Spring 5-13-2014

Carbon Sequestration in Tidal Salt Marshes and Mangrove Ecosystems

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This Master's Project

CARBON SEQUESTRATION IN TIDAL SALT MARSHES AND MANGROVE ECOSYSTEMS

by

Carlos Eduardo Quintana Alcántara

is submitted in partial fulfillment of the requirements
for the degree of:

Master of Science
in
Environmental Management

at the

University of San Francisco

Submitted: __________________________

Received: __________________________

Carlos E. Quintana Date

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Abstract

Wetlands are dynamic systems that provide support to vital environmental functions and services. Wetlands take part in the global carbon cycle by holding organic carbon in biomass, soils and sediments. In recent years, the wetland carbon sequestration capacity has been researched worldwide due to the increase of the concentrations of greenhouse gasses implicated in global warming and climate change. Although coastal wetlands release the greenhouse gasses carbon dioxide, methane and nitrous oxide, these ecosystems maintain high performance in offsetting significant amounts of atmospheric carbon. This paper investigated the carbon sequestration capacity of coastal wetland ecosystems summarizing the environmental conditions and factors associated with carbon fixation, production and storage in tidal salt marshes and mangrove ecosystems. The results showed that both coastal wetland ecosystems types are a significant carbon pools compared to other terrestrial ecosystems. Global estimations indicated that carbon storage in coastal wetlands ranges from 0.4 to 8.9 Pg C. Tidal salt marshes and mangroves store a large amount of carbon in their soils, which was estimated in the range of 0.02 to 4.9 Pg C. Moreover, the estimates of carbon storage in mangrove biomass ranged between 1.22 to 4.98 Pg C, and from 0.007 to 0.02 Pg C in tidal marsh biomass. Additionally, annual carbon storage was estimated to range from 4.6 to 8.7 Tg C yr\(^{-1}\) for tidal salt marshes, and to 52 Tg C yr\(^{-1}\) for mangrove ecosystems. Environmental and hydrologic conditions including salinity gradients and tidal regimes play a crucial role in the biogeochemistry cycling of carbon, methane and nitrous oxide in coastal wetlands. My review of methane and nitrous oxide production and emission indicated that tidal salt marshes and mangrove ecosystems are not a substantial source of these greenhouse gases. It is recommended that the protection and restoration of coastal salt marshes and mangroves should help to maintain their potential as natural carbon reservoirs and avoid becoming sources of atmospheric carbon. Wetland restoration efforts need of adequate policies, available funds, and social commitments. Financial aid obtained from the trading of wetland carbon credits could contribute to improve restoration activities and research projects in these ecosystems.
1. Introduction

Climate change is a global concern due to the increase of greenhouse gases including carbon dioxide (CO$_2$), methane (CH$_4$) and nitrous oxide (N$_2$O) that play a crucial role in global warming (Cowie 2007). The Intergovernmental Panel on Climate Change (IPCC) reported that the global atmospheric concentration of CO$_2$ has increased from a pre-industrial value of about 280 ppm to values of 379 ppm in 2005, and 391 ppm in 2011 (Solomon et al. 2007, Stocker et al. 2014). The National Oceanic and Atmospheric Administration (NOAA) reported that the annual average of CO$_2$ was 393.82 ppm in 2012 and 396.48 ppm in 2013$^1$. In addition, the global atmospheric concentration of CH$_4$ has increased from a pre-industrial value of about 715 ppb to 1774 ppb in 2005 and 1803 ppb in 2011. Moreover, the global atmospheric N$_2$O concentration increased from a pre-industrial value of about 270 ppb to 319 ppb in 2005 and 324 ppb in 2011 (Solomon et al. 2007, Stocker et al. 2014). Several mitigation measures have been considered to deal with this increase of greenhouse gases. One of them is the natural process performed by ecosystems such as forests, oceans, and wetlands. In recent years, researchers have focused their attention on the ability of wetlands to offset atmospheric CO$_2$ by storing it as carbon in plants and sediments (Zedler 2012).

Wetlands are dynamic and highly productive ecosystems characterized by aquatic and terrestrial components that provide a variety of ecological functions and services. The total area of wetlands worldwide is estimated to be around 7 x 10$^6$ to 9 x 10$^6$ km$^2$ and represents about 4 to 8% of the land surface area globally (Mitsch and Gosselink 2000, Mitsch et al. 2012). Wetlands provide ecological functions that include food web support associated with natural processes such as nutrient cycling, primary productivity, and nitrogen removal. Ecological services provided by wetlands to humans include carbon sequestration, soil and water quality filtration, flood prevention, climate change mitigation, and many others (Chabreck 1988, Keddy 2000, Richardson et al. 2001, Čížková et al. 2011, McKee et al. 2012). Economic values of wetlands include supply of natural goods such as grains, timber, fibers and fish. Moreover, wetlands offer recreational, cultural and educational opportunities for visitors and local communities (Mitsch and Gosselink 2000, LePage 2011, Zedler 2012).

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$^1$ Data from Mauna Loa Observatory provided by NOAA-ESRL in the United States, and presented in CO$_2$ Now.org website (http://co2now.org/) Accessed on 3/25/14.
The organization and types of wetlands depends on location in the landscape such as coastal or inland, water characteristics such as salt or freshwater, floods in the wetlands and marshes, and the types of dominant vegetation. Along the coastlines, there is a particular wetland type associated with estuaries connected to the ocean known as coastal wetlands (Chabreck 1988, Watson 2012). Coastal wetlands include tidal salt marshes in temperate climates, mangrove ecosystems in tropical climates, and tidal brackish and freshwater marshes in both. Figures 1 and 2 show a tidal salt and mangrove ecosystems. It is estimated that 3.2 x 10^6 ha of coastal wetlands are found in the United States of which almost 1.9 x 10^6 ha of the total are tidal salt marshes, and 0.5 x 10^6 ha are mangrove ecosystems (Mitsch and Gosselink 2000). Both fresh and saltwater wetlands are of ecological importance because they provide habitat for a variety of aquatic invertebrates, fish, and shorebirds. Moreover, coastal wetlands are an essential part of the Pacific Flyway where thousands of migratory birds stop to rest, feed and breed on their migratory route.

Figure 1 Mangrove ecosystem
Source: www.flmnh.ufl.edu
Coastal wetlands have been severely altered by filling, hydrological alterations, degradation of habitat, and external contamination primarily by human activities such as agricultural activities, housing developments, and urban and industrial runoff. These activities have decreased biodiversity by affecting their vital habitat and have disrupted ecological functions and services of coastal wetlands. Recently, restoration efforts have been implemented to recover their ecological functions and services such as improvement of water quality, reestablishment of vegetation, habitat, and carbon sequestration in wetlands (Steere et al. 2001, King et al. 2009, Palaima 2012).

The carbon sequestration function of coastal wetlands offers a potential to mitigate the increase of the atmospheric CO₂, which is associated with the increase of global warming. The

Figure 2 Tidal salt marsh ecosystem
Source: www.water.epa.gov
exchange of carbon in coastal wetlands is a complex process between wetland vegetation and soils. Vegetation assimilates atmospheric CO₂ by the photosynthesis process and stores it as organic carbon in plant tissues. Wetland ecosystems also have a capacity to store large quantities of carbon in sediments, aboveground and belowground organic matter, and peat deposits. The potential of wetlands as carbon sinks is affected by wetland characteristics such as the water level fluctuations, salinity, primary production and decomposition of organic matters, climatic conditions, microbial activities, and vegetation communities (Adhikari et al. 2009). In contrast to the carbon sequestration capacity, under anaerobic conditions wetlands release CH₄ and N₂O. Assessing the potential of wetlands to mitigate the increase in greenhouse gases comprises analyzing the net balance between carbon sequestration and emission of CH₄ and N₂O.

Carbon sequestration functioning performed by coastal wetlands can provide ecological opportunities in further wetland restoration efforts because of the capacity to sequester and store carbon. In addition, the developing of protocols for quantifying carbon sequestration in wetlands along with the feasibility of implementing a market for trading credits of wetland carbon could guarantee financial aid. These incomes can be used to protect, enhance and restore wetlands because of their importance as carbon reservoirs (Adhikari et al. 2009).

The goal of this paper was to analyze the carbon sequestration potential of tidal salt marshes in temperate climates and the mangrove ecosystems in tropical climates. A comparative analysis was used to evaluate and discuss the carbon estimates and rates including the exchange of carbon, methane and nitrous oxide from coastal wetland ecosystems. The following objectives were proposed to achieve the research’s goal. (a) Describe the carbon sequestration function in coastal wetlands including the factors and conditions that affect carbon sequestration. (b) Discuss factors and conditions that are associated with the production and emission of CH₄ and N₂O from coastal wetlands. (c) Present the estimates of the carbon sequestration rates and CH₄ and N₂O emission rates of tidal salt marshes and the mangrove ecosystems, and (d) Discuss an effective balance of fluxes of gases into the atmosphere focusing on carbon sequestration capacity of tidal salt marshes and the mangrove ecosystems.
2. **Methods**

I surveyed first data from the literature to study coastal wetland ecosystems and their characteristics, structure, and biotic and abiotic factors that have influence in ecosystem functioning. Additionally, I included summaries of papers and reports about the carbon sequestration function and estimates of carbon sequestration rates, sediment accumulation and net primary production in both wetland ecosystems in general, and in tidal salt marshes and mangrove ecosystems in particular. Moreover, I searched literature related to the production and emission of carbon, methane and nitrous oxide and fluxes estimates in coastal wetlands.

Fundamentally, I sought for information online at USF Gleeson library, and Google scholar, a search engine. I collected information on online databases such as Scopus, ScienceDirect, Willey Online Library, JSTOR, and EBSCOhost. All of them are large databases that provide peer-reviewed literature, scientific papers, and scholarly literature across many disciplines and sources, including theses, books, abstracts, reports, and articles. I also selected Mendeley, a free reference manager, to organize my literature review, references and citations.

Following my research proposal, I studied and elaborated a depth overview of the current knowledge about qualitative and quantitative biogeophysical aspects in tidal salt marshes and mangrove ecosystems. Using the best available data in the published literature, I reviewed, compiled, studied and compared the data relevant and the estimates of carbon sequestration, sediment burial, carbon pool, and biomass production in both tidal salt marshes and mangrove ecosystems. Moreover, I selected several papers containing tables and figures showing relevant data about carbon sequestration function in coastal wetland ecosystems. In addition, I also provided tables that contain the results of my analysis.

3. **Coastal wetland ecosystems and carbon sequestration function**

Located near coastline around the world, coastal wetlands include tidal salt marshes, tidal freshwater marshes and mangrove ecosystems (Figure 3). The periodic flow of tides and the salinity of the water characterize these wetland ecosystems. Approaching to the environmental conditions and abiotic and biotic factors of coastal wetlands will contribute to a
better understanding of wetland ecosystem processes and functioning such as the carbon sequestration function.

According to Mitsch and Gosselink (2000), wetland ecosystems comprise three main components: (1) hydrology characterized by water level, flow regime, and frequency and duration of inundation over time; (2) physicochemical environment that include sediments, reduced or oxidized soil conditions, and chemical reactions in the soil; and (3) biota characterized by plants, animals, and microbial organisms that are tolerant of flooding and wet conditions (Figure 4). These three components are interrelated, and affected directly by climate and basin geomorphology conditions. In addition, the physiochemical environment affects the hydrology. A biotic feedback also affects the physiochemical environment and the hydrology.

a. Abiotic and biotic factors associated to coastal wetland ecosystems

Abiotic and biotic factors such as temperature, salinity, water regimes, nutrients, sediments, soil composition, plants and animals, and primary productivity determine the
structure and function of coastal wetlands. The factors also are associated with the geophysical and chemical conditions that characterize these coastal ecosystems and their high productivity (Keddy 2000).

Describing the geomorphology of tidal wetland Rabenhorst (2001) mentioned three basic geomorphic types of tidal marshes. Estuarine marshes form in alluvial sediments deposited along tidally influenced rivers and streams. The mineral contain depend on the magnitude of erosion and deposition of mineral soil from upstream and the organic production within the marsh. Submerge coastal marshes form in marshes soils behind barrier island systems. Plants growth provides organic matter and the sandy soil provides the mineral sediments. Submerged upland marshes form in marsh soils overlying by rise in sea level over time. Here the O horizons are thinnest at the upland margin of the marsh (Figure 5) (Rabenhorst 2001).

Figure 4 Wetland ecosystem components.
Source: Mitsch and Gosselink (2000)
Mangrove geomorphologic settings are associated with waves, tides and rivers. There are five types of mangrove in according to the combination of these factors: river dominated, tide dominated, wave dominated barrier lagoon, composite river and wave dominated, and drowned bedrock valley (Figure 6) (Mitsch and Gosselink 2000). Tidal action has a crucial role in the hydrology in coastal wetland. Tidal is an important subsidy to salt marsh and mangrove that have influence in the physiographic, chemical and biological process. Based on marsh elevation and tidal regimen, salt marshes are divided into two main zones, the upper marsh (high marsh) that floods irregularly, and the intertidal lower marsh (low marsh) that floods daily (Figure 7). Tidal are associated to develop of tidal creeks and pannes, and sediments settings. Tidal creeks are developed into the lower marshes and serve to conduct compounds and nutrients between

Figure 5 Tidal marsh types classified according to geomorphologic settings
Source: Rabenhorst (2001)
marshes and their adjacent body of water. Also called sand barrens, pannes describe bare, exposed and water filled depressions in the marsh. Water flooding contributes to accumulate sediments into salt marsh systems (Mitsch and Gosselink 2000).

In accordance to their physical hydrologic conditions, mangroves ecosystems are classified in four types. (a) Fringe mangroves founded along protected shorelines and canals, rivers and lagoons. They accumulate organic materials due to the low tides and dense prop roots. (b) Riverine mangroves are high productive mangrove forest supporting tall mangrove trees of 16-23 m. They are found along the edges of coastal rivers and creeks. (c) Basin mangroves occur in inland depression or basins often behind fringe mangrove wetlands. They have high salinity soils with low redox potential because of the less frequent flushing by tides. Finally, (d) dwarf or scrub mangroves are isolated and low-productivity mangrove wetlands because of lack of nutrients and freshwater inflows. Scattered and small mangrove trees often less than 2 m tall dominate these systems (Mitsch and Gosselink 2000).
Salinity is a dominant abiotic factor in the productivity and the plant selection of salt tolerant species in the salt marsh and mangrove ecosystems. A wide range of salinity is influenced by seasons, tidal regimen and flooding, precipitations, zonation and elevations of wetlands, and freshwater inflows. For example, salt marshes salinities range from 10 to 60 ppt, in tidal creeks salinity range from 20 to 33 ppt, in fringe mangroves soil salinity ranges from 39 to 59 ppt, in riverine mangrove salinity ranges from 10 to 20 ppt, and in basin mangrove the soil salinity is more than 50 ppt. In addition, mangrove soils are often acidic because of the highly reduced conditions and the accumulation of reduced sulfides (Keddy 2000, Mitsch and Gosselink 2000).

In coastal wetlands, hydrologic conditions are strongly associated to salinity and tidal regimes that affect the wetland structure and function. Mitsch and Gosselink (2000) indicated that the tides act as a stress by causing submergence, saline soils, and soil anaerobiosis as well as a subsidy by removing excess salts, reestablish aerobic conditions, and providing nutrients. Base on elevation and the frequency of tidal inundation marshes are zoned as low marsh areas that are inundated frequently, middle marshes, or high marsh areas that are inundated less frequently (Rabenhorst 2001).

The accumulation of mineral sediments in marsh soils is associated to tidal regimens and sediment sources. Sandy sediments are transported a short distance in an estuary and added to marsh soils near from their sources because sand grain are large and require more transport
energy. However, silt and clay sediments are more easy transported and added to marsh soils at great distances from their origins because they are finer-textured particles (Rabenhorst 2001). Areas of lower elevation that are submerged more frequently have great potential for receiving sediment. However, the velocity of inundation may be affected due to the high sedimentation rate increases slightly the elevation influencing its energy and transport capacity (Rabenhorst 2001).

Hydrology has impacts on soil anaerobiosis condition, nutrient availability, primary production, organic accumulation, decomposition activity, as well as the type of biota, species composition and richness, which are hosted in wetlands (Mitsch and Gosselink 2000). Hydrologic and biogeochemical processes regulate exchanges of energy, water fluxes, and nutrients, which have a strong influence on wetland soils and functioning.

Wetland soils contain minerals and organic matters (OM) that are key elements in the productivity operate by microbes, animals and plants. Two main categories of soils are classified according to the materials that form them. Organic soils contain significant organic matter from animal and plant matter and nutrients decomposed by soil microorganisms such as bacteria and fungi. They are known as “peats” or “mucks”, and they are black, porous, and light in weight (Sprecher 2001). The accumulation of significant amounts of soil organic matter is the result of slow organic matter decomposition that is associated with the prolonged saturated and anaerobic conditions on wetland soils (Keddy 2000, Craft 2001, Kolka and Thompson 2006). Minerals soils contain different amounts of sand, silt, and clay from rocks and transported by wind, water and landslide (Sprecher 2001).

b. The biological structure of coastal wetland ecosystems

Biodiversity in coastal wetland ecosystems includes microorganisms, invertebrates, vegetation, vertebrates and fish. As a feature of these ecosystems, hydrology and biogeochemistry factors have influence in wetland biota (Figure 4).

In coastal wetlands, resident flora and fauna are associated to the saline environment such as halophytes plant that develop and complete their life cycles in high salinity concentrations (Mendelssohn and Batzer 2006). Hydrologic conditions associated to the
periodic flooding and hydrodynamics also are crucial determinants of the structure of species composition, species richness, and productivity of wetland plant communities.

Salt marsh plants adapt to survival under submerged conditions. The distribution of vegetation communities also are related to elevation gradients, frequency of tidal inundation and salinity gradients (Rabenhorst 2001). The periodic inundation of salt water brings oxygen, nutrients, sediments, and ionic solutes such as K, Ca, Mg, and S, which are required by plants (Rabenhorst 2001, Sharitz and Pennings 2006). However, the density and structure of marsh vegetation may affect the movement of floodwater across the marshes and the accumulation of mineral sediments (Rabenhorst 2001).

Herbaceous emergent vegetation such as the *Spartina* species is dominant in salt marshes and woody forest trees are dominant in mangrove ecosystem located in tropical areas (Craft 2001, Chmura 2009). The diversity of plant communities is modest - about ten to twenty species in salt marshes. They must cope with the stress of flooding and salinity (Keddy 2000). Salt marshes exhibit unique plant zonation based on hydrology and salinity levels. The tall form of *Spartina alterniflora* Loisel (smooth cordgrass) dominates low marshes. The short form of *S. alterniflora*, *Salicornia* spp. and *Distichlis* spp. are in the middle marshes. High marshes include *S. patens* (saltmeadow cordgrass), *Distichlis spicata* (spikegrass), *Iva frutescens* (marsh elder), *Juncus gerardi* (blackgrass), *J. roemerianus* (black rush) *Limonium carolinianum* (sea lavender), and other genera such as *Arthrocnemum*, *Artriplex*, and *Suaeda* (Mitsch and Gosselink 2000).

The stress of flooding soils and salinity conduce to have a simple flora in mangrove wetlands. Around the world, there are about 50 species of mangrove classified in 12 genera, and 10 of them are in the Americas. In south Florida, characteristic species dominates the hydrologic types of mangrove wetlands. Fringe mangroves are dominated by red mangrove (*Rhizophora mangle*) that contains abundant prop roots. Numerous red mangroves, black mangrove (*Avicennia germinans*) and white mangrove (*Laguncularia racemosa*) dominate riverine mangrove wetlands. Basin wetlands host all of three species being black mangroves the most common. The mangrove fern (*Acrostichum* spp.) is found as understory genus in many wetlands. The non-mangrove specie Buttonwood (*Conocarpus erecta*) is associated with mangrove habitats. (Mitsch and Gosselink 2000, Craft 2001).
Fauna diversity takes an essential part in the productivity and functioning of wetland coastal ecosystems. Benthic microalgal communities found in salt marshes are functional component of these ecosystems (Chmura 2009). The algal flora includes diatoms of the genera Navicula, Nitzschia and Amphora; cyanobacteria of the genera Chroococcus, Anacystis, Anabaena, Nostoc, and others; and yellow and green algae of the genera Euglena, Rhizoclonium, and Enteromopha (Mitsch and Gosselink 2000).

Bacteria and fungi aid in both aerobic and anaerobic decomposition of organic matter and produces physical and chemical characteristics unique to hydric soils. Bacteria are primarily agents for organic matter decomposition in anaerobic conditions. Fungi are the dominant agents of decomposition in aerobic conditions (Craft 2001). Decomposition in aerobic conditions uses oxygen (O₂) as an electron acceptor to convert organic carbon to CO₂. Decomposition in anaerobic conditions occurs in inundated or saturated wetland soils using oxidized iron (Fe³⁺) and manganese (Mn⁴⁺), nitrate (NO₃⁻) and sulfate (SO₄²⁻) as electron acceptors (Craft 2001).

Decomposition process is more slowly in anaerobic and oxygen limited conditions than aerobic decomposition. Temperature, pH, organic matter quality, nutrients, and the availability of electron acceptors are other factors that regulate organic matter by microorganisms in wetland soils. Craft (2001) believes that microorganisms have different tolerance to temperature but microbial activity and decomposition increase with temperature ranges of 35 to 40°C. An optimum pH and decomposition activity in most bacteria occur in the pH range of 6 to 8 and for fungi is in the pH 4 to 6.

In addition to microbial communities, a variety of animal enhances wetland productivity and functioning. Invertebrate communities are important for sustaining the significant secondary productivity of salt marshes and mangroves. High densities and diversity microflora of invertebrates living on soil salt marshes are associated to the highest secondary production of the ecosystems (Craft 2001, Chmura 2009).

Invertebrates promote growth of salt marsh plants by acting as ecosystem engineers reducing physical stress or increasing nutrients availability (Sharitz and Pennings 2006). Animals including detritus-feeding invertebrates, larval and adult insects and parasitic plants,
amphibians and reptiles, and herbivores such as mammals, birds, crabs, and snails can have a strong effect on plant productivity and species composition (Mitsch and Gosselink 2000, Craft 2001, Sharitz and Pennings 2006, Chmura 2009).

Soil invertebrates found in mangrove forests include nematodes, polychaete worms, and fiddler crabs. Surface-dwelling invertebrates include amphipods, harpacticoid copepods, and bivalves. The aerial or prop roots of mangroves that provide support to mangrove trees are important faunal habitat for barnacles, fiddler crabs, and other filter feeders that attach themselves to the prop roots (Mitsch and Gosselink 2000, Craft 2001). Table 1 shows vegetation and invertebrates found in salt marshes and mangroves.

| Table 1 Main species of plants and invertebrates in coastal wetlands |
| Sources: Data from Mitsch and Gosselink (2000) and Craft (2001) |

<table>
<thead>
<tr>
<th>Vegetation</th>
<th>Invertebrates</th>
</tr>
</thead>
<tbody>
<tr>
<td>Meiofauna (organism 63 to 500 μm in diameter)</td>
<td>Macrofauna (organisms greater than 500μm in diameter)</td>
</tr>
<tr>
<td><strong>Salt Marshes</strong></td>
<td>Oligochaete worms, fiddler crabs, mud crabs (<em>Sesarma</em> spp.) periwinkle (gastropod), snails, ribbed mussel, oyster (<em>Crassostrea virginica</em>), hard clam (<em>Mercenaria mecanaria</em>)</td>
</tr>
<tr>
<td><strong>Mangrove forest</strong></td>
<td></td>
</tr>
<tr>
<td>Red (<em>Rhizophora mangle</em>), black (<em>Avicennia germinans</em>) and white (<em>Laguncularia racemosa</em>) mangroves</td>
<td>Nematodes, amphipods, copepods, polychaete worms</td>
</tr>
<tr>
<td>Fiddler crabs, oysters, barnacles (<em>Balanus</em> spp.)</td>
<td></td>
</tr>
</tbody>
</table>

c. Photosynthesis and primary production in coastal wetland ecosystems

The global carbon cycle is the biogeochemical pathway that comprises a series of exchange events among the geosphere, biosphere, hydrosphere, and atmosphere of the Earth. The carbon cycle describes the movement, recycled and reused of carbon making possible to sustain the life on the Earth. Wetland ecosystems play a crucial role in the global carbon cycle
by acting as carbon pools due to their carbon sequestration function. It is estimated the Earth’s soil pool in 2,500 Pg C from this about 20 to 30 % is stored in wetlands (Mitsch et al. 2012). Wetland is associated to production and accumulation of soil organic matter (Figure 8).

In coastal wetlands, the vegetation adapted to saline habitat and tidal regimes takes CO₂ from the atmosphere and converted it in organic carbon compounds through the process of photosynthesis. Then, the organic compounds are stored as energy in plants and biomass in wetlands. Some carbon also returns to the atmosphere as CO₂ and CH₄ by respiration activities (McLeod et al. 2011).

![Figure 8 Wetland carbon cycle](source: Collins and Kuehl (2001))

Several factors such as temperature, water availability and saline conditions, transpiration rate, and light intensity affect the photosynthesis process in wetland plants. According to Mitsch and Gosselink (2000), saline habitats are stressful environments for wetland plant species. An important adaptation for that condition is the C₄ biochemical pathway of photosynthesis developed by plants. The first product of C₄ plant CO₂ fixation is a four-carbon compound. There are also C₃ plants, which produce a three-carbon compound from CO₂ incorporation. C₄ plants are more effective at CO₂ fixation than most C₃ plants because of their ability to fix CO₂ in the dark and to remove CO₂ from the air until
concentrations fall below 20 ppm compared to 30 to 80 ppm in C\textsubscript{3} plants. C\textsubscript{4} plants are also more efficient because they have low photorespiration rates, operate efficiently even in the most intense sunlight, have low rates of carbon fixation, and require low amounts of water per unit of carbon fixed (Mitsch and Gosselink 2000).

A comparison of photosynthesis attributes of two salt marsh species, a C\textsubscript{4} plant \textit{Spartina alterniflora} and a C\textsubscript{3} plant \textit{Juncus roemerianus}, showed that the C\textsubscript{4} plant has a higher rate of photosynthesis, and a lower CO\textsubscript{2} concentration in the leaf when photosynthesis is reduced to zero. In addition, the C\textsubscript{4} plant has a lower respiration rate in the light, a higher temperature optimum, and two times more water efficiency than C\textsubscript{3} plant (Table 2) (Mitsch and Gosselink 2000).

\begin{table}[h]
\centering
\begin{tabular}{|l|c|c|}
\hline
Photosynthetic Characteristics & \textit{Spartina alterniflora} & \textit{Juncus roemerianus} \\
\hline
Maximum seasonal net photosynthetic rate [mg CO\textsubscript{2}/cm\textsuperscript{2} s\textsuperscript{-1} (month)] & 90 (Sept.) & 65 (July) & 60 (March) \\
\hline
Photosynthetic light response (Fig. 7-8) & Nonsaturating & Saturating & Nonsaturating \\
\hline
CO\textsubscript{2} compensation concentration (ppm) & 12 & 84 & 84 \\
\hline
Photorespiration at 21\% O\textsubscript{2} (mg CO\textsubscript{2} cm\textsuperscript{-2} s\textsuperscript{-1}) & 6.7 & 18.2 & 9.1 \\
\hline
(% of photosynthesis) & (11) & (40) & (54) \\
\hline
Temperature optimum (summer) (\degree C) & 30–35 & 30–35 & 25 \\
\hline
Water use efficiency (mg CO\textsubscript{2}/g H\textsubscript{2}O) & 15 & 12–15 & 8–9 \\
\hline
\end{tabular}
\caption{Comparison of C\textsubscript{4} plants and C\textsubscript{3} plants.}
\label{table:photosynthesis}
\end{table}

Source: Mitsch and Gosselink (2000)

Coastal wetland ecosystems are considered to be among the most productive ecosystems in the world due to their tidal regimes (Mitsch and Gosselink 2000, Craft 2001, Sharitz and Pennings 2006, Bouillon et al. 2008). It is estimated that tidal marshes produce annually around 80 metric tons per hectare of plant material (8,000 g m\textsuperscript{-2} yr\textsuperscript{-1}) in the southern
Coastal Plan of North America (Mitsch and Gosselink 2000). The marsh grasses, the mud algae and the phytoplankton of the tidal creeks contribute to the high coastal wetland productivity.

Net primary productivity (NPP) and decomposition process are regulated by hydroperiod or duration of inundation through controlling oxygen availability for aerobic respiration and nutrient inputs from floodwaters. The input of nutrients and sediments coming from both freshwater and tidal creeks and sloughs improves soil conditions and increases plant growth. Nutrient availability such as nitrogen (N), phosphorus (P), iron (Fe), calcium (Ca) can enhance or limit NPP and decomposition. Abiotic stressors such as salinity, hydrogen sulfide (H$_2$S), acidity, and factors such as light, air and soil temperature also can regulated productivity and decomposition in coastal systems (Craft 2001).

Length of the growing season and the intense photosynthesis processes enhancing plant growth are another conditions controlling or contributing to high marsh productivity. Salt tidal marshes and mangrove forest are among the most productive ecosystems. Comparing the average annual aboveground net primary productivity in various freshwater and estuarine wetlands, the NPP of salt marshes ranges approximately from 1000 to 2000 g m$^{-2}$ yr$^{-1}$ and the NPP of mangrove systems ranges approximately from 400 to 1200 g m$^{-2}$ yr$^{-1}$ (Figure 9).

![Figure 9](image_url)

*Figure 9 Aboveground net primary productivity (ANPP) in wetland ecosystems*

Source Sharitz and Penning (2006)
Sharitz and Penning (2006) cited that estimates of the aboveground annual net primary productivity (ANPP). In Louisiana marshes, the estimates of salt marsh cordgrass (*Spartina alterniflora*) ranges from 1381 g m$^{-2}$ yr$^{-1}$ (by Hopkinson et al. 1980) to 2,895 g m$^{-2}$ yr$^{-1}$ (by White et al. 1980). The estimates of the belowground production of *S. alterniflora* vary from 220 to 2,500 g m$^{-2}$ yr$^{-1}$ adjacent to tidal flow to 420 – 6,200 g m$^{-2}$ yr$^{-1}$ on the marsh flat (by Good et al. 1982). The high aboveground annual net primary productivity of coastal mangrove forest depends on the hydrologic conditions and landscape positions (Bouillon et al. 2009). In Mexico, Day et al. (1996) estimated that the ANPP in riverine mangrove wetlands are higher in about 2,458 g m$^{-2}$ yr$^{-1}$ in than the fringe mangrove estimated in 1,607 g m$^{-2}$ yr$^{-1}$, and basin mangrove wetlands calculated about 399 to 695 g m$^{-2}$ yr$^{-1}$ (cited by Sharitz and Penning 2006). In addition, algae in salt marsh provide a significant amount of the primary production. The estimates of annual benthic microalgal production range from 10 to 60% of vascular plant productivity in the Atlantic and Gulf of Mexico coasts to 75 to 140% of vascular plant production in southern California (Sharitz and Pennings 2006, Chmura 2009).

4. **Carbon balance in coastal wetland ecosystems**

The carbon budget refers to the measurement of carbon fluxes between coastal wetland systems and the atmosphere. Coastal wetlands have the ability to sequestrate significant amounts of carbon from the atmosphere. Carbon is fixed into vegetation and is stored in the form of bulky organic matter, peats and sediment in wetland soils. Climate and geomorphology conditions affect the amounts of organic carbon accumulated in the above ground biomass and below ground biomass in coastal wetlands. Carbon also is released back to atmosphere by respiration activities of plants and animals. Figure 10 shows the main process and spot of carbon sequestration function in coastal wetlands.

The carbon balance in coastal wetlands comprises the estimates of carbon sequestration rates, the net primary productivity and sedimentation rates. In addition, the research of carbon balance should contribute to understand the sequestration potential of atmospheric CO$_2$ and the carbon sink capacity of coastal wetlands.
a. Carbon sequestration rate in tidal salt marshes and mangrove ecosystems

Research has been conducted worldwide to quantify carbon sequestration rates and soil accretion in coastal wetland ecosystems and soil carbon pools. The literature review shows that researches had reported differences in their estimations of carbon sequestration on wetlands systems, as well as on tidal marshes and mangrove ecosystems\(^2\). Estimations of soil carbon may consider the wetland areas because the carbon density in soil is associated to volume and spatial extensions of the ecosystems. The depth of carbon in soil also is a crucial component in calculations of volume of carbon storage on ecosystems. Even when there are no precise data of the extent of tidal marshes and mangroves areas, researchers have used different estimates of ecosystem areas in their calculations (Siikamäki et al. 2012).

\(^2\) Units for carbon are expressed in Megagram (Mg) = \(10^6\) g; Teragram (Tg) = \(10^{12}\) g; Petagram (Pg) = \(10^{15}\) g
Bridgham et al. (2006) estimated the global carbon sequestration rate for wetlands in 137 Tg C yr\(^{-1}\). They also estimated the North American carbon sequestration rate for wetlands in 57.2 Tg C yr\(^{-1}\); and for the USA, the carbon sequestration rate for wetlands was estimated in 17.3 Tg C yr\(^{-1}\) (Table 3).

Chmura et al. (2003) calculated the average annual of the global carbon sequestration by tidal saline wetlands in 42.6 ± 4.0 Tg C yr\(^{-1}\), using the documented value of 203 x 10\(^3\) km\(^2\) of global wetland area. Duarte et al. (2005) estimated the carbon sequestration rate in 60.4 Tg C yr\(^{-1}\) for a global area of tidal marshes ranges from 220 to 400 x 10\(^3\) km\(^2\) (Table 4).

Bridgham et al. (2006) also estimated the carbon sequestration rates in 4.6 Tg C yr\(^{-1}\) for global tidal marshes, 4.8 Tg C yr\(^{-1}\) for North American tidal marshes, and 4.4 Tg C yr\(^{-1}\) for USA tidal marshes (Table 3). Reports and papers published in recent years estimated the carbon sequestration for global tidal salt marshes from 5 to 87 Tg C yr\(^{-1}\) (Nellemann et al. 2009, Mcleod

Table 3 Global, North American, and USA carbon estimates on tidal marshes and mangroves

<table>
<thead>
<tr>
<th></th>
<th>Wetland area (x 10(^3) km(^2))</th>
<th>Carbon sequestration (Tg C yr(^{-1}))</th>
<th>Soil carbon pool (Pg C)</th>
<th>Plant carbon pool (Pg C)</th>
<th>Total carbon pool (Soil+plants) (Pg C)</th>
<th>Net carbon balance (Tg C yr(^{-1}))</th>
<th>CH(_4) flux from wetlands (Tg CH(_4) yr(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Global</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tidal Marsh</td>
<td>22</td>
<td>4.6</td>
<td>0.43</td>
<td>0.007</td>
<td>0.437</td>
<td>4.6</td>
<td>0.028</td>
</tr>
<tr>
<td>Mangrove</td>
<td>181</td>
<td>38</td>
<td>4.9</td>
<td>4.0</td>
<td>8.9</td>
<td>38</td>
<td>0.20</td>
</tr>
<tr>
<td>Total wetlands</td>
<td>5,961</td>
<td>137</td>
<td>513</td>
<td>15.5</td>
<td>528.3</td>
<td>-68</td>
<td>105</td>
</tr>
<tr>
<td><strong>North American</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tidal Marsh</td>
<td>22</td>
<td>4.8</td>
<td>0.44</td>
<td>0.037</td>
<td>0.477</td>
<td>4.8</td>
<td>0.028</td>
</tr>
<tr>
<td>Mangrove</td>
<td>8</td>
<td>2.1</td>
<td>0.19</td>
<td>0.074</td>
<td>0.264</td>
<td>2.1</td>
<td>0.011</td>
</tr>
<tr>
<td>Total wetlands</td>
<td>2,463</td>
<td>57.2</td>
<td>215</td>
<td>4.9</td>
<td>219.9</td>
<td>49.2</td>
<td>9.4</td>
</tr>
<tr>
<td><strong>USA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Tidal Marsh</td>
<td>20</td>
<td>4.4</td>
<td>0.40</td>
<td>0.034</td>
<td>0.434</td>
<td>4.4</td>
<td>0.026</td>
</tr>
<tr>
<td>Mangrove</td>
<td>3</td>
<td>0.50</td>
<td>0.061</td>
<td>0.024</td>
<td>0.085</td>
<td>0.5</td>
<td>0.004</td>
</tr>
<tr>
<td>Total wetlands</td>
<td>431</td>
<td>17.3</td>
<td>19.6</td>
<td>1.5</td>
<td>21.1</td>
<td>9.5</td>
<td>3.1</td>
</tr>
</tbody>
</table>

- Total wetlands includes permafrost peatlands, non-permafrost peatlands, freshwater mineral soil, tidal marsh, mangrove, and mudflat
- Pg = Petagram = 10\(^{15}\) g; Tg = Teragram = 10\(^{12}\) g
- Source: Data from Bridgham et al. (2006)
Estimates of carbon sequestration globally by mangroves range from 16 to 52 Tg C yr\(^{-1}\) (Table 5) (Siikamäki et al. 2012). Twilley et al. (1992) estimated the carbon sequestration in 24 Tg C yr\(^{-1}\) for mangrove. Duarte et al. (2005) estimated the global carbon sequestration for mangrove in a range from 23.6 to 28 Tg C yr\(^{-1}\). Other recent estimates of carbon sequestration for mangrove system globally range from 18.4 to 34 Tg C yr\(^{-1}\) (Bouillon et al. 2008, Nellemann et al. 2009, Mcleod et al. 2011).

### Table 4 Global estimates of carbon sequestration in tidal salt marshes

<table>
<thead>
<tr>
<th>Data sources</th>
<th>Area of wetland (x 10^3 km(^2))</th>
<th>Carbon sequestration (Tg C yr(^{-1}))</th>
<th>Long-term rate of carbon accumulation in sediment (g C m(^{-2}) yr(^{-1}))</th>
<th>Soil carbon density (g C cm(^{-3}))</th>
<th>Carbon pool in soil (Tg C)</th>
<th>Carbon pool in biomass (Tg C)</th>
<th>Total carbon pool (Tg C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chmura et al. (2003)</td>
<td>203</td>
<td>42 ± 4.0</td>
<td>210</td>
<td>0.089 ± 0.003</td>
<td>430 ± 30</td>
<td></td>
<td>430 ± 30</td>
</tr>
<tr>
<td>Duarte et al. (2009)</td>
<td>220 – 400</td>
<td>60.4</td>
<td>151</td>
<td></td>
<td></td>
<td></td>
<td>437</td>
</tr>
<tr>
<td>Bridgham et al. (2006)</td>
<td>220</td>
<td>4.6</td>
<td>430</td>
<td>7</td>
<td>437</td>
<td></td>
<td>437</td>
</tr>
<tr>
<td>Laffoley et al. (2009)</td>
<td></td>
<td></td>
<td>210</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nellemann et al. (2009)</td>
<td>400 – 800</td>
<td>60.4 – 70</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mcleod et al. (2011)</td>
<td>220 – 400</td>
<td>5.87</td>
<td>218 ± 24</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Siikamäki et al. (2012)</td>
<td>51</td>
<td>11</td>
<td></td>
<td>190</td>
<td>17.20</td>
<td>210</td>
<td></td>
</tr>
</tbody>
</table>

**Tg = Tegaram = 10^12 g**

Bridgham et al. (2006) estimated the global wetland carbon sequestration rates in 38 Tg C yr\(^{-1}\) for mangroves, 2.1 Tg C yr\(^{-1}\) for North American mangroves, and 0.50 Tg C yr\(^{-1}\) for USA mangrove systems (Table 3). Furthermore, North American estuarine wetlands dominated by herbaceous vegetation (tidal marshes), mangroves, and unvegetated (mud flats) have a sequester fate of 10.2 Tg C yr\(^{-1}\). This estimate represents about 21% of the North American total for all wetland types even though they comprise only a small area of 45 x 10^3 km\(^2\) compared to 2,463 x 10^3 km\(^2\) of the North American total for all types of wetland. This carbon sequestration rate is about 10 times higher on an area than other wetland systems because of high sedimentation rates and soil carbon content, and constant burial due to the rise in sea level (Bridgham et al. 2006).
b. Carbon accumulation rate in tidal salt marshes and mangrove ecosystems

Different from non-tidal systems, soil carbon storage in tidal wetlands occurs through burial of organic matter driven by the rise in sea level. Thus, coastal wetlands intermittently sequester and then store carbon in their soils. This feature enhances the value of tidal marshes and mangroves as natural carbon sinks. The carbon accumulation rates in soils are estimated using data from changes in soil surface elevation and carbon density. Mitsch et al. (2010) reported that the global carbon accumulation rates of boreal, temperate, and tropical wetlands ranges from 8 to 480 g C m\(^{-2}\) yr\(^{-1}\). This range may be due to different techniques used for the estimations. Chmura et al. (2003) estimated the average of global carbon accumulation rate in sediments for tidal salt marshes to be 210 ± 20 g CO\(_2\) m\(^{-2}\) yr\(^{-1}\). They indicated that this value is greater compared to the carbon accumulation by peatlands that are 20–30 g CO\(_2\) m\(^{-2}\) yr\(^{-1}\) reported by Roulet (2000). Other long-term rate of carbon accumulation in sediment reported in the literature ranges from 151 to 218 ± 24 g C m\(^{-2}\) yr\(^{-1}\) (Duarte et al. 2005, Laffoley and Grimsditch 2009, Mcleod et al. 2011).
Methods such as $^{137}$Cs dating and $^{210}$Pb dating are used to assess accumulation above a clay marker horizon, which combine vertical soil accretion rates over periods ranging from 1–100 years (Bridgham et al. 2006). Using this method, Bridgham et al. (2006) calculated the rates of North American estuarine soil carbon accumulation to be $3.3 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ for Mexican mangroves; $1.8 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ for U.S. mangroves; $2.2 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ for tidal marshes in the U.S.; and $2.1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ for tidal marshes in Canada and Alaska. Other estimates reported in the literature calculated $190 \pm 40 \text{ g C m}^{-2} \text{ yr}^{-1}$ for North American salt marshes, $240 \pm 30 \text{ 40 g C m}^{-2} \text{ yr}^{-1}$ for North American brackish marshes, and $86 – 387 \text{ 40 g C m}^{-2} \text{ yr}^{-1}$ for Florida Everglades (Mitsch et al. 2012).

Globally estimates of long-term rate of carbon accumulation in mangrove sediments ranges from 139 to $266 \pm 39 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Duarte et al. 2005, Mcleod et al. 2011). Choi and Wang (2004) indicated that in coastal wetlands of northwest Florida the short-term carbon accumulation rates of the surface peat ranges from 42 to $193 \text{ g C m}^{-2} \text{ yr}^{-1}$ in the low marsh; from $18 \text{ to } 184 \text{ g C m}^{-2} \text{ yr}^{-1}$ in middle marsh; and from approximately 50 to $181 \text{ g C m}^{-2} \text{ yr}^{-1}$ in high marsh. The slow and continuous decomposition of organic matter in the peat over time explains the higher current rates of carbon accumulation in comparison with the longer-term rates (Choi and Wang 2004). Estimations of carbon stock using the carbon inventory and the radioactive decay of radiocarbon ($^{14}$C) in peat soils of a coastal wetland in northwest Florida indicated that the long-term carbon accumulation rates have decreased with time from $130 \pm 9 \text{ g C m}^{-2} \text{ yr}^{-1}$ over the last century to $13 \pm 2 \text{ g C m}^{-2} \text{ yr}^{-1}$ over the millennium timescale (Choi and Wang 2004).

c. Carbon pool in tidal salt marshes and mangrove ecosystems

Chmura et al. (2003) estimated the amount of carbon stored globally in soils of 154 sites of mangroves and salt marshes from the western and eastern Atlantic and Pacific coasts, the Indian Ocean, Mediterranean Ocean, and Gulf of Mexico. They indicated that the average of soil carbon density for mangrove ecosystems was estimated as $0.055 \pm 0.004 \text{ g cm}^{-3}$. This value was higher than the average for salt marshes estimated as $0.039 \pm 0.003 \text{ g cm}^{-3}$. The consistently warmer temperatures and higher rainfall characteristic of the tropical climates in which mangroves ecosystems are located can explain their higher productivity.
Bridgham et al. (2006) estimated the global carbon pool as 528.5 Pg C in wetlands (Table 3). Globally estimates of carbon pool in tidal marsh soils ranges from 430 to 1190 Tg C (Bridgham et al. 2006, Siikamäki et al. 2012). The carbon pool in tidal marsh biomass ranges from 7 Tg C (Bridgham et al. 2006) to 20 Tg C (Siikamäki et al. 2012). Moreover, globally the total carbon pool in tidal marshes was estimated from 437 Tg C (Bridgham et al. 2006) to 2010 Tg C (Siikamäki et al. 2012).

In North American, the total carbon pool was estimated as 219.9 Pg C in wetlands. Moreover, in North American tidal marshes, the total carbon pool was estimated as 0.477 Pg C, of them 0.037 was in plants and 0.44 was in soil carbon pool. For the conterminous USA, the total carbon pool in wetlands was estimated as 21.1 Pg C. Also, for USA tidal marshes, the total carbon pool was estimated as 0.434 Pg, of them, 0.034 Pg C was in plants and 0.40 Pg C was in soils (Table 3) (Bridgham et al. 2006).

Furthermore, the annual amount of soil carbon sequestration based on existing soil vertical accretion and carbon sink data in Louisiana coastal wetlands were calculated to be 2.96 x 10^6 metric tons of carbon. This value of the carbon sequestered considers that 300 g C m^-2 yr^-1 in the marsh soil at an average accretion rate of 1 cm yr^-1, which is multiplying by the total of 988,888 ha of Louisiana marshes (DeLaune and White 2012).

Mangrove vegetation also are valuable sinks of carbon due to the biogeochemical activities, higher rates of net primary productivity, biomass accumulation, and carbon in soils that depend on climatic and hydrologic factors (Bouillion et al. 2008). Twilley et al. (1992) estimated that the net ecosystem production in mangroves globally in about 180 Tg C yr^-1 using a total areal extent of mangrove of 24.00 x 10^6 ha. From this value, the carbon accumulated in mangrove sediments was estimated as 20 Tg C yr^-1, and total wood production for mangroves was as 160 Tg C yr^-1 (Twilley et al. 1992). A reassessment of global mangrove primary production estimated at about 218 ± 72 Tg C yr^-1 using an areal extent of mangrove of 160,000 km^2. The wood production was estimated at about 66.7 ± 39.6 Tg C yr^-1 (Bouillion et al. 2008).

Globally the estimates of carbon pool in mangrove soils ranges from 20 Tg C (Twilley et al. 1992) to 4900 Tg C (Bridgham et al. 2006). Carbon pool in mangrove soils for North American was estimated as 0.19 Pg C, and for USA was estimated as 0.0.061 Pg C. Estimates related to
carbon pool in mangrove biomass ranges from 1220 Tg C (Laffoley and Grimsditch 2009) to 4980 Tg C (Chmura et al. 2003). Twilley et al. (1992) reported the global estimate of carbon stored in mangrove biomass based on 0.45 g C/g dry as about 4.03 Pg C, of this, 2.34 Pg C was estimated in aboveground biomass, and 1.69 Pg C in belowground biomass. Bridgham et al. (2006) estimated the globally carbon pool in mangrove vegetation as 4.0 Pg C. For North American, the carbon pool in biomass was estimated as 0.074 Pg C, and as 0.024 Pg C for USA in mangrove vegetation (Table 3). Figure 11 shows global averages of salt marshes and mangroves ecosystems.

Globally the total carbon pool including soil and biomass carbon storage for mangrove systems were estimated as 5000 ± 400 Tg C (Chmura et al. 2003), 8900 Tg C (Bridgham et al. 2006), and 6510 Tg C (Siikamäki et al. 2012). Moreover, the total carbon pool for North American mangrove systems was estimated as 0.264 Pg C, and for the USA was estimated as 0.085 Pg C (Bridgham et al. 2006).

![Figure 11 Global averages of carbon pools of coastal habitats](image)

**Figure 11 Global averages of carbon pools of coastal habitats**

Tropical forests are included for comparison

Sources: Murray et al. (2011)

d. Net carbon in tidal salt marshes and mangrove ecosystems

Net carbon is associated to the difference between carbons absorbed and emitted by an ecosystem. Coastal wetlands could be a source of carbon when emissions exceed
sequestration. Thus, tidal salt marshes or mangrove ecosystems are a carbon sink when sequestration is greater than emissions. Organic carbon exported from coastal wetland ecosystems to the ocean and adjacent environments as well as the estimates of respiration processes may take into account into the global carbon balance. The global estimate of respiration in mangrove systems was 373 Tg C yr$^{-1}$, and 804 Tg C yr$^{-1}$ for salt marshes. Subtracting these values from the gross primary production, the global net ecosystem production were 44 Tg C yr$^{-1}$ for mangrove, and 624 Tg C yr$^{-1}$ for salt marshes (Duarte et al. 2005).

Bridgham et al. (2006) calculated the net carbon balance for wetlands as the sum of sequestration in current wetlands, oxidation in former wetlands, and plant carbon sequestration. Oxidation in former wetlands refers to emissions from wetlands that have been converted to non-wetland uses or conversion among wetland types due to human influence. The uncertainty about this value is greater because there are not data available. Therefore, they estimated the net carbon balance for North American wetlands as 49.2 Tg C yr$^{-1}$, and as 9.5 Tg C yr$^{-1}$ for USA wetlands (Table 3).

Bouillon et al. (2008) present a comprehensive synthesis of carbon fluxes in mangrove ecosystems. The global mangrove net primary production that comprises litter fall, wood, and root production was estimated to be approximately 218 ± 72 Tg C a$^{-1}$. Comparing this global primary production with the best estimates of various carbon sinks such as organic carbon export, sediment burial, and mineralization, more of 50% of the carbon fixed by mangrove vegetation is unaccounted for. As is shown in the figure 12 the estimation of the unaccounted carbon sink is approximately 112 ± 85 Tg C a$^{-1}$, which is equivalent in magnitude to 30–40% of the global riverine organic carbon input to the coastal zone (Bouillon et al. 2008).

In conclusion, the above estimates suggest that a significant amount of atmospheric carbon is stored in coastal wetland ecosystems, with higher carbon accumulation in tropical climates where mangrove systems are located. The total estimate of the carbon budget in coastal wetlands must include the production and flow of gases into the atmosphere such as the CO$_2$ respired by biota, the CH$_4$ and N$_2$O produced by methanogenesis and oxidation processes.
5. **Methane and nitrous oxide emissions in coastal wetland ecosystems**

Coastal wetland ecosystems are sources of two greenhouse gases responsible for global warming CH₄ and N₂O. The availability of the carbon produced and stored into wetland soil and sediments is the main source to the production of CH₄ and N₂O. For this reason, the CH₄ and N₂O fluxes from wetland ecosystems were included in this study relate to the carbon sequestration capacity (Keddy 2000).
a. Methane production and emission in tidal salt marshes and mangrove ecosystems

Wetlands represent significant natural sources of CH\textsubscript{4}. It is calculated that wetlands release annually between 20 to 40% of the total global CH\textsubscript{4} emissions. The estimates of 54–72% of the total global CH\textsubscript{4} emission are calculated from the anthropogenic sources including natural emissions from livestock and farming activities, biomass burning, landfills and other waste management, and fossil fuel production (Kang et al. 2012, Bridgham et al. 2013). Thus, figure 13 shows the global estimates from anthropogenic sources (including rice fields) and natural sources (including freshwater and wetlands). The estimates of rice, freshwater and wetlands also are presented separately. The estimates of wetland emission range from 80 to 280 Tg CH\textsubscript{4} yr\textsuperscript{-1}, with a median of 164 Tg CH\textsubscript{4} yr\textsuperscript{-1} (Bridgham et al. 2013).

![Figure 13 Global methane sources.](image)

Source: Bridgham et al. (2013) Horizontal lines are the median for each category.

The amount of CH\textsubscript{4} emitted is the balance between CH\textsubscript{4} production, oxidation, and transportation. CH\textsubscript{4} production is a methanogenesis process mediated by methanogens in anaerobic conditions. CH\textsubscript{4} oxidation is methanotrophy process conducted by methanotrophic
bacteria (MOB) under aerobic conditions. Vegetation conducts CH4 transportation within the soil and from the wetlands to the atmosphere (Kang et al. 2012). Figure 14 shows the CH4 cycling in wetland ecosystems. White boxes show the pools of carbon. Solid arrows show the progressive mineralization of these carbon pools by the identified microbial processes or groups. Dotted lines indicate carbon inputs from the plant community and dashed lines represent the flux of the gaseous end-products of these processes (CH4 and CO2) into the atmosphere.

Figure 14 Methane cycling in wetland ecosystems.
Source: Bridgham et al. (2013).
Plants conduct O\textsubscript{2} form the atmosphere to the soil facilitating the CH\textsubscript{4} oxidation, and regulate CH\textsubscript{4} transport from the soil to the atmosphere via diffusion and ebullition by bubble release. Vegetation also has influence in the production and consumption of CH\textsubscript{4} by methanogens, which are an obligate anaerobic Archaea group of prokaryotes.

According to Mitsch and Gosselink (2000), a comparison of CH\textsubscript{4} production between freshwater and marine wetlands shows different rates. Marshes and swamps freshwater wetlands show the rate of methanogenesis up to 500 mg C m\textsuperscript{-2} day\textsuperscript{-1}, which is higher than the rate of salt marshes and mangrove marine wetlands that are estimated to be up to 100 mg C m\textsuperscript{-2} day\textsuperscript{-1}. This difference is due to the low amounts of sulfate available as oxidizable substrate in freshwater wetlands (Mitsch and Gosselink 2000).

In coastal marshes, the estimate of CH\textsubscript{4} productions of S. alterniflora ranges from 2.94 to 3.95 μg CH\textsubscript{4} kg\textsuperscript{-1} day\textsuperscript{-1}, which is significantly higher than that in the bare mudflat as indicated by Yuan et al. (2014). In addition, plant biomass represents carbon substrate available for methanogenesis and emissions. Consequently, enhancing primary productivity would increase CH\textsubscript{4} emissions (Kang et al. 2012, Bridgham et al. 2013, Yuan et al. 2014).

Diverse efforts have been made to measure global CH\textsubscript{4} emissions from wetlands. One methodological approach quantifies CH\textsubscript{4} emission into the atmosphere by considering both the plant net primary production (NPP) and the soil organic matter decomposition (SOMD) as substrates needed by methanogens to perform the methanogenesis process (Levy et al. 2012).

At the regional scale, Cao et al. (1996) estimated the rate of CH\textsubscript{4} emission from natural wetlands using a CH\textsubscript{4} emission model based on the NPP, the SOMD and the balance between CH\textsubscript{4} production and oxidation. The calculated NPP of tropical swamps was 820 g C m\textsuperscript{-2} yr\textsuperscript{-1} higher than the NPP of northern bogs calculated as 45 g C m\textsuperscript{-2} yr\textsuperscript{-1} (Cao et al. 1996).

In addition, the rate of SOMD transformed to CH\textsubscript{4}-C annually varied from 2.0 to 40.5%, with a mean of 14.2% over all wetlands. They also estimated that of the CH\textsubscript{4} produced, 70 to 80% was oxidized to CO\textsubscript{2} by methanogens, and the remaining CH\textsubscript{4} was emitted into the atmosphere. The estimate of NPP-C decomposed as CH\textsubscript{4} and released into the atmosphere was 4.2% in temperate wetlands, and 4.8% in tropical wetlands, both were higher than the 2.3% in northern wetlands (Cao et al. 1996).
The variation of CH₄ emission at the regional and global scales depends on factors such as climate patterns, temperature, wetland distribution in latitude and longitude, type of vegetation and productivity, carbon supply, seasons, water availability, soil conditions, and complex biological processes over CH₄ flux rates (Kang et al. 2012). For instance, the CH₄ flux rates are related to latitude, temperature, NPP and SOMD (Figure 15). In the tropics at latitude of 30°N and 30°S, the distribution of CH₄ emission from wetlands are high, and this estimate is related to high temperature, NPP, and SOMD (Cao et al. 1996).

![Figure 15 Latitudinal distribution of temperature, net primary production, and CH₄ emission](image)

Source: Cao et al. (1996)
Temperature (°C), net primary production (NPP g C m⁻² yr⁻¹), and methane emission (10⁻¹ g CH₄ m⁻² yr⁻¹)

The seasonality of wetland inundation has importance to consider the estimates of CH₄ emissions rates in the regional categories of wetland systems (Figure 16). In northern wetlands, the CH₄ emissions were insignificant in winter, while the CH₄ emissions were on average between 40 and 60 mg CH₄ m⁻² day⁻¹ in summer. Moreover, in temperate wetlands the CH₄ emission rates ranged from 100 to 400 mg CH₄ m⁻² day⁻¹ in summer, while in the other seasons the rate varied from 20 to 150 mg CH₄ m⁻² day⁻¹. These seasonal variations are related to change in temperature, NPP, and SOMD (Cao et al. 1996).
Cao et al. (1996) specified that the regional wetland location and area size, and the type of vegetation are associated to the variation of the estimates of CH$_4$ flux rates. The CH$_4$ flux rates of northern wetlands were low, 16.7 Tg yr$^{-1}$ to inundated wetland and 6.6 Tg yr$^{-1}$ to moist and dry tundra wetlands, but their area of wetlands was large. Both the inundated and moist/dry wetlands contribute in 25% of the total global emission; however, the inundated wetland area alone is 53% the total global area. The CH$_4$ flux rate of temperate wetlands was 17.2 Tg yr$^{-1}$, which is much higher than the northern wetlands, but its area of 5.8 x10$^{11}$ m$^2$ is smaller. Moreover, the CH$_4$ flux rate of tropical wetlands was 51.4 Tg yr$^{-1}$ representing 56% of the total global emission. Even when the area of tropical wetland was less than that of northern wetland, the mean CH$_4$ flux rate was much higher (Cao et al. 1996).

Globally, the estimates of total emission of CH$_4$ into the atmosphere from natural wetlands was calculated by Cao et al. (1996) as 92 Tg CH$_4$ yr$^{-1}$, distributed as 85 Tg yr$^{-1}$ for inundated wetlands, and 7 Tg yr$^{-1}$ for moist/dry tundra wetlands. Mitsch et al. (2010) indicated
estimates values of 115 and 145 Tg CH$_4$ yr$^{-1}$ for natural wetlands. Kang et al. (2011) estimated the annual emission in a range of 120 – 240 Tg CH$_4$ yr$^{-1}$. Bridgham et al. (2013) estimated from published studies the global average emission of 166 Tg CH$_4$ yr$^{-1}$.

Bridgham et al. (2006) reported that North American wetlands CH$_4$ flux as 9.4 Tg CH$_4$ yr$^{-1}$; 0.028 Tg CH$_4$ yr$^{-1}$ release by tidal marshes, and 0.11 Tg CH$_4$ yr$^{-1}$ release by mangrove ecosystems. In addition, the CH$_4$ fluxes from the conterminous USA wetlands were estimated as 3.1 Tg CH$_4$ yr$^{-1}$; 0.026 Tg CH$_4$ yr$^{-1}$ release by tidal marshes, and 0.004 Tg CH$_4$ yr$^{-1}$ release by mangrove systems (Table 3). Other estimates of the methane emission rate in mangrove forests are from 0.0473 to 0.3245 g CH$_4$ m$^{-2}$ day$^{-1}$ for South India mangroves; from 0.00019 to 0.00052 g CH$_4$ m$^{-2}$ day$^{-1}$ for Ranong mangroves (Thailand); and from 0.0004– 0.0082 g CH$_4$ m$^{-2}$ day$^{-1}$ for mangroves in southeastern coast of Puerto Rico (Suratman 2008). In addition, methane emission rates for mangrove wetlands in Queensland (Australia) were estimated from 20 to 350 μg CH$_4$ m$^{-2}$ hr$^{-1}$ (Kreuzwieser et al. 2003), and CH$_4$ emissions from a mangrove ecosystem of the Cauvery delta (Muthupet) in South India were estimated in a range from 18.99 to 37.53 mg CH$_4$ m$^{-2}$ day$^{-1}$ (Krithika et al. 2008).

b. Nitrous oxide production and emission in tidal salt marshes and mangrove ecosystems

Nitrification and denitrification processes produce N$_2$O in wetland systems (Kayranli et al. 2010). Nitrification process is the transformation of ammonium (NH$_4^+$) to nitrate (NO$_3^-$) by ammonia oxidizing bacteria under aerobic conditions, producing N$_2$O as a by-product. Denitrification process is the transformation of NO$_3^-$ to N$_2$ by denitrifying bacteria under anaerobic soil conditions, producing N$_2$O (Livesley and Andrusiak 2012).

Wetland ecosystems are not a significant contributor of global N$_2$O fluxes into the atmosphere. There are a few global estimates of the N$_2$O emission from wetland ecosystems, and some of them were reported from coastal mangroves. Global emissions from mangroves range from 0.004 to 0.17 Tg N yr$^{-1}$ (Anderson et al. 2010); and N$_2$O emission for coastal mangrove ecosystems was estimated in 0.076 Tg N yr$^{-1}$ (Barnes et al. 2006, Anderson et al. 2010). Furthermore, N$_2$O emission rates for mangrove wetlands in Queensland (Australia) were
estimated from -2 to 14 μg N m\(^{-2}\) hr\(^{-1}\) (Kreuzwieser et al. 2003), and N\(_2\)O emission from a mangrove ecosystem of the Cauvery delta (Muthupet) in South India were estimated in a range from 0.41 to 0.80 mg N m\(^{-2}\) day\(^{-1}\) (Krithika et al. 2008).

Environmental conditions and climate factors facilitate the production and emission of N\(_2\)O from ecosystems. Livesley and Andrusiak (2012) indicated that the availability of N and the oxygen state of the soil/sediment determine the nitrification and denitrification processes by bacteria. Thus, they estimated small N\(_2\)O emissions under 3 μg N m\(^{-2}\) h\(^{-1}\) from the mangrove sediments in Westernport Bay (Australia). The low emissions are associated to a small nitrogen load and therefore small NO\(_3^−\) and NH\(_4^+\) pools in the soil (Livesley and Andrusiak 2012).

In addition, N\(_2\)O emissions are associated with seasonality and primary production in wetland ecosystems. Thus, in summer, N\(_2\)O emissions were greatest from the salt marsh due to the greater sediment NH\(_4^+\) concentrations (Livesley and Andrusiak 2012). Moreover, in winter the N\(_2\)O fluxes being high because of the release of stored N\(_2\)O (Kayranli et al. 2010).

Furthermore, mangrove ecosystems are small contributors to coastal N\(_2\)O emissions but could dominate coastal CH\(_4\) emissions (Barnes et al. 2006). As a reference, table 3 shows slightly higher CH\(_4\) compared to N\(_2\)O emissions, that is, 34.5 μmol m\(^{-2}\) hr\(^{-1}\) and 1.3 μmol m\(^{-2}\) hr\(^{-1}\) respectively from mangrove creek during the wet season compared to during the dry season at Wright Myo, India (Barnes et al. 2006).

Table 6 Methane and nitrous oxide emission fluxes (μmol m\(^{-2}\) hr\(^{-1}\)) at Wright Myo (India).
Source: Barnes et al. 2006.
Observations during January 2004 (dry season) and July 2004 (wet season)

<table>
<thead>
<tr>
<th></th>
<th>Mangrove Creek</th>
<th>Mangrove Forest</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Wet Season</td>
<td>Dry Season</td>
</tr>
<tr>
<td>N(_2)O</td>
<td>1.3(^b)</td>
<td>0.23(^c)</td>
</tr>
<tr>
<td>CH(_4)</td>
<td>34.5(^b)</td>
<td>23.0(^c)</td>
</tr>
</tbody>
</table>

\(^a\)Static chamber estimates.
\(^b\)Floating chamber estimates.
\(^c\)Estimated from the Clark et al. [1995] relation.
6. Discussion

Coastal wetland ecosystems play a vital role in the global carbon cycling. These systems sequester, transform, and store atmospheric carbon. In addition, they are sources of CH$_4$ and N$_2$O. Environmental conditions, climatic factors, and biogeochemical processes are responsible for the production, storage and emission of CO$_2$, CH$_4$ and N$_2$O gasses from wetland ecosystems (Abril and Borges 2005).

Tidal salt marshes and mangroves are substantial carbon pools. Global estimates of carbon in tidal marshes show ranges of between 400 to 2010 Tg C. The estimate of carbon in mangroves ranges between 4600 to 8900 Tg C (Table 7). Global estimates on coastal wetland ecosystems also indicates that mangrove system contain a significant amount of carbon in soil and biomass Soil carbon pool on mangrove systems ranges from 20 to 4900 Tg C. That estimate is higher than the carbon storage on tidal marshes, whose estimates range from 430 to 1990 Tg C (Table 7).

<table>
<thead>
<tr>
<th>Global estimates</th>
<th>Tidal Salt Marshes</th>
<th>Mangrove ecosystems</th>
</tr>
</thead>
<tbody>
<tr>
<td>Area of wetland (km$^2$)</td>
<td>51,000 – 203,000</td>
<td>137,000 – 300,000</td>
</tr>
<tr>
<td>Carbon sequestration (Tg C yr$^{-1}$)</td>
<td>4.6 – 87</td>
<td>6 – 52</td>
</tr>
<tr>
<td>Carbon accumulation rate in sediment (g C m$^{-2}$ yr$^{-1}$)</td>
<td>151 – 242</td>
<td>139 – 265</td>
</tr>
<tr>
<td>Soil carbon density (g C cm$^{-3}$)</td>
<td>0.036 – 0.039</td>
<td>0.028 – 0.059</td>
</tr>
<tr>
<td>Soil carbon pool (Tg C)</td>
<td>430 – 1990</td>
<td>20 – 4900</td>
</tr>
<tr>
<td>Wood production (Tg C yr$^{-1}$)</td>
<td>7 – 20</td>
<td>1220 – 4980</td>
</tr>
<tr>
<td>Biomass carbon pool (Tg C)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total Carbon Pool (Tg C)</td>
<td>400 – 2010</td>
<td>4800 – 8900</td>
</tr>
</tbody>
</table>

Data from Twilley et al. (1992), Chmura et al. (2003), Duarte et al. (2005), Bridgham et al. (2006), Bouillon et al. (2008), Laffoley et al. (2009), Neillmann et al. (2009), Mcleod et al. (2011), Siikamäki et al. (2012)

Tg = Teragram =10$^{12}$ g

Mangrove systems also are a significant biomass carbon pool with estimates ranging from 1220 to 4980 Tg C. These estimates are almost 90-fold higher than the estimates of carbon biomass on tidal salt marshes, which range from 7 to 20 Tg C. The results suggest that mangrove forests are an important component in the global carbon cycle (Bouillon et al. 2008).
An explanation for these results is that mangrove vegetation communities are favored by the tropical climate contributing to the high net primary production on these ecosystems. Moreover, the lower carbon levels in tidal marshes are related to the dominant herbaceous plants that do not accumulate carbon in wood as is the case of mangrove trees (Bridgham et al. 2006).

Furthermore, comparing the carbon accumulation rate in sediments of both tidal marshes and mangroves shows a little difference between their estimates of 151 to 242 g C m\(^{-2}\) yr\(^{-1}\) for tidal marshes and 139 to 265 g C m\(^{-2}\) yr\(^{-1}\) for mangroves. These rates are enhanced by the tidal regimens that contribute to the vertical sediment accretion in both ecosystems. Tidal floodwaters also contribute by reducing the potential for aerobic decomposition due to the saturates soils. Consequently, there is a low rate of anaerobic decomposition that enables the accumulation of organic matter and carbon in soils (Chmura 2009). Moreover, aboveground biomass enhances to trapping nutrients and depositing sediments in salt marshes (Siikamäki et al. 2012). It is important to highlight that sediments represent long-term carbon pool (millennial) compared to biomass that store carbon over a short time (decennial) because of the natural death of vegetation (Mcleod et al. 2011).

A research on soil carbon sequestration at tropical and temperate wetlands using radiometric dating of the soil cores with \(^{137}\)Cs reported that the average of soil carbon content on tropical wetlands was 111 g C kg\(^{-1}\), which was twice more than the temperate wetlands calculated in 54 g C kg\(^{-1}\) soil (Mitsch et al. 2010). That difference is explained by the fact that tropical wetlands are more productive than temperate wetlands, and that the high temperatures in the tropics lead to more rapid organic matter decomposition (Bouillon et al. 2009, Mitsch et al. 2010). The results also suggest that humid tropical wetlands sequester more carbon than do similar wetlands in the temperate zone (Mitsch et al. 2010).

Globally, tidal salt marshes and mangrove ecosystems are important natural carbon pools compared to terrestrial forest ecosystems. For instance, the estimate of the annual carbon accumulation rate in sediments of both tidal marshes and mangrove systems ranges from 139 to 265 g C m\(^{-2}\), whereas by contrast, the estimates of carbon burial rate of temperate, tropical and boreal forest ranges from 3.5 to 6.1 g C m\(^{-2}\) yr\(^{-1}\) (Mcleod et al. 2011). Figure 17
shows mean long-term rates of C sequestration in soils on tropical, boreal and temperate terrestrial forests, and in sediments on vegetated coastal ecosystems. According to the data for salt marshes, and mangrove, coastal wetland systems hold high carbon burial rates; almost they are 10-fold higher than the terrestrial forests carbon burial rates.

![Figure 17 Carbon sequestration rate in soils in terrestrial forest and coastal ecosystems](image)

Figure 17 Carbon sequestration rate in soils in terrestrial forest and coastal ecosystems
Uppermost point of bars indicate maximum rates of accumulation. Note the logarithmic scale of the y-axis.
Source: Mcleod et al. (2011)

Although wetlands are a significant carbon sink and the annual carbon sequestration is higher than other natural ecosystems, researchers are concerned that wetlands are a source of methane and therefore they have influence in the increasing of greenhouse gasses (Cao et al. 1996, Abril and Borges 2005, Chmura 2009, Kang et al. 2012, Siikamäki et al. 2012, Mitsch et al. 2012, Bridgham et al. 2013). Here, it was indicated that wetlands are responsible for 20 to 40 % of the total CH₄ emissions at the global level. According to the information presented in this paper, the global methane emission in wetlands ranges from 92 to 166 Tg CH₄ yr⁻¹ (Table 8). Although it was not found in the literature review conducted for the preparation of this research current estimates of methane emissions in tidal marshes and mangroves worldwide,
Researchers indicate that the CH₄ emission for both coastal wetland ecosystems are small in relation to the amount of carbon sequestered and stored in them (Abril and Borges 2005, Siikamäki et al. 2012, Livesley and Andrusiak 2012). In addition, the production of methane decreases in coastal wetlands when the gradient of salinity increases, as well as, because of the presence of significant amounts of sulfates that reduce the activity of methanogens (Chmura et al. 2003, Mitsch et al. 2010, Livesley and Andrusiak 2012).

In the review of the published literature, no paper was found that relates carbon sequestration and methane emissions in coastal wetlands that contributes to investigate the significance of these wetlands as sources of methane. Instead, what was studied was a research article of carbon sequestration and methane emission in temperate and tropical freshwater wetlands (Mitsch et al. 2012).

Mitsch and colleagues developed a dynamic simulation model to calculate the net exchange of carbon sequestration and methane emissions in wetlands. They applied their model in seven created and natural temperate and tropical wetlands in Costa Rica, Ohio, and Botswana (Table 9). The results of carbon sequestration indicated that the four natural tropical wetlands located in Costa Rica and Botswana sequestered 42–306 g C m⁻² yr⁻¹, with an average sequestration rate of 129 g C m⁻² yr⁻¹. A natural temperate wetland in Ohio sequestered 143 g C
m² yr⁻¹. The two created flow-through temperate wetlands in Ohio were significantly higher at 219 g C m² yr⁻¹ for the planted, and 267 g C m² yr⁻¹ for unplanted wetlands. The measurement of methane emission indicated that emission were highest in the tropics. In Costa Rica, the methane emissions were higher in the natural tropical wetland (220 - 263 g C m² yr⁻¹) than the flow-through tropical wetland (33 g C m² yr⁻¹). In Ohio, the methane emission was highest in the natural temperate wetland (57 g C m² yr⁻¹) than the emissions in the two created marshes, which methane emissions were estimated in an average 30 g C m² yr⁻¹.

Table 9 Carbon sequestration and methane emission in temperate and tropical wetlands
Source: Mitsch et al. (2012)

<table>
<thead>
<tr>
<th>Climate</th>
<th>Humid temperate</th>
<th>Natural flow-through wetland</th>
<th>Humid tropical</th>
<th>Natural isolated wetland</th>
<th>Dry tropical</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland type</td>
<td>Old Woman Creek, Ohio</td>
<td>Oldentangy River Wetlands, Ohio</td>
<td>Earth University, Costa Rica</td>
<td>La Selva, Costa Rica</td>
<td>Okechobee, Florida</td>
</tr>
<tr>
<td>Wetland name</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>F CO₂, Carbon sequestration, g C m⁻² year⁻¹</td>
<td>-243 (overall)</td>
<td>-143</td>
<td>-306</td>
<td>-84</td>
<td>-84</td>
</tr>
<tr>
<td>F CH₄, Methane emissions, g C m⁻² year⁻¹</td>
<td>+30 ± 14 (125)</td>
<td>+57 ± 18</td>
<td>+33 ± 5</td>
<td>+220 ± 64</td>
<td>+263 ± 64</td>
</tr>
<tr>
<td>Net carbon exchange with atmosphere, g C m⁻² year⁻¹</td>
<td>-213 (overall)</td>
<td>-257 (Wetland 1)</td>
<td>-86</td>
<td>-273</td>
<td>+136</td>
</tr>
<tr>
<td>F CO₂/F CH₄</td>
<td>8.1:1</td>
<td>2.5:1</td>
<td>9.3:1</td>
<td>0.3:1</td>
<td>0.3:1</td>
</tr>
<tr>
<td>Carbon dioxide sequestration, g CO₂ m⁻² year⁻¹</td>
<td>-891 (overall)</td>
<td>-524</td>
<td>-1122</td>
<td>-308</td>
<td>-308</td>
</tr>
<tr>
<td>Methane emissions, g CH₄ m⁻² year⁻¹</td>
<td>+76 ± 24</td>
<td>+44 ± 6</td>
<td>+293 ± 86</td>
<td>+350 ± 86</td>
<td>+96 ± 11</td>
</tr>
<tr>
<td>CO₂/CH₄ ratio</td>
<td>22.3:1 (overall)</td>
<td>7:1</td>
<td>25:1</td>
<td>1:1:1</td>
<td>0.9:1</td>
</tr>
<tr>
<td>CO₂/CH₄ ratio, 100-year simulation</td>
<td>223:1 (overall)</td>
<td>157:1</td>
<td>171:1</td>
<td>13:1:1</td>
<td>11:5</td>
</tr>
<tr>
<td>Sink, year</td>
<td>0 (overall)</td>
<td>31</td>
<td>0</td>
<td>214</td>
<td>255</td>
</tr>
</tbody>
</table>

The year at which the wetland goes from being a net radiative force to a net radiative sink is also indicated for each wetland. Negative signs indicate carbon fluxes out of the wetlands; positive signs indicate carbon fluxes into the wetlands.

After applying the dynamic simulation model, Mitsch and colleagues reported that methane emissions became irrelevant within 300 years compared to carbon sequestration in wetlands. The simulated amount, called CO₂ equivalent, is carbon dioxide plus 25 times methane. They defined the carbon dioxide equivalent (CO₂eq) as CO₂eq = CO₂ + (GWPₘ x MCH₄). MCH₄ is the atmospheric methane, and the methane global warming potential (GWPₘ) is 25.
Policy makers use this GWP ratio to compare methane and carbon dioxide fluxes. Thus, the results indicated that in the tropical flow-through wetland, 255 kg of CO₂ is taken out of the atmosphere for every kg of CH₄ release into the atmosphere after 100 years. For the created temperate wetlands averaged together, the ratio is 223 kg of CO₂ for every kg of CH₄ released. For these three wetland simulations, the net CO₂ becomes negative almost immediately and consequently they become sinks of greenhouse gases almost from the beginning. The natural temperate wetland, with a ratio of 71:1 after 100 years, becomes a sink of greenhouse gases after 31 simulated years. Moreover, the three natural tropical wetlands become sinks after 140–255 simulated years.

Mitsch and colleagues also applied the dynamic simulation model for 14 additional wetlands. The results indicated that twelve of them became net sinks of carbon, with ratios above 25:1 well within 100 years (Mitsch et al. 2012). They also estimated the carbon sequestration in the world’s wetlands on 1,280 Tg C yr⁻¹, and the average of carbon sequestration in 118 g C m⁻² yr⁻¹. In addition, the methane emission from the world’s wetlands was estimated in 448 Tg C yr⁻¹. On the balance between soil carbon sequestration and methane emissions, the world’s wetlands are significant sinks of carbon of 830 Tg C yr⁻¹. Therefore, they concluded that most wetlands are net carbon sinks and not radiative sources of climate change, even when methane emissions are considered (Mitsch et al. 2012).

As it was indicated, tidal salt marshes and mangrove ecosystem play a role in the exchange of methane and nitrous oxide and carbon dioxide. Even when flows of methane and nitrogen from coastal wetland ecosystems are minimal (CH₄) or insignificant (N₂O), here it is presented a research to understand the role of these three greenhouse gasses in the global carbon cycle. The soil-atmosphere exchange was studied along the transect in the melaleuca woodland, salt marsh and into mangroves in the Westernport Bay (Australia) (Livesley and Andrusiak 2012). Results of the soil/sediment exchange of CO₂, CH₄ and N₂O measured seasonally along the transects indicated that the melaleuca woodland soil was a constant CH₄ sink of approximately -25 μg C m⁻² hr⁻¹ (Figure 18).

The salt marsh was a weak CH₄ source, less of 5 μg C m⁻² hr⁻¹, and in the mangrove sediments where emissions of up to 375 μg C m⁻² hr⁻¹ in summer. In addition, all three
ecosystems were a small N\textsubscript{2}O source of under 10 \(\mu\text{g} \text{ N m}^{-2} \text{ hr}^{-1}\). Moreover, the soil CO\textsubscript{2} emissions were significant from the melaleuca woodland soil and salt marsh sediment but small from the mangrove sediments, and there was a general increase in spring and summer months. Sediment carbon density was significant in the salt marsh than the mangrove. Salt marsh sediment carbon density was 168 Mg C hr\(^{-1}\), an estimate considered slightly greater than the mangrove sediment carbon density of 145 Mg C hr\(^{-1}\).

![Seasonal N\textsubscript{2}O, CH\textsubscript{4} and CO\textsubscript{2} exchange rates in Westernport Bay (Australia)](image)

**Figure 18** Seasonal N\textsubscript{2}O, CH\textsubscript{4} and CO\textsubscript{2} exchange rates in Westernport Bay (Australia)

Measured were along transects from melaleuca woodland to salt marsh and into mangrove woodland in May (autumn) and December (summer) in 2009.

Source: Livesley and Andrusiak (2012)

Livesley and Andrusiak highlighted that the small N\textsubscript{2}O emission from the mangrove were not surprising considering the small NO\textsubscript{3} and NH\textsubscript{4} pools. They also indicated that the sediment CH\textsubscript{4} flux correlated with salinity, pneumatophore number and the redox potential of sediment water at depth. Moreover, they affirmed that their study has shown that tidal salt ecosystems in temperate zones are not a large source of CH\textsubscript{4} or N\textsubscript{2}O, but their sediments are a significant C stock.

Costal wetland ecosystems host a valuable biome and provide ecosystem services to local, regional and global communities. However, these ecosystems are threatened by human
activities such as agricultural practices, logging, deforestation, engineering and urban development, and by the impact of climate change such as rise sea level (Duarte et al. 2005, Mcleod et al. 2011, Mitsch et al. 2012, Chmura 2013). Thus, the wetlands vulnerability comes up when land-use change, habitat loss, and logging and fire interact with global and regional climate change forcings.

Estimates of global loss of carbon pools indicated that the percentage of mangroves loss at the global level was between 30 to 50 % (since 1940s) and 20 % (since 1980s). The annual rate of global loss was estimated around between 0.7 to 3 %. In the case of salt marshes, the percentage of global loss was estimated to be 25% (since 1800s), with an annual rate of global loss of between 1 to 2 % (Mcleod et al. 2011). Prolonged disturbance had changed wetland structure, nutrient dynamics, and biodiversity composition. Disturbance had also affected essential ecosystem services associated with global carbon cycles such as carbon sequestration, carbon density in soils, sediments and biomass, and carbon fluxes into the atmosphere. For instance, land use changes affect the carbon storage, climate regulation, hydrologic balance and biodiversity in wetland ecosystem. In addition, tidal salt marshes face many stressors including invasions of exotic species and pollution by excessive nutrients, pesticides, herbicides, heavy metals and organic compounds released into coastal waters. These stressors may disrupt components of the ecosystem and impact the carbon storage capacity that depends on the sustainability of marsh accumulation and the maintenance of vegetation cover (Chmura 2009).

Furthermore, coastal wetlands could shift from the net carbon sink to a net carbon source. As it was highlighted, tidal salt marshes and mangroves store about 10.80 Pg C yr\(^{-1}\), mainly that carbon is store in soils. However, land conversion and deforestation of mangrove and tidal marsh ecosystems by human and natural interventions cause carbon emissions from these ecosystems. Current threats to mangrove ecosystems are attributed to human pressures such as over-harvesting for timber and fuel wood production, reclamation for aquaculture and salt ponds (Bouillon et al. 2009). Estimates of global carbon emissions from mangroves loss indicated that currently about 33.5 million tons of carbon are released annually, and the estimation of global carbon emission from tidal salt marshes are calculated to be around 10.5 million tons of carbon per year (Siikamäki et al. 2012).
7. **Recommendations**

Preserving wetland ecosystems in their natural state is the best method for conserving their biodiversity and habitats, and maintaining critical functions, services and goods that they provide. However, when it is not possible to maintain the ecosystems without human disturbances such as changes in land use or natural causes such as natural disasters and severe climate events, it is necessary to implement restoration actions. Ecological restoration seeks to recover as many environmental components, ecological functions, and services of the ecosystems affected. Efforts to protect and restore natural carbon pools may contribute to avoiding carbon emissions and offset important greenhouses gas emissions of CO\textsubscript{2}, CH\textsubscript{4}, and N\textsubscript{2}O from wetlands (Crooks et al. 2010b).

Applying recovery efforts requires knowledge of the dynamics and complexity of the ecosystems. External factors that cause stress or disturbances within ecosystems also need to be understood to ensure desirable characteristics are restored to maintain sustainable and productive ecosystems into the future (Hobbs and Harris 2001). An adaptive management approach together with monitoring and assessment activities are key measures that should be implanted to ensure restoration success and sustainability of restored ecosystems. The adaptive approach is valuable because it prioritizes experimentation to identify problems and constraints throughout the life of a restoration project, which leads to a better understanding of ecosystem recovery and a more successful ecosystem restoration (Zedler et al. 2012).

Besides ecological principles, scientific concepts, and restoration processes, wetland restoration efforts require adequate policies, available funds, and social commitments. Government policy that promotes restoration efforts plays a crucial role in the success of ecosystem restoration. Therefore, governments should promote and regulate laws to provide the necessary funding for restoration projects. Governments should promote environmental research and development of new technologies that provide the best restoration management practices. Moreover, governments should develop policies to promote research projects beyond national borders for large-scale projects that require long-term assessments and internationally funding (Zedler et al. 2012).
Promoting research on carbon sequestration in coastal wetland ecosystems can contribute to addressing the knowledge gaps, overcome the lack of data and uncertainties, and improve understanding of the driving forces affecting carbon sequestration rates (Chmura et al. 2003, Bouillon et al. 2008, Bridgham et al. 2013). Although global carbon sequestration rates are high for vegetated coastal ecosystems, these rates vary among locations, reflecting the wide range of factors that affect the magnitude of any given carbon sink. Development of local, regional and global maps, remote sensing and aerial photography of coastal wetland ecosystems will contribute to obtaining proper extents of mangrove and tidal salt marshes. High quality mapping will help to quantify carbon sequestration potential, estimate the soil carbon density, and set accurate values for carbon pools. Maps can help in conservation planning, adaptive management and restoration efforts as well. Setting standard methods for measuring carbon sequestration, sediment carbon rates, and above and belowground biomass will help to improve the estimates of carbon pools (Mcleod et al. 2011).

Bouillon et al. (2009) proposed management recommendations to enhance the potential of mangroves as a carbon sink, both on short and longer time-scales. They also proposed the development of guidelines for mangrove management and rehabilitation to guarantee biomass production during forest establishment, growth, and accumulation of carbon in mangrove sediments. Another one of their recommendations is the development of a strategy to improve the ability to estimate and forecast mangrove carbon and nutrient cycling patterns with simulation models. In association with field studies, the modeling approach may enhance assessment of the effects of environmental change and disturbance on ecological processes. In addition, modeling will enhance monitoring the sustainability of mangrove resources and evaluating the impacts on the role of mangrove forest in the global carbon cycle.

Chmura (2009) proposed management recommendations to maintain and enhance carbon storage capacity. He indicated that even when tidal salt marshes are protected from direct impacts, such as dredging and filling in many regions guaranteeing the sustainability of protected marshes requires protection from indirect impacts as well. Thus, Chmura proposed that the implementation of programs to protect wetlands should include activities that evaluate water and sediment discharges within the estuarine watershed. In addition, programs
may assess the loss of suspended sediments that are associated with the decrease in the ability of tidal marshes to keep elevations with rising sea level. Finally, he proposed that the construction of terrestrial buffer zones could help to reduce nutrient enrichment of salt marshes, a threat to the marsh carbon sink and the sustainability of the ecosystems.

Coastal wetlands face threats of habitat destruction and as a result critical ecosystem functions and services such as carbon sequestration are affected. The implementation of proposals to prevent the destruction of these ecosystems through financial incentives for protecting them is necessary. The recognition of the ability of coastal ecosystems to sequester carbon from the atmosphere and the capacity to store significant amounts of carbon that can be released as CO$_2$ upon disturbance could be connected with carbon financial incentives (Murray et al. 2011). Proposals to protect and provide financial aid to restoration projects can be enacted by assigning a monetary value for each unit area of ecosystem. Financial incentives for conservation coastal wetland ecosystems and emission offsets is a promising option for the protection of coastal wetland ecosystems (Siikamäki et al. 2012, Grimsditch et al. 2013).

Trading systems or carbon markets that promote financial incentives are needed to persuade managers of coastal ecosystems to avoid habitat conversion and the possibility that wetland ecosystems will change from greenhouse gases sink to sources. Murray et al. (2011) They suggest that assuming carbon prices of $0 to $30 t CO2e the gross financial returns to avoided habitat-conversion projects fall anywhere between $0 and $37,000 per hectare of protected habitat. For example, at a carbon price of $15/t CO2e, in the case of mangroves ecosystems containing the largest carbon pools, the average gross returns are over $18,000/ha for oceanic ecosystems and over $13,000/ha for estuarine ecosystems. In comparison, the average gross returns for salt marshes are approximately $8,000/ha.

Initiatives to develop carbon-offset projects are elaborate in the U. S. For example, Fisher and Huo presented a Duke Carbon Offsets Initiative. They developed a business plan for a potential blue carbon project in North Carolina for preserving and revitalizing coastal ecosystems that stored carbon as a way of offsetting the Duke University carbon emissions. They also estimated the offset price as more than $20 per ton of carbon (Fisher and Huo 2012). The feasibility of the projects for coastal ecosystem carbon, known as a blue carbon, depends
on the environmental factors and conditions in tidal salt marshes and mangrove areas where projects will be implemented. Kraft and colleagues developed a cost-benefit analysis to characterize the current state of blue carbon offset opportunities in North Carolina and Louisiana for the Duke Carbon Offset Initiative. They indicated that the unit cost of a blue carbon project in North Carolina is 170 times greater than the cost in Louisiana because the initial investment should include the development of previous wetland restoration project in North Carolina. In addition, their analysis of sea level rise impacts indicates that Louisiana has a higher carbon sequestration rate than North Carolina due to a smaller critical tidal range when sea level rises from 0.1 to 1 cm yr$^{-1}$ (Kraft et al. 2013).

Initiatives to develop protocol to quantify carbon sequestration and reduction of carbon emission form coastal ecosystems are underway. Since 2010, the Restore America’s Estuaries organization (RAE) is leading a national effort to develop a protocol for tidal wetlands greenhouse gases offsets. The agency indicated that the protocol should provide guidelines to calculate, report, and verify greenhouse gas emission-reductions associated with offset projects. The protocol also should provide a framework for implementing tidal wetlands projects to generate offset credits that are likely to be recognized by current climate markets, registries, and under emerging climate change laws and regulations (Crooks et al. 2010a).

In September 2012, the American Carbon Registry (ACR), certified the “Restoration of Degraded Deltaic Wetlands of the Mississippi Delta” developed by Tierra Resources. The document provides guidelines on how to calculate, report, and verify greenhouse gas reductions for the creation of wetland carbon credits. In December 2013, The Blue Carbon Initiative announced that Restore America’s Estuaries submitted “Greenhouse Gas Accounting Methods for Tidal Wetland and Seagrass Restoration” to the Verified Carbon Standard to begin

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3 The Restore America’s Estuaries (REA) is a nonprofit organization working to protect and restore the vital habitats of estuaries.

4 The American Carbon Registry (ACR) is a leading non-profit U.S. carbon market standard and registry agency. Tierra Resources is a private agency created to conserve, protect, and restore coastal wetland ecosystems by creating innovative solutions that support investment into blue carbon.

the approval process\textsuperscript{6}. The document contains a methodology to meet the eligibility conditions to calculate net greenhouse gas benefits and receive carbon credits for tidal wetland and seagrass restoration projects\textsuperscript{7}. In California, the state California Coastal Conservancy is promoting the development of a wetland carbon sequestration protocol for coastal and delta wetlands in California\textsuperscript{8}.

8. Conclusion

Coastal wetland ecosystems are significant carbon pool although they are only about 3 – 5% of the world’s wetlands. As I had indicated earlier in this paper, global estimates for carbon in coastal wetland range from 0.4 to 8.9 Pg C. I had also shown that tidal salt marshes and mangrove soils store a large proportion of carbon. For my literature review, global estimates of soil carbon in tidal salt marshes and mangroves range from 0.02 to 4.9 Pg C. Coastal wetlands are high primary productive systems, principally in the tropics where mangroves are located. Thus, mangrove vegetation stores higher amounts of carbon in both aboveground and belowground biomass (1.22 – 4.98 Pg C) than tidal salt marshes plants (0.007 – 0.02 Pg C). Moreover, it is estimated that the annual carbon storage is in a ranges from 4.6 to 8.7 Tg C yr\textsuperscript{-1} for tidal salt marshes and from 6 to 52 Tg C yr\textsuperscript{-1} for mangrove ecosystems.

As I indicated in this study, tidal salt marshes and mangrove not only sequester CO\textsubscript{2}, which is a dangerous greenhouse gas, but also they emit CO\textsubscript{2}, CH\textsubscript{4} and N\textsubscript{2}O. According to a few cases related to wetland exchange of CO\textsubscript{2}, CH\textsubscript{4}, N\textsubscript{2}O discussed here; tidal salt marshes and mangrove are not a substantial source of CH\textsubscript{4} or N\textsubscript{2}O compared to CH\textsubscript{4} production and emission from freshwater wetlands.

In addition, coastal wetland ecosystems face threats from human and natural actions. Thus, it was indicated that wetlands could be severely affected by climate change. At the global level, the estimates indicated that the annual rate of mangrove loss is about 0.7 to 3% and that

\textsuperscript{6} The Blue Carbon Initiative is a global program working to mitigate climate change through the restoration and sustainable use of coastal and marine ecosystems. The Verified Carbon Standard is the world’s leading voluntary greenhouse gas program created to promote a quality assurance in voluntary carbon markets.


of tidal salt marshes is between 1 to 2 %. Drastic changes in these ecosystems could release large amounts of carbon stored in soils and biomass.

As was discussed in this paper, protecting and restoring tidal salt marshes and mangrove globally may be necessary practices to maintain these large natural carbon pools, which have high rates of carbon sedimentation, low rates of CO₂ and CH₄ emissions, and insignificant N₂O flux into the atmosphere. Restoration actions in these ecosystems could benefit from funds obtained from the trade of carbon credits. The development of protocols to determine a standard measure about carbon uptake, monetary values, and any financial incentives for carbon storage in wetlands are valuable alternatives that can contribute to consolidate wetland carbon market. Although there are data gaps and uncertainties about estimates, I found this research relevant to improve the knowledge about the importance of coastal wetland ecosystems as carbon pools, as well as the vulnerability of these ecosystems including a considerable potential to become sources of atmospheric carbon if they are highly disturbed.
Literature cited


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