# Global carbon sequestration in tidal, saline wetland soils

Gail L. Chmura[1], Shimon C. Anisfeld,[2] Donald R. Cahoon,[3,4] and James C. Lynch[3,4]

[1]*Department of Geography and Centre for Climate and Global Change Research, McGill University, 805 Sherbrooke St. W; Montreal, QC H3A 2K6 Canada*

[2]*Yale School of Forestry and Environmental Studies, 370 Prospect St., New Haven, CT 06511 USA*

[3]*US Geological Survey, National Wetlands Research Center, 700 Cajundome Blvd., Lafayette, LA 70506 USA*
[4]Present address: US Geological Survey, Patuxent Wildlife Research Center, 11510 American Holly Dr., Laurel, MD 20708 USA

received 16 April 2002
MS#2002GB0019A

## Abstract

Wetlands represent the largest component of the terrestrial biological carbon pool, and thus play an important role in global carbon cycles. Most global carbon budgets however, have focused on dry land ecosystems that extend over large areas, and have not accounted for the many small, scattered carbon-storing ecosystems such as tidal saline wetlands. We compiled data for 154 sites in mangroves and salt marshes from the western and eastern Atlantic and Pacific coasts, as well as the Indian Ocean, Mediterranean Ocean, and Gulf of Mexico. The set of sites spans a latitudinal range from 22.4°S in the Indian Ocean to 55.5°N in the northeastern Atlantic. The average soil C density of mangrove swamps ($0.055 \pm 0.004$ g cm$^{-3}$) is significantly higher than the salt marsh average ($0.039 \pm 0.003$ g cm$^{-3}$). Soil C density in mangrove swamps and *Spartina patens* marshes declines with increasing average annual temperature, probably due to increased decay rates at higher temperatures. In contrast, carbon sequestration rates were not significantly different between mangrove swamps and salt marshes. Variability in sediment accumulation rates within marshes is a major control of carbon sequestration rates masking any relationship with climatic parameters. Globally, these combined wetlands store at least 44.6 Tg C yr$^{-1}$, and probably more as detailed areal inventories are not available for salt marshes in China and South America. Much attention has been given to the role of freshwater wetlands, particularly northern peatlands, as carbon sinks. In contrast to peatlands, salt marshes and mangroves release negligible amounts of greenhouse gases and store more carbon per unit area.

## 1. Introduction

Wetlands represent the largest component of the terrestrial biological carbon pool [*Dixon and Krankina*, 1995] and thus play an important role in global carbon cycles [*Sahagian and Melack*, 1998]. Most global carbon budgets, however, have focused on dry land ecosystems that extend over large areas, and have not accounted for the many small, scattered carbon-storing ecosystems such as mangrove swamps and salt marshes [*Atjay et al.*, 1979, *Olson et al.*, 1983]. Syntheses that do include wetlands typically exclude tidal saline wetlands because there have been no empirically based estimates of their carbon storage potential.

In this study we used published and our own unpublished data to estimate the amount of carbon stored globally in soils of salt marshes and mangrove swamps. We then examine spatial patterns in carbon density and storage with respect to climate parameters, as well as local variability, to determine which are important controls.

Tidal saline wetlands (TSWs), i.e., salt marshes and mangrove swamps, are found on sheltered marine coastlines. The former, dominated by herbaceous vegetation, exist in climates ranging from arctic to subtropical. Mangrove swamps replace salt marshes in the subtropics, around 25° N and S, and are dominated by woody vegetation [*Mitsch and Gosselink*, 2000]. Mangrove swamps and salt marshes are intertidal ecosystems; in order to persist, their surface elevations must increase with rising sea level.

Both types of wetlands are noted for exceptional rates of production, rivaling that of productive agricultural lands [*Odum*, 1959]. Root to shoot ratios of salt marsh plants range from 1.4 to 50 (see review in *Smith et al.*, 1979), thus a large portion of the primary production is found in belowground biomass that contributes to vertically extensive deposits, as great as 8 m deep [e.g., *Scott and Greenberg*, 1983]. Mangrove deposits can attain comparable depths [e.g., *Woodroffe et al.*, 1993]. In mangrove swamps peat formation primarily occurs through deposition and slow turnover of mangrove roots as aboveground tissues rapidly decay or are transported from the system [*Middleton and McKee*, 2001].

The global importance of wetlands as carbon sinks is widely recognized [*Adams et al.*, 1990; *Watson et al.*, 2000]. Due to their great expanse, the role of peatlands as carbon sinks has received the greatest attention by researchers [*Roulet*, 2000], who report rates of soil carbon sequestration from 20-30 g C $m^{-2}$ $yr^{-1}$. However, decomposition of peatland soils results in high rates of $CH_4$ flux [*Bartlett and Harris*, 1993], reducing their value as a means to moderate greenhouse warming. The soil chemistry and carbon accumulation patterns of TSWs differ in several respects from those of peatlands or other freshwater wetlands. For one thing, carbon concentrations in TSWs are often lower than in peatlands, since tidal wetlands can receive significant inputs of fine-grained minerals (through tidal exchange with adjacent coastal waters), which dilute the inputs of organic matter from above- and belowground production. On the other hand, rates of soil accumulation in tidal wetlands tend to be higher than in peatlands, so net carbon sequestration is potentially substantial. Perhaps most important, the presence of abundant sulfate in TSW soils hinders $CH_4$ production, so these ecosystems are considered to be negligible sources of $CH_4$, if not $CH_4$ sinks [*Bartlett and Harris*, 1993; *Magenheimer et al.*, 1996; *Giani et al.*, 1996]. Studies of gas fluxes in TSWs suggest that emissions of the greenhouse gas $N_2O$ are also negligible [*Smith et al.* 1983; *DeLaune et al.*, 1990].

## 2. Methods

We found 26 studies (Tables 1 and 2) that reported soil carbon densities or parameters necessary for calculation of soil carbon densities (soil bulk density and % soil organic matter or

% soil carbon) in TSWs. From these studies (and our unpublished data) we compiled data for 154 sites in TSWs from the western and eastern Atlantic and Pacific coasts, as well as the Indian Ocean, Mediterranean Ocean, and Gulf of Mexico (Table 1 and 2). The set of sites spans a latitudinal range from 22.4°S in the Indian Ocean [*Naidoo*, 1980] to 55.5°N in the northeastern Atlantic [*Morris and Jensen*, 1998]. Most of the data found (~75%) were from salt marshes (Table 1). Some of the measurements came from the same estuary or a contiguous wetland area, allowing us to compare local variability to large-scale variability.

In most cases the carbon densities reported were derived from measurements of loss-on-ignition (LOI), but the Walkley Black and dichromate digestion methods were also used, as well as % carbon determined from carbon analyzers. LOI measurements of mangrove soils were transformed to organic carbon by dividing by a factor of 1.724 [*Allen*, 1974], but for salt marsh soils we applied the quadratic relationship specific to salt marshes reported by *Craft et al.* [1991]:

$$\%\text{organic carbon} = (0.04)\ \text{LOI} + (0.0025)\text{LOI}^2.$$

If data were reported as % carbon using other methods we transformed values to be consistent with our data set.

Many investigators also reported rates of vertical soil accumulation, allowing us to compile carbon accumulation rates for 124 sites. (This includes the Australian mangroves for which only rates of carbon accumulation were reported [*Brunskill et al*., 2002].) Vertical soil accumulation rates represent averages over variable periods, from 1 to ~100 yr. The depth of the maximum concentration of $^{137}$Cs (associated with peak fall-out in 1963 [*DeLaune et al*., 1978]) or the pattern of unsupported $^{210}$Pb with depth was employed to determine long term average rates of vertical accretion. Where reports made available both rates and carbon densities, we averaged carbon density over the dated depths.

At 13 sites (Florida mangroves, Louisiana salt marshes, and Bay of Fundy salt marshes) we employed clay marker horizons [*Cahoon and Turner*, 1989; *Chmura et al.* 2000] to determine soil accumulation rates. Measurement of the thickness of soil accumulated over these horizons in a given period provides a short-term accretion rate. Although a few samples were lost because of erosion, we used only positive accretion values in our calculations. To estimate rates of carbon accumulation we multiplied rates by carbon density of the surface 2 cm of paired soil samples.

A global inventory of mangrove area was compiled by Spaulding et al. [1997], who estimated 181,000 km$^2$ of mangrove swamps. No single global inventory of tidal salt marshes has been published. Regional or national salt marsh inventories are available for Canada [*Letourneau and Jean*, 1996*; Hanson and Calkins*, 1996], Europe [*Dijkema*, 1987], the United States [*Field et al*., 1991], and South Africa [*O'Callaghan*, 1990]. Together these regions hold approximately 22,000 km$^2$ of salt marsh (Table 3). We found no data on the extent of salt marshes on the temperate coasts of Asia, South America, and Australia, but we expect these to be substantial.

For most sites we were able to locate nearby meteorological stations for which climate normals, average monthly minima and maxima computed over at least three decades, were available. Where possible we located stations at low elevations and avoided large urban areas. Because data was compiled by various agencies such as the U.S. Weather Service (http://ggweather.com, http://cirrus.dnr.state.sc.us), the Meteorological Service of Canada (www.msc-smc.ec.gc.ca), Mexico’s Servicio Meteorológico Nacional (http://smn.cna.gob.mx), the Australian Commonwealth Bureau of Meteorology (www.bom.gov.au), the South African

Weather Service (www.weathersa.co.za), and the Met Office of the U.K. (www.metoffice.com), the period over which the averages were calculated varies (Tables 1 and 2).

## 3. Results and Discussion

### 3.1 Climatic Controls

The average soil carbon density of all sites is 0.043 ±0.002 g $cm^{-3}$. A t-test for difference of means ($P < 0.05$) shows that the average soil carbon density of mangroves swamps is significantly higher than the salt marsh average, 0.055 ±0.004 vs. 0.039 ±0.003 g $cm^{-3}$. This difference could be due simply to higher productivity in warmer climates (*Turner*, 1976), yet average annual temperature explains only a small amount (<25%) of the variability in soil carbon density in the entire data set (Table 4).

Production and decay rates could vary with plant species so we tested average annual temperature as a predictor for carbon density in soils of salt marshes dominated by *Spartina patens* and those dominated by *Spartina alterniflora* – all on the Northwest Atlantic and Gulf of Mexico (Figure 1). Rather than increase with temperature, soil carbon density in both vegetation types decreases with increasing average annual temperature, as well as annual maximum and minimum averages (Table 4). Only in the *S. patens* marshes do climate parameters explain a significant portion of the variability in soil carbon density. This relationship may be driven by a cluster of sites on the Gulf of Mexico that have low C density and high average annual temperature, but it is accepted that soil carbon decreases with average annual temperatures in terrestrial soils, presumably due to stimulated microbial decay [*Schimel et al.* 1994].

Climate parameters explain more of the variability in mangrove soils (Figure 2). Here carbon density also decreases with increasing temperatures (Table 4).

Globally, rates of carbon sequestration average 210 ±20 g $m^{-2}$ $yr^{-1}$. A t-test ($P < 0.05$) shows no significant difference between average rates of carbon sequestration in mangrove and salt marsh systems (Figure 3). Average annual temperature explains only 5% of the variability in rates of carbon sequestration, and generally, no significant portion of the variability is explained by temperature when our data is analyzed by wetland type (Table 4). The exception is soil of *S. alterniflora* salt marshes for which C accumulation rates decline with increasing average annual minimum temperature. Thus, thermal controls on decomposition rates may be a factor in C accumulation rates, but regional or local factors must be the dominant controls on rates of carbon sequestration in TSW soils.

### 3.2 Local Controls

What is most noticeable about the data is the high variability within a given region, such as the 14 salt marsh sites on the Connecticut coast of Long Island Sound (~ 41ºN) and the 22 salt marsh sites on the Bay of Fundy (45.1 – 45.9ºN), as well as the 25 salt marsh (29 – 30ºN) and 20 mangrove (25 – 26ºN) sites on the northern Gulf of Mexico (Figure 3). Much of this variability can be explained by differences in suspended sediment supply and tidal water flooding.

The range in carbon densities of individual surface samples (0-2 cm) from single wetlands is broad with respect to the global range (Figure 4). There are also significant differences (t-test, $P < 0.05$) in carbon density or carbon accumulation rates within distinct zones of single mangrove wetlands or salt marshes.

At Shark River, Florida fringe mangroves (those adjacent to coastal waters) have lower soil carbon density than soils in basin mangroves located more distant from open waters. This relationship, however, is reversed for rates of soil carbon accumulation, as rates in fringe mangroves are significantly higher.

The pattern seen in mangroves is repeated in salt marsh soils. In the Bay of Fundy, soil carbon densities can be significantly greater in the *S. patens* zone, where elevations are higher, tidal flooding is less frequent, and suspended sediment supply is lower with respect to the *S. alterniflora* zone [*Chmura et al.*, 2001]. Because sediment deposition is more rapid at lower marsh elevations, soil and soil carbon accumulation rates are significantly higher there. It is likely that sediment deposition enhances carbon sequestration by trapping organic matter from both macrophytes and microflora growing on the soil surfaces [*Connor et al.*, 2001]. Our Louisiana example actually comes from two sites on the Mississippi Delta, that are 105 km apart. The average soil carbon density of the Louisiana *S. alterniflora* marsh is significantly higher than the *S. patens* marsh, but greater accretion rates (1.6 times greater in the *S. patens*) are enough to balance carbon accumulation rates in the two marshes.

### 3.3 Global Stocks and Rates

Because there is no significant difference in carbon sequestration rates by ecosystem type (mangrove swamp or salt marsh) or climatic regime, we calculate an overall average rate of carbon sequestration per unit area: 210 g $CO_2$ $m^{-2}$ $yr^{-1}$. This is an order of magnitude greater than C sequestration by peatlands (20 – 30 g $CO_2$ $m^{-2}$ $yr^{-1}$) [*Roulet*, 2000]. Using the documented value of 203 x $10^3$ $km^2$ of global wetland area (which is an underestimate, as discussed above), this means that at least 42.6 ±4.0 Tg C are sequestered by TSWs each year. Using the TSW area estimate from the U.S. wetland inventory [*Field et al*., 1991] we can assess the importance of the TSW carbon sink with respect to the total carbon sink estimated for the conterminous U.S. [*Pacala et al.*, 2001]. At a magnitude of 5 Tg C $yr^{-1}$, the TSWs would make up roughly 1 to 2% of the carbon sink (300 to 580 Tg C $yr^{-1}$) previously estimated for the conterminous U.S.

Assessment of the potential value of TSWs as an enhanced carbon sink in the future must include consideration of methane as well, with a global warming potential of 23 times that of $CO_2$ (over a 100 year time horizon [*Ramaswamy* et al. 2001]). Methane flux in TSWs has not been studied to the same degree as in peatlands, where a range of 9.6-13.0 g $CH_4$ $m^{-2}$ $yr^{-1}$ has been reported [*Bartlett and Harris*, 1993]. The presence in TSWs of abundant $SO_4^{-2}$ for organic matter decomposition through sulfate reduction means that $CH_4$ production is expected to be considerably smaller than in peatlands. Measurements indicate that methane flux in TSWs appears to decrease with increasing salinity and increase with temperature [*Bartlett et al.* 1987, *Magenheimer et al.* 1996]. Although fluxes as high as 22 g $CH_4$ $m^{-2}$ $yr^{-1}$ have been measured in some TSW sites [e.g., *Bartlett et al*. 1987], other TSWs have been reported to be methane sinks [e.g., *Giani et al.* 1996]. The combination of greater C burial and possibly lower $CH_4$ emissions means that TSWs could be more valuable as C sinks per unit area than peatlands if anthropogenic activity or natural processes were to increase ecosystem $CO_2$ assimilation and burial [*Whiting and Chanton* 2001]. This could occur, for example, as a response to an increase in the rate of sea level rise [*Morris et al.*, 2001], nitrogen fertilization, or global area.

As depths of TSW soil deposits are variable, we estimate the carbon stored in only the surface 0.5 m of soil. Salt marsh surface deposits store 430 ±30 Tg C, while mangrove deposits store another 5,000 ±400 Tg C. Although adequate inventories have not been made, it is likely that average soil depths are closer to 1 m, and the magnitude of carbon storage is probably ≥10000 Tg C.

Projected climate changes caused by greenhouse warming are expected to alter processes related to carbon storage in wetlands. Higher temperatures should increase primary production, but also increase decomposition rates in wetland soils. In TSWs, the net effect is expected to be minor in light of the limited relationship between annual average temperatures and soil carbon sequestration rate. Regional increases in aridity will result in lower water tables in inland peatlands and freshwater wetlands, increasing decomposition and release of $CO_2$ and $CH_4$ [*Gorham*, 1991]. This effect is not expected in tidal wetlands, as water tables are controlled by tidal flooding regimes, but increases in aridity great enough to cause shifts in TSWs from vegetated systems to salt flats would result in local losses of this carbon sink. However, salt marshes exist in areas with high evapotranspiration, such as the Tijuana Estuary on the Mexico/USA border [*Cahoon et al*., 1996] and the Rhone Delta [*Hensel et al.,* 1999] where soil carbon accumulation rates are 343 and 161 g $m^{-2}$ $yr^{-1}$, respectively.

Greenhouse warming is likely to have the greatest impact on TSWs through an acceleration in rates of sea-level rise. Since TSWs vertically accumulate soil roughly in equilibrium with sea-level rise, rates of soil carbon sequestration and the magnitude of the soil carbon pool also will increase. In addition, TSWs can expand inland over terrestrial soils that have a lower carbon storage capacity. However, there is a limit to the rates at which TSWs can vertically accrete, and submerged salt marsh peats found on the inner Scotian shelf [*Shaw and Forbes*, 1990] provide striking evidence that rapid sea level rise exceeded the rate of marsh elevation increase during the early Holocene. Where there is an accretion deficit, soil surfaces become submerged and edges of the remaining wetland are subject to lateral erosion, releasing carbon stored from their deposits. Wetland loss is expected to be particularly prevalent where coastal development limits the landward migration of the wetland [*Working Group on Sea Level Rise and Wetland Systems*, 1997] or where disturbances to hydrologic or sedimentological regimes prevent the wetland from adjusting to sea level rise [e.g., *Templet and Meyer-Anrendt*, 1988; *Kearney and Stevenson*, 1991].

## Acknowledgements

Support for this research came from Quebec's FCAR, the Natural Sciences and Engineering Research Council of Canada, the Climate Change Action Fund of Canada, the United States Global Change Research Program, and the Long Island Sound Research Fund of the Connecticut Department of Environmental Protection (grant CWF319-R). The comments of J. Adams and two anonymous reviewers helped to improve this manuscript.

## References

Adams, J.M., H. Faure, L. Faure-Denard, J.M. McGlade, and F.I. Woodward, Increases in terrestrial carbon storage from the last glacial maximum to the present, *Nature,* 348, 711-714, 1990.

Allen, S. E., *Chemical Analysis of Ecological Materials*, Blackwell Scientific, London, 1974.

Alongi, D.M., F.Tirendi, P. Dixon, L.A. Trott, and G.J. Brunskill, Sources, sinks, and export of organic carbon fluxes in adjacent tropical nearshore sediments, *Marine Ecology Progress Series,* 56, 133-144, 1999.

Anisfeld, S.C., M.J. Tobin, and G. Benoit,. Sedimentation rates in flow-restricted and restored salt marshes in Long Island Sound, *Estuaries*, 22, 231-244, 1999.

Atjay, G.L., P. Ketner, and P. Duvigneaud, Terrestrial primary production and phytomass. in *The Global Carbon Cycle*, edited by B. Bolin, E.T. Degens, S. Kempe, and P. Ketner, pp. 129-181, John Wiley & Sons, Chichester, 1979.

Bartlett, K.B. and R.C. Harris, Review and assessment of methane emissions form wetlands. *Chemosphere,* 26, 261-320, 1993.

Brunskill, G.J., Zagorskis, and J. Pfitzner, Carbon burial rates in sediments, and a carbon mass balance, of the Herbert River region of the Great Barrier Reef continental shelf, north Queensland, Australia, *Estuar. Coast. Shelf Sci.,* 54, 677-700, 2002.

Bryant, J.C. and R. H. Chabreck, Effects of impoundment on vertical accretion of coastal marsh, *Estuaries,* 21, 416-422, 1998.

Cahoon, D.R., Recent accretion in two managed marsh impoundments in coastal Louisiana, *Ecol. Appl.,* 41, 166-176, 1994.

Cahoon, D.R. and J.C. Lynch, Vertical accretion and shallow subsidence in a mangrove forest of southwestern Florida, U.S.A., *Mangroves and Salt Marshes,* 1, 173-186, 1997.

Cahoon, D.R. and R.E. Turner, Accretion and canal impacts in a rapidly subsiding wetland II. feldspar marker horizon technique, *Estuaries,* 12, 260-268, 1989.

Cahoon, D.R., J.C. Lynch, and A. Powell, Marsh vertical accretion in a Southern California estuary, U.S.A., *Estuar. Coast. Shelf Sci.,* 43, 19-32, 1996.

Callaway, J.C., R.D. DeLaune, and W.H. Patrick, Jr., Sediment accretion rates from four coastal wetlands along the Gulf of Mexico, *J. Coastal Res,* 13, 181-191, 1997.

Callaway, J.C., R.D. DeLaune, and W.H. Patrick, Jr., Chernobyl $^{137}$Cs used to determine sediment accretion rates at selected northern European wetlands, *Limnol. Oceanogr.,* 41, 444-450, 1996.

Cardona, P. and L. Botero, Soil characteristics and vegetation structure in a heavily deteriorated mangrove forest in the Caribbean coast of Colombia, *Biotropica,* 30, 24-34, 1998.

Chen, R. and R.R. Twilley, A simulation model of organic matter and nutrient accumulation in mangrove wetland soils, *Biogeochem.,* 44, 93-118, 1999.

Chmura, G.L. and Hung, G. *in review*. Controls on salt marsh accretion: a test in salt marshes of Eastern Canadian. *Estuaries*

Chmura, G.L., A. Coffey, and R. Crago, Variation in surface sediment deposition on salt marshes in the Bay of Fundy, *J. Coastal Res.,* 17, 221-227, 2001.

Church, J.A., J.M. Gregory, P. Huybrechts, M. Kuhn, K. Lambeck, M.T. Nhuan, D. Qin, and P.L. Woodworth, Chapter 11 Changes in sea level, in *Climate Change 2001: The Scientific Basis Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC)* edited by Houghton, J.T., Y. Ding, D.J. Griggs, M. Noguer, P. J. van der Linden and D. Xiaosu, Cambridge University Press, UK. 2001.

Connor, R.F., G.L. Chmura, and C.B. Beecher, Carbon accumulation in Bay of Fundy salt marshes: implications for restoration of reclaimed marshes, *Global Biogeochem. Cy.,* 15, 943-954, 2001.

Craft, C.B., E.D. Seneca, and S.W. Broome, Loss on ignition and Kjeldahl digestion for estimating organic carbon and total nitrogen in estuarine marsh soils: calibration with dry combustion, *Estuaries,* 14, 175-179, 1991.

Craft, C.B., E.D. Seneca, and S.W. Broome, Vertical accretion in microtidal regularly and irregularly flooded estuarine marshes, *Estua.r Coast. Shelf Sci.,* 37, 371-386, 1993.

DeLaune, R.D., W.H. Patrick, Jr., and R.J. Buresh, Sedimentation rates determined by $^{137}$Cs dating in a rapidly accretion salt marsh, *Nature,* 275, 532-533, 1978.

DeLaune, R.D., W.H. Patrick, Jr., C.W. Lindau, and C.J. Smith, in *Soils and the Greenhouse Effect*, edited by A.F. Bouwman, pp. 497-502, Wiley, New York, 1990.

Dijkema, K.S., Geography of salt marshes in Europe, *Z. Geomorphol.,* 31, 489-499, 1987.

Dixon, R.K. and O.N. Krankina, Can the terrestrial biosphere be managed to conserve and sequester carbon? In: Carbon sequestration in the biosphere: Processes and products. *NATO ASI Series. Series 1. Global Environmental Change*, 33, 153 – 179, 1995.

Field, D.W., A.J. Reyer, P.V. Genovese, and B.D. Shearer, *Coastal Wetlands of the United States,* U.S. Department of Commerce, NOAA, NOS, 1991.

French, J.R. and T. Spencer, Dynamics of sedimentation in a tide-dominated backbarrier salt marsh, Norfolk, UK, *Mar. Geol.,* 110, 315-331, 1993.

Giani, L., K. Dittrich, A. Martsfeld-Hartmann, and G. Peters, Methanogenesis in saltmarsh soils of the North Sea coast of Germany, *Eur. J. Soil Sci.,* 47, 175-182, 1996.

Gorham, E., Northern peatlands: role in the carbon cycle and probably responses to climatic warming, *Ecol. Appl.,* 1, 182-195, 1991.

Hanson, A.R. and L. Calkins, Wetlands of the Maritime Provinces: revised documentation for the wetlands inventory, *Technical Report 267,* Canadian Wildlife Service Atlantic Region, Sackville, New Brunswick, 1996.

Hatton, R. S., Aspects of marsh accretion and geochemistry: Barataria Basin, Louisiana, M.S..thesis, Louisiana State University, 1981.

Hensel, P.F., J.W. Day Jr., and D. Pont, Wetland vertical accretion and soil elevation change in the Rhone River Delta, France: the importance of riverine flooding, *J. Coastal Res.,* 15, 668-681, 1999.

Houghton, R.A., E.A. Davidson, and G.M. Woodwell, Missing sinks, feedbacks, and understanding the role of terrestrial ecosystems in the global carbon cycle. *Global Biogeochem. Cy.,* 12, 25-34, 1998.

Kearney, M.S. and J.C. Stevenson, Island land loss and marsh vertical accretion rate evidence for historical sea-level changes in Chesapeake Bay, *J Coast. Res.,* 7, 403-415, 1991.

Letourneau, G. and M. Jean, Cartographie des marais, marecages et herbiers aquatiques le long du Saint-Laurent par teledetection aeroportee, *Environment Canada Rapport ST-61*. Centre Saint-Laurent, 1996.

Lynch, J. C., *Sedimentation and nutrient accumulation on mangrove ecosystems of the Gulf of Mexico*. M.S. thesis, University of Southwestern Louisiana, 1989.

McCaffrey, R.J. and J. Thomson, A record of the accumulation of sediment trace metals in a Connecticut salt marsh, *Advances in Geophysics,* 22, 165-236, 1980.

Magenheimer, J.F., T.R. Moore, G.L. Chmura, and R. Daoust, Methane and carbon dioxide flux from a macrotidal salt marsh, Bay of Fundy, New Brunswick, *Estuaries,* 19, 139-145, 1996.

Markewich, H.W., L.D. Britsch, G.R. Buell, D.L. Dillon, C.M. Fraticelli, T.L. Fries, J.P. McGeehin, J.B. Pracht, J.A. Robbins, B.M. Samuel, and J.H. Wrenn, Carbon storage and late Holocene chronostratigraphy of a Mississippi River Deltaic Marsh, St. Bernard Parish, Louisiana, *U.S. Geological Survey Open-File Report 98-36,* 1998.

Middleton, B.A. and K.L. McKee, Degradation of mangrove tissues and implications for peat formation in Belizean island forests, *J. Ecol.*, 89, 818-828, 2001.

Mitsch, W.J. and J.G. Gosselink, *Wetlands*, John Wiley & Sons, New York, 2000.

Morris, J.T. and A. Jensen, The carbon balance of grazed and non-grazed *Spartina anglica* saltmarshes at Skalligen, Denmark, *J. Ecol.,* 86, 229-242, 1998.

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon, Responses of coastal wetlands to rising sea level, *Ecol.,* 83, 2869-2877, 2002.

Naidoo, G., Mangrove soils of the Beachwood Area, Durban, *Journal of South African Botany,* 46, 293-304, 1980.

O'Callaghan, M. Saltmarshes – a highly specialized environment, *Custos,* 18, 58-60, 1990.

Odum, E. P., *Fundamentals of Ecology,* W.B. Saunders Company, Philadelphia, 1959.

Oenema O. and R.D. Delaune, Accretion rates in salt marshes in the Eastern Scheldt, south-west Netherlands, *Estuar. Coast. Shelf Sci.,* 26, 379-394, 1988.

Olson, J.S., J.A. Watts, and L.J. Allison, *Carbon in Live Vegetation of Major World Ecosystems*. Oak Ridge, Tennessee: Oak ridge National Laboratory, ORNL-5862, 1983.

Orson, R.A., R.S. Warren, and W.A. Niering, Interpreting sea level rise and rates of vertical marsh accretion in a Southern New England tidal salt marsh, *Estuar. Coast. Shelf Sci.*, 47, 419-429, 1998.

Pacala, S.W. G.C. Hurtt, R.A. Houghton, R.A. Birdsey, L. Heath, E.T. Sundquist, R.F. Stallard, D. Baker, P. Peylin, P. Moorcroft, J. Caspersen, E. Shevliakova, M.E. Harmon, S.-M. Fan, J.L. Sarmiento, C. Goodale, C.B. Field, M. Gloor and D. Schimel*,* Consistent land- and atmosphere-based U.S. carbon sink estimates, *Science,* 292, 2316-2320, 2001.

Patrick, W.H., Jr. and R.D. DeLaune, Subsidence, accretion, and sea level rise in South San Francisco Bay marshes, *Limnol. Oceanogr.,* 35, 1389-1395, 1990.

Ramaswamy, V., O. Boucher, J. Haigh, D. Hauglustaine, J. Haywood, G. Myhre, T. Nakajima, G.Y. Shi, S. Solomon, Radiative forcing of climate, in *Climate Change 2001: The Scientific Basis, Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change*, edited by J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P.J. vander Linden, X. Dui, K. Maskell, and C.A. Johnson, Cambridge University Press, Cambridge, 2001.

Roman, C.T., A. Peck, J.R. Allen, J.W. King, and P.G. Appleby, Accretion of a New England (U.S.A.) salt marsh in response to inlet migration, storms, and sea-level rise, *Estuarine Coast. Shelf Sci.,* 45, 717-727, 1997.

Ross, M.S., M.S. Meeder, J.P. Sah, P.L. Ruiz, and G.J. Telesnicki, The Southeast Saline Everglades revisited: 50 years of coastal vegetation change, *J. Vegetation Sci.,* 11, 101-112, 2000.

Roulet, N.T., Peatlands, carbon storage, greenhouse gases, and the Kyoto Protocol: prospects, and significance for Canada, *Wetlands,* 20, 605-615, 2000.

Sahagian, D. and J. Melack, Global wetland distribution and functional characterization: trace gases and the hydrologic cycle, *IGBP report* 46, 1988.

Schimel, D.S., B.H. Braswell, E.A. Holland, R. McKeown, D.S. Ojima, T.H. Painter, W.J. Parton, and A.R. Townsend, Climatic, edaphic and biotic controls over carbon and turnover of carbon in soils. *Global Biogeochem. Cyc.*, 8, 279-293.

Schindler, D.W., The mysterious missing sink, *Nature,* 398, 105-107, 1999.

Scott, D.B. and D.A. Greenberg, Relative sea level rise and tidal development in the Fundy tidal system, *Can. J. Earth Sci.,* 20, 154-1564, 1983.

Shaw, J. and D.L. Forbes, Short- and long-term relative sea level trends in Atlantic Canada, in *Proceedings, Canadian Coastal Conference, Kingston, Ontario May 8-11, 1990*, edited by M.H. Davies, pp. 291-306, National Research Council of Canada, Ottawa, 199xx.

Smith, C.J., R.D. DeLaune, and W.H. Patrick, Jr., Nitrous oxide emission from Gulf Coast wetlands, *Geochim. Cosmochim. Ac.,* 47, 1805-1814, 1983.

Smith, K.K., R.E. Good, and N.F. Good, Production dynamics for above and belowground components of a New Jersey *Spartina alterniflora* tidal marsh, *Est. Coast. Mar. Sci.*, 9, 189-201, 1979.

Spaulding, M.D., F. Blasco, and C.D. Field, *World Mangrove Atlas,* The International Society for Mangrove Ecosystems, Okinawa, Japan, 1997.

Templet, P. H. and K. Meyer-Arendt, Louisiana wetland loss: a regional water management approach to the problem. *Environ. Mgt*. 12, 181-192, 1988.

Turner, E.G., Geographic variations in salt marsh macrophyte production: A review, *Contribut. Mar. Science*, 20, 47-68, 1976.

Watson, R.T., I.R. Noble, B. Bolin, N.H. Ravindranath, D.J. Verardo, and D.J. Dokken, *Land use, land-use change, and forestry*, Intergovernmental Panel on Climate Change Special Report, 2000, www,grida.no/climate/ipcc/land_use/223.htm.

Whiting, G.J. and J.P. Chanton, Greenhouse carbon balance of wetlands: methane emission versus carbon sequestration. *Tellus* 53B, 521-528, 2001.

Woodroffe, C.D., M.E. Mulrennan, and J. Chappell, Estuarine infill and coastal progradation, southern van Diemen Gulf, northern Australia. *Sed. Geol.*, 83, 257-275, 1993.

Working Group on Sea Level Rise and Wetland Systems, Forecasting the fate of coastal wetlands, *EOS* 78(25):257/262, 1997.

Table 1. Soil carbon density, soil carbon accumulation rates and climate normals at salt marsh sites. Carbon values are calculated according to the formula of Craft et al. [1991]

| location: site name or core number, |  |  | density | rate | C data | avg annual temp °C |  |  | normals |
|---|---|---|---|---|---|---|---|---|---|
| state/province, country | latitude | longitude | g cm$^{-3}$ | g m$^{-2}$ yr$^{-1}$ | source* | min | max | overall | period** |
| *Gulf of Mexico* | *°N* | *°W* |  |  |  |  |  |  |  |
| Aransas, TX | 28.4 | 96.8 | 0.040 | 178 | 1 | 17.2 | 25.3 | 21.2 | 1 |
| Fina la Terre, LA | 29.0 | 91.0 | 0.027 | 136 | 2 | 15.5 | 24.5 | 20.0 | 1 |
| Fina la Terre, LA | 29.0 | 91.0 | 0.018 | 18 | 2 | 15.5 | 24.5 | 20.0 | 1 |
| San Bernard, TX | 29.1 | 95.6 | 0.033 | 203 | 1 | 16.6 | 25.3 | 20.9 | 1 |
| Old Oyster Bayou, LA | 29.3 | 91.1 | 0.019 | 84 | 3 | 15.5 | 24.5 | 20.0 | 1 |
| Bayou Chitigue, LA | 29.3 | 90.6 | 0.016 | 516 | 3 | 15.4 | 25.3 | 20.4 | 1 |
| Rockefeller Refuge, LA | 29.5 | 92.7 | 0.028 | 309 | 2 | 15.2 | 25.2 | 20.2 | 1 |
| Rockefeller Refuge, LA | 29.5 | 92.7 | 0.033 | 27 | 2 | 15.2 | 25.2 | 20.2 | 1 |
| Lafourche Parish, LA | 29.5 | 90.3 | 0.019 | 186 | 4 | 15.4 | 25.3 | 20.4 | 1 |
| Cameron Parish, LA | 29.5 | 93.2 | 0.010 | 41 | 4 | 15.9 | 24.7 | 20.3 | 2 |
| Cameron Parish, LA | 29.5 | 93.2 | 0.010 | 115 | 4 | 15.9 | 24.7 | 20.3 | 2 |
| Barataria Basin, LA | 29.5 | 90.0 | 0.013 | 185 | 5 | 15.4 | 25.3 | 20.4 | 1 |
| Barataria Basin, LA | 29.5 | 90.0 | 0.012 | 71 | 5 | 15.4 | 25.3 | 20.4 | 1 |
| Barataria Basin, LA | 29.5 | 90.0 | 0.012 | 93 | 5 | 15.4 | 25.3 | 20.4 | 1 |
| unit 1, Marsh Island Refuge, LA | 29.5 | 91.9 | 0.110 | 318 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| unit 1, Marsh Island Refuge, LA | 29.5 | 91.9 | 0.109 | 763 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| unit 15, Rockefeller Wildlife Refuge, LA | 29.6 | 92.7 | 0.120 | 349 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| unit 15, Rockefeller Wildlife Refuge, LA | 29.6 | 92.7 | 0.119 | 657 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| Three Bayous, LA | 29.6 | 90.1 | 0.014 | 116 | 3 | 15.2 | 25.2 | 20.2 | 1 |
| Rockefeller Wildlife Refuge unit 14, LA | 29.7 | 92.7 | 0.116 | 337 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| Rockefeller Wildlife Refuge unit 14, LA | 29.7 | 92.7 | 0.093 | 448 | 6 | 15.2 | 25.2 | 20.2 | 1 |
| McFaddin National Wildlife Refuge, TX | 29.7 | 94.1 | 0.012 | 95 | 3 | 15.1 | 25.6 | 20.4 | 1 |
| Sabine National Wildlife Refuge unit 3, LA | 29.9 | 93.5 | 0.190 | 1713 | 6 | 15.9 | 24.7 | 20.3 | 1 |
| Sabine National Wildlife Refuge unit 3, LA | 29.9 | 93.5 | 0.121 | 714 | 6 | 15.9 | 24.7 | 20.3 | 1 |
| St. Bernard Parish, LA | 30.0 | 89.9 | 0.028 | 140 | 7 |  |  |  |  |
| St. Marks, FL | 30.1 | 84.2 | 0.025 | 44 | 3 | 14.1 | 25.6 | 19.9 | 2 |
| Biloxi Bay, MS | 30.4 | 88.9 | 0.027 | 153 | 1 | 15.0 | 24.4 | 19.7 | 2 |
| *Northeastern Atlantic* | *°N* | *°E* |  |  |  |  |  |  |  |
| St. Annaland, The Netherlands | 51.5 | 4.1 | 0.041 | 277 | 8 |  |  |  |  |
| St. Annaland, The Netherlands | 51.5 | 4.1 | 0.041 | 139 | 8 |  |  |  |  |
| Scheldt, The Netherlands | 51.5 | 4.1 | 0.029 | 587 | 9 |  |  |  |  |
| Scheldt, The Netherlands | 51.5 | 4.1 | 0.020 | 650 | 9 |  |  |  |  |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Dengie Marsh, England, UK | 51.7 | 0.9 | 0.041 | 187 | 8 | 7.2 | 12.9 | 10.1 | 2 |
| Dengie Marsh, England, UK | 51.7 | 0.9 | 0.041 | 139 | 8 | 7.2 | 12.9 | 10.1 | 2 |
| Dengie Marsh, England, UK | 51.7 | 0.9 | 0.041 | 159 | 8 | 7.2 | 12.9 | 10.1 | 2 |
| Dengie Marsh, England, UK | 51.7 | 0.9 | 0.041 | 110 | 8 | 7.2 | 12.9 | 10.1 | 2 |
| Hut marsh, UK | 53.0 | 0.7 | 0.027 | 165 | 10 | | | | |
| Hut marsh, UK | 53.0 | 0.7 | 0.027 | 77 | 10 | | | | |
| Skallingen, Denmark | 55.5 | 8.4 | 0.021 | | 11 | | | | |
| Skallingen, Denmark | 55.5 | 8.4 | 0.027 | | 11 | | | | |
| *Mediterranean* | *°N* | *°E* | | | | | | | |
| Rhone Delta, France | 43.3 | 4.6 | 0.073 | 161 | 12 | | | | |
| *Northeastern Pacific* | *°N* | *°W* | | | | | | | |
| Tijuana Slough, CA | 32.5 | 117.1 | 0.018 | 343 | 13 | 12.8 | 22.4 | 17.6 | 1 |
| Tijuana Slough, CA | 32.6 | 117.1 | 0.017 | 43 | 14 | 12.8 | 22.4 | 17.6 | 1 |
| Tijuana Slough, CA | 32.6 | 117.1 | 0.040 | | 14 | 12.8 | 22.4 | 17.6 | 1 |
| Alviso, San Francisco Bay, CA | 37.5 | 122.0 | 0.009 | 385 | 15 | 10.4 | 20.6 | 15.5 | 1 |
| Bird Island, San Francisco Bay, CA | 37.6 | 122.2 | 0.014 | 54 | 15 | 10.4 | 20.6 | 15.5 | 1 |
| Uculet, BC | 48.9 | 125.5 | 0.017 | | 16 | 5.5 | 12.7 | 9.1 | 4 |
| *Northwestern Atlantic* | *°N* | *°W* | | | | | | | |
| Cedar Island National Wildlife Refuge, NC | 35.0 | 76.4 | 0.022 | 70 | 3 | 12.1 | 22.0 | 17.0 | 2 |
| Oregon Inlet, NC | 35.9 | 75.6 | | 59 | 17 | 12.3 | 20.8 | 16.6 | 1 |
| Oregon Inlet, NC | 35.9 | 75.6 | | 21 | 17 | 12.3 | 20.8 | 16.6 | 1 |
| Jacob's Creek, NC | 35.3 | 76.8 | | 146 | 17 | 12.3 | 20.8 | 16.6 | 1 |
| Jacob's Creek, NC | 35.3 | 76.8 | | 107 | 17 | 12.3 | 20.8 | 16.6 | 1 |
| MC4, Chesapeake Bay, MD | 38.3 | 75.9 | 0.040 | 308 | 18 | 8.7 | 20.2 | 14.4 | 1 |
| MCL8, Chesapeake Bay, MD | 38.3 | 75.9 | 0.027 | 213 | 18 | 8.7 | 20.2 | 14.4 | 1 |
| MCL15, Chesapeake Bay, MD | 38.3 | 75.9 | 0.044 | 340 | 18 | 8.7 | 20.2 | 14.4 | 1 |
| Sybil 1, CT | 41.2 | 72.6 | 0.054 | 136 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Hoadley 1, CT | 41.2 | 72.0 | 0.037 | 154 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Hoadley 2, CT | 41.2 | 72.0 | 0.040 | 169 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Hoadley 3, CT | 41.2 | 72.0 | 0.035 | 114 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| East River 1, CT | 41.2 | 72.7 | 0.030 | 134 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| East River 2, CT | 41.2 | 72.7 | 0.060 | 204 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Sluice 1, CT | 41.2 | 72.7 | 0.026 | 99 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Sluice Core 2, CT | 41.2 | 72.7 | 0.045 | 85 | 19 | 5.3 | 15.2 | 10.3 | 1 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Leetes 1, CT | 41.2 | 72.7 | 0.039 | 153 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Leetes 2, CT | 41.2 | 72.7 | 0.030 | 93 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Sybil 2, CT | 41.2 | 72.6 | 0.029 | 72 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Sybil 3, CT | 41.2 | 72.6 | 0.046 | 116 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Branford River 1, CT | 41.2 | 72.6 | 0.029 | 182 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Branford River 2, CT | 41.2 | 72.6 | 0.026 | 181 | 19 | 5.3 | 15.2 | 10.3 | 1 |
| Farm River, CT, CT | 41.2 | 72.9 | 0.025 | 70 | 20 | 5.3 | 15.2 | 10.3 | 1 |
| Bloom's Point, Little Narragansett Bay, CT | 41.3 | 71.9 | 0.036 | 62 | 21 | 5.3 | 15.2 | 10.3 | 1 |
| Inlet 1, Nauset Bay, MA | 41.5 | 70.0 | 0.028 | 105 | 22 | 5.7 | 14.0 | 9.8 | 1 |
| Nauset Bay, MA | 41.5 | 70.0 | 0.041 | 155 | 22 | 5.7 | 14.0 | 9.8 | 1 |
| Wells National Estuarine Research Reserve, ME | 43.3 | 70.5 | 0.020 | | 16 | 1.5 | 12.5 | 7.0 | 1 |
| Dipper a, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.048 | 85 | 23 | -0.2 | 9.8 | 4.8 | 5 |
| Dipper d, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.033 | 63 | 23 | -0.2 | 9.8 | 4.8 | 5 |
| Little Lepreau, Bay of Fundy, NB | 45.1 | 66.5 | 0.059 | 80 | 23 | -0.2 | 9.8 | 4.8 | 5 |
| Chance Harbour, Bay of Fundy, NB | 45.1 | 66.3 | 0.038 | 72 | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH SA 3, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.035 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH SA 2, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.034 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH SA1, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.036 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH Sp3, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.047 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH Sp2, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.036 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| DH Sp1, Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.043 | | 23 | -0.2 | 9.8 | 4.8 | 5 |
| Bocabec River, Bay of Fundy, NB | 45.1 | 67.0 | 0.034 | 456 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Bocabec River, Bay of Fundy, NB | 45.1 | 67.0 | 0.046 | 113 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.030 | 445 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Dipper Harbour, Bay of Fundy, NB | 45.1 | 66.4 | 0.033 | 94 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Cape Enrage, Bay of Fundy, NB | 45.6 | 64.8 | 0.018 | 582 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Cape Enrage, Bay of Fundy, NB | 45.6 | 64.8 | 0.023 | 186 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Lorneville, Bay of Fundy, NB | 45.2 | 66.2 | 0.028 | 277 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Lorneville, Bay of Fundy, NB | 45.2 | 66.2 | 0.033 | 330 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| St. Martins, Bay of Fundy, NB | 45.3 | 65.5 | 0.027 | 265 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| St. Martins, Bay of Fundy, NB | 45.9 | 65.5 | 0.024 | 928 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Wood Point, Bay of Fundy, NB | 45.8 | 64.4 | 0.026 | 264 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Wood Point, Bay of Fundy, NB | 45.8 | 64.4 | 0.025 | 253 | 16 | -0.2 | 9.8 | 4.8 | 5 |
| Kouchigouguacis Lagoon, Gulf of St. Lawrence, NB | 46.7 | 64.9 | 0.031 | 102 | 24 | 0.6 | 10.0 | 5.3 | 6 |
| Bay St-Louis, Gulf of St. Lawrence, NB | 46.8 | 64.9 | 0.032 | 93 | 24 | 0.6 | 10.0 | 5.3 | 6 |
| Tabusintac Bay, Gulf of St. Lawrence, NB | 47.4 | 65.0 | 0.033 | 66 | 24 | 0.6 | 10.0 | 5.3 | 6 |
| Malpeque Bay, Gulf of St. Lawrence, PEI | 46.5 | 63.7 | 0.029 | 71 | 24 | 0.9 | | | 3 |
| Brackley Bay Gulf of St. Lawrence, PEI | 46.4 | 63.2 | 0.035 | 89 | 24 | 0.9 | | | 3 |
| Pubnico Harbour, Gulf of Maine, NS | 43.6 | 65.3 | 0.041 | 113 | 24 | 2.8 | 10.7 | 6.8 | 7 |
| Cheboque Harbour, Gulf of Maine, NS | 43.8 | 66.1 | 0.045 | 75 | 24 | 2.8 | 10.7 | 6.8 | 7 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Little River Harbour, Gulf of Maine, NS | 43.7 | 66.1 | 0.078 | 304 | 24 | 2.8 | 10.7 | 6.8 | 7 |
| Cole Harbour, NS | 44.7 | 63.4 | 0.042 | 161 | 24 | 0.4 | 11.6 | 6.0 | 1 |
| Lawrencetown Lake, NS | 44.7 | 63.4 | 0.024 | 60 | 24 | 0.4 | 11.6 | 6.0 | 1 |
| Chezzetcook Inlet, NS | 44.7 | 63.4 | 0.038 | 106 | 24 | 0.4 | 11.6 | 6.0 | 1 |
| Rustico Bay, PEI | 46.4 | 63.2 | 0.034 | 125 | 24 | 0.9 | | | 3 |

*1) Callaway et al., 1997; 2) Cahoon, 1994; 3) Cahoon and Lynch, unpublished; 4) Cahoon and Turner, 1989; 5) Hatton, 1981; 6) Bryant and Chabreck; 1998; 7) Markewich et al., 1998; 8) Callaway et al., 1996; 9) Oenema and Delaune; 1988; 10) French and Spencer, 1993; 11) Morris and Jensen, 1998; 12) Hensel et al., 1999; 13) Cahoon et al., 1996; 14) Cahoon, unpublished; 15) Patrick and DeLaune, 1990; 16) Chmura, unpublished; 17) Craft et al., 1993; 18) Kearney and Stevenson, 1991; 19) Anisfeld, unpublished; 20) McCaffrey and Thomson, 1980; 21) Orson et al., 1998; 22) Roman et al., 1997; 23) Connor et al., 2001; 24) Chmura and Hung, in review.

**Climate normals were calculated over different periods: 1) 1961-1990, 2) 1971-2000, 3) over history of station (93 yr), 4) 1957/59-1990, 5) 1946-1990, 6) 1965-1990, 7) 1940-1990, and 8) 1951-1980.

Table 2. Soil carbon density, soil carbon accumulation rates and climate normals at mangrove sites.

| location: site name or core number, | | | density | rate | C data | avg annual temp °C | | | normals |
|---|---|---|---|---|---|---|---|---|---|
| state/province, country | latitude | longitude | g cm$^{-3}$ | g m$^{-2}$ yr$^{-1}$ | source* | min | max | overall | period** |
| *Gulf of Mexico* | *°N* | *°W* | | | | | | | |
| CAR, Columbia | 11.0 | 74.2 | 0.071 | | 4 | | | | |
| RIN, Columbia | 11.0 | 74.2 | 0.061 | | 4 | | | | |
| HON, Columbia | 11.0 | 74.2 | 0.058 | | 4 | | | | |
| Terminos Lagoon, Boca Chica, Mexico | 18.7 | 91.5 | 0.047 | 308 | 5 | 21.3 | 31.1 | 26.2 | 3 |
| Terminos Lagoon,Estero Pargo, Mexico | 18.7 | 91.5 | 0.052 | 194 | 5 | 21.3 | 31.1 | 26.2 | 3 |
| Terminos Lagoon,Estero Pargo, Mexico | 18.7 | 91.5 | 0.058 | 146 | 5 | 21.3 | 31.1 | 26.2 | 3 |
| Terminos Lagoon, Boca Chica, Mexico | 18.7 | 91.5 | 0.051 | 654 | 5 | 21.3 | 31.1 | 26.2 | 3 |
| FL keys: Lignumvitae, to Key Largo | 25.0 | 80.6 | 0.036 | 143 | 1 | 21.7 | 28.7 | 25.2 | 2 |
| FL keys: Lignumvitae, to Key Largo | 25.0 | 80.6 | 0.037 | 100 | 1 | 21.7 | 28.7 | 25.2 | 2 |
| S1, Shark River Estuary, FL | 25.0 | 80.8 | 0.051 | | 6 | 18.9 | 28.3 | 23.6 | 2 |
| S3, Shark River Estuary, FL | 25.0 | 81.1 | 0.039 | | 6 | 18.9 | 28.3 | 23.6 | 2 |
| S4, Shark River Estuary, FL | 25.0 | 81.1 | 0.046 | | 6 | 18.9 | 28.3 | 23.6 | 2 |
| S6, Shark River Estuary, FL | 25.0 | 81.1 | 0.050 | | 6 | 18.9 | 28.3 | 23.6 | 2 |
| Rookery Bay, FL (Fringe) | 26.0 | 81.7 | 0.036 | 265 | 7 | 18.9 | 28.3 | 23.6 | 2 |
| Rookery Bay, FL (Basin) | 26.0 | 81.7 | 0.066 | 381 | 7 | 18.9 | 28.3 | 23.6 | 2 |
| Rookery Bay, FL (Exposed Island) | 26.0 | 81.7 | 0.052 | 338 | 7 | 18.9 | 28.3 | 23.6 | 2 |
| Rookery Bay, FL (Sheltered Island) | 26.0 | 81.8 | 0.049 | 222 | 7 | 18.9 | 28.3 | 23.6 | 2 |
| southeast Everglades, FL | 25.3 | 80.6 | 0.040 | | 8 | 17.9 | 28.5 | 23.2 | 2 |
| southeast Everglades, FL | 25.3 | 80.6 | 0.033 | | 8 | 17.9 | 28.5 | 23.2 | 2 |
| southeast Everglades, FL | 25.3 | 80.6 | 0.027 | | 8 | 17.9 | 28.5 | 23.2 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.043 | 142 | 5 | 17.6 | 29.4 | 23.5 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.050 | 154 | 5 | 17.6 | 29.4 | 23.5 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.044 | 154 | 5 | 17.6 | 29.4 | 23.5 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.067 | 170 | 5 | 17.6 | 29.4 | 23.5 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.024 | 20 | 2 | 17.6 | 29.4 | 23.5 | 2 |
| Rookery Bay, FL | 26.0 | 81.7 | 0.033 | 39 | 2 | 17.6 | 29.4 | 23.5 | 2 |
| *Pacific and Indian Ocean* | *°N* | *°E* | | | | | | | |
| Kosrae | 5.3 | 163.0 | 0.023 | | 3 | 22.7 | 31.2 | 17.0 | 2 |
| Kosrae | 5.3 | 163.0 | 0.040 | | 3 | 22.7 | 31.2 | 16.6 | 1 |
| Kosrae | 5.3 | 163.0 | 0.031 | | 3 | 22.7 | 31.2 | 16.6 | 1 |

| | °S | °E | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| HM 2, Hinchinbrook Channel, Australia | 18.5 | 146.3 | | 67 | 9 | 18.8 | 28.8 | 16.6 | 1 |
| HMF 3, Hinchinbrook Channel, Australia | 18.5 | 146.3 | | 48 | 9 | 18.8 | 28.8 | 16.6 | 1 |
| HMF 4, Hinchinbrook Channel, Australia | 18.5 | 146.3 | | 336 | 9 | 18.8 | 28.8 | 14.4 | 1 |
| core 576, Herbert River region, Australia | 18.5 | 146.3 | | 26 | 10 | 18.8 | 28.8 | 14.4 | 1 |
| core 577, Herbert River region, Australia | 18.5 | 146.3 | | 168 | 10 | 18.8 | 28.8 | 14.4 | 1 |
| core 582, Herbert River region, Australia | 18.5 | 146.3 | | 84 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| core 583, Herbert River region, Australia | 18.5 | 146.3 | | 336 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| core 584, Herbert River region, Australia | 18.5 | 146.3 | | 300 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| core 585, Herbert River region, Australia | 18.5 | 146.3 | | 100 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| core 586, Herbert River region, Australia | 18.5 | 146.3 | | 71 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| core 587, Herbert River region, Australia | 18.5 | 146.3 | | 97 | 10 | 18.8 | 28.8 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.107 | | 11 | 17.0 | 25.0 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.105 | | 11 | 17.0 | 25.0 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.115 | | 11 | 17.0 | 25.0 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.109 | | 11 | 17.0 | 25.0 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.097 | | 11 | 17.0 | 25.0 | 10.3 | 1 |
| Umengi estuary, Durban, South Africa | 22.4 | 31.0 | 0.106 | | 11 | 17.0 | 25.0 | 10.3 | 1 |

*1) Callaway et al., 1997; 2) Cahoon and Lynch, unpublished; 3) Cahoon, unpublished; 4) Cardona and Botero, 1998; 5) Lynch, 1989; 6) Chen and Twilley, 1999; 7) Cahoon, and Lynch, 1997; 8) Ross et al., 2000; 9) Alongi et al., 1999; 10) Brunskill et al., 2002); 11) Naidoo, 1980.

**Climate normals were calculated over different periods: 1) 1961-1990, 2) 1971-2000, and 3) 1951-1980.

Table 3. Area of salt marsh reported.

| region | $km^2$ | sources |
|---|---|---|
| United States | 19,265 | Field et al., 1991 |
| Europe and Scandinavia | 2,302 | Dijkema, 1987 |
| Canada | 328 | Letourneau and Jean, 1996<br>Hanson and Calkins, 1996<br>Wetlands International Inventory |
| Tunisia | 59 | Wetlands International Inventory |
| Morrocco | 34 | Wetlands International Inventory |
| South Africa | 170 | O'Callaghan, 1990 |
| total | 21,988 | |

Table 4. Results of simple linear regression of soil carbon density and rate of sequestration to average annual temperatures at salt marshes and mangrove swamp sites (R = coefficient of correlation, P = probablility, N = sample number).

| | Avg Annual Temp | | | Annual Max | | | Annual Min | | |
|---|---|---|---|---|---|---|---|---|---|
| | R | P | N | R | P | N | R | P | N |
| *Carbon density* | | | | | | | | | |
| all sites | 0.23 | * | 122 | 0.21 | * | 122 | 0.25 | ** | 122 |
| mangroves | -0.70 | *** | 33 | -0.80 | *** | 33 | -0.49 | *** | 33 |
| salt marshes | 0.19 | ns | 90 | 0.18 | ns | 90 | 0.20 | ns | 93 |
| *S. alterniflora* marshes | -0.21 | ns | 20 | -0.21 | ns | 20 | -0.20 | ns | 20 |
| *S. patens* marshes | -0.50 | *** | 21 | -0.54 | *** | 32 | -0.54 | *** | 35 |
| *Rate of carbon sequestration* | | | | | | | | | |
| all sites | 0.05 | * | 108 | 0.06 | ns | 113 | 0.08 | ns | 113 |
| mangroves | 0.35 | ns | 28 | 0.24 | ns | 28 | 0.34 | ns | 28 |
| salt marshes | 0.14 | ns | 85 | 0.14 | ns | 85 | 0.15 | ns | 88 |
| *S. alterniflora* marshes | -0.45 | ns | 19 | -0.44 | ns | 19 | -0.47 | * | 20 |
| *S. patens* marshes | -0.13 | ns | 28 | -0.13 | ns | 28 | -0.11 | ns | 31 |

*P<0.05, **P<0.01, ***P<0.005, ns P>0.05

Regressions run using SPSS 11.0.

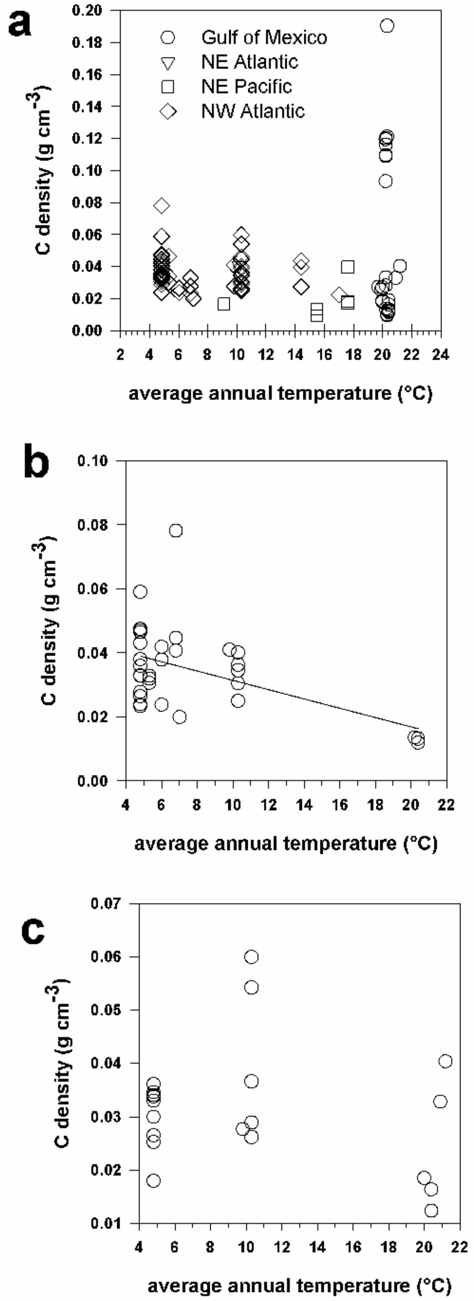
## Figure Captions

Figure 1. Relationship of soil carbon density to annual average temperature in soils of **a**) all salt marshes, **b**) *Spartina patens* marshes, and **c**) *Spartina alterniflora* marshes.

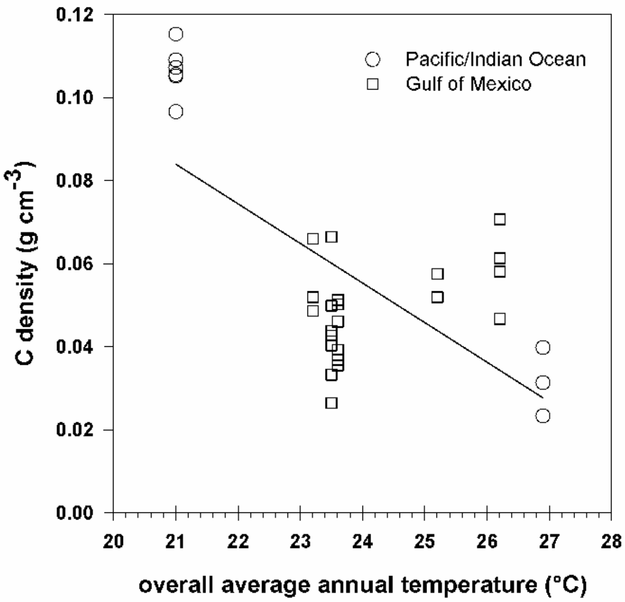
Figure 2. Relationship of soil carbon density to annual average temperature in soils of mangrove swamps of the Gulf of Mexico and Indian/South Pacific Ocean.

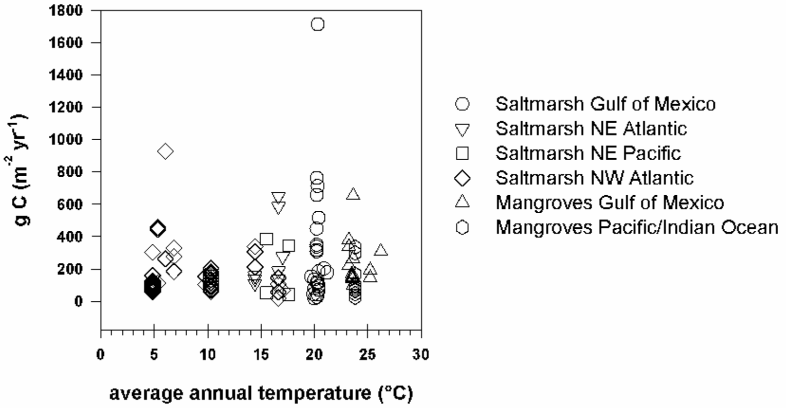
Figure 3. Relationship of soil carbon accumulation rates to annual average temperature in soils of all tidal saline wetlands.

Figure 4. Local variability in carbon storage in tidal saline wetland soils; **a**) C density in surface soils (0-2 cm depth); **b)** C accumulation rate in surface soils (0-2 cm depth). Vertical bars represents range of values, ● = average, averages from same wetland are connected. Averages within wetlands are significantly different if labeled with different letters. 1 and 2, fringe and basin mangroves, Shark River, FL; 3, *S. alterniflora* marsh, Old Oyster Bayou, LA; 4, *S. patens* marsh, Three Bayous, LA; 5 and 6, low and high elevation *S. alterniflora* zone, Bocabec River, Bay of Fundy, NB; 7, *S. patens* zone, Bocabec River; 8 and 9, low and high elevation *S. alterniflora* zone, Dipper Harbour, Bay of Fundy, NB; 10, *S. patens* zone, Dipper Harbour; 11 and 12, low and high elevation *S. alterniflora* zone, Cape Enrage, Bay of Fundy, NB; 13, *S. patens* zone, Cape Enrage.



Chmura et al fig 1


○ Pacific/Indian Ocean
□ Gulf of Mexico
C density (g cm$^{-3}$)
0.12
0.10
0.08
0.06
0.04
0.02
0.00
20
21
22
23
24
25
26
27
28
overall average annual temperature (°C)


○ Saltmarsh Gulf of Mexico
▽ Saltmarsh NE Atlantic
□ Saltmarsh NE Pacific
◇ Saltmarsh NW Atlantic
△ Mangroves Gulf of Mexico
⬡ Mangroves Pacific/Indian Ocean
g C (m$^{-2}$ yr$^{-1}$)
1800
1600
1400
1200
1000
800
600
400
200
0
0
5
10
15
20
25
30
average annual temperature (°C)

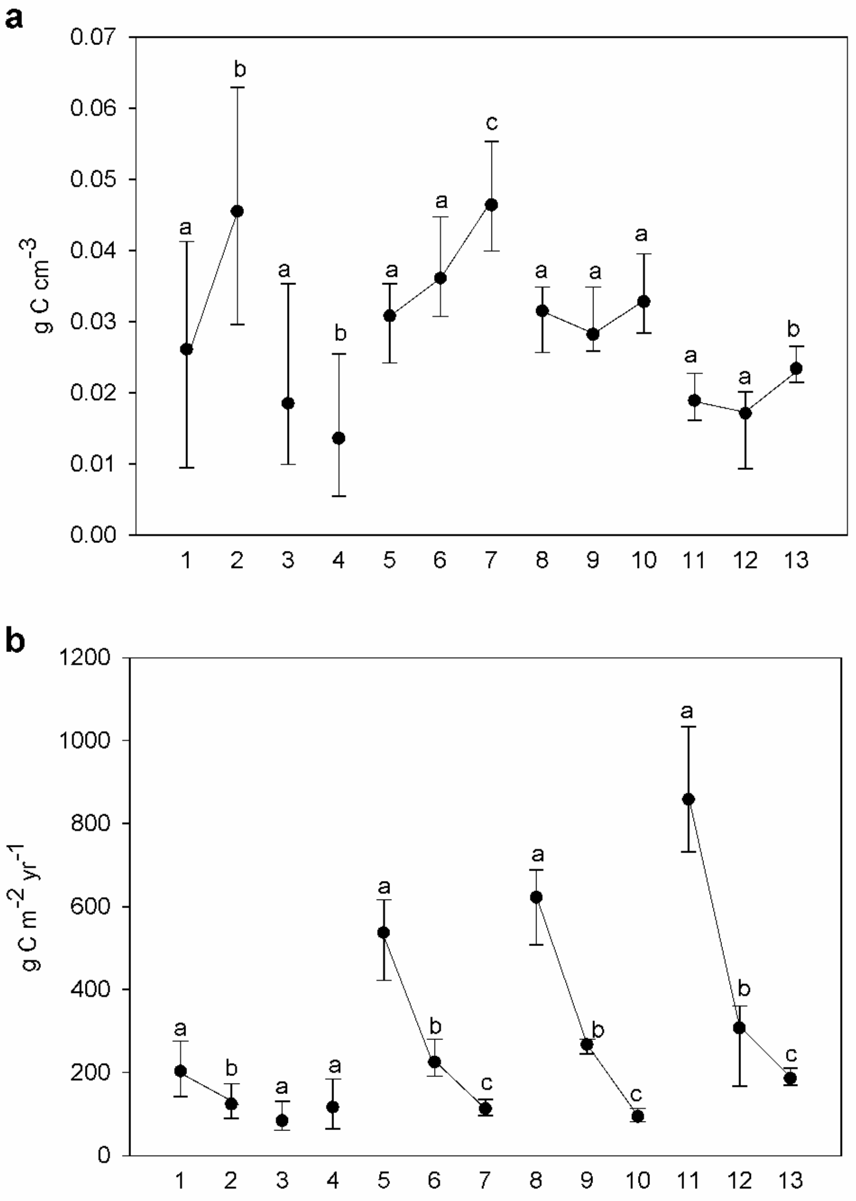

a
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
g C cm$^{-3}$
a
b
a
b
a
a
c
a
a
a
a
a
b
1
2
3
4
5
6
7
8
9
10
11
12
13
b
1200
1000
800
600
400
200
0
g C m$^{-2}$ yr$^{-1}$
a
b
a
a
a
b
c
a
b
c
a
b
c
1
2
3
4
5
6
7
8
9
10
11
12
13

fig 4