notes: **To be completed**			

TIER 2 AND 3

Little information is available to conduct Tier 2 and Tier 3 soil C stock analyses for Lands Converted to IMS wetlands unless in the Prairie Pothole Region of Canada and the United States (see below). Outside of this region single studies have been done in New York (Ballantine and Schneider 2009), Virginia (Wolf et al. 2011), North Carolina (Morse et al. 2012), South Carolina (Wiggington et al. 2000), Florida (Schipper and Reddy 1994), Louisiana (Hunter et al. 2008), Ohio (Zhang and Mitsch 2007), Nebraska (Meyer et al. 2008), and Oregon (Pfeifer-Meister 2008) within the United States, and in Denmark (Herbst et al. 2011) and China (Lu et al. 2007). Although there are several studies listed from the east coast of the United States, the wetland types are very different ranging from riparian bottomland hardwood ecosystems in South Carolina (e.g. Wiggington et al. 2000) to marsh ecosystems in New York (Ballantine and Schneider 2009).

Due to the number of studies that have assessed soil C stock changes resulting from the restoration of prairie pothole wetlands, a direct rate change for this Region is reasonable. Most recently Badiou et al. (2011) calculated the mean soil C sequestration rate for the Canadian part of the Prairie Pothole Region to be 2.70 Mg C ha^{-1} yr^{-1} which is slightly lower than the 3.05 Mg C ha^{-1} yr^{-1} found by Euliss et al. (2006) for semi-permanent prairie potholes in the United States part of the Prairie Pothole Region. Based on the large number of wetlands assessed in both studies, 2.7 to 3.1 Mg C ha^{-1} yr^{-1} are reasonable bounds for estimating soil C stock changes in prairie pothole wetlands. For IMS wetlands other than prairie potholes, estimation of soil carbon stock changes due to management activity requires a two-step approach. Stocks are first estimated in previous land use according to their respective matrix of vegetation, climate and management practices and methodologies described in the appropriate Chapters of Volume 4 of the 2006 Guidelines (e.g. Chapter 4 if the original land use is Forest). Then stocks are estimated in the newly wetted soils employing the methods described in Section 5.2.3 (Soil Carbon).

5.4 COMPLETENESS, TIME SERIES CONSISTENCY, QA/QC, AND REPORTING AND DOCUMENTATION

Consistent reporting is a major issue for IMS wetlands because multiple activities or land uses may occur. In addition to managed peatlands and flooded land already given in IPCC 2006, a complete carbon inventory on this land use should include CO_2, and non-CO2 emissions and removals from Wetlands Converted to Other land uses and activities of wetland management (i.e. biomass burning, harvesting).

The countries selecting other land uses in inland mineral wetlands should not change during the whole reporting period to avoid double accounting. For example, if a forested wetland has been reported as a forest it should be reported as a forest during the whole series. It is suggested that flooded lands, peatlands, and coastal wetlands are clearly excluded from IMS wetlands and this separation is applied consistently throughout the reporting period.

It is good practice to disaggregate the type of IMS wetlands according to national circumstances and employ national emission factors if possible. Carbon stocks and fluxes are highly variable for wetland type. It is also good practice to apply methods that separate riparian ecosystems from upland ecosystems considering the usually larger biomass stocks in riparian ecosystems.

5.5 FUTURE METHODOLOGICAL GUIDANCE

IMS wetlands are significant compartments in carbon cycle. However, accounting carbon emissions and removals is challenging due to diversity in this land use. The diversity is not only caused by soil and climatic conditions but also seasonality of water table. Changes in water table affect CO_2 and CH_4 emissions and N_2O in some cases considerably. Besides, mineral wetlands are reported under other land uses when they fit under the definition of forest, agriculture, or grassland.

It is clear that removal of biomass in wetlands with human activities like harvesting, or grazing would affect the stocks or fluxes of carbon different than upland conditions. Particular effort should be employed to differentiate multiple uses in relation with wetlands (i.e. forested wetlands, wet grasslands) for future methodological improvements.

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