4 CHAPTER 4

COASTAL WETLANDS

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4.1 INTRODUCTION

This chapter provides guidance on estimating and reporting anthropogenic greenhouse gas (GHG) emissions and removals from managed coastal wetlands.

Coastal wetlands are defined here as organic and mineral soils vegetated by vascular plants (eg. marsh grasses, seagrasses, mangroves) that are covered or saturated for all or part of the year by tidal freshwater or salt water (>0.5ppt). The boundary of coastal wetlands is recognized as the landward extent of tidal inundation and extending seaward to the depth of vascular vegetation. This definition is very similar to a recent definition in a specialized treatise on coastal wetlands (Perillo *et al.* 2009). In the 2006 IPCC Guidelines for National Greenhouse Gas Inventories, wetlands were defined as any land that is covered or saturated with water for all or part of the year, and that does not fall into the Forest Land, Cropland, or Grassland categories. While this definition should still be used for classifying land and for reporting emissions, the guidance in this chapter applies to all coastal wetlands whatever IPCC land class they fall into.

Coastal wetlands are important habitats for both ecological and economic reasons, providing numerous ecosystem services for humanity, including food, fuel, and building material (Walters et al 2008; Geden et al 2009). Coastal wetlands play a wide spectrum of roles as nursery grounds for many terrestrial and marine species, in maintaining coastal water quality, in preventing or ameliorating land erosion, in sequestering carbon and other elements, and as important transformers of nutrients often derived from land and sea. Coastal wetlands are found on all continents and at all latitudes, covering more than 1,250,000 km^2 of coastline worldwide, and are vital links between land and the open ocean. It is estimated that globally up to 50% of historic natural coastal wetlands have been degraded or converted to other land uses by human activities. Conversion of coastal wetlands continues.

For ease of use, methods and guidance are given in three discrete sections: *Management Changes in Coastal Wetlands* (Section 4.3), *Drainage of Coastal Wetlands* (Section 4.4), *Rewetting and Restoration of Coastal Wetlands* (Section 4.5), and as defined in Table 4.1. This chapter covers coastal wetlands that have been cleared or degraded, restored, created or constructed, and where resource extraction occurs with little or no apparent degradation (e.g., selective logging of mangrove trees).

TABLE 4.1
SECTIONS ADDRESSING MAJOR GREENHOUSE GAS EMISSIONS FROM MANAGED COASTAL WETLANDS

Land-use category/GHG	Section
Management Changes in in Coastal Wetlands (This section covers coastal wetland converted to other land uses in which the vegetation is altered (e.g. such as harvesting mangrove trees), or through disruption of natural drainage, and/or modifications to soils.	
CO_2	Section 4.3.2, 4.3.3, 4.3.4
CH_4	Section 4.3.5
N_2O	Section 4.3.5
Drainage of Coastal Wetlands (This section covers wetlands that have been drained in which the soil water table is lowered, such as conversion for arable agriculture and urban settlement).	
CO_2	Section 4.4.2, 4.4.3, 4.4.4
CH_4	Section 4.4.5
N_2O	Section 4.4.5
Rewetting and Restoration of Coastal Wetlands (This section covers rewetting of coastal wetlands that occurs where hydrology has been altered but soil is at appropriate elevation for establishment of functioning vegetation).	
CO_2	Section 4.5.2, 4.5.3, 4.5.4
CH_4	Section 4.5.5
N_2O	Section 4.5.5

Human Activities and Sub-categories

Coastal wetlands are managed for a wide variety of purposes, and for guidance, we have categorized wetland type further in relation to the specific type of human impact in Table 4.2.

TABLE 4.2
CLASSIFICATION OF KNOWN HUMAN IMPACTS ON COASTAL WETLANDS, THEIR LAND-USE CHANGE, AND THE CORRESPONDING SECTION IN THIS CHAPTER FOR GUIDANCE

MANAGEMENT CHANGES IN COASTAL WETLANDS: CHAPTER SECTION: 4.3	
Activity	**Sub-category**
Aquaculture	Fish or Shrimp Ponds; Fish Cages
Salt Production	
Extraction	Dredging, Ports, Harbors; Beach Nourishment
Nutrient Enrichment	
Nutrient Management	Addition; Removal
Hydrological/Sediment Diversion	Impoundments; Barriers
Nutrient Management	Addition; Removal
Fire Management	
Harvesting	Fish, Shellfish; Wood and Non-Wood Products; Grass/Forbs
Recreation	Boat Anchoring, Mooring; Boating; Hunting
Grazing (Soil + Fauna)	Cattle, Sheep, Camel, Horse
DRAINAGE OF COASTAL WETLANDS (W-D): CHAPTER SECTION: 4.4	
Modified Vegetation	Afforestation/Reforestation; Alteration of Vegetation
Hydrological/Sediment Diversion	Impoundments; Barriers
Ditching	Vector Control; Access Regulation
Agriculture	Arable Grass; Pasture
Draining and Filling	Settlement; Urbanization
Salt Production	
REWETTING AND RESTORATION OF COASTAL WETLANDS (D-W): CHAPTER SECTION: 4.5	
Restored Hydrology	Rewetting of drained soils; Restored Tidal (frequency), Connectivity
Revegetation	Afforestation/Reforestation; Reseeding; Alteration of Vegetation
Restoration through sediment modifications or enhancement	Sediment Removal; Ditch Filling; Sediment Resupply

Methodological assumptions and issues more specific to the three wetland-use categories are discussed in the corresponding sections of this chapter. Readers are referred to Chapter 2 (Generic Methodologies Applicable to Multiple Land-Use Categories) of Volume 4 of the 2006 IPCC Guidelines for the basic equations to estimate greenhouse gas emissions but guidance is duplicated here for the convenience of the inventory compiler. The sections below are intended to provide carbon emissions and removals utilizing approaches based on emissions and changes in carbon stocks.

4.2 GENERAL METHODOLOGICAL FRAMEWORK

The general methods used follow the guidance given in the 2006 Guidelines, Volume 4, especially chapter 2. This section highlights some specific considerations for coastal wetlands.

The general methodological framework and generic methods described herein can be applied when reporting removals and emissions associated with the land use activity changes in *Management Changes of Coastal Wetlands* (Section 4.3), *Drainage of Coastal Wetlands* (Section 4.4), *Rewetting and Restoration of Coastal Wetlands* (Section 4.5). The specific use of the generic methods described in Section 4.2 is provided under each of these sub-section headings. The reader is referred to Table 4.2 for guidance related to a specific activity and the subsection in which it is included. Following the general approach of the 2006 GLs, land use activity changes can result in emissions and removals in each carbon pool (soil, biomass, dead organic matter) and in non-CO2 emissions. These changes are summed within each pool.

EQUATION 4.1

ANNUAL CARBON STOCK CHANGES FOR COASTAL WETLANDS AS A SUM OF CHANGES IN ALL POOLS AND LAND USE ACTIVITY CHANGES

$$\Delta C_{CW} = \Delta C_M + \Delta C_D + \Delta C_R$$

$$\Delta C_i = \Delta C_{ABi} + \Delta C_{BBi} + \Delta C_{DWi} + \Delta C_{LIi} + \Delta C_{SOi} + \Delta C_{HWPi}$$

Where:

ΔC_{CW} = carbon stock changes for Coastal Wetlands

ΔC_M = Management Changes in Coastal Wetlands
ΔC_D = Drainage of Coastal Wetlands
ΔC_R = Rewetting and Restoration of Coastal Wetlands

ΔC_{ABi} = carbon stock changes for aboveground biomass
ΔC_{BBi} = carbon stock changes for belowground biomass
ΔC_{DWi} = carbon stock changes for dead wood
ΔC_{LIi} = carbon stock changes for litter
ΔC_{SOi} = carbon stock changes for soil
ΔC_{HWPi} = carbon stock changes for harvested wood products

Emissions and removals of CO_2 are based on changes in ecosystem C stocks for each management change for each pool of Coastal Wetlands using either the *Gain-Loss Method* (2006 GL, Vol. 4, Eq. 2.4) or the *Stock-Difference Method* (2006 GL, Vol. 4, Eq. 2.5). Non-CO_2 emissions are estimated with an emission rate from a source directly to the atmosphere (Volume 4, pg. 2.10). CO_2 emissions may also be estimated using an emission rate but is here considered as a parameter within the soil pool estimate. The use of each methodological approach is based on the activity that is reported. Clear guidance on using these methods are provided under each of the subsections detailed in this Chapter. Refer to Table 4.2 for guidance related to a specific activity and the subsection in which it is included. The guidance has been developed to allow the inventory compiler the flexibility to report on an activity that may not be included in this methodological guidance of this Chapter, especially where Tier 2 methods should be applied. It is *good practice* to report neither over- or under-estimates and to perform validation checks that all pools have been considered and to avoid possible omissions or double-counting (eg. transition matrices Table 2.1, pg. 2.19).

The choice of Tier level for estimating emissions and removals associated with management changes, drainage or rewetting and restoration in coastal wetlands is determined following a decision tree with decisions based on available data and key category analysis (Figure 4.1). The decision tree applies similarly for each of the C pools and non-CO2 emissions as for activity data.

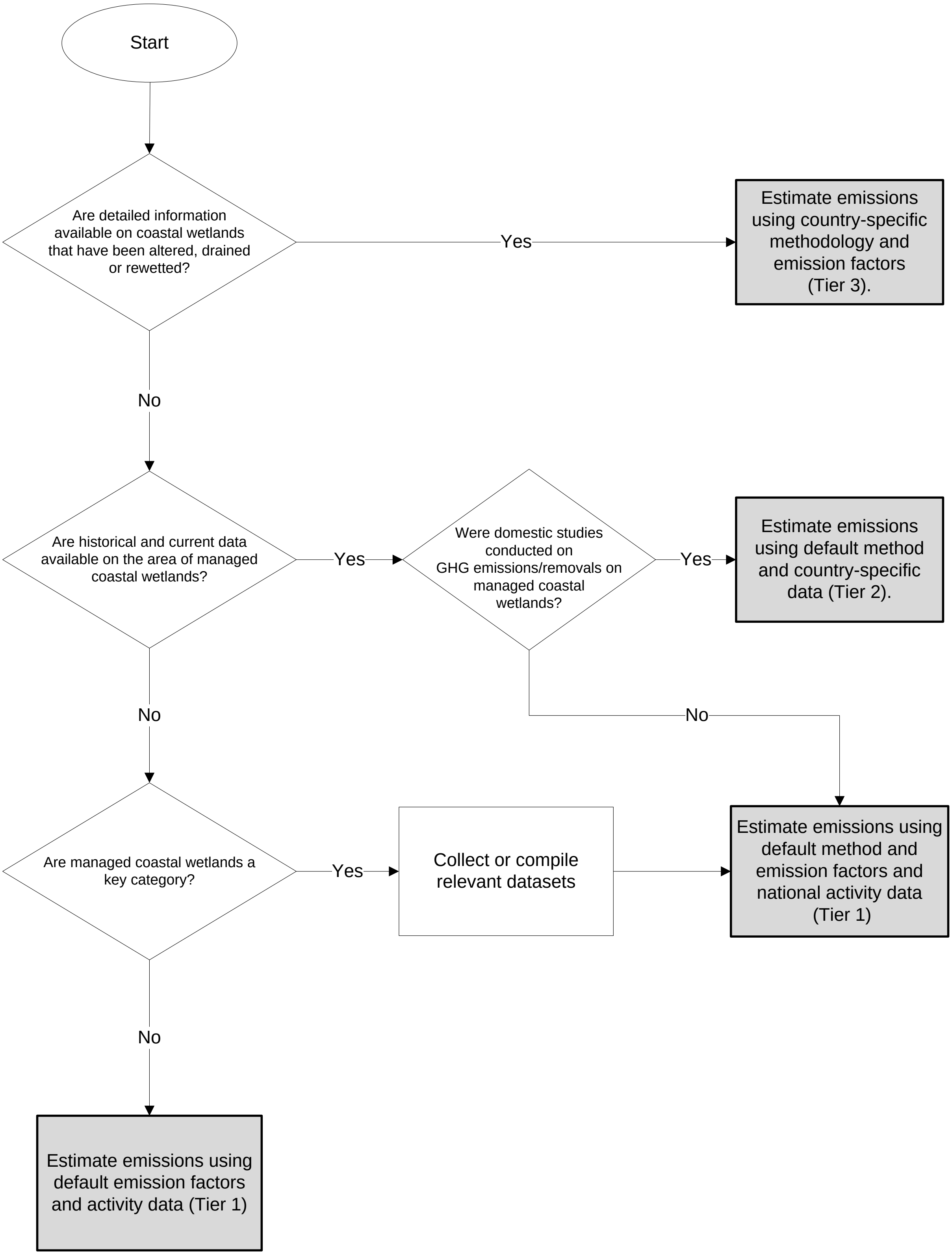

Figure 4.1 Decision tree for identification of appropriate Tier to estimate changes in carbon emissions and removals in Coastal Wetlands.

4.2.1 Change in soil carbon

Activities associated with land-use change in coastal wetlands can influence organic, mineral and inorganic stocks of C in soils. While data on inorganic stocks of C (i.e. dissolved inorganic carbon) are not sufficient to provide generic methodologies, the methodological approach provided here takes into account the possible emissions and removals associated with changes in C of organic and mineral soils.

Previous guidance on C in organic soils (Chapter 2, pg. 2.28 and Chapter 7) omitted C gains that are possible through coastal wetland restoration. This supplement includes good practice methodologies to report on C gains and C losses. Activities in coastal wetlands such as restoration where C gains should be reported are presented along with methodologies to estimate CH_4 losses. This is especially important for some coastal wetlands where CH4 emissions are low or considered to be negligible (Bridgham et al, 2006; Poffenbarger et al 2011) and for higher Tier methods.

TIER 1

Many land-use (i.e. management) activities specific to coastal wetlands can result in changes to soil carbon stocks. For mineral and organic soils, the generic default method is the same at Tier 1 level of estimation. As mineral and organic soils are difficult to differentiate using activity data that would likely be used for coastal wetlands at Tier 1 level of estimation, the following methodology assumes they are aggregated. The equation for estimating the total change in soil C stocks has been adapted from Equation 2.24 in the 2006 GLs to include C gain and is given in Equation 4.2. Depending on management activity, the parameters used to determine emissions and removals from coastal wetland soils vary. Thus, inventory compilers are thus referred to the activity sections 4.3-4.5 (below) for more detailed guidance and for emission/removal and stock change factors specific to those activities.

EQUATION 4.2

ANNUAL CHANGE IN CARBON STOCKS IN SOILS

$$\Delta C_{SO} = \Delta C_{MIN/ORG} + \Delta C_{INORGANIC}$$

Where:

ΔC_{SO} = annual change in carbon stocks in soils, tonnes C yr^{-1}

$\Delta C_{MIN/ORG}$ = annual change in organic carbon stocks in mineral and organic soils, tonnes C yr^{-1}

$\Delta C_{Inorganic}$= annual change in inorganic carbon stocks from soils, tonnes C yr^{-1} (assumed to be zero unless using a Tier 3 approach)

Soil organic C in mineral soils is assumed to exist to a default depth of 1m. While changes in soil organic C in organic soils can occur throughout the depth profile, they are assumed to occur within 1 m for Tier 1 estimates; in wetlands, carbon content in soils generally declines with increasing depth. Residue/litter C stocks are not included because they are addressed by estimating dead organic matter stocks.

In a similar fashion to the methodological approaches applied for soils in the 2006 GL, it is possible that countries will use different Tiers to estimate emissions and removals from C stocks in mineral and organic soils and inorganic C. The inventory compiler can refer to the Decision Tree found in Figures 2.4 and 2.5 on pgs. 2.32 and 2.33 of the AFOLU Volume for guidance on Tier levels that should be applied.

Depending on management activity, C changes in organic soils may be determined by: 1) assigning an annual emission or removal factor that estimates losses or gains of C following an activity (Equation 4.3) or 2) stock change factor approach (Equation 4.4). *Whether approach 1 or 2 is used for estimating C change in organic soils, non-CO_2 emissions must also be reported.*

Tier 1, Approach 1 – Annual Emission/Removal Factor

EQUATION 4.3

ANNUAL CHANGE IN CARBON STOCKS IN SOILS

$$\Delta C_{MIN/ORG} = G_{MIN/ORG} - L_{MIN}/_{ORG}$$

$$G_{Min/Org} = \sum_{c,s,e} (A_{c,s,e} \bullet RF_{LUc,s,e}) \qquad L_{Min/Org} = \sum_{c,s,e} (A_{c,s,e} \bullet EF_{LUc,s,e})$$

Where:

$G_{Min/Org}$ = annual carbon gain from rewetted or restored soils, tonnes C yr^{-1}

$L_{Min/Org}$ = annual carbon loss from drained soils, tonnes C yr^{-1}

A = land area of drained organic soils

c = climate zones

s = salinity level (if applicable)

e = ecosystem type (i.e. mangrove, tidal freshwater wetland, seagrass, salt marsh; oceanic, estuarine)

$RF_{LUc,s,e}$ = removal factor for each land-use subsystem resulting in rewetting or restoration (i.e. management activity) disaggregated climate, salinity level and ecosystem type

$EF_{LUc,s,e}$ = emission factor for each land-use subsystem resulting in drainage (i.e. management activity) disaggregated climate, salinity level and ecosystem type

Tier 1, Approach 2 – Stock Change Factor

In the case where coastal wetlands can be disaggregated by soil type (e.g. where land areas are classified by soil type and higher level ecosystem type classifications are not available), the stock change factor approach to estimating the change in C in organic soils is the appropriate Tier 1 default method.

For all soils, the default methods are based on changes in soil C stocks over a finite period of time (40 year transition period). The change is calculated based on C stock after the management changes relative to the carbon stock in a reference condition (i.e native vegetation that is not degraded or otherwise altered). At Tier 1 the following assumptions are made:

(i) Over time, soil organic C reaches a spatially-averaged, stable equilibrium value specific to the management state. This implies C fluxes approach zero in the equilibrium state; and

(ii) Soil organic C stock changes during the transition to a new equilibrium occur linearly.

While soils in wetlands subject to tidal flooding continuously accumulate Carbon (Chmura et al. 2003), this rate of accumulation can be small in the absence of changes in water level. For the purpose of Tier 1 estimation however, the assumption is a 40 year transition period, with C changes occurring linearly, until equilibrium at the end of the transition period. Under some management conditions, such as those that include drainage, any accumulation stops and soils may tend towards an equilibrium C content. Emissions for CO_2 from coastal organic soils are comparable with emissions from terrestrial organic soils for a given management practice and climate zone (e.g. Deverel and Rojstaczer, 1996; Deverel and Loughton, 2010; Yit et al., 2011, Zenello et al., 2011). Although soil C changes in response to management changes may better be described by other simple model fits, assumption (ii) greatly simplifies the Tier 1 methodology.

Using the default method, changes in soil C stocks are computed over an inventory time period, likely established based on the years in which the management activity data are collected. Thus, management activity data collected in 1990, 1995, 2000 would correspond to inventory periods 1990-1995 and 1995-2000. The stock change factor approach described here takes the difference in SOC between periods of change in management activities based on multiplying the reference C stocks by the stock change factors. Annual rates of carbon stock change are then estimated using a *Stock-Difference Method* (i.e. as the difference in stocks at two points in time divided by the time dependence of the stock change factors.

The following was adapted from Equation 2.25 (Volume 4, Chapter 2):

EQUATION 4.4

ANNUAL CHANGE IN ORGANIC CARBON STOCKS IN COASTAL WETLAND SOILS

$$\Delta C_{MIN/ORG} = (SOC_{M/O,a} - SOC_{M/O,b})/D$$

$$SOC = \sum_{c,s,e} (SOC_{REFc,s,e} \bullet F_{LUc,s,e} \bullet F_{MGc,s,e} \bullet A_{c,s,e})$$

$\Delta C_{MIN/ORG}$ = annual change in C stocks in coastal wetland soils, tonnes C yr^{-1}

$SOC_{M/O,a}$ = equilibrium soil organic carbon stock following a change in management tonnes C

$SOC_{M/O,b}$ = equilibrium soil organic carbon stock before a change in management tonnes C

D = number of years for transition to take place, default = 40 yr

c = climate zones

s = salinity level (if applicable)

e = ecosystem type (i.e. mangrove, tidal freshwater wetland, seagrass, tidal salt marsh; oceanic and estuarine)

SOC_{REF} = the reference carbon stock, tonnes C ha $^{-1}$

F_{LU} = stock change factor for land-use subsystem (i.e. management activity)

F_{MG} = stock change factor for management regime (i.e. intensity)

A = land area of the stratum being estimated, ha

$SOC_{M/O,a}$ and $SOC_{M/O,b}$ are calculated using the SOC equation above where the reference carbon stocks and stock change factors are assigned according to the land-use subsystem, management regime (if applicable) and corresponding areas at each point in time. The time dependence of stock change factors (D) is the default time period for transition between equilibrium SOC values (40 years). The time dependence considers assumptions made in computing the stock change factors F_{LU} and F_{MG}. In disaggregating land area (A), all land in a stratum should have common biophysical conditions and management history over the inventory time period. For wetlands on mineral soils, assume that 50% of organic material held within the top 1 meter of soils is emitted over a 40 year period and that the remaining carbon is recalcitrant. For example, temperate wetland converted to arable land or settlement EF = (0.5 * 351)/40 Mg C ha^{-1} yr^{-1} over a 40 year period.

Inventory calculations are based on land areas that are disaggregated by climate zones, salinity level and ecosystem type where stock change factors for management activities apply. All levels of disaggregation MAY or MAY NOT be included in estimation of soil C stock change as not all management activities apply in all ecosystems at all salinity levels at any Tier level approach. However, where a management activity is a key category, countries should disaggregate to the level that will provide the best estimation possible utilizing country-specific data.

See land-use subsections 4.3 – 4.5 for tables of soil stock change factors where applicable. Refer to pg. 2.34, Box 2.1 for more information on alternative formulations of Equations 4.5 and 4.6 when using different land representation approaches for activity data. Methods are presented for separating stocks and stock change factors at these levels of disaggregation based on available data or for a Tier 2 or higher level of estimation.

TABLE 4.3. DEFAULT VALUES FOR CARBON STORED IN COASTAL WETLANDS (SOC_{REF}) BASED UPON AVERAGE CARBON DENSITY MEASURED IN TOP 1 M.

Climate Zone	Ecosystem	Number of Observations	Mean (Mg C Ha^{-1})	Range (Mg C ha^{-1})
Boreal and Temperate	Tidal Marshes	134	351	47-1900[1]
Tropical and Sub tropical	Mangroves	94	485	155-1150[1]
Global	Seagrass	89	194.2±20.2	9 – 628[1]

Sources: Chmura et al . 2003, Silfleet et al., 2011, Donato et al., 2011; Fourqurean et al., 2012; Data for tidal marsh soil stocks is derived from the following locations (with number of observations): Northeast Canada (37), Gulf of Mexico (Louisiana, Texas, Mississippi) (26), New England (20), Chesapeake Bay (12), California (6), North Carolina (6), UK, (6), California (5), Florida (5), Netherlands (4), Denmark (2), Rhone Delta, France (1), British Columbia, Canada (1). Data for mangrove soil stocks is derived from the following locations (with number of observations): Orinoco Delta, Venezuela (40), Florida, USA (12), Kosrae (7), Sulawesi (6), Borneo (5), Yap (5), Boca Chica, Mexico (4), Umengi estuary, South Africa (4); Pohnpei Island, Micronesia (3); Columbia (3); Sundarbans, Bangladesh (2); Java (2), Palau (1).

TIER 2

Many land-use (i.e. management) activities specific to coastal wetlands can result in changes to carbon stocks in both organic and mineral soils. Tier 2 methods assume that mineral and organic soils can be disaggregated within ecosystem type. Other methods may be more appropriate depending on national circumstances. Regardless of the method used, Tier 2 and higher level of estimation require country-specific data which, to the extent possible, should be disaggregated by climate type and ecosystem type. The equation for estimating the total change in soil C stocks has been adapted from Equation 2.24 in the 2006 GLs to include C gain and is given in Equation 4.5:

EQUATION 4.5
ANNUAL CHANGE IN CARBON STOCKS IN SOILS

$$\Delta C_{SO} = \Delta C_{MINERAL} + \Delta C_{ORGANIC} + \Delta C_{INORGANIC}$$

Where:

ΔC_{SO} = annual change in carbon stocks in soils, tonnes C yr^{-1}

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

$\Delta C_{Organic}$ = annual change in organic carbon stocks in organic soils, tonnes C yr^{-1}

$\Delta C_{Inorganic}$= annual change in inorganic carbon stocks from soils, tonnes C yr^{-1} (assumed to be zero unless using a Tier 3 approach)

For Tier 2 methods, soil organic C in mineral soils is assumed to exist to a default depth of 1m. For wetlands on mineral soils, assume that 50% of organic material held within the top 1 meter of soils is emitted over a 40 year period and that the remaining carbon is recalcitrant. Residue/litter C stocks are not included because they are addressed by estimating dead organic matter stocks. While changes in soil organic C in organic soils can occur throughout the depth profile, carbon content in soils generally declines with increasing depth, but should also be considered. In disaggregating land area (A), all land in a stratum should have common biophysical conditions and management history over the inventory time period. For both mineral and organic soils, Tier 2 methods should apply different model assumptions about the fit of the decay or accumulation curves. Compilers should also develop national factors and parameters to match their national conditions.

In a similar fashion to the methodological approaches applied for soils in the 2006 GL, it is possible that countries will use different Tiers to estimate emissions and removals from C stocks in mineral and organic soils and inorganic C. The inventory compiler can refer to the Decision Tree found in Figures 2.4 and 2.5 on pgs. 2.32 and 2.33 of the AFOLU Volume for guidance on Tier levels that should be applied.

4.2.2 Change in biomass carbon stocks

Changes in biomass stocks follow guidance provided in the 2006 GL, Vol. 4, Chapter 2 and Chapter 4 for coastal wetland ecosystem types with woody biomass (i.e. mangroves and some salt marshes) and Chapter 2 and Chapter 6 for coastal wetland ecosystem types with non-woody biomass (most salt marshes and seagrasses). While biomass stock changes are typically not estimated for ecosystems with non-woody biomass at the Tier 1 level which are based on a steady-state assumption, it is good practice to evaluate the appropriateness of this assumption, and to report changes in non-woody biomass when it is not met or the management activity is associated with a key category. This assumption and steady-state consideration for non-woody biomass follows that for Grasslands in the 2006 GL and applies for non-woody coastal wetland ecosystems.

The decision tree in, Figure 4.1 provides guidance for selecting the appropriate Tier level for the implementation of estimation procedures for biomass associated with the rewetting of previously drained coastal wetlands. Estimation of changes in biomass requires an estimate of changes in above-ground vegetation and changes in below-ground biomass. Countries should use the highest Tier possible given national circumstances. It is *good practice* to use a Tier 2 or Tier 3 approach if carbon emissions and removals in coastal wetlands is a key category or if the sub-category of biomass is considered significant, based on principles outlined in Volume 1, Chapter 4.

Biomass stock changes can be estimated with either **Gain-Loss** or **Stock-Difference** methods as presented in Equations 2.7 and 2.8, Volume 4, Chapter 2, and summarized below.

Gain-Loss Method *(see Equation 2.7):* This method involves estimating the area of each type of conversion associated with rewetting of land that was previously a coastal wetland and the average annual transfer into and out of biomass stocks. This requires: (i) an estimate of the area under *Rewetting of Coastal Wetlands* according to different ecosystem type, different species within a particular ecosystem that could be related to climate or salinity or other ecological zoning of vegetation, disturbance regime, management regime, or other factors significantly affecting biomass carbon pools; (ii) the quantity of biomass accumulating in the biomass stocks; and (iii) the quantity of biomass lost from the biomass stocks on per hectare basis according to different type of vegetation that grows after rewetting due to the restoration of hydrology.

Stock-Difference Method *(see Equation 2.8):* This method involves estimating the area of each type of land conversion associated with change in the biomass stocks at two periods of time, t_1 and t_2. The biomass stock changes for the inventory year are obtained by dividing the stock changes by the period (years) between two measurements. The Stock-Difference Method is feasible for countries that have periodic inventories, and is more suitable for countries adopting Tier 3 methods. This method may not be well suited to regions with very variable climates and may produce spurious results unless annual inventories can be made.

TIER 1

The Tier 1 method, when combined with default biomass growth rates, or change in stocks, for a management activity allows any country to calculate the annual increase in biomass, using estimates of area and mean annual biomass increment for each stratum. In the case of coastal wetlands, these strata include possible disaggregation by climate, ecosystem type and salinity level. All levels of disaggregation may not be applicable**.** The Tier 1 level only considers biomass changes associated with woody vegetation. Biomass changes in non-woody vegetation are considered negligible except in the case of higher level Tier estimations or where there is a change in intensity of management in which case the assumption of biomass at an approximate steady-state (i.e., carbon accumulation through plant growth is roughly balanced by losses through grazing, decomposition and fire) is not valid, and the carbon stock changes can be significant. If it is reasonable to assume that coastal wetlands are not part of a key source, a country may apply the Tier 1 assumption of no change in biomass. However, if information is available to develop reliable estimates of rates of change in biomass, a country may use a higher Tier, even if C stock changes in coastal wetlands are not a key source, particularly if management changes are likely.

The change in biomass carbon stock on under Tier 1 should be estimated using Equation 2.15 of the 2006 guidelines. The average carbon stock change is equal to the sum of increase in carbon stock due to biomass growth following conversion, changes due to actual conversion (difference between biomass stocks before and after conversion), and decrease in carbon stocks due to losses after conversion. As a simplification for Tier 1, it is assumed that all biomass is lost immediately from the land after a conversion (Equation 2.16), even when there is no abrupt change. In this case the residual biomass (B_{AFTER}) is assumed to be zero, (i.e., the vegetation that characterizes the previous state, for example scrubland on drained wetland, is cleared of all vegetation before the new, coastal wetland state is established). In this case there is no transfer of biomass from the biomass pool to the dead wood pool, for example. Default values for mangrove, seagrass and salt marsh biomass prior to conversion can be found in Tables 4.4-4.8).

TIER 2

The Tier 2 calculations differ structurally in a number of ways from Tier 1. First, Tier 2 estimates use the two phase approach described earlier. Tier 2 relies on country-specific estimates of the biomass in initial and final land uses rather than the defaults, as in Tier 1. Land area estimates may be disaggregated at higher resolution spatial scales than in Tier 1 to capture country wide regional variations within and between the different coastal wetland ecosystems that result from the rewetting. Second, for Tier 2 countries may modify the assumption that biomass immediately following conversion is zero. This enables countries to take into account land-use transitions where some, but not all, vegetation from the original land use is removed. In addition, under Tier 2, it is possible to account for biomass accumulation following establishment of the rewetted coastal ecosystem over a several year period (rather than accounting all biomass stock change in the year of conversion) if data are available to estimate the time to full biomass establishment and the annual stock changes. Third, under Tier 2, it is *good practice* to apportion transfers of carbon between pools. Some of the rewetted coastal ecosystems may not contain significant carbon in the dead wood or litter pools, but woody biomass may persist that had previously accumulated during a particular land use on a previously drained coastal wetland. If the rate of land conversion is more or less constant, the assumption that all carbon in theses pools was lost at the time of conversion would be a reasonable first approximation. Where the rate of land conversion varies over time, it is appropriate to try to account for the transfer and release of carbon from litter, dead wood, and soil carbon pools. It is therefore necessary to distinguish immediate losses due to the conversion activities from the losses that occur in the years following the land conversion.

The immediate and abrupt carbon stock change in biomass due to rewetting of previously drainage of coastal wetlands under Tiers 2 and 3 will be estimated using Equation 2.16 in Chapter 2, where B_{AFTER} is assumed to be zero. During the transition period, pools that gain or lose C often have a non-linear loss or accumulation curve that can be represented through successive transition matrices. For Tier 2, a linear change function can be assumed. For a Tier 3 approach based upon these methods, it is *good practice* to use the true shapes of the curves. These curves are to be applied to each cohort that is under transition during the reporting year to estimate the annual change in the biomass carbon pools. For the estimation of changes in biomass carbon during the transition phase, two methods are suggested. The equations, related to the ***Gain-Loss Method*** and ***Stock-Difference Method*** are the same as those used in Chapter 2 of the 2006 guidelines.

TIER 3

Tier 3 methods are used where countries have country-specific emission factors, and substantial national data. Country-defined methodology may be based on detailed inventories of permanent sample plots for each coastal wetland ecosystem created through rewetting and/or models. For Tier 3, countries should develop their own methodologies and parameters for estimating changes in biomass. These methodologies may be derived from methods specified above, or may be based on other approaches. The method used needs to be clearly documented.

Tier 3 involves inventory systems using statistically-based sampling of biomass over time and/or process models, stratified type of coastal wetland ecosystem by management regime, climate and salinity. For example, if a previously drained coastal wetland was being used as pastureland and it was converted to salt march when inundated by seawater, validated species-specific growth models that incorporate management effects such as grazing intensity and fertilization, with corresponding data on management activities, could be used to estimate net changes biomass in the rewetted coastal ecosystem, over time. Models, together with periodic sampling-based biomass estimates, could be applied to estimate stock changes to make spatial extrapolations for areas of rewetting of previously drained coastal wetland.

Key criteria in selecting appropriate models include the ability to represent all of the ecosystem conversions and management practices that are represented in the activity data. It is critical that the model be validated with independent observations from country-specific or region-specific field locations that are representative of the variability of local environmental change as well as climate, salinity, soil and vegetation management systems in the country.

If possible, spatially explicit area estimates should be used to facilitate complete coverage of the converted area represented by the rewetting of previously drained coastal wetlands and ensure that areas are not over- or underestimated. Furthermore, spatially explicit area estimates can be related to locally relevant carbon accumulation and removal rates, and restocking and management impacts, improving the accuracy of estimates.

Regardless of Tier estimation method used, it is *good practice*, if possible, to develop and use a disturbance matrix (Example, 2006GL, Vol. 4, Table 2.1) for each biomass, dead organic matter and soil carbon pool, the proportion of the carbon remaining in that pool, and the proportions transferred to other pools, to harvested wood products and to the atmosphere, during the disturbance event. The proportions in each row always sum to 1 to ensure conservation of carbon. The value entered in cell A is the proportion of above-ground biomass remaining after a disturbance (or 1 – *fd*, where *fd*, the fraction of biomass lost in the disturbance, is defined in Equation 2.14, Volume 4, Chap 2). The Tier 1 assumption is that all of *fd* is emitted in the year of disturbance: therefore the value entered in cell F is *fd*. For higher Tiers, only the proportion emitted in the year is entered in cell F and the remainder is added to cells B and C in the case of fire, and B, C, and E in the case of harvest. It is *good practice* to develop disturbance matrix even under Tier 1 to ensure that all carbon pool transfers are considered, though all biomass carbon is assumed to be emitted in the year of land conversion. It is important to note that some of the transfers could be small or insignificant.

TABLE 4.4 RATIO OF BELOW-GROUND BIOMASS TO ABOVE-GROUND BIOMASS (R)				
Domain	**Vegetation type**	**R mean [tonne root d.m. (tonne shoot d.m.)$^{-1}$**	**R range or error [tonne root d.m. (tonne shoot d.m.)$^{-1}$**	**References**
Tropical	Mangrove	0.67	0.53-0.77	Donato et al. 2012 ; Kauffman et al. 2011 ; Komiyama et al. 2008 Liao et al. 1991 ; Lin and Lin 1988 ; Lin and Lu 1990
	Salt marsh			
	Seagrass (global average)	1.4	0.2-5.5/1.2	Duarte & Chiscano 1999
Subtropical/low Temperate	Mangrove	0.89	0.35-1.43	Komiyama et al. 2008 Lin and Lin 1988
	Salt marsh (low marsh, high marsh)	0.42, 3.4	±16%*, ±230%*	Buresh et al 1980; Darby and Turner 2008; Edwards and Millis 2005; Dunton et al 2001
Temperate	Salt marsh	0.91	±27%*	Whindham 2001
Mediterranean	Salt marsh	2.5	±89%*	Curco et al 2002; Neves et al 2007
Boreal	Salt marsh			
*95% CI				

TABLE 4.5 DEFAULT BIOMASS CONVERSION AND EXPANSION FACTORS (BCEF), TONNES BIOMASS (M^3 OF WOOD VOLUME)$^{-1}$ (IPCC 2006)

BCEF for expansion of merchantable growing stock volume to above-ground biomass ($BCEF_S$), for conversion of net annual increment ($BCEF_I$) and for conversion of wood and fuelwood removal volume to above-ground biomass removal ($BCEF_R$)

Climatic zone	**Forest type**	**BCEF**	**Growing stock level (m^3)**							
Humid Tropical			<10	11-20	21-40	41-60	61-80	80-120	120-200	>200
	natural forests	$BCEF_S$	**9.0** (4.0-12.0)	**4.0** (2.5-4.5)	**2.8** (1.4-3.4)	**2.05** (1.2-2.5)	**1.7** (1.2-2.2)	**1.5** (1.0-1.8)	**1.3** (0.9-1.6)	**0.95** (0.7-1.1)
		$BCEF_I$	4.5	1.6	1.1	0.93	0.9	0.87	0.86	0.85
		$BCEF_R$	10.0	4.44	3.11	2.28	1.89	1.67	1.44	1.05

Note: Lower values of the ranges for BCEFs apply if growing stock definition includes branches, stem tops and cull trees; upper values apply if branches and tops are not part of growing stock, minimum top diameters in the definition of growing stock are large, inventoried volume falls near the lower category limit or basic wood densities are relatively high. Continuous graphs, functional forms and updates with new studies can be found at the forest- and climate- change website at: http://www.fao.org/forestry/

Average BCEF for inhomogeneous forests should be derived as far as possible as weighted averages. It is good practice to justify the factors chosen. To apply BCEFI, an estimate of the current average growing stock is necessary. It can be derived from FRA 2005 at http://www.fao.org/forestry/

BCEFR values are derived by dividing BCEFS by 0.9

Sources: Brown S. *et al*., 1989; Brown S. and A. Lugo, 1992; Brown S., 2002; Fang J.Y., 2001.

TABLE 4.6 DEFAULT VALUES FOR ABOVE-GROUND BIOMASS OF MANGROVE AND TIDAL SALT MARSH WETLANDS (TO BE COMPLETED)

Domain	Vegetation type	Continent	Species	Above-ground biomass (tonnes d.m. ha^{-1})	References
Tropical	Mangrove	Pacific and Australia	*A. marina*	144.5	Briggs 1977
			B. gymnorrhiza	305.5 (97.6-436.4)	Komiyama et al. 2008
			B. sexangula	179.9 (79.0-279.0)	
			R. apiculata	249.2 (40.7-460.0)	
			Sonneratia spp.	225.2 (169.1-281.2)	
			Mixed mangrove forests	107.1 (57.0-172.0)	
			Mixed mangrove forest	392.0 (262-522)	Donato et al. 2012
			B. gymnorrhiza	47.3 (37.2-76.7)	Liao et al. 1991
		Atlantic and Africa	*C. tagal*	40.1	
			L. racemosa	51.7 (31.5-71.8)	
			R. mangle	147.5 (62.9-233.0)	
			R. mucronata	249.0	
			Mixed mangrove forests	151.2 (56.0-315.0)	
			A. alba	22.5 (21.1-23.3)	Mitra et al. 2011
	Salt marsh				
Subtropical	Mangrove	Pacific and Australia	*A. marina*	341.0	Mackey 1993
		Atlantic and Africa	*B. gymnorrhiza*	94.5	Komiyama et al. 2008
Subtropical	Mangrove	Pacific and Australia	*A. marina*	341.0	Mackey 1993
	Salt marsh				
Temperate	Salt marsh				
Boreal	Salt marsh				

TABLE 4.7 DEFAULT VALUES FOR ABOVE-GROUND GROWTH IN MANGROVE AND TIDAL SALT MARSH WETLANDS

Climate Zone	Country	Species	Net growth in above-ground biomass (tonnes d.m. ha^{-1} yr^{-1})	

Tropical	Cuba	*R. mangle*	1.6[a]	
		germinans	5.9	
		L. racemosa	5.4	
	Bangladesh	*Sonneratia apetala*	12.5	
		S. caseolaris	26.4	
		A. officinalis	7.6	
		A. marina	4.4	
		A. alba	2.1	
		gymnorrhiza	0.6	
		Bruguiera sexangula	0.1	
		Excoecaria agallocha	4.7	
		Xylocarpus moluccensis	0.5	
	Malaysia	*B. parviflora*	27.4	
		R. apiculata, B. gymnorrhiza	8.7	
	Micronesia	*Mixed species*	4.2	
	Vietnam	*R. apiculata*	4.9	
		R. apiculata	19.0	
		K. candel	5.3	
		K. candel	13.4	
	Thailand	*R. apiculata*	15.7	
		R. apiculata	10.6	
	China	K. candel	24.4	
	Dominican Republic	Mixed species (*R. mangle, A. germinans, L. racemosa*)	19.7	
	Guadeloupe	Mixed species (*R. mangle, A. germinans, L. racemosa*)	21.2 (fringe), 6.2 (dwarf)	
	Hawaii	*R. mangle*	29.1	
	Sri Lanka	*R. mucronata, A. marina*	11.0	
Source: Table 2.9 in Alongi 2010				

TABLE 4.8 DEFAULT VALUES FOR ABOVE-GROUND BIOMASS AND NET GROWTH OF SEAGRASSES

Seagrass (Species)	Number of observations	Mean (max biomass, g C m^{-2})	Std Dev (max biomass, g C m^{-2})
Amphibolis antarctica	1	660	
Cymodocea nodosa	14	81	81
Cymodocea rotundata	3	10	6
Cymodocea serrulata	4	31	17
Enhalus acoroides	6	92	69
Halodule uninervis	6	115	242
Halodule wrightii	17	49	26
Halophila ovalis	7	15	7
Heterozostera tasmanica	5	103	47
Phyllospadix	1	1	
Posidonia oceanica	8	213	132
Ruppia maritima	3	56	19
Syringodium filiforme	2	45	30
Syringodium isoetifolium	3	22	16
Thalassia hemprichii	3	73	57
Thalassia testudinum	12	342	750
Thalassodendron ciliatum	2	53	7
Zostera capricorni	1	96	
Zostera marina	43	139	140
Zostera noltii	5	36	26

Source: Duarte and Chiscano 1999

4.2.3 Change in Dead Organic Matter Carbon Stocks

The dead organic matter pool in coastal wetlands includes coarse woody debris, dead roots as well as fine litter materials, and varies depending on tidal inundation and frequency, as well as soil oxidation and vegetation cover. Fine litter can be exported with tidal activity while senesced woody biomass is buried or decomposes *in-situ* to influence soil organic C stocks. Due to the extent of tidal action, accumulation of fine litter is difficult to estimate because exports can be large. Turnover of course woody debris and woody material is also difficult to estimate because of slow decay rates. Partitioning these materials from peat soils also poses estimation difficulties. However, as coverage in this chapter is meant to be applicable to all coastal wetland types, especially mangrove, tidal salt marsh, tidal freshwater marsh, and seagrass ecosystems, coastal wetlands covered in this chapter include both wetlands of herbaceous and woody vegetation cover and thus approaches for including changes in dead organic matter C stocks. Countries are encouraged to use Tier 2 or higher methods when changes in dead organic matter C stocks are a key category.

FORESTED COASTAL WETLANDS

For ecosystems with woody vegetation, management activities that result in vegetation disturbance or recovery will influence C emissions and removals. Following Tier 1 methodologies, changes in dead organic matter C stocks with a management activity that result in conversion of another land-use to a forested coastal wetland (e.g. restoration) or conversion of a forested coastal wetland to another land-use (e.g. clearing for aquaculture), general methodologies developed for forestland conversion should be followed. All C in DOM stocks are considered lost in the year of conversion when converting to another land-use category, management regime or disturbance event. Thus, the carbon in biomass killed during a disturbance or management event (less removal of harvested wood products (HWP), depending on how HWP is treated) is assumed to be released entirely to the atmosphere in the year of the event. When conversion to a forested coastal wetland occurs, dead organic matter stocks are considered to start at 0 with linear accumulation over a transition period. In forestland conversions, this is estimated as 40 yr. The transition period from live woody biomass to dead wood (i.e. course woody debris) is dependent on biomass turnover which would vary as a result of soil oxidation, nutrient availability, and climate domain as well as time since other types of disturbance. During transition periods, changes in dead organic matter stocks following land-use conversion require that the area subject to land-use change be tracked for the duration of the transition period on an annual time step.

When applying Tier 1, harvesting or other vegetation disturbance that results in a change from a forested coastal wetland to a non-forested coastal wetland should also be estimated Regardless of whether the vegetation disturbance results in another coastal wetland type.

The methods for estimating changes in forested coastal wetlands follow generic equations for dead organic matter stock changes as provided in Volume 4, Chapter 2 of the 2006 GL. both changes in dead wood and litter should be reported and summed to obtain changes in total dead organic matter.

In Tier 1 estimations, a land-use conversion or management activity that results in a disturbance to vegetation follows the assumption that all C from biomass loss results in a CO_2 emission to the atmosphere. Conversely, for a change that results in a conversion to forested coastal wetland from non-forested coastal wetland, the Carbon stock in dead wood and litter is assumed to increase linearly from 0. At the Tier 1 level of estimation, litter and dead wood exports (due to tidal activity) are considered to be unchanged before and after the conversion or activity. Countries using a Tier 1 approach can use the default dead wood and litter stocks in Table 4.8. The difference between litter carbon stocks were not significantly different for tropical and subtropical mangrove sites and they were thus aggregated. Countries with dead organic matter stocks as a component of key category land use or management activities should use Tier 2 or higher methods and country estimates to report C emissions and removals. In these cases, exports of dead wood and litter should also be estimated and reported.

TABLE 4.9 TIER 1 DEFAULT VALUES FOR DEAD WOOD AND LITTER CARBON STOCKS

Climate	Dead wood carbon stocks[E] (tonnes C ha^{-1})		Litter carbon stocks[F] (tonnes C ha^{-1})	
Tropical[A]	estuarine	13.2 (3.4)	estuarine[C]	0.037 (0.009); 0.026 - 0.053
	oceanic	28.0 (5.7)		
Subtropical[B]	estuarine	9.9 (2.1)		
NE Brazil (tropical, urbanized watershed)			Estuarine	0.192 (0.144)
Mexico (tropical, coastal lagoon)			Estuarine	0.106 (0.032)

Source: Donato et al. 2011, Robertson and Daniel 1989, Krauss et al. 2005, Twilley et al. 1986, Chen et al. 2008, Imgraben and Dittman 2008, Twilley et al 1997, Flores-Verdugoet al.1987, Silva et al 1998; [A]Tropical dead wood stocks are estimates for Indo-Pacific mangroves only; [B]Subtropical dead wood stocks are estimates for S. Florida mangroves only; [C]Litter C stocks for tropical and subtropical estuarine mangroves were aggregated; [E]standard error; [F]95% CI

NON-FORESTED COASTAL WETLANDS

For coastal ecosystems with non-woody vegetation, Tier 1 methodologies for changes in dead organic matter are followed assuming no change in the dead organic matter C stocks. This is the default assumption in the case of any land-use conversion involving lands that are not forested. Where a land use activity results in no change to the forested vegetation, Tier 1 default of 0 is similarly employed. Thus, Tier 1 value for $\Delta C_{DOM} = 0$ in these cases.

Tier 2 methodologies should employ Equation 2.17 & 2.23, Volume 4, Chap 2 of the 2006 GLs where management activities or land-use results in changes to herbaceous vegetation. For instance, after a management activity that results in a change from non-vegetated or dominant vegetation with different aboveground biomass stocks, conversion of aboveground biomass to dead organic matter as litter can be significant. Countries should use in such cases national estimates for litter stocks for non-forested coastal wetlands disaggregated by climate, vegetation type and salinity where applicable.

4.2.4 Non-CO_2 Emissions

Some coastal wetlands in intertidal settings are regularly flooded by incoming tides. In this situation, their soils, altering between anoxic and aerobic conditions, favor nitrification, denitrification and methanogenesis. The reduced and anoxic environments enable methanogens, sulfurgens and others bacteria to decompose part of the organic matter with CH_4 as a product. Nitrous oxide is produced naturally in soils through the processes of nitrification and denitrification. Nitrification is the aerobic microbial oxidation of ammonium to nitrate and denitrification is the anaerobic microbial reduction of nitrate to nitrogen gas (N_2). Nitrous oxide is a gaseous intermediate in the reaction sequence of denitrification and a by-product of nitrification that leaks from microbial cells into the soil and ultimately into the atmosphere.

Non-CO_2 emissions (i.e. CH_4 and N_2O) are covered in the following subsections for reporting of specific management activities in coastal wetlands. There are two fundamental assumptions: (1) under a given set of management conditions, soils tend toward equilibrium non-CO_2 gas emissions; and (2) coastal vegetation is not a source of these non-CO_2 gases. Gas fluxes from coastal wetland soils vary greatly in time and space. Despite great variations the assumptions of gas fluxes being in equilibrium will make estimations easier. Non-CO_2 emissions are estimated using Equation 2.6, Vol. 4, Chapter 2.

4.2.5 Choice of activity data

To account for changes in soil C stocks associated with Coastal Wetlands, countries need to have, at a minimum, estimates of wetland areas at the beginning and end of the inventory time period and classification of new land use and land use activities (management data). If land-use and management data are limited, aggregate data, such as FAO statistics on mangroves (Food and Agricultural Organization of the United Nations, 2007), can be used as a starting point, along with knowledge of country experts about the approximate distribution of land management systems. Land use classes must be stratified according to climate regions and major vegetation or soil types, which could either be based on default or country-specific classifications. This can be accomplished with overlays of land use on suitable soil and climate maps, or classifying climate zones based upon published climate norms (e.g., www.ncdc.noaa.gov/oa/wdc/index.php?name=worldweatherrecords).

All Tiers require estimates of wetland areas. The same area data should be used for biomass calculations, dead organic matter, and the soil carbon estimates. If necessary, area data used in the soils analysis can be aggregated to match the spatial scale required for lower order estimates of biomass; however, at higher Tiers, stratification should take account of major soil types. Area data should be obtained using the methods described in Chapter 3 of the 2006 Guidelines. Cross-checks should be made to ensure complete and consistent representation of annually converted lands in order to avoid possible omissions or double counting. If possible, data should be disaggregated according to the general climatic categories and type of ecosystem that results from the rewetting of previously drained coastal wetlands. Tier 3 inventories will require more comprehensive information on the establishment of the coastal ecosystems as a result of rewetting and inundation by seawater, with refined soil classes, climates, and spatial and temporal resolution. All changes having occurred over the number of years selected as the transition period should be included with transitions older than the transition period (to be defined on a country specific basis) reported as a subdivision of the category of land use that results after rewetting of previously drained coastal wetlands. Higher Tiers require greater detail and the areas of land that have significant change in biomass and are under-going conversion and they should be identified separately. This implies that at least partial knowledge of the land-use change matrix, and therefore, where Approaches 1 and 2 from Chapter 3 of the 2006 Guidelines are used to estimate land area are being used, supplementary surveys may be needed to identify the relevant area of land being converted. As pointed out in Chapter 3 of the 2006 Guidelines, where surveys are being set up, it will often be more accurate to seek to establish directly areas undergoing conversion than to estimate these from the differences in total land areas under particular uses at different times.

TIER 1

Estimates of wetland areas disaggregated by activity type. The methodology assumes that area estimates are based on a one-year time frame, after they are transferred to the coastal ecosystem. If area estimates are assessed over longer time frames, they should be converted to average annual areas to match the carbon stock values used. If countries do not have these data, partial samples may be extrapolated to the entire land base or historic estimates of conversions may be extrapolated over time based on the judgement of country experts. Tier 1 approaches may use average annual rates of conversion and estimated areas in place of direct estimates. For mangroves, Tier 1 uses data of area of mangroves, which can be obtained through national statistics, conservation agencies and survey and mapping agencies. If no country data are available, aggregate information can be obtained from international data. The inventory also requires data on wood removals, including fuelwood removals and biomass losses due to disturbances, in order to calculate biomass stock changes and carbon pool transfers.

TIER 2

It is *good practice* to use actual area estimates for all possible transitions. Complete reporting can be accomplished either through analysis of periodic remotely sensed images of land-use and land-cover patterns, and/or periodic ground-based sampling of land-use patterns, or hybrid inventory systems. For mangroves, Tier 2 uses country-defined national data sets, according to different mangrove types, climate, management systems, and regions, with a resolution sufficient to ensure appropriate representation of individually relevant land areas.

TIER 3

Activity data used in Tier 3 calculations should provide a full accounting of all possible transitions disaggregated to account for different conditions within a country. Disaggregation can occur along political boundaries (county, province, etc.), biome area, climate zone, or on a combination of these parameters. In many cases countries may have information on multi-year trends in land conversion (from periodic sample based or remotely sensed inventories of land use and land cover). For mangroves, Tier 3 uses country-specific data on managed mangroves from different sources, notably mangrove national inventories, registers of land-use changes, or remote sensing. These data should be disaggregated along climate, soil, and vegetation types.

4.2.6 Uncertainty assessment

SOIL

Three broad sources of uncertainty exist in soil C inventories: 1) uncertainties in land-use and management activity and environmental data; 2) uncertainties in reference soil C stocks if using a Tier 1 or 2 approach (mineral soils only); and 3) uncertainties in the stock change/emission factors for Tier 1 or 2 approaches, model structure/parameter error for Tier 3 model-based approaches, or measurement error/sampling variability associated with Tier 3 measurement-based inventories. In general, precision of an inventory is increased and confidence ranges are smaller with more sampling to estimate values for the three broad categories, while reducing bias (i.e., improve accuracy) is more likely to occur through the development of a higher Tier inventory that incorporates country-specific information.

For Tier 1, uncertainty estimates are provided with the reference C stocks in the first footnote in Table 4.3 and in respective sections for emission and stock change factors. Uncertainties in land-use and management data will need to be addressed by the inventory compiler, and then combined with uncertainties for the default factors and reference C stocks using an appropriate method, such as simple error propagation equations. If using aggregate land-use area statistics for activity data (e.g., FAO data), the inventory agency may have to apply a default level of uncertainty for the land area estimates (+50%). However, it is *good practice* for the inventory compiler to derive uncertainties from country-specific activity data instead of using a default level.

Default reference C stocks and stock change factors for mineral and organic soils and emission factors for organic soils can have inherently high uncertainties, particularly bias, when applied to specific countries. Defaults represent globally averaged values of land-use and management impacts or reference C stocks that may vary from region-specific values (Powers et al. 2004; Ogle et al. 2006). Bias can be reduced by deriving country-specific factors using a Tier 2 method or by developing a Tier 3 country-specific estimation system. The underlying basis for higher Tier approaches will be experiments in the country or neighbouring regions that address the effect of land use and management on soil C. In addition, it is *good practice* to further minimize bias by accounting for significant within-country differences in land-use and management impacts, such as variation among climate regions and/or soil types, even at the expense of reduced precision in the factor estimates (Ogle et al. 2006).

Bias is considered more problematic for reporting stock changes because it is not necessarily captured in the uncertainty range (i.e., the true stock change may be outside of the reported uncertainty range if there is significant bias in the factors).

BIOMASS

As biomass methods follow that of the 2006 GL, uncertainties described there also pertain to biomass estimation methods and default data described in this supplement chapter on Coastal Wetlands. See Chapter 7, this supplement for further detail.

DEAD ORGANIC MATTER

In general, the magnitude of uncertainty in dead organic matter pools is larger than the uncertainty in biomass estimates because much less data are typically available for DOM pools compared to biomass pools. Uncertainties in area estimates made using the approaches suggested in Chapter 3, Volume 4 are indicated in Table 3.7 and uncertainties in assessing dead organic matter carbon stock changes may be several times larger than the uncertainty of biomass stock change estimates using default coefficients.

Although relatively few estimates of uncertainty, in changes in carbon stock in DOM pools, are available in the literature or elsewhere, several sources of uncertainty can be identified for the estimates of changes in carbon stock in dead organic matter pools. First, the assumption that carbon stocks in DOM are zero in non-forest land is not always justified. Underestimating the true initial DOM stock size will lead to overestimates of the true accumulation rates. Second, the default values for litter and dead wood carbon stock sizes are likely to be biased by being based upon estimates from land that was Forest Land for a long period of time. Thus the stock sizes at the end of the transition period may be overestimated, again, leading to overestimates of the accumulation rates. Third, the default transition period may be too long for litter carbon stocks, leading to underestimates of the true accumulation rates. For the dead wood pool, however, the current default assumption of a 40-year transition period is likely to be too short. Thus, the rate of carbon accumulation in the dead wood pool may be overestimated. Finally, litter and dead wood that may have otherwise accumulated can be exported through tidal advection thus also resulting in an overestimation of the carbon accumulation rate in the dead organic matter pool.

NON-CO_2 EMISSIONS

As non-CO2 emission methods follow that of the 2006 GL, uncertainties described there also pertain to non-CO_2 estimation methods and default data described in this supplement chapter on Coastal Wetlands. See Chapter 7, this supplement for further detail.

4.3 MANAGEMENT CHANGES IN COASTAL WETLANDS

This section describes various human activities that impact of coastal wetlands either through disruption of natural drainage and / or modification to vegetation or soils.

4.3.1 Description of Activities

Aquaculture (shrimp ponds, fish ponds, fish cages). Loss of marshes mangroves and seagrass meadows has been associated with a range of aquaculture practices including bivalve and fish farming and shrimp ponds. Commercial bivalve farming can either simply augment the natural density of bivalves in seagrass meadows or, be associated with the creation of structures fixed directly to, or above the sediment that house the bivalves (Dumbauld et al. 2009). Bivalve farming may be locally important, for example 90% of the clams consumed in Portugal are produced in one location, the Ria Formosa (Guimarães et al. 2012). Clam farming, as practiced here, takes place in intertidal seagrass meadows where seagrasses are deliberately removed by ploughing. Less intensive culture, where bivalve are left to grow naturally in the seagrass bed can be more sustainable, although competition between cultured bivalves and eelgrass occurs. Physical disturbance is obviously an issue with on-ground shellfish culture (Dumbauld et al. 2009).

Shrimp and fish ponds are constructed by clearing ground, leveling and excavation of surface soils to build containing berms within which waters are held. Carbon from cleared biomass and excavated soils is lost from storage (Lovelock et al., 2011). Shrimp and fish ponds on organic coastal soils have often resulted in pollution through the buildup of acid sulfate soil conditions and release of acidic water.

Fish farming activities are frequently reported to deteriorate their immediate surrounding environment: uneaten food and fish faeces can result in nutrient over-enrichment of the water column and organic enrichment of the sediments beneath and adjacent to fish farms. Nutrient over-enrichment of seagrass meadows has led to water and sediment quality deterioration, decreased light, algal overgrowth, seagrass shading and eventually seagrass die-off (Apostolaki et al. 2010; Apostolaki et al. 2012). Human activities, such as construction of shrimp ponds, have also been shown to have a deleterious effect on seagrass distribution (Yang and Yang 2009).

Other activities which will result in similar partial drainage of wetland soils and carbon loss are excavation and piling of soils. Excavation of channels (generally <1 m) for mosquito control and canals (tens of meters wide) for boat access results in drainage of the excavated soil (if piled in embankments) and an increase in depth to water table in wetland areas adjacent to the channels or canals.

Salt production. The Ramsar convention recognizes "salt exploitation sites" or solar salterns as a type of wetland. These are sites where salt is produced along marine coasts by evaporating tidal water, or in inland regions by evaporating water from brine springs. If solar salterns have displaced tidal marshes or mangroves it must be assumed that these were at salinities >18 ppt. Many solar salterns have long histories, on the order of centuries (Thiery and Puente 2002), and some are undergoing restoration. However, there is a report of relatively recent establishment of a salt production system displacing mangroves on the coast of Java (Sukardjo 1993). Establishment of the production system required clearing vegetation, levelling, diking and constructing channels over a 496 ha area. Some of the production sites in both in San Francisco Bay and on the Camargue have been abandoned and restoration activities are underway. Emission factors will vary for each stage: (1) construction,(2) maintenance, and (3) restoration.

Extraction (Dredging, Ports, Harbors, Beach Nourishment). Dredging excavates soil from the seafloor causing adverse effects on seagrass meadows and tidal wetlands that often lead to their loss, either directly by physical removal and/or burial, or indirectly through lack of light due to high turbidity (Erftemeijer and Robin Lewis III 2006). Although the purpose for dredging is varied, the main documented removal of seagrass has been associated with the deepening and maintainance of navigation channels and harbor entrances (Larkum and West 1990; Da Silva et al. 2004) and sand and gravel excavation for construction and land reclamation (Zainal et al. 1993). Other activities that cause direct loss of seagrass due to their excavation and removal are mining for cement production (Penn 1981), for the improvement of bathing for tourists at beach hotels (Daby 2003) and dredge-fill activities during the deployment of pipes and cables for gas, water transport and communication (Badalamenti et al. 2011). Enhanced sediment supply from construction, beach stabilization, dredging and excess siltation from changes in land catchments may enhance suspended sediment loads. Smothering, due to the settling of sediment on seagrass leaves and burial, at higher sedimentation rates can result in a major die back (Cyrus et al. 2008) and/or seagrass loss, with the capacity of seagrass species to withstand sediment burial being species specific and size-dependent (Cabaço et al. 2008). The re-suspended sediment also reduces light availability (Onuf 1994) and the capacity of seagrass species to survive below its minimum light requirement is again species specific depending, in this case, on the amount of below-ground biomass (Erftemeijer and Robin Lewis III 2006).

Nutrient enrichment. The production of greenhouse gases from soils are mainly due to microbiological processes controlled by several biological, chemical and physical factors in soil, especially the nutrients (Allen et al. 2007; Huang et al. 2009; Chen et al. 2010). Therefore, the changes in soil and water borne nutrients affect the microbiological

Anthropogenic N, as nitrate, is the dominant input from rivers and groundwater to many coastal ecosystems (Moseman-Valtierra et al. 2011). There have also been many examples showing that estuaries directly or indirectly (through organic nitrogen mineralization) receive a large amount of ammonium (Dai et al. 2008).With nutrient enrichment, low oxygen availability due to rapid consumption of oxygen by aerobic microorganisms promotes anaerobic microbial processes, including those that lead to emissions of N_2O and CH_4. Strauss and Lamberti (2000) found that the addition of organic carbon increased microbial respiration rates, but decreased nitrification rates in a hydric ecosystem. Increased anthropogenic nutrient inputs are known to increase these microbial processes in coastal water and soil, and hence enhance the emissions of greenhouse gases from wetlands to the atmosphere (Purvaja and Ramesh 2001; Kreuzwieser et al. 2003).

Decline of seagrass populations has been observed in many estuarine embayments, and the declines are often associated with anthropogenic nutrient loading (Waycott et al. 2009). Increased availability of nutrients in enriched embayments may lead to blooms of macroalgae, phytoplankton, and epiphytes, all of which shade seagrass, reducing the light available for photosynthesis and decreasing seagrass productivity. The N load per estuary that relates to complete loss of seagrass has been predicted to be at loading values between 100 and 175 kgN ha^{-1} yr^{-1} (Steward and Green 2007; Latimer and Rigo 2010).

Nutrient management. Nutrient availability is known to mediate plant community structure in coastal wetlands. Numerous studies concerning plant production suggest that productivity may be limited by availability of nitrogen (Loveland and Ungar 1983). Many fertilization experiments have been conducted in tidal salt marshes as the salt marshes are generally nitrogen limited (Pennings et al. 2002). Fertilization of salt marsh vegetation with nitrogen leads to an increase in plant height, photosynthetic rates, biomass production (Leendertse et al. 1997; Pennings et al. 2002). The high primary productivity in tidal salt marshes is known to translate into exceptionally high rates of carbon sequestration (Chmura et al. 2003).

Up to now, tentative studies have demonstrated that nitrogen fertilization in salt marsh results in increased N_2O emission, based on simulative/control experiments, with inorganic nitrogen as fertilizer. However, this research shows inconsistent effects of nitrogen fertilization on CO_2 or CH_4 emission. Moseman-Valtierra et al. (2011) found that short-term nitrogen additions can shift a coastal wetland from a sink to a source of N_2O, but the changes in CO_2 and CH_4 emission are not significant. Slightly higher CH_4 flux was recorded in *Spartina alterniflora* tidal salt marsh with N fertilizer (Ding et al. 2010), but there was no significant effect of N fertilization on CH_4 emission in the *S. salsa* wetland (Zhang et al. 2010); soil CH_4 and CO_2 production were inhibited by nitrogen (nitrate) addition (Wang et al. 2010). Changes in CO_2 and CH_4 emission seem much less intensive than N_2O due to N fertilization.

Hydrological/sediment diversion. Tidal wetlands in many coastal settings are impacted by upstream diversions of water and sediment. Declines in sediment supply reduce the capacity to maintain elevations against sea level rise, or in the case of deltas as the coastline subsides and conversion of vegetated wetlands to open water results. Reduction in water supply in arid areas may result in hyper salinization and death of vegetated coastal wetlands.

The capacity of tidal wetlands to maintain their elevation against sea level rise and natural coastal subsidence (as found for example in large coastal deltas) is supported by the supply of mineral sediments, derived from nearby rivers, cliffs or seabeds. Reduction of sediment delivery brought about by construction of river dams or coastal structures can result in sediment starvation and erosion of coastal wetland and release of stored carbon.

Modification to hydrology may result in a response of natural wetlands. For example, diversion of freshwater supply to coastal mangroves has been linked to increased salinization of wetland soils leading to the death of mangrove vegetation (Baumann et al. 1984; McKee and Mendelssohn 1988).

Fire management. Fire is used in tidal wetlands as a tool for remediation of oil spills (Baustian et al. 2010) and to encourage growth of marsh species favorable to target wildlife (Mitchell et al. 2006; Owens et al. 2007; Leonard et al. 2010). A common practice is to set fires during periods when the marsh is saturated to avoid burning of soil organic matter (Gabry and Afton, 2001).

Harvesting (**Marine resources (fish/shellfish), wood, non-wood forest products.**

1.Non-wood products. Physical disturbance is obviously an issue with harvesting marine resources such as bivalves. On-ground shellfish culture and harvest methods can result in trampling and disturbance from boat wakes and propeller scars in shallow waters (Dumbauld et al. 2009). Intertidal seagrass beds can be particularly impacted by bait collection, through digging and trenching as well as pumping of sediment for prawns (Pillay et al. 2010) and associated trampling. These activities have been implicated in the loss of 40% of seagrass in Langebaan, part of the Western Cape, which was once one of the major areas of seagrass cover in South Africa (Barnes and Ellwood 2011).

Harvest of aboveground biomass in tidal marshes dominated by herbaceous vegetation is assumed to have negligible impacts on carbon stocks due to relatively rapid turnover rates. However, where tidal wetlands are dominated by woody vegetation (e.g., shrubs in marshes of mediterranean climates) change in carbon stocks must be calculated. Although plant-mediated transport of methane has been documented in wetlands (Kreuzwieser et al. 2003; Cheng et al. 2007) available data is inadequate to develop emissions factors for methane. There is no evidence to suggest a change in nitrous oxide emissions with harvests.

2.Wood products. Mangrove harvesting is a ubiquitous activity in all areas of the tropics. Mangrove wood is harvested locally for fuelwood, charcoal and construction (Ellsion and Farnsworth 1996, Walters et al 2008) Bark removal and selective harvesting of individual trees also occurs, but with lower impact on biomass stocks .

Other Activities

1.Recreation (Boating and associated activities). Associated with increasing coastal populations is an increased threat to seagrass beds from recreational boaters. Propeller scars are formed when boat propellers strike bottom in shallow seagrass beds and destroy leaves, roots, and rhizomes. The disturbance extends beyond the formation of the physical scar because exposed sediments are more readily suspended by erosional forces, and increased turbidity limits growth of the light sensitive seagrasses surrounding the scar (Martin et al. 2008). One of the most prevalent and destructive effects of recreational boating is the destabilization of the seabed due to anchoring particularly the degradation of seagrass habitats, which are particularly sensitive to structural damage from dragging anchors. In some *Thalassia testudinum*-dominated systems, vessel damage may persist for years or decades, and even small scars may leave seagrass habitat susceptible to severe erosion by wind and wave-driven currents and storms (Hammerstrom et al. 2007).

2.Grazing (fauna and soil). Cattle, horses, goats, buffalo, and sheep all have been deliberately pastured on undrained tidal wetlandsMesleard et al. 1991; Sukardjo 1993; Burnside et al. 2007; Yu and Chmura 2010; Olson et al. 2011). Most studies of grazed marsh have focused on plant diversity and very few have reported data that can be used to assess carbon stocks or potential greenhouse gas emissions associated with these activities from tidal wetlands soils.

4.3.2 Specific Methodological Approach

Management activities described in this section can affect one or many carbon pools as well as non-CO_2 emissions. For instance, aquaculture can effect changes in soil, biomass and DOM pools and in non-CO_2 emissions. However, with no changes in water table or frequency of inundation, the tier 1 assumption is that there are only affects to non-CO_2 emissions for nutrient enrichment/management. These effects are summarized in Table 4.10. When reporting on coastal wetland management changes, activity data for different management changes are required where changes in coastal wetlands are a key category.

TABLE 4.10. TIER 1 METHODOLOGICAL CONSIDERATIONS FOR MANAGEMENT CHANGES IN COASTAL WETLANDS

Activity	CO_2			Non-CO_2
	Soil	Biomass	DOM	
Aquaculture (ΔC_{AQ})	X	X	X	X
Salt Production (ΔC_{SP})	X	X	X	
Extraction (ΔC_{EXT})	X	X	X	
Nutrient Enrichment (ΔC_{NE})*				X
Nutrient Management (ΔC_{NM})*				X
Hydrologic/Sediment Diversion (ΔC_{HSD})				
Harvesting (ΔC_{H})		X	X	
Other Activities				

*Inventory compiler is directed to section 4.3.6 on non-CO_2 emissions. C pool changes are assumed to be zero at Tier 1 level for nutrient management and nutrient enrichment.

Thus, in order to insure full coverage of C pools for each activity, equation 4.6 should be employed.

EQUATION 4.6

ANNUAL CARBON STOCK CHANGES FOR MANAGEMENT CHANGES IN COASTAL WETLANDS AS A SUM OF POOL CHANGES ASSOCIATED WITH LAND USE ACTIVITIES

$$\Delta C_M = \Delta C_{AQ} + \Delta C_{SP} + \Delta C_{EXT} + \Delta C_{NE} + \Delta C_{NM} + \Delta C_{HSD} + \Delta C_H$$

Where:

ΔC_M = Management Changes in Coastal Wetlands

ΔC_{AQ} = carbon stock changes for aquaculture
ΔC_{SP} = carbon stock changes for salt production
ΔC_{EXT} = carbon stock changes for extraction
ΔC_{NE} = carbon stock changes for nutrient enrichment (assumed to be zero for C pools)
ΔC_{NM} = carbon stock changes for nutrient management (assumed to be zero for C pools)
ΔC_{HSD} = carbon stock changes for hydrologic/sediment diversion
ΔC_H = carbon stock changes for harvesting

In the cases of aquaculture, salt production and extraction, C pool changes for soil, biomass and dead organic matter must be estimated and summed at Tier 1 level. Additionally, in the case of aquaculture, the inventory compiler is referred to section 4.3.6 for guidance on estimating non-CO_2 emissions associated with nutrient enrichment. Where changes in CO_2 emissions or removals do not occur in soil, biomass or dead organic matter C pools (i.e. nutrient enrichment and nutrient management), the Tier 1 assumption is that the value for the parameter is zero. However, the inventory compiler is directed to section 4.3.6 for methodological guidance on non-CO_2 emissions associated with changes in CH_4 and N_2O. While hydrologic and sediment diversions are excepted to have important changes in soil and biomass C stocks, the literature is not yet developed enough to provide default emission factors. Finally, with regard to harvesting, changes in the soil C or DOM pools are assumed to be zero at the Tier 1 level of estimation, and only changes in biomass are reported.

4.3.3 Soil Carbon

This section deals with anthropogenic impacts to wetland soil organic C stocks, primarily C emissions from soil and soil C storage, by activities affecting soil drainage either through modification of the water table, mechanical disturbance to soils, and disruption to mineral sediment supply. General information and guidance for estimating changes in soil C stocks are provided in Section 4.2.

Total emissions from soil C stocks for *Management Changes in Coastal Wetlands* are estimated using Equation 4.3 to account for soil C losses or possible soil C gains. Mangrove or salt marsh should be further disaggregated by soil type for Tier 2 estimation.

4.3.3.1 CHOICE OF METHOD

For *Management Changes in Coastal Wetlands*, soils are maintained wet (though may be temporarily dried during the land-use transition phase) and changes in C storage result from land-use activity. Mechanical disturbance associated with pond construction for aquaculture are assumed to release carbon from the top 1 meter of the soil profile. As the water table is maintained at or above the graded soil surface further emissions of soil carbon are assumed to be zero.

For wetlands that are not revegetated (e.g. aquacultural ponds, marshes denuded of vegetation) further soil C input is assumed to be zero. For aquacultural ponds labile carbon will be supplied to accumulating muds but this is assumed to be released through pond management practices or through respiration.

4.3.3.2 CHOICE OF EMISSION/REMOVAL FACTORS

MINERAL/ORGANIC SOILS

Tier 1

Changes in stocks of organic carbon are estimated following Equation 4.3 to estimated C losses. The methodology stratifies managed organic soils by climate region and ecosystem type and assigns an annual emission rate. Land areas are multiplied by the emission factor and then summed to derive annual C emissions.

Tier 2

The Tier 2 approach also applies Equation 4.3, but country-specific information is incorporated to better specify emission factors, climate regions, and/or the land management classification system.

Tier 3

Tier 3 approaches for organic carbon use dynamic models and/or measurement networks. A variety of models designed to simulate soil carbon dynamics exist for terrestrial soils (for example, see review by Poeplau et al. 2011), but none have been applied to coastal wetlands. Key criteria in developing an appropriate model for these purposes include its capability of representing belowground production, soil organic matter turnover, and vertical soil accumulation rates that are compatible with the availability of country-wide input data, but more research is needed; and that the model sufficiently represents stock changes based on comparisons with experimental data.

TABLE 4.11. EMISSION FACTORS FOR COASTAL WETLAND SOILS CHANGED THROUGH AQUACULTURE, SALT PRODUCTION OR EXTRACTION

Climate Zone	Ecosystem	Mean (Mg C ha^{-1} yr^{-1})	Range (Mg C ha^{-1} yr^{-1})
Boreal and Temperate	Tidal Marshes	8.75	1.18-47.50
Tropical and Sub tropical	Mangroves	12.13	3.88-28.75

Sources: Chmura et al . 2003, Silfleet et al., 2011, Donato et al., 2011; Lovelock et al. 2011; Fourqurean et al., 2012.

4.3.4 Biomass

This section provides guidance for estimating carbon stock changes in biomass for *Management Changes in Coastal Wetlands* including changing cover in vegetation, effects of nutrient additions and the effects of management. The concepts underlying carbon stock changes in biomass of *Management Changes in Coastal Wetlands* are tied to management practices. Larger amounts can accumulate in the woody component of vegetation in mangroves, and in root biomass and in soils of all three coastal wetland ecosystems. In mangroves gains include total (aboveground and belowground) biomass growth, but additional losses are incurred due to fuelwood removal/harvest and gathering. When such losses occur, belowground biomass is also reduced and transformed to dead organic matter (DOM).

The decision tree in Figure 4.1 provides guidance on the choice of Tiers for reporting changes in biomass C stocks. Because data on below-ground biomass are often lacking for specific ecosystems, a simplified approach based upon below-ground to above-ground biomass ratios is used. With this approach, estimates of below-ground biomass are closely tied to estimates of above-ground biomass. Hence, for simplicity, above-ground and belowground biomasses are combined for estimation and reporting.

Although the methods for estimating biomass changes are conceptually similar among the ecosystems that constitute *Management Changes in Coastal Wetlands*, there are some aspects that are unique to each of the ecosystems. In particular, mangrove stands allometric relationships differ from those used in Forest Land and consideration must also be taken of differences between mangrove species growing in different geographic regions.

4.3.4.1 CHOICE OF METHOD

General considerations relevant to, and the generic equations that can be used for, the estimation of biomass changes for are provided in Chapters 2 and 4 of Volume 4. These generic equations can be used but more specific consideration needs to be taken when estimating biomass stocks in association with *Management Changes in Coastal Wetlands.* Depending on the Tier used and data availability, coastal wetlands can be disaggregated by ecosystem, climate, salinity level and management system.

TIER 1

A Tier 1 approach generally assumes no change in biomass in *Management Changes in Coastal Wetlands* except those ecosystems where woody vegetation is prevalent. In any coastal wetland ecosystem, where there is no change in either type or intensity of management, biomass will be in an approximate steady-state (i.e. carbon accumulation through plant/tree growth is roughly balanced by losses through herbivory and decomposition). However, where management changes are occurring over time (e.g., through introduction of aquaculture, physical disturbance, soil removal, nutrient enrichment or other practices) carbon stock changes can be significant. If it is reasonable to assume that any of the coastal wetland ecosystems are not a key source, a country may apply the Tier 1 assumption of no change in biomass. However, if information is available to develop reliable estimates of rates of change in biomass in *Management Changes in Coastal Wetlands*, a country may use a higher Tier, even if *Management Changes in Coastal Wetlands* is not a key source, particularly if management changes are likely, in which case default data in Tables 4.5-4.8 can be applied.

TIER 2

Tier 2 allows for estimation of changes in biomass when *Management Changes in Coastal Wetlands* is a key category. Two methods are suggested for estimating the carbon stock change in biomass. ***Gain-Loss and Stock-Difference Methods.***

TIER 3

Tier 3 methods are used where countries have country-specific emission factors, and substantial national data. Country-defined methodology may be based on detailed inventories of permanent sample plots for their Coastal ecosystems, and/or models. For Tier 3, countries should develop their own methodologies and parameters for estimating changes in biomass. These methodologies may be derived from equations specified above, or may be based on other approaches. The method used needs to be clearly documented.

4.3.4.2 CHOICE OF EMISSION/REMOVAL FACTORS

Emission and removal factors that are required to estimate the changes in biomass resulting from management include biomass growth rate, loss of biomass, and expansion factor for below-ground biomass.

TIER 1

Default factors in Table 4.4-4.8 can be used to estimate change in woody biomass at Tier 1 level. If management changes are significant, but management changes are not a key category, then non woody biomass can be estimated using default data in Tables 4.4-4.8

TIER 2

It is *good practice* to use country-specific data on biomass C stocks for different mangrove, seagrass and salt marsh species that constitute a particular biogeographic region. Country-specific values for *net* biomass increment as well as losses from harvested mangroves and decomposition rates, in the case of the Gain-Loss Method, or the net change in biomass stocks, in the case of the Stock-Difference Method can be derived from country-specific data, taking into account the ecosystem type. Estimating below-ground biomass is also an important component of biomass surveys of coastal wetland ecosystems and expansion factors are provided to estimate belowground biomass from aboveground biomass. Table 4.4 provides default root-to-shoot ratios (all vegetation) for mangrove, seagrass and salt marsh ecosystems in the major climate zones of the world (IPCC climate zones are the same as those reported in Annex 3A.5, 2006 GL as well as the classification of climate zones more appropriate for seagrass species). These values can be used as defaults when countries do not have more specific information to develop country-specific ratios.

TIER 3

Tier 3 approaches consist of using a combination of dynamic models and inventory measurements of biomass stock changes. This approach does not employ simple stock changes or emission factors *per se*. Estimates of emissions/removals using model-based approaches are derived from the interaction of multiple equations that estimate the net change of biomass stocks within the models. Models, jointly with periodic sampling-based stock estimates could be applied to estimate stock changes or inputs and outputs as in Tier 2 to make spatial extrapolations for coastal wetland areas.

4.3.5 Dead Organic Matter

Changes in dead organic matter resulting from land-use conversion or management activities that occur in forested coastal wetlands are estimated at the Tier 1 level because they represent potentially large C emissions to or removals from the atmosphere. Dead organic matter pools under conditions of low soil oxidation-reduction potential under saturated conditions, especially at high latitudes, can be large. Conversions that result in previously forested coastal wetlands shifting to non-forested wetlands can have large implications for C estimates. In non-forested coastal wetlands, the default assumption is that no changes in stocks occur at the Tier 1 level of estimation. For key categories or extensive management activities occurring in either forested or non-forested coastal wetlands, Tier 2 and higher estimation methods should be used and these values reported. In these cases, stock changes should be disaggregated relative to climate, vegetation type and salinity where applicable.

4.3.5.1 CHOICE OF METHOD

TIERS 1, 2 AND 3

The method for estimating changes in dead organic matter stocks are presented in Equations 2.18 and 2.19 (Volume 4, Chapter 2). The method estimates the change in dead organic matter stocks applying the Stock-Difference Method where the change in dead organic matter stocks are estimated at two points in time relative to the period before and after the land-use conversion or management activity. This method can be applied for forested coastal wetlands at a Tier 1 level or at a Tier 2 level using country estimates where changes occur in either forested or non-forested coastal wetlands. Tier 1 default stocks for forested coastal wetlands are provided in Table 4.9. The default for changes in dead organic matter stocks at the Tier 1 level of estimation is 0 for non-forested coastal wetlands. Tier 2 level of estimation should be used when extensive management activities lead to key category land-use conversions. The *Gain-Loss Method* and better estimates of transition times may also be more appropriate for Tier 2 or higher level of estimation to reduce uncertainties. Estimates of dead wood and litter exports due to tidal advection should also be considered.

4.3.5.2 CHOICE OF EMISSION/REMOVAL FACTORS

TIER 1

Countries using a Tier 1 method require data on the default dead wood and litter carbon stocks as defined in Table 4.8. The Tier 1 assumption is that carbon stocks in litter and dead wood pools in all non-forested coastal wetlands and conversion among non-forest land-use categories are zero. The use of equation 2.18 is recommended for Tier 1 level estimation. For lands converted to forested coastal wetland, the carbon stocks in dead wood and litter pools are assumed to increase linearly over the transition period T (default is 20 years for both litter and dead wood C stocks). Thus, the annual rate of increase is estimated as the ratio between the difference in carbon stocks in the DOM pools in the non-forest and forest categories, and the numbers of years in the transition period T. Conversions from forested coastal wetland to non-forested coastal wetland are assumed to be emitted in the year of conversion.

TIERS 2 AND 3

The higher Tier methods described above (Equation 4.18) and in Chapter 2 will allow for more robust estimates when applied to national data. Additional requirements may arise if the assumption that carbon stocks in dead wood and litter pools of non-forest land-use categories are zero cannot be justified, such as in some agro-forestry systems or other circumstances where intensive management activities have occurred.

4.3.6 Non-CO_2 emissions

This section provides methods for estimating the emissions of two non-CO_2 gases from coastal wetlands under different management schemes, but with altered Hydrology. Non-CO_2 gas emissions from seagrass are not taken into account due to limited data. Management activities include: construction of canals for transportation, ditching for vector control, harvesting, grazing, and construction and maintenance of solar salterns (salt production facilities), nutrient management and nutrient enrichment. Although the wetland hydrology remains unchanged (or the characteristics of the hydrological regime that maintain its wetland status), the management activities alter the environmental factors that regulating the non-CO_2 gas productions and emissions, e.g. the substrate availability, hence leading to changes in gas emissions.

Some management activities, such as shrimp/fish culturing, nutrient enrichment and nutrient management, result in nutrient/organic enrichment in the water and soil, and affect the microbiological mechanisms responsible for CH_4 and N_2O production. But for some other activities, such as salt production, no reports are available on N_2O emissions. Despite notable levels of sulfate reduction, methanogenesis occurs in the saltern pond sediments but no significant methane oxidation has been measured (Conrad et al. 1995).

Coastal wetlands subject to intensive nutrient loading may also be sources of CH_4 emissions. However, in the absence of intensive nutrient loading only coastal wetlands with salinity of soil porewater <15 are likely to be sources of CH_4 emissions.

4.3.6.1 CHOICE OF METHOD AND EMISSION FACTOR

Emissions of non-CO_2 greenhouse gases from coastal wetlands with altered Hydrology are associated with data on specific activities within this land-use category, and are calculated as the sum of these wetlands with different management activities. A generic method for estimating the emission of non-CO_2 gas is provided in Equation 2.6, Vol. 4, Chapter 2.

TIER 1

The default method to estimate the non-CO_2 gases emissions from coastal wetlands is to multiply the area of these wetlands by the gases emission rates.

Methane

In Tier 1 method, the CH_4 emissions due to activities associated with partial drainage are not considered and the emission factors are assumed to be zero. Default emission factors for Tier 1 method are provided in Table 4.12.

TABLE 4.12 EMISSION FACTORS FOR CH_4 FROM COASTAL WETLANDS WITH ALTERED HYDROLOGY

Wetland	Default EF (tonnes CH_4-C ha^{-1} y^{-1})	EF Range (tonnes CH_4-C ha^{-1} y^{-1})	Error	Reference
N-fertilized wetland	To be completed			
Nutrient-enriched wetland	11.5	0-54		Sotomayor et al., 1994, Purvaja and Ramesh 2001, Ye and Lu, 2001, Allen et al., 2007, Chauhan et al., 2008, Krithika et al., 2008, Liikanen et al., 2009, Allen et al., 2010, Chen et al., 2010, Tong et al., 2010, Adams et al., 2011, Ramesh et al., 2011
Impounded wetlands	To be completed			

Nitrous oxide

Tier 1 method estimates N_2O emissions from coastal wetlands with altered Hydrology based on the assumption that the N_2O gas was negligible from those wetlands without nutrient loading. Default N_2O emission factors for Tier 1 method are provided in Table 4.13.

TABLE 4.13 EMISSION FACTORS FOR N_2O EMISSION FROM COASTAL WETLANDS				
Wetland	**Default EF ($kg\ ha^{-1}\ y^{-1}$)**	**EF Range ($kg\ ha^{-1}\ yr^{-1}$)**	**Error**	**Reference**
N-fertilized wetland	49.89	0.77~200.43		Muñoz-Hincapié et al., 2002, Kreuzwieser et al., 2003, Moseman-Valtierra et al., 2011, Kenny et al., 2004, Lindau and DeLaune, 1991
Nutrient-enriched wetland	10.52	−0.35~91.85		Kreuzwieser et al., 2003, Corredor et al., 1999, Liikanen et al., 2009, Adams et al., 2011, Bauza et al., 2002, Krithika et al., 2008, Alongi et al., 2005, Chen et al., 2010, Allen et al., 2007, Wang et al., 2007
Impounded wetlands	To be completed			

TIER 2

The Tier 2 method requires country-specific emission factors that account for climate, vegetation type, and intensity of nutrient input.

For nutrient management, emissions of non-CO_2 gases from nutrient-managed wetlands could also be estimated based on the annual amount of fertilizer N (activity data) applied to wetland soils and the emission factor (transfer rate of fertilizer N to non-CO_2 gases by weight).

TIER 3

The Tier 3 method involves a comprehensive understanding and representation of the dynamics of non-CO_2 gas emission factors, taking account of parameters such as season, soil texture, fertilizer composition (for Nutrient management), tidal elevation and tidal cycle (high tide vs. low tide). Field research should be carefully carried out at representative sites for emission factors estimation. The countries are encouraged to setup own methodology or model based on the N input amount for estimation of emission factors.

4.4 DRAINAGE OF COASTAL WETLANDS

4.4.1 Description of Activities

Tidal wetlands have been diked and drained to create land for agriculture (pasture and croplands) and settlement since before the eleventh century (Gedan et al. 2009). The practice, sometimes referred to as "reclamation", continues today on many of the world’s coastlines, for example countries such as Korea (e.g., Byun et al. 2004) and China (e.g., Sun et al. 2003) where drained wetlands may be used for modern industrial complexes. On some diked coasts groundwater below reclaimed former-wetlands is pumped out to maintain the water table at an optimum level below a dry soil surface while on other coasts drainage is achieved through a system of ditches and tidal gates (opening only at low tide). A third means to deal with high water tables is to increase the surface soil elevation with fill, a practice most common when wetlands are transformed to settled lands.

Rates of organic matter decomposition in soil below the water table are lower than in aerated soils above, because anaerobic decomposition is less efficient (e.g., Reddy and Patrick 1975; Kristensen et al. 1995). Thus, reduction in the degree of soil saturation during drainage should increase rates of decomposition, resulting in loss of soil carbon stored as a result of wetland plant production (Armentano and Menges, 1986). This response will vary regionally with climate (Pozo and Colino 1992) and locally with soil salinity, soil texture, and the quantity of labile organic matter available in the drained soil (e.g., Heminga et al. 1998, Setia et al. 2011), the latter can be related to varied chemical composition tissues of plant species (e.g., Buth 1987).

4.4.2 Soil carbon

This section deals with anthropogenic impacts to wetland soil organic C stocks, primarily C emissions from soil and soil C storage, by activities affecting soil drainage through modification of the water table,. This section provides guidelines for estimating carbon stock changes in soils of coastal wetlands due to drainage which can increase rates of decomposition. Land conversions following Coastal Wetland drainage (e.g., agriculture, settlement, aquaculture), will impact soil carbon stocks and these must be considered in calculation of overall carbon stocks.

4.4.2.1 CHOICE OF METHOD

Inventories can be developed using a Tier 1, 2 or 3 approaches, with each successive Tier requiring more details and resources than the previous one. It is also possible that countries will use different tiers to prepare estimates for the separate sub-categories of soil C (i.e., soil organic C stocks changes in mineral and organic soils; and stock changes associated with soil inorganic C pools). A decision trees is provided (Figure 4.1) to assist inventory compilers with the selection of the appropriate tier for their soil C inventory.

4.4.2.2 CHOICE OF EMISSION AND STOCK CHANGE FACTORS

MINERAL/ORGANIC SOILS

Tier 1

For all soils, the estimation method is based on emission of C from soil (with mineral and organic soils aggregated) over a finite (40 yr) period following changes in management that impact soil organic C storage. After a finite transition period, one can assume a steady state change in stock assuming no change in water level that may result in a net accumulation of soil C after the transition period. The method assumes that land area data is available to disaggregate coastal wetlands by ecosystem type to estimate losses due to drainage activities.

Equation 4.3 (Chapter 2) is used to estimate change in soil organic C stocks. Note that area of exposed bedrock in wetlands and open water channels are not included. In practice, country-specific data on land management activity should be obtained and classified into appropriate land management systems, and then stratified by ecosystem type. Table 4.14 provides annual rates of C emission for soils of drained coastal wetlands.

If land area data disaggregated by ecosystem type is not available, countries can employ Equation 4.4, using stock change factor data provided in Table 4.15.

Tier 2

Equation 4.4 is used to estimate change in soil organic C stocks, disaggregated by soil type (mineral and organic) in coastal wetland soils by subtracting the C stock in the last year of an inventory time period (SOC_0) from the C stock at the beginning of the inventory time period ($SOC_{O/M,b}$) and dividing by the time dependence of the stock change factors (D). Note that area of exposed bedrock in wetlands and open water channels are not included in the soil C stock calculation (assume a stock of 0). In practice, country-specific data on land management activity should be obtained and classified into appropriate land management systems, and then stratified by IPCC climate regions and soil types. Soil organic C stocks (SOC) are estimated for each time period in the inventory using default reference carbon stocks (SOC_{ref}) and default stock change factors (F_{LU}, F_{MG}). If country-specific stock change factors are not available, countries-specific stocks can be used with stock change factors in Table 4.15.

A tier 2 method might also utilize the emission factor approach if country-specific data, disaggregated by soil type, climate, ecosystem type, and/or salinity level, are available.

Tier 3

Tier 3 approaches do not employ simple stock change factor *per se*, but rather use dynamic models and/or detailed soil C inventory measurements as the basis for estimating annual stock changes. Estimates of stock changes using model-based approaches are computed from the coupled equations that estimate the net change of soil carbon as described in section 4.2.2.2.

TABLE 4.14 ANNUAL EMISSION FACTORS (EF) FOR DRAINED SOILS FOR TIER 1 ESTIMATION

Climate Zone	**Wetland Type**	**Default ($Mg\ C\ ha^{-1}\ yr^{-1}$)**	**Range ($Mg\ C\ ha^{-1}\ yr^{-1}$)**
Boreal and Temperate	Tidal Marshes	8.75	1.18-47.50
Tropical and Subtropical	Mangrove	12.13	3.88-28.75
Review sources: Chmura et al . 2003, Silfleet et al., 2011, Donato et al., 2011; Fourqurean et al., 2012.			

TABLE 4.15 RELATIVE STOCK CHANGE FACTORS FOR DRAINED COASTAL WETLANDS FOR TIER 1 LEVEL ESTIMATION (WHERE DISAGGREGATION BY SOIL TYPE IS POSSIBLE OR TO SUPPLEMENT TIER 2 ESTIMATION)

Soil Type	Wetland Type	Default Stock Change Factor (unitless)	Reference
Organic	Tidal Marsh	0	
	Mangrove	0	Lovelock et al 2011
Mineral	Tidal Marsh	0.5	
	Mangrove	0.5	Lovelock et al 2011

Note: Stock change factors for mineral soils are assumed based on a value of 50% C stock of coastal wetland soil i.e. (0.5 * 351)/40 Mg C ha^{-1} yr^{-1} over a 40 year period where 0.5 = 50% C stock of coastal wetland soil.

4.4.3 Biomass

This section provides guidelines for estimating carbon stock changes in biomass due to the conversion of coastal wetlands, mostly mangroves and tidal salt marshes, due to drainage. During drainage of coastal wetlands, changes in the carbon stock of biomass can result from the loss of existing vegetation and replacement with vegetation that is more characteristic of the dried soils. This scenario is different than that described in Coastal wetlands with unaltered or partly modified hydrology, where the vegetation of the existing wetland remains the same but generally experiences changes in biomass due to different management activities.

Conversion of coastal wetlands during drainage often results in the transfer of carbon from one pool to another. All transfers must be accounted during the transition to a new steady state. For example, when converting mangroves via drainage, trees can be felled and a portion of the above-ground biomass is transferred to harvested wood products and/or the dead organic matter pool. In addition, a portion of the below-ground biomass is transferred to the soil organic matter pool, etc.

Estimating changes in carbon stocks in biomass for coastal wetlands, converted by drainage, can require a two-phase approach if there are different time-scales for vegetation loss from existing wetlands and replacement by new, different vegetation in the drained soils. There can be an abrupt change in biomass, if say mangrove trees are felled prior to or concurrent with drainage. This abrupt change can be treated as phase 1 and in-line with the 2006 guidelines be estimated at the year of conversion. The second phase accounts for a gradual biomass loss and gain during a transition period to a new steady state system. At some time after drainage the new vegetation that develops on the drained coastal wetland should be comparable to that in the natural setting and should then be considered under, and accounted for in this new category of land-use. The transition period following conversion by drainage will not be the same for different coastal ecosystems, Default values of 10 years are to be used for drained mangroves and tidal salt marshes respectively for remaining in the transitional category. Countries can determine more appropriate transition periods, at their discretion, that more specifically quantify the particular type of drainage and ecosystem that is affected. The values of coefficients determining the rate of emissions may depend on the transition period used. In-line with the 2006 guidelines, wetlands converted by drainage should be accounted for with Phase 1 methods in the year that they are converted, with Phase 2 methods being used for all subsequent transitional years until the new vegetation achieves steady state. At the end of the transitional period, the land area currently being accounted for now comes under consideration of the new category, which has resulted from the drainage of the particular coastal wetland. If there is no abrupt change in biomass associated with the conversion, and a gradual transition occurs between the two different vegetation types, Phase 1 accounting would not be appropriate and the whole conversion accounting can be undertaken using Phase 2 methods. Conversely there may be a rapid conversion with little or no opportunity available for subsequent re-vegetation on drained soils. This usually occurs when there is partial drainage of wetland soils, such as when excavation of water-logged soils is piled onto the existing soil surface during ditching and canalisation. If the excavation is more frequent than the time taken for new vegetation to colonize the soil, the whole accounting for biomass change can be treated with Phase 1 methods.

When the Phase 1 method is implemented, it is *good practice* to apportion transfers of carbon between pools. The immediate impacts of drained coastal wetland conversion activities on the five carbon stocks can be summarized in a “disturbance matrix”. The disturbance matrix describes the retention, transfers and releases of carbon in the pools in the original ecosystem following conversion due to drainage. A disturbance matrix defines for each pool the proportion that remains in that pool and the proportion that is transferred to other pools. A small number of transfers are possible, and are outlined in the disturbance matrix in Table 2.1 of Chapter 2 in the 2006 guidelines. If the rate of conversion of coastal wetland by drainage is more or less constant, the assumption that all carbon in theses pools was lost at the time of conversion would be a reasonable first approximation.

Where the rate of conversion of coastal wetland by drainage varies over time, it is *good practice* to account for the transfer and release of carbon among the different carbon pools and ensure that all carbon is accounted. In cases where there is an immediate and abrupt carbon stock change in biomass due to conversion of coastal wetlands, the effect of this conversion will be estimated using Equation 2.16 in Chapter 2 of the 2006 guidelines, with the default assumption that B_{After}=0. During the transition period, pools that gain or lose C often have a non-linear loss or accumulation curve that can be represented through successive transition matrices. If the true shapes of the curves are known, these curves can be applied to each cohort that is under transition during the reporting year to estimate the annual emission or removal by the specific pool. If the shape of the curve is unknown, countries may simplify and use a linear decay function to estimate pool changes.

4.4.3.1 CHOICE OF METHOD

The decision tree (Figure 1) provides guidance for selecting the appropriate tier level for the implementation of estimation procedures for biomass associated with the drainage of coastal wetlands. Estimation of changes in biomass requires an estimate of changes in above-ground vegetation and changes in below-ground biomass. Countries should use the highest tier possible given national circumstances. It is *good practice* to use a Tier 2 or Tier 3 approach if carbon emissions and removals in coastal wetlands converted by drainage is a key category and if the sub-category of biomass is considered significant, based on principles outlined in Volume 1, Chapter 4.

TIER 1

The change in biomass carbon stock on *Drainage of Coastal Wetlands* under Tier 1 should be estimated using Equation 2.15 of the 2006 guidelines. The average carbon stock change is equal to the sum of increase in carbon stock due to biomass growth following conversion, changes due to actual conversion (difference between biomass stocks before and after conversion), and decrease in carbon stocks due to losses after conversion. As a simplification for Tier 1, it is assumed that all biomass is lost immediately from the previous coastal wetland ecosystem after conversion (Equation 2.16) , even when there is no abrupt change, and residual biomass (B_{AFTER}) is thus assumed to be zero (i.e., the wetland ecosystem is cleared of all vegetation before a new category of land is established).

In this case there is no transfer of biomass from the biomass pool to the dead wood pool, for example. Default values for mangrove, seagrass and tidal salt marsh biomass prior to conversion can be found in this chapter, Tables 4.6 and 4.7).

Additionally, it is assumed that each new category of land use established after drainage of coastal wetlands achieves its steady-state biomass during the first year following conversion. Thus, for Tier 1, there are no stock changes associated with Phase 2, though the drained coastal wetlands that are converted to a new category should be retained in the conversion category for the 20 year transition period because the soil stocks will take longer to reach equilibrium. Emissions and uptakes from biomass during Phase 2 of the calculation are therefore zero. If there are significant management changes during the transition phase, countries can account for the impacts of this on C stocks in biomass using the appropriate Tier 2 methods for any category remaining in the same category. It is *good practice* to account for all ecosystems within coastal wetlands converted by drainage to another category. Thus, a separate calculation must be done for each ecosystem conversion.

TIER 2

The Tier 2 calculations differ structurally in a number of ways from Tier 1. First, Tier 2 estimates use the two phase approach described earlier. Tier 2 relies on some country-specific estimates of the biomass in initial and final land uses rather than the defaults, as in Tier 1. Area estimates for *Drainage of Coastal Wetlands* are disaggregated at higher resolution spatial scales than in Tier 1 to capture regional variations within and between the different coastal ecosystems of the country. Second, for Tier 2 countries may modify the assumption that biomass immediately following conversion is zero. This enables countries to take into account land-use transitions where some, but not all, vegetation from the original land use is removed. In addition, under Tier 2, it is possible to account for biomass accumulation following establishment of a new land use category over a several year period (rather than accounting all biomass stock change in the year of conversion) if data are available to estimate the time to full biomass establishment and the annual stock changes. Third, under Tier 2, it is *good practice* to apportion transfers of carbon between pools. Some of the land categories developed after coastal wetland drainage do not contain significant carbon in the dead wood or litter pools, but dead wood may persist for a number of years in land that is replacing mangroves or woody biomass may persist that had previously accumulated in salt marshes. If the rate of land conversion is more or less constant, the assumption that all carbon in theses pools was lost at the time of conversion would be a reasonable first approximation. Where the rate of land conversion varies over time, it is appropriate to try to account for the transfer and release of carbon from litter, dead wood, and soil carbon pools. It is therefore necessary to distinguish immediate losses due to the conversion activities from the losses that occur in the years following the land conversion.

The immediate and abrupt carbon stock change in biomass due to drainage of coastal wetlands under Tiers 2 and 3 will be estimated using Equation 2.16 in Chapter 2, where B_{AFTER} is assumed to be zero. During the transition period, pools that gain or lose C often have a non-linear loss or accumulation curve that can be represented through successive transition matrices. For Tier 2, a linear change function can be assumed. For a Tier 3 approach based upon these methods, it is *good practice* to use the true shapes of the curves. These curves are to be applied to each cohort that is under transition during the reporting year to estimate the annual change in the biomass carbon pools. For the estimation of changes in biomass carbon during the transition phase, two methods are suggested. The equations, related to the *Gain-Loss Method* and *Stock-Difference Method* are the same as those used in Chapter 2 of the 2006 guidelines. The equations have also been presented in this chapter (Eqs 4.9 and 4.10 respectively).

TIER 3

Tier 3 methods are used where countries have country-specific emission factors, and substantial national data. Country-defined methodology may be based on detailed inventories of permanent sample plots for teach coastal wetland ecosystem and/or models. For Tier 3, countries should develop their own methodologies and parameters for estimating changes in biomass. These methodologies may be derived from methods specified above, or may be based on other approaches. The method used needs to be clearly documented. Tier 3 involves inventory systems using statistically-based sampling of biomass over time and/or process models, stratified by what type of land use the coastal wetlands have been converted to, changes in vegetation type within each land use, climate, and management regime. For example, if the drained coastal wetland was converted to grassland, validated species-specific growth models that incorporate management effects such as grazing intensity, fire, liming, and fertilization, with corresponding data on management activities, could be used to estimate net changes in grassland biomass overtime. Models, together with periodic sampling-based biomass estimates, could be applied to estimate stock changes to make spatial extrapolations for areas of drained coastal wetland.

Key criteria in selecting appropriate models include the ability to represent all of the ecosystem conversions and management practices that are represented in the activity data. It is critical that the model be validated with independent observations from country-specific or region-specific field locations that are representative of the variability of local environmental change as well as climate, soil and vegetation management systems in the country.

If possible, spatially explicit area estimates should be used to facilitate complete coverage of the converted area of coastal wetland and ensure that areas are not over- or underestimated. Furthermore, spatially explicit area estimates can be related to locally relevant carbon accumulation and removal rates, and restocking and management impacts, improving the accuracy of estimates.

4.4.3.2 CHOICE OF EMISSION/REMOVAL FACTORS

TIER 1

Tier 1 methods require estimates of the biomass of the land use before conversion and after conversion. It is assumed that all biomass is cleared when preparing a coastal wetland site for drainage and alternative land use, thus, the default for biomass immediately after conversion is 0 tonne ha^{-1}. Default values for biomass before conversion can be found in this Chapter (Tables 4.6 and 4.7). For biomass following (not immediately after) conversion can be found in the appropriate chapter of the 2006 guidelines.

TIER 2

It is *good practice* to use country-specific estimates for biomass stocks and emissions/removals due to land conversion, and also include estimates of on-site and off-site losses due to burning and decay following land conversion due to drainage of coastal wetlands. These improvements can take the form of systematic studies of carbon content and emissions and removals associated with land uses and land-use conversions within the country or region and are-examination of default assumptions in light of country-specific conditions.

Region-specific or country-specific data on biomass for growth in the vegetation that characterises the drained wetland are needed for a Tier 2 approach. These can be obtained from the existing description of conversion to Grassland, Cropland and Forest Land in the 2006 guidelines. General guidance on survey and sampling techniques for biomass inventories is given in Chapter 3 in Annex 3A.3. If possible accurate knowledge of the dynamics of below-ground biomass should be obtained for accounting for carbon stockchanges when coastal wetlands are drained to provide land for other uses. Estimation of below-ground biomass in the converted land should be available in Grassland, Cropland and Forest Land Chapters in the 2006 guidelines.

Estimation of below-ground biomass in the pre-existing coastal wetland can be an important component of the calculated stock change. Field measurements are laborious and difficult and thus expansion factors to estimate below-ground biomass from above-ground biomass are often used. Below-ground biomass to above-ground biomass ratios appropriate to each coastal wetland ecosystem should be used. Even within a coastal ecosystem, below-ground biomass to above-ground biomass ratios vary at both individual species and community scales. Thus, it is recommended to use, as far as possible, empirically-derived root-to-shoot ratios specific to each ecosystem type and region. Tables 4.4 provide default root-to-shoot ratios for mangroves, tidal salt marsh and seagrass meadows for different regions. These data can be used as defaults when countries do not have more regionally specific information to develop country specific values.

TIER 3

Tier 3 approaches consist of using a combination of dynamic models and inventory measurements of biomass stock changes. This approach does not employ simple stock changes or emission factors *per se*. Estimates of emissions/removals using model-based approaches are derived from the interaction of multiple equations that estimate the net change of biomass stocks within the models. Models can be used, together with periodic sampling-based stock estimates, to estimate stock changes or inputs and outputs. For example, validated species-specific growth models that incorporate management effects in the drained coastal wetland with corresponding data on management activities, could be used to estimate net changes in biomass in the converted land over time.

4.4.4 Dead Organic Matter

Changes in dead organic matter resulting from drainage activities that occur in forested coastal wetlands are estimated at the tier 1 level because they represent potentially large C emissions to or removals from the atmosphere. Drainage can increase the oxidation rate of dead wood and litter that occurred in the pre-drainage condition. In non-forested coastal wetlands, the default assumption is that no changes in stocks occur at the tier 1 level of estimation. When drainage of coastal wetlands is a key category in both forested or non-forested coastal wetlands, tier 2 and higher estimation methods should be used and these values reported. In all cases, stock changes should be disaggregated relative to climate and vegetation type where applicable.

4.4.4.1 CHOICE OF METHOD

TIERS 1, 2 AND 3

The method for estimating changes in dead organic matter stocks are presented in Equation 4.15. The method estimates the change in dead organic matter stocks applying the Stock-Difference Method where the change in dead organic matter stocks are estimated at two points in time relative to the period before and after the land-use conversion or management activity. This method can be applied for forested coastal wetlands at a Tier 1 level or at a tier 2 level using country estimates where changes occur in either forested or non-forested coastal wetlands. For drainage of forested coastal wetlands, the Gain-Loss Method may also be applied as decay rates in a drained condition are readily estimated. Tier 1 default stocks for forested coastal wetlands are provided in Table 4.6. The default for changes in dead organic matter stocks at the Tier 1 level of estimation is 0 for non-forested coastal wetlands. Tier 2 level of estimation should be used when extensive management activities lead to key category land-use conversions. Better estimates of transition times may also be more appropriate for Tier 2 or higher levels to reduce uncertainties. Estimates of dead wood and litter exports due to tidal advection would be considered negligible under a drained condition.

4.4.4.2 CHOICE OF EMISSION/REMOVAL FACTORS

TIER 1

Countries using a Tier 1 method require data on the default dead wood and litter carbon stocks as defined in Table 4.8. The Tier 1 assumption is that carbon stocks in litter and dead wood pools in all non-forested coastal wetlands and conversion among non-forest land-use categories are zero. For lands converted to forested coastal wetland, the carbon stocks in dead wood and litter pools are assumed to increase linearly over the transition period T (default is 20 yr for both litter and dead wood C stocks). Thus, the annual rate of increase is estimated as the ratio between the difference in carbon stocks in the DOM pools in the non-forest and forest categories, and the numbers of years in the transition period T. Conversions from forested coastal wetland to non-forested coastal wetland are assumed to be emitted in the year of conversion.

TIERS 2 AND 3

The higher Tier methods described above (Equation 4.8) and in Chapter 2 will allow for more robust estimates when applied to national data. Additional requirements may arise if the assumption that carbon stocks in dead wood and litter pools of non-forest land-use categories are zero cannot be justified, such as in some agro-forestry systems or other circumstances where intensive management activities have occurred.

4.4.5 Non-CO_2 emissions

This section provides guidelines for estimating non-CO_2 gas emissions from coastal wetlands, mostly mangroves and tidal marshes, being drained by way of conversing them to other dry land uses. The land use following Coastal Wetland drainage (e.g., agriculture, settlement, aquaculture), will have a major impact on emissions of non-CO_2 greenhouse gases from drained Coastal Wetlands. Non-CO_2 emissions from these dry land uses are dependent upon the type of management that follows drainage of coastal wetlands, and are covered in GL2006 (e.g. Chapter 11 of Volume 4).

Conversely to rewetting of coastal wetlands, drainage of coastal wetlands increases the surface area for aerobic respiration and chemical oxidation of organic matter, resulting in a reduction of CH_4 production but enhanced CH_4 oxidation by methanotrophic bacteria in the top aerated soil layer. CH_4 emissions from aerobic soils are negligible. Under this aerobic soil, N_2O production was attributed mainly to nitrification, and the production and emission rate of N_2O is largely controlled by the provision of nitrogen by mineralization.

Tier 1 method for estimation of the gas emissions from drained wetlands assumes that the emissions could be ignored, and the emission factors for CH_4 and N_2O from drained wetland is assigned to be zero. Tier 2 method requires country specific data that account for climate, vegetation type and soil texture. A comprehensive understanding and representation of the dynamics of non-CO_2 gas emission factors during the drainage, based on field measurement, is involved in higher Tier method (Tier 3).

4.4.5.1 CHOICE OF METHOD AND EMISSION FACTOR

Non-CO2 emissions are dependent upon the type of management that follows drainage of coastal wetlands.

Methods described in Chapter 11 of Volume 4 of the 2006 supplement should be consulted for when land use of drained coastal wetlands includes application of:

- synthetic N fertilisers;
- organic N applied as fertiliser (e.g., animal manure, compost, sewage sludge, rendering waste);
- urine and dung N deposited on pasture, range and paddock by grazing animals;
- N in crop residues (above-ground and below-ground), including from N-fixing crops and from forages

Livestock production on drained coastal wetlands can result in methane emissions from enteric fermentation and both CH_4 and N_2O emissions from livestock manure management systems. The methods for estimating CH_4 and N_2O emissions from these activities are provided in Chapter 10 Emissions from Livestock and Manure Management of Volume 4.

TIER 1

Tier 1 method for estimation of the gas emissions from drained wetlands assumes that the emissions could be ignored, and the emission factors for CH_4 and N_2O from drained wetland is assigned to be zero.

TIER 2 AND TIER 3

Country-specific emission factors should be used in Tier 2 methods. Tier 2 method requires country specific data that account for climate, vegetation type. a comprehensive understanding and representation of the dynamics of non-CO_2 gas emission factors, based on field measurement, is involved in higher Tier method (Tier 3), taking account of parameters such as season, soil texture, tidal elevation and tidal cycle (high tide vs. low tide).

4.5 REWETTING AND RESTORATION OF COASTAL WETLANDS

4.5.1 Description of activities

Rewetting. Rewetting of coastal wetlands occurs where hydrology has been altered but the soil is at an appropriate elevation for the establishment of functioning vegetation. Wetland restoration through reconnection of hydrology halts any ongoing emissions of soil carbon and upon the reestablishment of vegetation reinitiated autochthenous carbon sequestration. Once vegetated is reestablished rates of removals are comparable with natural marsh reference sites (Craft et al., 2003). Restored hydrological connectivity, through actions that remove tidal restrictions or otherwise modify water delivery or discharge, can also result in increased soil storage and productivity of coastal wetland plant species (Harris et al 2010). The rewetted condition covered in this section results where off-site hydrologic modifications reverse drainage or impoundments or other obstructions to hydrologic flow are removed (i.e. levee breach).

Restoration (vegetation and soils). Restoration generally refers to re-creating habitat that was present prior to the current land use within recorded time, but here we also include afforestation in this definition.

1.Afforestation/reforestation. Afforestation and reforestation are common restoration activities in coastal regions to conserve biodiversity and to mitigate disasters from tsunami and cyclones. Improved estimates of C storage in mangroves have recently been obtained at global/regional (Donato et al. 2011) and local (Fujimoto et al. 1999; Ren et al. 2009; Kauffman et al. 2011; Donato et al. 2012; Zhang et al. 2012) scales, which makes it possible to quantify carbon gain due to mangrove afforestation and reforestation. It has been shown that mangroves contain 2-3 times higher carbon pools than in nearby upland forests in tropical Pacific regions (Donato et al. 2012). Similarly, the sediments of mangrove forests in southern China sequester large quantities of soil organic C during mangrove restoration (Zhang et al. 2012). Recognition of the C sequestration value of mangrove forests and other vegetated coastal wetlands provides a strong argument for their protection and restoration (Chmura et al. 2003; Irving et al. 2010; Crooks et al. 2011).

2.Revegetation. Seagrasses are known as ‘ecosystem engineers’. For example, they reduce current velocity and attenuate wave activity, promote soil stabilization and as a result enhance sediment trapping, reduce resuspension that cause accumulation of fine sediments and organic matter in seagrass beds relative to adjacent bare areas. Loss of seagrass cover due to a range of anthropogenic activities leads to de-stabilisation of the sediment, reduction in C sequestration and soil carbon content. Restoration of seagrass meadows can restore these functions, but before recolonization of bare sediment can occur certain habitat conditions must be met, including adequate light levels and the availability of donor material. Donor material can be provided via transplantation or from seed banks or can occur naturally with donor material supplied from adjacent vegetated areas. Restoration occurs slowly, for example a 3-5 yr time lag between nutrient load reduction and initiation of seagrass recolonization has been observed (Vaudrey et al. 2010), with a further 12-15 yr time lag before seagrass biomass attains a relatively stable distribution. Over larger spatial scales, C accumulation rates in seagrass meadows are significant (Duarte et al. 2011; Kennedy et al. 2012).

Creation and restoration with sediment modifications including sediment resupply. These activities include restoration that results from sediment removal or ditch filling or other modifications of the soil and vegetation (hydrology is unaltered), sediment/soil elevation or vegetation that is modified). These activities include: a) *sediment removal* and b) *sediment enhancement* including active sediment amendments. A recent synthesis of wetland restoration studies suggests that wetlands with a higher degree of connectivity with allochthonous inputs is an important factor driving high rates of recovery after restoration in coastal systems such as riverine and tidal marshes (Ballantine and Schneider 2009). In these systems, wetland restoration may be accelerated by inputs of mineral sediment and organic particles (Fennessy et al. 1994, Morgan and Short 2002, Anderson et al. 2005). Examples of sediment modification include the use of sediment slurries of dredge material to fill containment levees, water ponds and deteriorating brackish marshes (LaPeyre et al 2009; Llewellyn and LaPeyre 2011). Deteriorating marshes in tidal systems often result as a function of sediment diversions or embankments that restrict sediment supply that are replaced by shallow ponds (Barras et al 2003). Other factors have been implicated that cause coastal erosion and subsidence (Dahl 2011), but the result can be decreased plant productivity and plant mortality (Baumann et al. 1984; McKee and Mendelssohn 1988).

4.5.2 Soil Carbon

This section deals with anthropogenic impacts to wetland soil organic C stocks, primarily C emissions from soil and soil C storage, by activities affecting soil drainage through rewetting by modification of the water table or restoration of natural hydrology and restoration of vegetation and soils. Guidelines are provided for estimating carbon stock changes in soils of drained coastal wetlands where the hydrological regime has been restored. Restoration of the soil water table and tidal flooding regime in coastal wetlands are assumed, over the long term, to increase rates of soil organic C and increase the net stock of C through return of vertical soil accretion (e.g. Connor et al. 2001). Rates of organic matter decomposition in soil below the restored water table will be lower than that in the previously aerated soil, because anaerobic decomposition will dominate below the water table, and this process is less efficient than aerobic decomposition (e.g., Reddy and Patrick 1975; Kristensen et al. 1995). Belowground biomass is a major contributor to soil carbon stocks, and this production generally occurs within the upper 15 cm of soil in tidal marshes. There may be a lag time until maximum belowground biomass and soil carbon concentrations are reached. It is likely that reference carbon stocks of will be representative of carbon stocks obtained after native wetland vegetation has been reestablished under restored conditions. These responses will vary locally within a wetland and may vary regionally with climate, but as yet there evidence for differences in magnitude of these responses is inadequate. Where this information is available it can be used for refined calculations in upper tiers.

General information and guidance for estimating changes in soil C stocks are provided in Section 4.2 (including generic equations), and this section should be read before proceeding with a consideration of specific guidelines dealing with coastal wetland soil C stocks.

Total change in soil C stocks for *Rewetting and Restoration of Coastal Wetlands* is estimated using Equation 4.2, which combines the change in soil organic C stocks for mineral soils and organic soils (including C gains); and stock changes associated with soil inorganic C pools (if estimated at Tier 3). Changes in stock may also be calculated by input-output approaches though requires additional balancing of dissolved organic carbon emissions along with balancing atmospheric fluxes.

For *Rewetting and Restoration of Coastal Wetlands*, soils are maintained wet (or dry periodically in response to natural hydrological cycles). *Rewetting and Restoration of Coastal Wetlands* may occur through managed activities (such as rewetting of soils in areas disconnected from natural flows by structures) or the removal of barriers to allow for natural natural hydrological connections. . As the water table is maintained at or above the graded soil surface further emissions of soil carbon are assumed to be zero.

4.5.2.1 CHOICE OF METHOD

Inventories can be developed using a Tier 1, 2 or 3 approaches with each successive Tier requiring more details and resources than the previous one. It is also possible that countries will use different tiers to prepare estimates for the separate sub-categories of soil C (i.e., soil organic C stocks changes in mineral and organic soils; and stock changes associated with soil inorganic C pools). A decision tree is provided in Figure 4.1 to assist inventory compilers with the selection of the appropriate tier for their soil C inventory.

4.5.2.2 CHOICE OF EMISSION/REMOVAL FACTORS

MINERAL/ORGANIC SOILS

Tier 1

For all soils, the estimation method is based on changes in soil organic C stocks over a finite (40 yr) period following changes in management that impact soil organic C storage. After a finite transition period, one can assume a steady state change in stock at Tier 1 level of estimation assuming no change in water level.

Equation 4.3 is used to estimate change in soil organic C stocks. Note that area of exposed bedrock in wetlands and open water channels are not included. In practice, country-specific data on land management activity should be obtained and classified into appropriate land management systems, and then stratified by ecosystem type. Table 4.16 provides annual rates of C removal for soils of rewetted and hydrologically restored coastal wetlands.

Tier 2

The Tier 2 method for soils also uses Equation 4.3 but the inventory approach is further developed with country-specific information to better specify emission/removal factors for climate regions, soil types, and/or the land management activities. Tier 2 also employs models to better approximate accumulation or decay rates over time, beyond the linear assumptions made in application of Tier 1 methods.

Tier 3

Tier 3 approaches use dynamic models and/or detailed soil C inventory measurements as the basis for estimating annual stock changes. Estimates of stock changes using model-based approaches are computed from the coupled equations that estimate the net change of soil carbon as described in section 4.3.2.2.

TABLE 4.16 ANNUAL REMOVAL FACTORS FOR REWETTED AND RESTORED SOILS FOR TIER 1 ESTIMATION

Climate Zone	Ecosystem	Number of Observations	Mean (Mg C ha^{-1} yr^{-1})	Range (Mg C ha^{-1} yr^{-1})
Boreal and Temperate	Tidal Marshes	126	2.06	
Tropical	Mangroves	39	1.32	
Global	Seagrass*	17	0.5±0.5	0.04 - 2

Source: Chmura et al . 2003, ,Silfleet et al., 2011, Duarte et al., 2011; *Data only includes measurements representative of ≥1 yr sediment accumulation

Data for tidal marsh soil stocks is derived from the following locations (with number of observations): Louisiana (34), Northeast Canada (31), New England (20), Continental Europe (9), Chesapeake Bay (8), UK, (6), California (5), North Carolina (5), Florida (4), Honk Kong, China (1)

Data for mangrove soil stocks is derived from the following locations (with number of observations), Florida Keys, USA (15), Queensland, Australia (12), Estero Pargo, Mexico (4), Hong Kong, China (2), Pohnpei Island, Micronesia (2), Victoria, Australia (1), Malaysia (1), Puerto Rico (1), South Africa (1).

4.5.3 Biomass

This section provides guidelines for estimating carbon stock changes in biomass due to the reversion from previously drained coastal wetlands, through hydrological change, back to an ecosystem with fully saturated soil and restoration through revegetation or reforestation. These conversions are relevant to mangrove and salt marsh coastal ecosystems. Vegetation that characterizes for example agricultural land, protected by dikes will be lost when the land is inundated by seawater. Over a period of time new vegetation will colonise and expand within the coastal wetland or, in the case of mangroves, re-vegetation may be accelerated through replanting. This scenario is similar to that described during drainage of coastal wetlands in as far as they both should be considered as a two process conversion. If rewetting or restoration of previously drained coastal wetlands results in the transfer of carbon from one pool to another, all transfers must be accounted for during the transition to a new steady state.

4.5.3.1 CHOICE OF METHOD

TIER 1

A Tier 1 approach generally assumes no change in biomass in *Rewetting and Restoration of Coastal Wetlands* except those ecosystems where woody vegetation is prevalent. In any coastal wetland ecosystem, where there is no change in either type or intensity of management, biomass will be in an approximate steady-state (i.e. carbon accumulation through plant/tree growth is roughly balanced by losses through herbivory and decomposition). However, where management changes are occurring over time (e.g., through introduction of aquaculture, physical disturbance, soil removal, nutrient enrichment or other practices) carbon stock changes can be significant. If it is reasonable to assume that any of the coastal wetland ecosystems are not a key source, a country may apply the Tier 1 assumption of no change in biomass. However, if information is available to develop reliable estimates of rates of change in biomass in *Rewetting and Restoration of Coastal Wetlands*, a country may use a higher Tier, even if *Rewetting and Restoration of Coastal Wetlands* is not a key source, particularly if management changes are likely, in which case default data in Tables 4.5-4.8 can be applied.

TIER 2

Estimating changes in carbon stocks in biomass for any rewetted coastal wetland ecosystem can require a two-phase approach if there are different time-scales for vegetation loss associated with the drained coastal wetland land use and replacement by vegetation tolerant of the saline, waterlogged conditions in the rewetted soil or restored ecosystem. There can be an abrupt change in biomass, which can be treated as phase 1 and in-line with the 2006 guidelines be estimated at the year of conversion. The second phase accounts for a gradual biomass loss and gain during a transition period to a new steady state system. At some time after rewetting the new vegetation that develops on the coastal wetland ecosystem should be comparable to that found in its natural setting and then should be considered under, and accounted for under *Rewetting and Restoration of Coastal Wetlands*. The transition period following rewetting and conversion to coastal wetland will vary depending on the how the restoration of the hydrology is managed and countries should determine transition periods that specifically relate to the particular type of rewetting and ecosystem that is affected. The values of coefficients determining the rate of emissions may depend on the transition period used. In line with the 2006 guidelines, rewetted coastal wetlands should be accounted for with Phase 1 methods in the year that they are converted, with Phase 2 methods being used for all subsequent transitional years until the new vegetation in the rewetted coastal ecosystem achieves steady state. At the end of the transitional period, the land area should be accounted for under *Rewetting and Restoration of Coastal Wetlands.* If there is no abrupt change in biomass associated with the conversion, for example if hydrology is restored to existing tilled fields or abandoned shrimp ponds or a gradual transition occurs between two different vegetation types, Phase 1 accounting would not be appropriate and the whole conversion accounting can be undertaken using Phase 2 methods.

In cases where there is an immediate and abrupt carbon stock change in biomass on the land that is being rewetted the effect of this conversion will be estimated using Equation 2.16 in Chapter 2 of the 2006 guidelines, with the default assumption that B_{After}=0. During the transition period, pools that gain or lose C often have a non-linear loss or accumulation curve that can be represented through successive transition matrices. If the true shapes of the curves are known, these curves can be applied to each cohort that is under transition during the reporting year to estimate the annual emission or removal by the specific pool. If the shape of the curve is unknown, countries may simplify and use a linear decay function to estimate pool changes. A disturbance matrix as described in section 4.3.3.1 is a useful tool for *good practice* implementation.

4.5.3.2 CHOICE OF EMISSION/REMOVAL FACTORS

TIER 1

Tier 1 methods require estimates of the biomass of the land use before conversion and after conversion. It is assumed that there is no biomass (as in abandoned shrimp ponds) or that all biomass is cleared when preparing land for rewetting and restore a coastal wetland. Thus, the default for biomass immediately after conversion is 0 tonne ha^{-1}. Default values for biomass before conversion can be found in the appropriate chapters (e.g. Croplands, Grasslands) of the 2006 guidelines. Default values for biomass and net growth, following (not immediately after) rewetting and conversion back to a coastal ecosystem, can be found in Table 4.6 and 4.7.

TIER 2

It is *good practice* to use country-specific estimates for biomass stocks and emissions/removals due to land conversion, and also include estimates of on-site and off-site losses due to burning and decay following land conversion by rewetting of previously drained coastal wetlands. These improvements can take the form of systematic studies of carbon content and emissions and removals associated with land uses and land-use conversions within the country or region and a re-examination of default assumptions in light of country-specific conditions.

Region-specific or country-specific data on biomass for growth in the vegetation that characterises the rewetted coastal ecosystem are needed for a Tier 2 approach. These can be obtained from Table 4.6. General guidance on survey and sampling techniques for biomass inventories is given in Volume 4, Chapter 3 (Annex 3A.3).

If possible accurate knowledge of the dynamics of below-ground biomass should be obtained for accounting for carbon stock changes when land is rewetted and converted to a coastal wetland. Estimation of changes in below-ground biomass during conversion from the land use in the previously drained coastal wetland to rewetted coastal ecosystem can be an important component of the calculated stock change. Field measurements are laborious and difficult and thus expansion factors to estimate below-ground biomass from above-ground biomass are often used. Default ratios can be found in Table 4.4.

Below-ground biomass to above-ground biomass ratios can vary at both individual species and community scales. Thus, it is recommended to use, as far as possible, empirically-derived root-to-shoot ratios specific to each land use type, ecosystem and region are used and can be found in the appropriate chapters (e.g. Croplands, Grasslands) of the 2006 guidelines or in this chapter for the rewetted coastal ecosystems (state location).. These data can be used as defaults when countries do not have more regionally specific information to develop country specific values.

TIER 3

Tier 3 approaches consist of using a combination of dynamic models and inventory measurements of biomass stock changes. This approach does not employ simple stock changes or emission factors *per se*. Estimates of emissions/removals using model-based approaches are derived from the interaction of multiple equations that estimate the net change of biomass stocks within the models. Models can be used, together with periodic sampling-based stock estimates, to estimate stock changes or inputs and outputs. For example, validated species-specific growth models that incorporate management effects in the drained coastal wetland with corresponding data on management activities, could be used to estimate net changes in biomass in the converted land over time.

4.5.4 Dead Organic matter

Changes in dead organic matter resulting from rewetting of forested coastal wetlands are estimated at the Tier 1 level because they represent potentially large C emissions to or removals from the atmosphere. In non-forested coastal wetlands, the default assumption is that no changes in stocks occur at the Tier 1 level of estimation. For key categories or extensive management activities occurring in either forested or non-forested coastal wetlands, Tier 2 and higher estimation methods should be used and these values reported. In these cases, stock changes should be disaggregated relative to climate, vegetation type and salinity where applicable.

4.5.4.1 CHOICE OF METHOD

TIERS 1, 2 & 3

The method for estimating changes in dead organic matter stocks are presented in Equations 2.18 and 2.19 (Vol. 4, Chapter 2). The method estimates the change in dead organic matter stocks applying the Stock-Difference Method where the change in dead organic matter stocks are estimated at two points in time relative to the period before and after the land-use conversion or management activity. This method can be applied for forested coastal wetlands at a Tier 1 level or at a Tier 2 level using country estimates where changes occur in either forested or non-forested coastal wetlands. Tier 1 default stocks for forested coastal wetlands are provided in Table 4.9. The default for changes in dead organic matter stocks at the tier 1 level of estimation is 0 for non-forested coastal wetlands. Tier 2 level of estimation should be used when extensive management activities lead to key category land-use conversions. The Gain-Loss Method and better estimates of transition times may also be more appropriate for Tier 2 or higher level of estimation to reduce uncertainties. Estimates of dead wood and litter exports due to tidal advection should also be considered.

4.5.4.2 CHOICE OF EMISSION/REMOVAL FACTORS

TIER 1

Countries using a Tier 1 method require data on the default dead wood and litter carbon stocks as defined in Table 4.9. The Tier 1 assumption is that carbon stocks in litter and dead wood pools in all non-forested coastal wetlands and conversion among non-forest land-use categories are zero. The use of equation 4.17 is recommended for Tier 1 estimation. For lands converted to forested coastal wetland, the carbon stocks in dead wood and litter pools are assumed to increase linearly over the transition period T (default is 20 yr for both litter and dead wood C stocks). Thus, the annual rate of increase is estimated as the ratio between the difference in carbon stocks in the DOM pools in the non-forest and forest categories, and the numbers of years in the transition period T. Conversions from forested coastal wetland to non-forested coastal wetland are assumed to be emitted in the year of conversion.

TIERS 2 AND 3

The higher Tier methods described above (Equation 4.18) and in Chapter 2 will allow for more robust estimates when applied to national data. Additional requirements may arise if the assumption that carbon stocks in dead wood and litter pools of non-forest land-use categories are zero cannot be justified, such as in some agro-forestry systems or other circumstances where intensive management activities have occurred.

4.5.5 Non-CO_2 emissions

These saturated conditions also enable methanogens, sulfate-reducers, and other bacteria to decompose part of the organic matter with CH_4 as a by-product. Nitrous oxide results mainly from microbial denitrification under this reduced environment, with rates depending mostly on the availability of nitrate. The degree of water saturation is one of the key factors controlling CH_4 and N_2O emissions.

Rewetting of Coastal Wetland soils shifts microbial decomposition from aerobic to anaerobic (responsible for CH4 emissions) regimes, generally reducing the potential for CH_4 emissions from rewetted coastal wetlands. If soil conditions prior to rewetting were aerobic, then it is likely that CH4 emissions will increase where tidal water salinities are <15 ppt. However, coastal wetlands subject to intensive nutrient loading may be sources of CH4 emissions. If tidal waters rewetting coastal wetlands are nutrient-enriched, then wetlands with salinities >15 ppt also are likely to be sources of CH_4 emissions.

One of the main controlling factors in this reaction is the availability of inorganic N in the soil. If, prior to rewetting, the area of coastal wetland was subject to increased N supply (e.g. in fertilized land) it may be a source of N_2O emissions during the rewetting. However, experimental studies have shown that the substrate N is rapidly depleted (Moseman-Valtierra et al. 2011), thus N_2O is likely to be negligible after the N is depleted and no continuous N is supplied. Contrarily, flooded coastal wetlands subjected to increased N supplies through ground water or tidal water may become sources of N_2O emissions.

This section provide methods for estimations of non-CO_2 emission from rewetted wetlands at the first conversion process, i.e. initial conversion from some other land use to a rewetted coastal wetlands ecosystem and transition to a new steady state. The estimations of non-CO_2 emissions from the second process, subsequent management of the rewetted coastal wetlands, are addressed in section 4.4.4 in this chapter.

4.5.5.1 CHOICE OF METHOD AND EMISSION FACTORS

TIER 1

Tier 1 method in this section estimates the non-CO_2 emissions by taking no account of the land use prior to rewetting. The emissions of N_2O and CH_4 depend on the nutrient/organic matter available in the tidal water.

If the rewetted lands are subjected to nutrient-enriched condition, N_2O and CH_4 emissions could be estimated using default emission factor similar to those in section 4.4.4 (Nutrient-enriched Wetlands).

In those lands rewetted by 'pristine' water, N_2O emission is assumed to be negligible, and the emission factors are assigned to be zero. For CH_4, its flux depends on salinity level in tidal marshes (Table 4.17), but polyhaline marshes approximate an average CH_4 flux of a 1 g m^{-2} yr^{-1} and thus assumed to be negligible at Tier 1 level of estimation (Poffenbarger et al 2011). In mangrove wetlands, average CH_4 emissions are around 3 (g m^{-2} y^{-1}) and are also considered negligible at Tier 1 estimation level.

TABLE 4.17 EMISSION FACTORS FOR CH_4 FROM "INTACT" COASTAL WETLANDS FOR TIER 1 ESTIMATION

Wetland Type	Salinity type	Default EF (tonnes CH_4-C ha^{-1} y^{-1})	EF Range (tonnes CH_4-C ha^{-1} y^{-1})	Error (95%CI)	Reference
Tidal Marsh	Fresh-Brackish	0.28	0.02-1.6	±78%	Poffenbarger et al 2011
	Polyhaline (>18ppt)	0	0-0.04	±94%	Poffenbarger et al 2011
Mangrove	Saline	0		±87%	Page and Dal 2010

TIER 2 AND TIER 3

Country-specific emission factors should be used in Tier 2 methods. In tier 3 method, the land use prior to rewetting, climate and vegetation type should be taken into account. A comprehensive understanding and representation of the dynamics of non-CO_2 gas emission factors, based on field measurement, is involved in Tier 3.

4.6 COMPLETENESS, TIMES SERIES CONSISTENCY, QUALITY ASSURANCE AND QUALITY CONTROL

4.6.1 Completeness

Completeness is a requirement for greenhouse gas inventories, and it is *good practice* to address all wetland carbon gain and losses. For completeness, it is *good practice* to include all carbon pools and non-CO_2 gases. The wetland area used for calculation for different carbon pools must be the same, and emissions from organic soils and emissions or removals attributed to land-use change on mineral soils should be estimated. Country-specific information should be incorporated into higher tier methodologies. A complete accounting of emissions and removals of CO_2 associated with *Management Changes in Coastal Wetlands, Drainage of Coastal Wetlands* and *Re-Wetting of Coastal Wetlands* is necessary. It is *good practice* that all losses from biomass carbon pools that result in transfers to dead organic matter pools are first accounted for as changes to biomass carbon stocks.

4.6.2 Developing a consistent time series

It is good practice to develop a consistent time series of inventories of anthropogenic emissions and removals of greenhouse gases for all wetland conversion categories using the guidance in Chapter 8 in this volume. Achieving time series data may require extrapolation or interpolation from longer time series data or from long term trends, as few long-term data for available for most coastal wetlands.

Consistent accounting over time of wetland areas included in biomass and soil C emissions and removals inventory requires that activity data be stratified by the common definitions of wetland type/ soil type. Wetlands subject to land-use change will not be lost or double-counted due to accounting errors resulting from inconsistent stratification of wetland types/ soil types. Ideally, the same protocol should be applied consistently every year in the time series, at the same level of disaggregation and where country-specific data are used, it is *good practice* to use the same values and methods for equivalent calculations throughout the time series.

New values should be included if the inventory capacity and information and data sources improve over time. It is *good practice* in these circumstances to consistently recalculate the earlier emissions and removals. Other changes during the time series need to be consistent to take account of new data or methods and their consistency with the earlier data. It is *good practice* to recalculate the entire time series of data if either the default values are changed; changes in wetland types need to be tracked for long periods of time.

It is *good practice* to use the same model parameter values for the entire time series and to recalculate the entire dataset if one of more of the parameters has changed. Failure to do so may result in either under-or over-estimates of the true changes in carbon and non-CO_2 gas emissions or removals.

4.6.3 Quality assurance and quality control

Different levels of precision and accuracy, and as a result, bias will invariably apply to a number of the values used to assess greenhouse gas inventories. Estimates are influenced by the quality and consistency of data and information available as well as knowledge gaps, all of which will vary among countries. Depending on the tier level used, estimates will be affected by different sources and degrees of error, such as sampling error.

It is *good practice* to execute quality control checks through Quality Assurance (QA) and Quality Control (QC) procedures as detailed in Chapter 8, and review the emission estimation procedures by experts. Additional quality control checks as outlined in Chapter 8 and quality assurance procedures may also be applicable. This is especially so if higher tier methods are used. It is *good practice* to supplement the general QA/QC related to data processing, handling, and reporting and documenting, with source-specific procedures. QA/QC procedures should be documented separately for *Management Changes in Coastal Wetlands, Drainage of Coastal Wetlands* and *Re-Wetting of Coastal Wetlands.*

Organizations and institutions which collect the data are responsible for reviewing data collection methods and all aspects of the data handling and analysis procedures, and ensure that they are done correctly, and are complete and consistent over time. It is important to document all procedures and processes as it enables reviewers to identify inaccuracy, gaps and to suggest improvements. Transparency is most important in other to ensure consistency and clarity of the processes and procedures over time.

All data should be checked against other reliable sources of information that are independent. Any differences or discrepancies must be documented, and consistency must be applied to total areas involved in the inventory to ensure that wetland area are neither ‘created’ or ‘lost’ overtime. When using country-specific data, the inventory compiler should compare these data to the IPCC default values or the Emissions Factor Database (EFDB) and detail any differences. These country-specific data must be of high quality, adequately described, and documented.

If factors are based on direct measurements (i.e., soil C content) the inventory agency should review the measurements to ensure that they are representative of the actual range of environmental conditions. It is *good practice* to review and, if necessary, revise the default assumptions and to compare model estimates with field measurements and other data sources.

4.6.4 Reporting and documentation

General requirements for reporting and documentation are set out in Chapter 8. It is *good practice* to archive and document all data and information applied to produce the national emissions/removals inventory. Definitions of all carbon pools should be included in the inventory, including evidence that these definitions have been applied consistently over time.

Documentation is necessary for demonstrating transparency, completeness, consistency of all data and methods for interpolating between samples, methods and years, and for recalculating and avoidance of possible double accounting or 'loss' of C inventory. Regardless of Tier methodology used, explanations are required for decisions regarding choice of methodology, approaches and use of default or other data. This is necessary to facilitate examination by independent third parties; inventories should include summaries of approaches and methods used and references to data sources so that the reported emissions estimates are transparent and can be retraced or recalculated.

All data sources, including default values, must be quoted. The scientific basis for any country-specific data and methods must be completely described and justified, as well as describing sources and magnitudes of uncertainty. This is especially so for any large-scale estimates as in these cases the statistical procedures should be described and well as the level of uncertainty

Differences between years in emissions should be explained and the possible reasons for these differences documented as much as possible.

4.7 FUTURE METHODOLOGICAL DEVELOPMENT

TBD

Annex 4.1

Mangrove soils

Lovelock et al., 2011 measured CO2 efflux from mangrove soils that had been cleared for up to 20 years on the islands of Twin Cays, Belize. Rates of emissions from disturbed soils display exponential decay. At years 1 and 20 Lovelock et al., 2011, document rates of emissions of 28.9 Mg C ha^{-1} yr^{-1} and 8.2 Mg C ha^{-1} yr^{-1}, respectively.

It has been shown that mangroves contain 2-3 times higher carbon pools than in nearby upland forests (830-1218 vs. 357-486 Mg C ha^{-1}) in tropical Pacific regions (Donato et al. 2012). Similarly, the sediments of mangrove forests in southern China sequester large quantities of soil organic C during mangrove restoration, with sediment total organic carbon densities of 90, 170 and 288 Mg C ha^{-1}, respectively, for barren sites, mangrove plantations, and natural mangrove forests (Zhang et al. 2012). In afforested *Sonneratia apetala* plantations, the average annual rate of total carbon storage accumulation at 4, 5, 8, and 10-year age was 5.0, 7.9, 8.7, and 8.4 Mg C ha^{-1} yr^{-1}, respectively (Ren et al. 2009).

Seagrass soils

Changes in carbon stocks during meadow recolonization have been assessed over different time scales (Pedersen et al. 2003; Barron et al. 2004; McGlathery et al. 2012). In general, no increase in sediment carbon pools can be discerned during early years (0-6 yr) of meadow development. There has only been one study over an extended period (9 yr), after which time there were quantifiable changes in the sediment carbon pool. The carbon stock in unvegetated sediment (1.4 Mg C ha^{-1}) was almost half that found in the restored meadow (2 Mg C ha^{-1}). Over larger spatial scales, C accumulation rates in seagrass meadows vary from 0.03 to 2 Mg C ha^{-1} yr^{-1} (range of which 50% can be attributed to seagrass organic matter (Kennedy et al. 2012) resulting 0.02 to 1 Mg C $ha^{-1}yr^{-1}$ (mean 0.54±0.5 Mg C ha-1, N=16) accumulation of seagrass C (Duarte et al. 2011).

References

Adams, C.A., Andrews, J.E., Jickells, T. (2011). Nitrous oxide and methane fluxes vs. carbon, nitrogen and phosphorous burial in new intertidal and saltmarsh sediments. *Science of the Total Environment* (in press)

Allen D.E., Dalal R.C., Rennenberg H., Meyer R.L., Reeves S. and Schmidt S. 2007. Spatial and temporal variation of nitrous oxide and methane flux between subtropical mangrove soils and the atmosphere. Soil Biology and Biochemistry 39: 622–631.

Allen, D., Dalal, RC, Rennenberg, H, Schmidt, S. 2011. Seasonal variation in nitrous oxide and methane emissions from subtropical estuary and coastal mangrove sediments, Australia. Plant Biology 13: 126-133.

Alongi, DM. 2010. The Energetics of Mangrove Forests. Springer Science.

Alongi, DM, Pfitzner, J, Trott, LA, Tirendi, F, Dixon, P, Klumpp, DW. 2005. Rapid sediment accumulation and microbial mineralization in forests of the mangrove Kandelia candel in the Jiulongjiang Estuary, China. Estuarine Coastal and Shelf Science 63: 605-618.

Anderson, C. J., W. J. Mitsch, and R. W. Nairn. 2005. Temporal and spatial development of surface soil conditions at two created riverine marshes. Journal of Environmental Quality 34:2072–2081.

Apostolaki ET, Holmer M, Marbà N, Karakassis I. 2010. Metabolic imbalance in coastal vegetated (*Posidonia oceanica*) and unvegetated benthic ecosystems. Ecosystems 13:459−471

Apostolaki ET, Holmer M, Marbà N, Karakassis I. 2011. Reduced carbon sequestration in a Mediterranean seagrass (Posidonia oceanica) ecosystem impactedby fish farming. Aquaculture Environment Interactions 2: 49-59.

Armentano, T. V. and Menges, E. S.: Patterns of change in the carbon balance of organic soil wetlands of the temperate zone, Journal of Ecology 74:755–774.

Badalamenti F, Alagna A., D'Anna G.a, Terlizzi A., Di Carlo G. 2011 The impact of dredge-fill on P*osidonia oceanica* seagrass meadows: Regression and patterns of recovery Marine Pollution Bulletin 62: 483–489.

Ballantine K. and Schneider R. 2009. Fifty-five years of soil development in restored freshwater depressional wetlands. Ecological Applications 19: 1467-1480.

Barras, J.A., S. Vebille, D. Britsch, S. Hartley, S. Hawes, J. Johnston, P. Kemp, Q. Kinler, A. Martucci, J. Porthouse, D. Reed, K. Roy, S. Sapkota, and, J. Suhayda. 2003. Historical and projected coastal Louisiana land changes: 1978–2050: USGS Open File Report 03-334, 29 pp.

Barron, C., Marbaa, N, Terrados, J, Kennedy, H, Duarte, CM. 2004. Community metabolism and carbon budget along a gradient of seagrass (Cymodocea nodosa) colonization. Limnology and Oceanography 49: 1642-1651.

Baumann, R.H., J.W. Day Jr., and C.A. Miller. 1984. Mississippi deltaic wetland survival: Sedimentation versus coastal submergence. Science 224: 1093–1095.

Baustian, J., Mendelssohn, I., Lin, Q. and Rapp J. 2011. In situ burning restores the ecological function and structure of an oil-impacted coastal marsh. Environmental Management 46: 781-789.

Bauza, J., Morell, J.M, Corredor, J.E. (2002). Biogeochemistry of nitrous oxide production in the red mangrove (Rhizophora mangle) forest soils. *Estuarine, Coastal and Shelf Science* 55: 697-704.

Bridgham, SD, Megonigal, JP, Keller, JK, Bliss, NB, Trettin, C. 2006. The carbon balance of north American wetlands. Wetlands 26:889-916.

Buresh, RJ, DeLaune, RD, Patrick, WH. 1980. Nitrogen and phosphorus distribution and utilization by Spartina alterniflora in a Louisiana Gulf Coast Marsh. Estuaries 3: 111-121.

Burnside N.G., Joyce C.B., Puurmann E. and Scott D.M. 2007. Use of vegetation classification and plant indicators to assess grazing abandonment in Estonian coastal wetlands. Journal of Vegetation Science 18: 645-654.

Buth, G.J.C. 1987. Decomposition of roots of three plant communities in a Dutch salt marsh. Aquatic Botany 29: 123-138.

Byun, D.S., Wanga, X.H. and Holloway, P.E. (2004) Tidal characteristic adjustment due to dyke and seawall construction in the Mokpo Coastal Zone, Korea *Estuarine, Coastal and Shelf Science* 59:185-196.

Chauhan, R, Ramanathan, AL, Adhya, TK. 2008. Assessment of methane and nitrous oxide flux from mangroves along the Eastern coast of India. Geofluids 8: 321-332.

Chen G.C., Tam N.F.Y. and Ye Y. 2010. Summer fluxes of atmospheric greenhouse gases N_2O, CH_4 and CO_2 from mangrove soil in South China. Science of the Total Environment 408: 2761–2767.

Chen G..C., Tam N.F.Y., Wong Y.S. and Ye Y. 2011. Effect of wastewater discharge on greenhouse gas fluxes from mangrove soils. Atmospheric Environment 45: 1110-1115.

Chen G.C., Ye Y., Lu C.Y. 2008. Seasonal variability of leaf litter removal by crabs in a *Kandelia candel* mangrove forest in Jiulongjiang Estuary, China. Estuarine, Coastal and Shelf Science 79: 701-706.

Cheng X., Peng R. Chen J., Luo Y., Zhang Q., An S., Chen J., and Li B. 2007. CH_4 and N_2O emissions from *Spartina alterniflora* and *Phragmites australis* in experimental mesocosms, Chemosphere 68:420-427.

Chmura G..L., Anisfeld S., Cahoon D. and Lynch J. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17: 1111 (1-12).

Connor, R.F., Chmura, G.L. and Beecher, C.B. 2001. Carbon accumulation in Bay of Fundy salt marshes: implications for restoration of reclaimed marshes. Global Biogeochemical Cycles 15(4): 943-954.

Corredor, JE, Morell, JM, Bauza, J. 1999. Atmospheric nitrous oxide fluxes from mangrove sediments. Marine Pollution Bulletin 38: 473-478.

Craft, C., Megongial, P, Broome, S., Stevenson, J. Freese, R., Cornell, J, Zheng, L, Sacco, J. 2003. The pace of ecosystem development of constructed Spartina alterniflora marshes. Ecological Applications 13: 1417-1432.

Crooks S. and others. 2011. Mitigating Climate Change through Restoration and Management of Coastal Wetlands and Near-shore Marine Ecosystems: Challenges and Opportunities. Environment Department Paper 121. Washington, DC: World Bank.

Curcó, A, Ibanez, C, Day, JW, Prat, N. 2002. Net primary production and decomposition of salt marshes of the Ebre Delta. Estuaries 25: 309-324.

Daby D. 2003 Effects of seagrass bed removal for tourism purposes in a Mauritian bay . Environmetal Pollution 125: 313-324.

Dahl, T.E. 2011. Status and trends of wetlands in the conterminous United States 2004 to 2009. U.S. Department of the Interior; Fish and Wildlife Service, Washington, D.C**.** 108 pp.

Dai M. and others. 2008. Nitrification and inorganic nitrogen distribution in a large perturbed river/estuarine system: the Pearl River Estuary, China. Biogeosciences 5: 1545–1585.

Da Silva J.F., Duck R.W., Catarino J.B. 2004. Seagrasses and sediment response to changing physical forcing in a coastal lagoon. Hydrology and Earth System Sciences 8: 151–159.

Darby, FQ and Turner, RE. 2008. Below- and aboveground Spartina alterniflora production in a Louisiana salt marsh. Estuaries and Coasts 31: 223-231.

Ding W., Zhang,Y., and Cai Z. 2010. Impact of permanent inundation on methane emissions from a *Spartina alterniflora* coastal salt marsh. Atmospheric Environment 44: 3894-3900.

Donato D, JB Kauffman, D Murdiyarso, S Kumianto, M Stidham, M Kanninen. 2011. Mangroves among the most carbon-rich forests in the tropics. Nature Geoscience 4: 293-297.

Donato D. and others. 2012. Whole-island carbon stocks in the tropical Pacific: Implications for mangrove conservation and upland restoration. Journal of Environmental Management 97:89-96.

Duarte C.M. and Chiscano C.L. 1999 Seagrass biomass and production: a reassessment. Aquatic Botany, 65, 159-174

Duarte, CM, Marba, N, Gacia, E, Fourqurean, JW, Beggins, J, Barron, C, Apostolaki, ET. 2011. Seagrass community metabolism: assessing the carbon sink capacity of seagrass meadows. Global Biogeochemical Cycles 24

Dumbauld B.R., Ruesink J.L., Rumrill S.S. 2009 The ecological role of bivalve shellfish aquaculture in the estuarine environment: A review with application to oyster and clam culture in West Coast (USA) estuaries. Aquaculture 290: 196–223.

Dunton, KH, Hardegree, B, Whitledge, TE. Response of estuarine marsh vegetation to interannual variation in precipitation. Estuaries 24: 851-861.

Edwards, KR and Mills, KP. 2005. Aboveground and belowground productivity of Spartina alterniflora (smooth cordgrass) in natural and created Louisiana salt marshes. Estuaries 28: 252-265.

Ellison, AM and Farnsworth, EJ. 1996. Anthropogenic disturbance of Caribbean mangrove ecosystems: past impacts, present trends, and future predictions. Biotropica 28: 549-565.

Erftemeijer P.L.A. and Lewis III, R.R. 2006 Environmental impacts of dredging on seagrasses: A review. Marine Pollution Bulletin 52:1553–1572.

Fennessy, M. S., C. C. Brueske, and W. J. Mitsch. 1994. Sediment deposition patterns in restored freshwater wetlands using sediment traps. Ecological Engineering 3:409–428.

Food and Agriculture Organization of the United Nations (FAO). 2003. Status and trends in mangrove area extent worldwide, FAO, Paris, 2003.

Food and Agriculture Organization of the United Nations. 2007. The World's Mangroves 1980-2005. *FAO Forestry Paper 153.*

Flores-Verdugo FJ, Day JW, Briseño-Dueñas R. 1987. Structure, litter fall, decomposition, and detritus dynamics of mangroves in a Mexican coastal lagoon with an ephemeral inlet. Marine Ecology Progress Series 35: 83-90.

Fujimoto, K and others. 1999. Below-ground carbon storage of Micronesian mangrove forests. Ecological Research 14:409-413.

Gabrey S.W. and Afton A.D. 2001. Plant community composition and biomass in Gulf Coast Chenier Plain marshes: Responses to winter burning and structural marsh management. Environmental Management 27: 281-293.

Gedan, K.B., Silliman, B. and Bertness, M.D. 2009. Centuries of Human-Driven Change in Salt Marsh Ecosystems. *Annual Review of Marine Science* 1:117–41.

Giri, R and others. 2010. Status and distribution of mangrove forests of the world using earth observation satellite data. Global Ecology and Biogeography 20: 154-159.

Guimarães H.M.E., Cunha A.H., Nzinga R.L., Marques J.F. 2012 The distribution of seagrass (*Zostera noltii*) in the Ria Formosa lagoon system and the implications of clam farming on its conservation. Journal for Nature Conservation 20: 30– 40

Hammerstrom K.K., Kenworthy W.J., Whitfield P.E. and Merello M.F. 2007 Response and recovery dynamics of seagrasses *Thalassia testudinum* and *Syringodium filiforme* and macroalgae in experimental motor vessel disturbances. Marine Ecology Progress Series 345: 83–92,

Harris R.J, Mildbrandt E.C., Everham E.M. and Bovard B.D. 2010. The effects of reduced tidal flushing on mangrove structure and function across a disturbance gradient. Estuaries and Coasts 33:1176-1185.

Hemminga, M.A., Kok, C.J. and de Munck, W. 1988. Decompostion of *Spartina anglica* roots and rhizomes in a salt marsh of the Westerschelde Estuary. Marine Ecology Progress Series 48:175-184.

Huang B., Yu K. and Gambrell R.P. 2009. Effects of ferric iron reduction and regeneration on nitrous oxide and methane emissions in a rice soil. Chemosphere 74: 481-486.

Imgraben S. and Dittmann S. 2008. Leaf litter dynamics and litter consumption in two temperate South Australian mangrove forests. Journal of Sea Research 59: 83-93.

Kauffman, J.B. C Heider & T G. Cole &K A. Dwire & Dl C. Donato. 2011. Ecosystem carbon stocks of Micronesian mangrove forests. Wetlands 31:343-352.

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Kenny, C., Yamulki, S., Blackwell, M., Maltby, E., French, P., Birgand, F. (2004). The release of nitrous oxide from the intertidal zones of two european estuaries in response to increased ammonium and nitrate loading. *Water, Air and Soil Pollution: Focus* 4: 61–66.

Krauss KW, TW Doyle, RR Twilley, TJ Smith, KT Whelan, JK Sullivan. 2005. Woody debris in the mangrove forests of south Florida. Biotropica 37: 9-15

Kreuzwieser J, Buchholz J, and Rennenberg H. 2003. Emission of methane and nitrous oxide by Australian Krithika, mangrove ecosystems. Plant Biology 5: 423–431

Kreuzwieser J., Purvaja, R. and Ramesh, R. 2008. Fluxes of methane and nitrous oxide from an Indian mangrove. Current Science 94: 218-224.

Kristensen, E., Bouillon, S. Dittmar, T. and Marchand, C. 1995. Organic carbon dynamics in mangrove ecosystems: a review. Aquatic Botany 89:201-219.

Krithika, K, Purvaja, R, Ramesh, R. 2008. Fluxes of methane and nitrous oxide from an Indian mangrove. Current Science 94: 218-224.

La Peyre MK, Gossman B, Piazza BR. 2009. Short- and Long-Term Response of Deteriorating Brackish Marshes and Open-Water Ponds to Sediment Enhancement by Thin-Layer Dredge Disposal. Estuaries and Coasts 32: 390-402.

Landsat TM. 2009. Photogrammetric Engineering and Remote Sensing 59: 415–421.

Larkum, A.W.D. and West R.J. 1990 Long-term changes of seagrass meadows in Botany Bay, Australia. Aquatic Botany 37: 55-70.

Latimer J.S. and Rego S.A. 2010. Empirical relationship between eelgrass extent and predicted watershed-derived nitrogen loading for shallow New England estuaries. Estuarine, Coastal and Shelf Science 90: 231-240.

Leendertse P.C., Rozema J. and Andrea A. 1997. Effects of nitrogen addition on the growth of the salt marsh grass *Elymus athericus*. Estuarine, Coastal and Shelf Science 3: 35-40.

Leonard C.A., Ahn C. and Birch D. 2010. Above- and below-ground vegetative responses to prescribed fire regimes in a Chesapeake Bay tidal brackish marsh. Journal of Ecology and Field Biology 33: 351-361.

Liao B.W., Zheng D.Z. and Zheng S.F. 1998. Biomass and Productivity of *Bruguiera gymnorrhiza* stand. Forest Research 4: 22-29 (in Chinese with English abstract).

Liikanen, A, Silvennoinen, H. Karvo, A, Rantakokko, P, Martikainen, PJ. 2009. Boreal Environment Research 14: 351-368.

Lin G.H. and Lin P. 1988. Study of energy in two mangrove communities, *Bruguera sexangula* and *Kandelia candel, in* China. Acta Phytoecologica et Geobotanica Sinica, 12: 31-39. (in Chinese with English abstract).

Lin P. and Lu C.Y. 1990. Biomass and productivity of *Bruguera sexangula* mangrove forest in Hainan Island, China. Journal of Xiamen University (Natural Science) 29: 209-213 (In Chinese with English abstract)

Lindau, CW, Delaune, RD. 1991. Dinitrogen and nitrous-oxide emission and entrapment in Spartina alterniflora salt marsh soils following addition of N-15 labelled ammonium and nitrate. Estuarine Coastal and Shelf Science 32: 161-172.

Llewellyn C and La Peyre M. 2011. Evaluating Ecological Equivalence of Created Marshes: Comparing Structural Indicators with Stable Isotope Indicators of Blue Crab Trophic Support. Estuaries and Coasts 34: 172-184.

Loveland D.G. and Ungar I.A. 1983. The effect of nitrogen fertilization on the production of halophytes in an inland salt marsh. *American Midland Naturalist 109: 346-354.*

Lovelock, CE, Ruess, RW, Feller, IC. CO2 efflux from cleared mangrove peat. PLOS One 6: 1-4.

Martin S.R., Onuf C.P.and Dunton K.H. 2008. Assessment of propeller and off-road vehicle scarring in seagrass beds and wind-tidal flats of the southwestern Gulf of Mexico. Botanica Marina 51: 79–91.

McGlathery, KJ, Reynolds, LK, Cole, LW, Orth, RJ, Marion, SR, Schwarzschild, A. 2012. Recovery trajectories during state change from bare sediment to eelgrass dominance. Marine Ecology Progress Series 448: 209-221.

McKee, K.L., and I.A. Mendelssohn. 1988. Spartina alterniflora dieback in Louisiana: Time-course investigation of soil waterlogging effects. Journal of Ecology 76: 509–521.

Mesléard F., Lepart J., Grillas P. and Mauchamp A. 1991. Effects of seasonal flooding and grazing on the vegetation of former ricefields in the Rhône delta (Southern France). Plant Ecology 145:101-114

Mitchell L.R., Gabrey S., Marra P.P. and Erwin R.M. 2006. Impacts of marsh management on coastal-marsh bird habitats. In R. Greenberg, J. Malonado and S. Droege (Eds.), Terrestrial Vertebrates of Tidal Marshes: Evolution, Ecology, and Conservation; Studies in Avian Biology (No.32**).** Cooper Ornithological Society. p. 155-175.

Morgan, P. A., and F. T. Short. 2002. Using functional trajectories to track constructed salt marsh development in the Great Bay Estuary, Maine/New Hampshire, USA. Restoration Ecology 10:461–473.

Moseman-Valtierra S., Gonzalez R., Kroeger K., Tang J., Chun W., Crusius J., Bratton J., Green A. and Shelton J. 2011. Short-term nitrogen additions can shift a coastal wetland from a sink to a source of N_2O. Atmospheric Environment 45: 4390-4397.

Muñoz-Hincapié, M., Morell, J.M., Corredor, J.E. (2002). Increase of nitrous oxide flux to the atmosphere upon nitrogen addition to red mangroves sediments. *Marine Pollution Bulletin* 44: 992–996.

Neves, JP, Ferreira, Simoes, MP, Gazarini, LC. 2007. Primary production and nutrient content in two salt marsh species, Atriplex portulacoides L. and Limoniastrum monopetalum L. in southern Portugal. Estuaries and Coasts 30: 459-468.

Olsen Y.S., Dausse A., Garbutt A., Ford H., Thoman, D.N. and Jones, D.L. 2011. Cattle grazing drives nitrogen and carbon cycling in a temperate salt marsh. Soil Biology and Biochemistry 43:531-541.

Onuf C.P., 1994. Seagrasses, dredging and light in Laguna Madre, Texas, USA. Estuarine Coastal and Shelf Science 39: 75–91.

Owens A.B., Proffitt C.E. and Grace J.B. 2007. Prescribed fire and cutting as tools for reducing woody plant succession in a created salt marsh. Wetlands Ecology and Management 15: 405-416.

Pedersen, MF, Duarte, CM, Cebrian, J. 1997. Rates of changes in organic matter and nutrient stocks during seagrass Cymodocea nodosa colonization and stand development. Marine Ecology Progress Series 159: 29-36.

Penn,N., 1981. The environmental consequences and management of coral sand dredging from seagrass beds in the Suva region, Fiji Islands. Bulletin of Marine Science 31:814-824.

Pennings S.C., Stanton L.E. and Stephen Brewer J. 2002. Nutrient effects on the composition of salt marsh plant communities along the southern Atlantic and Gulf coasts of the United States. Estuaries 25: 1164–1173.

Perillo G.M.E., Wolanski E., Cahoon D.R., Brinson M.M. 2009. Coastal Wetlands: An Integrated Ecosystem Approach. Elsevier, Amsterdam, 941 pp.

Poeplau, C., Axel, D., Vesterdal, L., Leifeld, J., Van Wesemael, B,. Schumacher, J. and Gensior, A. 2011. Temporal dynamics of soil organic carbon after land-use change in the temperate zone – carbon response functions as a model approach. Global Change Biology 17:2415-2427.

Poffenbarger, H.J, Needelman, B.A, and Megonigal, J.P. (2011). Salinity influence on methane emissions from tidal marshes. Wetlands 31(5):831-842.

Pozo, J. and Colino, R. 1992. Decomposition processes of *Spartina maritima* in a salt marsh of the Basque Country. Hydrobiologia 231:165-175.

Purvaja R. and Ramesh R. 2001. Natural and anthropogenic methane emission from coastal wetlands of South India. Environmental Management 27: 547-557.

Rajendran N. and Kathiresan K. 1996. Effect of effluent from a shrimp pond on shoot biomass of mangrove seedlings. Aquaculture Research 27: 745-747.

Ramesh, R., Purvaja, G. R., Parashar, D. C., Gupta, P. K., Mitra, A.P., 2011. Anthropogenic Forcing on Methane Efflux from Polluted Wetlands (Adyar River) of Madras City, India. *Ambio* 6: 369-374.

Ramos e Silva, CA, Oliveira SR, Rêgo RDP, Mozeto AA. 2007. Dynamics of phosphorus and nitrogen through litter fall and decomposition in a tropical mangrove forest. Marine Environmental Research 64: 425-534.

Reddy, K.R. and Patrick, W.H., Jr. (1975). Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. Soil Biology Biochemistry 7:87-94.

Ren H., Chen H., Li Z.A., Han W.D. 2009. Biomass accumulation and carbon storage of four different aged *Sonneratia apetala* plantations in Southern China. Plant and Soil 327:279-291.

Robertson A.I. and Daniel P.A. 1989. Decomposition and the annual flux of detritus from fallen timber in tropical mangrove forests. Limnology and Oceanography 34: 640-646.

Setia, R., Marschner, P., Baldock, J., Chittleborough, D. and Verma, V. 2011. Relationships between carbon dioxide emission and soil properties in salt-affected landscapes. Soil Biology and Biochemistry 43:667-674.

Silva CAR, Mozeto AA, Ovalle ARC. 1998. Distribution and fluxes as macrodetritus of phosphorus in red mangroves, Sepetiba Bay, Brazil. Mangroves and Salt Marshes 2:37-42.

Sotomayor, D, Corredor JE, Morell, JM. 1994. Methane flux from sediments along the southwestern coast of Puerto-Rico. Estuaries 140-147.

Steward J.S.., and Green W.C.G. 2007. Setting load limits for nutrients and suspended solids based upon seagrass depth-limit targets. Estuaries and Coasts 30: 657–670

Strauss E.A. and Lamberti G.A. 2000. Regulation of nitrification in aquatic sediments by organic carbon. Limnology and Oceanography 45: 1854-1859.

Sukardjo, S. 1993. The present status of the mangrove forests in the northern coast of West Java with special reference to the recent utilization. Southeast Asian Studies 31: 141-157.

Sun S, Cai Y, Tian X. 2003. Salt marsh vegetation change after a short-term tidal restriction in the Changjiang Estuary. *Wetlands* 23:257–66.

Thiery A. and Puente L. 2002. Crustacean assemblage and environmental characteristics of a man-made solar saltwork in southern France, with emphasis on anostracan (Branchiopoda) population dynamics. Hydrobiologia 486: 191-200.

Tong, C, Wang, WQ, Zeng, CS, Marrs, R. 2010. Methane (CH(4)) emission from a tidal marsh in the Min River estuary, southeast China. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 45: 506-516.

Twilley R.W., Lugo A.E., Patterson-Zucca C. 1986. Litter Production and Turnover in Basin Mangrove Forests in Southwest Florida. Ecology 67: 670-683

Twilley R.R, Pozo M., Garcia VH, Rivera-Monroy VH, Zambrano R, Bodero A. 1997. Litter dynamics in riverine mangrove forests in the Guayas River estuary, Ecuador. Oecologia 111: 109-122.

Vaudrey, JMP, Kemer, JN, Branco, BF, Short FT. 2010. Aquatic Botany 93: 237-243.

Walters, BB, Ronnback, P, Kovacs, JM, Crona, B, Hussain, SA, Badola, R, Primavera, JH, Barbier, E, Dahdouh-Guebas, F. 2008. Ethnobiology, socio-economics and management of mangrove forests: A review. Aquatic Botany 89: 220-236.

Wang, DQ, Chen, ZL, Wang, J, Xu, SY, Yang, HX, Chen, H, Yang, LY, Hu, L. 2007. Summer-time denitrification and nitrous oxide exchange in the intertidal zone of the Yangtze Estuary. Estuarine Coastal and Shelf Science 73: 43-53.

Wang, J., Dodla, S., DeLaune R., and Cook, R. L. (2010). Organic carbon transformation along a salinity gradient in Louisiana wetland soils. 2010 19th World Congress of Soil Science, Soil Solutions for a Changing World 1 – 6 August 2010, Brisbane, Australia.

West, RC. 1977. Tidal Salt-Marsh and Mangal Formations of Middle and South America Ecosystems of the World 1: Wet Coastal Ecosystems, Elsevier Scientific Publishing Co., New York 1977, p. 193-213. 16 fig, 1 tab, 92 ref.

Windham L. 2001. Comparison of biomass production and decomposition between Phragmites australis (common reed) and Spartina patens (salt hay grass) in brackish tidal marshes of New Jersey, USA. Wetlands 21: 179-188.

Yang D, and Yang C. 2009 Detection of seagrass distribution changes from 1991 to 2006 in Xincun Bay, Hainan, with satellite remote sensing. Sensors 2: 830-844.

Yu, O. and Chmura, G.L. 2010. Soil carbon is maintained under grazing in a St. Lawrence Estuary tidal marsh. Environmental Conservation 36: 312-320.

Zainal A.J.M., Dalby D.H. and Robinson, I.S., 1993. Monitoring marine ecological changes on the east coast of Bahrain with

Zhang, J.P. and others. 2012. Estimating change in sedimentary organic carbon content during mangrove restoration in southern China using carbon isotopic measurements. Pedosphere 22:58-66.

Zhang Y., Ding W., Cai Z., Valerie P., and Han F. 2010. Response of methane emission to invasion of *Spartina alterniflora* and exogenous N deposition in the coastal salt marsh. Atmospheric Environment 44: 4588-4594