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REVIEW

Estimating the Provision of Ecosystem Services by Gulf of Mexico Coastal Wetlands

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Abstract Gulf of Mexico (GOM) coastal wetlands contribute to human well-being by providing many ecosystem services. The GOM region continues to experience substantial losses of coastal wetlands, but the magnitude of reduction in ecosystem services resulting from the loss of GOM coastal wetlands is unknown. To gain an appreciation of the impact of GOM coastal wetland loss on ecosystem services, recent literature was reviewed to derive quantitative estimates of ecosystem services provided by GOM coastal wetlands. GOM coastal wetlands provide essential habitat for the production of juvenile shrimp, which supports the GOM's most valuable commercial fishery; protect coastal communities from storm surge; improve water quality by removing nitrogen from surface waters; and are valuable sinks for greenhouse gases due to high rates of carbon sequestration combined with low rates of methane emission. Using 1998 to 2004 as a baseline, the potential loss of ecosystem services associated with loss of coastal wetlands is presented. Additional research is needed to quantify wetland services at multiple geospatial and socioeconomic scales, to determine the effect of wetland loss on ecosystem services, and to demonstrate the impact of future management decisions on the capacity of GOM coastal wetlands to provide services that affect human well-being.

Keywords Carbon sequestration · Nitrogen removal · Shrimp fishery · Storm surge protection

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Introduction

Coastal wetlands provide a wide range of ecosystem services (e.g., fishery support, storm surge protection, water quality improvement, wildlife habitat provision, recreational opportunities, and carbon sequestration) that support human well-being. More than half of the coastal wetlands in the U.S. (excluding Alaska) are located in the Gulf of Mexico (GOM) region (Field et al. 1991; Fig. 1). An estimated 14 million ha of wetlands (approx. 50%) were lost in the GOM states between 1780 and 1980 (Dahl 1990) and GOM coastal wetlands continue to suffer the highest loss rates in the U.S. (25,000 ha yr^{-1}; Stedman and Dahl 2008). Rapid development, population growth, hurricanes, sea-level rise, subsidence, and, now, the 2010 GOM oil spill, will contribute to further loss and degradation of GOM coastal wetlands, thus reducing the capacity of these wetlands to provide valued ecosystem services.

Wetland benefits are recognized conceptually by the public, government, and environmental managers; however, quantitative estimates of ecosystem services provided by coastal wetlands are not easily incorporated into monetary values or cost-benefit schemes for making decisions related to wetland protection, restoration, or mitigation. Certain wetlands are included in the legal definition of U.S. waters and, therefore, protected under multiple U.S. laws and statutes (e.g., the Clean Water Act). In March 2008, the U.S. Environmental Protection Agency (USEPA) and U.S. Army Corps of Engineers (ACOE) released a final revised compensatory mitigation rule to require that a mitigation wetland "...should be located where it is most likely to successfully replace lost functions and services' (73 Fed. Reg. 19688 (2008-04-10)). To implement this new rule successfully, however, both agencies recognized the need for rigorous methods to define clearly measurable and quantifiable units of ecosystem services so that they will be defensible from

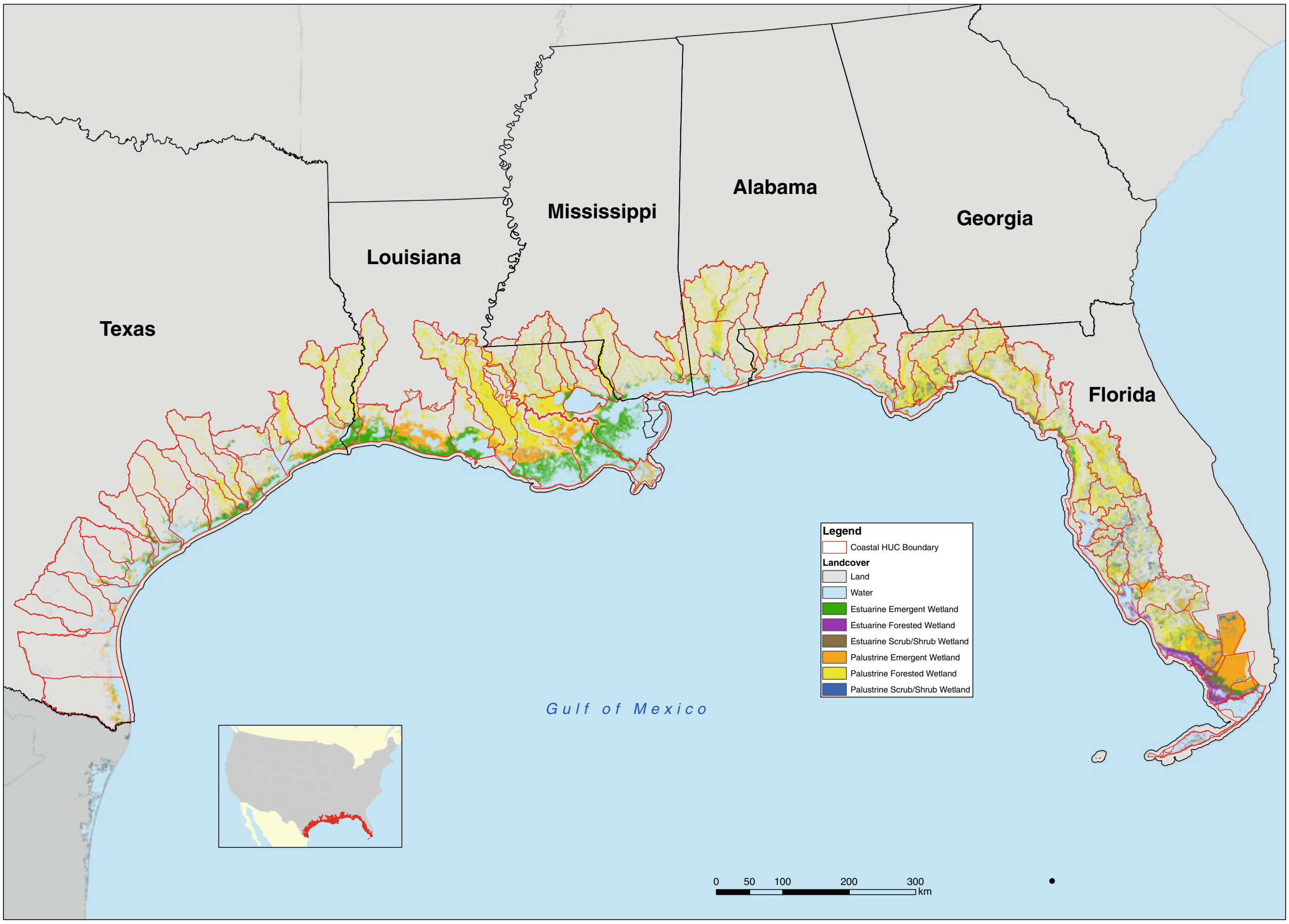

Fig. 1 Hydrologic units (8-digit HUCs) that represent coastal watersheds bordering the Gulf of Mexico and the distribution of wetlands by class within the coastal boundary

both ecological and economic perspectives (Boyd and Banzhaf 2007; de Groot et al. 2009; Ruhl et al. 2009). Wetland policies, planning, and management decisions will benefit from qualitative and quantitative information about the ecosystem services associated with different wetland classes, how provision of these services is impacted by stressors, and how changes in the flows of services from wetlands impact human communities (Ruhl et al. 2009).

The objective of this paper is to present a framework for quantification of ecosystem services for GOM coastal wetlands. While it is recognized that GOM coastal wetlands provide many ecosystem services, the focus of this paper is on four specific ecosystem services that could potentially be quantified as a stock or rate that is produced by or attributed to a unit of wetland area: (1) commercial and recreational fishery support, (2) storm surge protection, (3) nitrogen removal as a component of water quality regulation, and (4) carbon sequestration as a component of greenhouse gas regulation. Within the Millennium Ecosystem Assessment framework, fishery support is both a provisioning ecosystem service measured by the harvest of commercial or recreational fishery species and a supporting service measured by the area of suitable wetland habitat that supports growth and reproduction of fishery species (MEA 2005). Storm surge protection is an element of natural hazard regulation that can be indicated by the reduction in storm surge wave height or inundation provided by coastal wetlands. Nitrogen removal and carbon sequestration are supporting services that are necessary elements of water quality and climate regulation. Both of these services are measured as a quantity of nitrogen or carbon that is removed or stored by wetlands.

Literature from ca. 1990 to 2009 was reviewed to obtain quantitative estimates of ecosystem services where possible and to identify research needs that will enhance quantification of ecosystem services. Relevant literature was compiled from internet search engines (e.g., Google Scholar http://scholar.google.com/, Science Direct http://www.sciencedirect.com/, Elsevier http://www.elsevier.com/), professional societies (e.g., Society of Wetland Scientists http://www.sws.org/), agency internet sites (e.g., U.S. Environmental Protection Agency http://www.epa.gov/, U.S. Geological Survey http://www.

usgs.gov/, U.S. Fish and Wildlife Service http://www.fws.gov/, National Marine Fisheries Service http://www.st.nmfs.noaa.gov/st1/), and additional references contained within these publications. This review encompassed wetland classes that occur within GOM coastal watersheds: estuarine emergent (tidal marsh), estuarine shrub-scrub and forested (mangroves), palustrine emergent (freshwater marsh), and palustrine shrub-scrub and forested (forested swamps) (Cowardin et al. 1979; Fig. 1). Although not a complete list, combinations of the following key words were used in the literature searches: Gulf of Mexico, coastal, wetlands, marsh, mangrove, forested swamp, cypress, ecosystem service, function, benefits, fishery, shrimp, storm surge, hurricane, nutrients, nitrogen removal, wastewater treatment, carbon sequestration, soil carbon, and greenhouse gas. Publications were selected for inclusion in this review if the study was located in GOM coastal wetlands and presented evidence to support quantitative estimates of an ecosystem service or an ecological endpoint that could be translated to an ecosystem service.

Stedman and Dahl (2008) reported estimates of the total area of coastal wetlands in the GOM in 1998 and 2004 (Table 1). Areal estimates were reported for estuarine emergent, estuarine shrub (which includes estuarine forested), freshwater forested, freshwater shrub, and freshwater emergent wetlands. For the purposes of this review, "freshwater" is equivalent to "palustrine", and forested and shrub wetland area estimates were combined within their respective classes and reported here as "estuarine shrub" and "palustrine forested". Where possible, these reported areas were used to estimate the regional provision of ecosystem services by GOM coastal wetlands. For example, ecosystem services reported on an areal basis (e.g., quantity provided per ha of wetland) were multiplied by the wetland areas reported by Stedman and Dahl (2008) to estimate the potential magnitude of services provided by GOM coastal wetlands and to estimate the potential loss of services resulting from the reported loss of wetlands between 1998 and 2004.

Fishery Support

Estuarine emergent and shrub wetlands are widely considered to provide critical nursery habitat for coastal fishery species. To quantify the fishery support service for GOM coastal wetlands, the literature was initially reviewed to determine the ecological dependence of individual fishery species on GOM coastal wetlands. While many commercially or recreationally valuable fishery species utilize GOM estuarine emergent and shrub wetlands, the shrimp fishery was selected for illustration because it is the most valuable commercial fishery in the GOM and ample evidence exists to demonstrate the ecological relationships between estuarine emergent wetlands and the three species of penaeid shrimp (brown *Farfantepenaeus aztecus,* pink *Farfantepenaeus duorarum*, and white *Litopenaeus setiferus*) that make up the GOM shrimp fishery.

The GOM commercial shrimp fishery depends on the ability of GOM coastal wetlands to provide suitable habitat for survival, growth and reproduction of shrimp (Boesch and Turner 1984; Zimmerman et al. 2000; Minello et al. 2003). Turner (1977, 1992) first demonstrated this relationship as a log-linear correlation between the area of estuarine vegetated habitat (ha) and shrimp yields (kg yr^{-1}). While many subsequent studies from the GOM have shown that juvenile penaeid shrimp are found at higher densities in estuarine wetlands than in unvegetated habitat (Rozas and Minello 1998; Howe et al. 1999; Minello 1999; Fry 2008; Shervette and Gelwick 2008; see also Zimmerman et al. 2000 for a review of habitat-related shrimp densities), few studies have shown similar patterns in shrimp biomass or production (Herke et al. 1992; Zeug et al. 2007; Minello et al. 2008). Zimmerman et al. (2000) reviewed linkages between GOM estuarine emergent wetlands and shrimp productivity and found sufficient evidence that estuarine emergent wetlands not only contain higher densities of juvenile shrimp than other habitats but also provide essential requirements for survival and growth of juvenile shrimp.

The GOM commercial shrimp fishery is a valuable ecosystem service that is directly provided by the coastal waters of the GOM where shrimp are actually harvested. GOM estuarine wetlands provide a supporting ecosystem service because the shrimp fishery depends on the success of juvenile shrimp populations that rely on wetlands and other coastal habitats. Although commercial shrimp landings data provide estimates of this service, the challenge is to attribute a portion of the service to coastal wetlands. Quantifying the proportional contribution of GOM coastal

Table 1 Estimated area (ha) of estuarine and palustrine wetlands along the Gulf of Mexico coast (from Stedman and Dahl 2008)

Wetland Class	1998 (ha)	2004 (ha)	1998–2004 Loss (ha)
Estuarine emergent	982,969	965,127	17,842
Estuarine shrub	274,838	274,296	542
Palustrine emergent	1,124,913	1,104,812	20,101
Palustrine forested & shrub	3,744,884	3,604,418	140,466
Total	6,127,604	5,948,652	178,952

wetlands to the commercial shrimp fishery may be difficult for several reasons: 1) juvenile shrimp in the GOM utilize other coastal habitats in addition to wetlands, 2) temperature and salinity also affect shrimp production, 3) management of the fishery (i.e., limits and closures) and fishing effort affect shrimp yields, and 4) few studies have quantified the links between juvenile shrimp abundance, recruitment to the adult population, and shrimp landings. This ecosystem service (i.e., kg shrimp harvested per ha of wetland) has been quantified, however, by applying shrimp landings data to estimates of wetland area (Turner 1977, 1992; Zimmerman et al. 2000). Using shrimp landings data from NMFS (2009) and estimates of estuarine emergent wetland area for each GOM state, estimates of shrimp harvest attributed to estuarine wetlands ranged from 57 to 1,660 (mean = 241) kg ha^{-1} yr^{-1} (Table 2). These ecosystem service estimates are, at best, gross estimates with much uncertainty. Calculating this ecosystem service in this way overestimates the service provided by estuarine wetlands because it assumes that the entire shrimp harvest can be attributed to estuarine emergent wetlands, when it is known that other habitats also support GOM shrimp populations. On the other hand, because shrimp have an annual life cycle and the shrimp fishery is fully exploited in the GOM, these estimates may be much lower than the actual shrimp production that can be attributed to estuarine wetlands (Minello et al. 2008). While direct comparisons across time or geographic area cannot be made due to differences in wetland area estimation methods and temporal coverage, these estimates are comparable to those reported in the literature (71–185 kg ha^{-1} yr^{-1} from a Louisiana marsh-pond complex [Herke et al. 1992]; 237–401 kg ha^{-1} yr^{-1} from a Texas marsh complex [Minello et al. 2008]).

While accepted as dogma, the positive correlation between wetland area and commercial shrimp landings shown by Turner (1977, 1992) has not been reproduced successfully with more recent data. When compared geographically, for example, the GOM has more estuarine emergent wetlands than the south Atlantic and, consequently, higher shrimp yields (Turner 1992; Zimmerman et al. 2000). But when evaluated on a temporal basis, wetland loss in the GOM over decades has not resulted in decreased shrimp yields (Fig. 2; NMFS 2009). GOM shrimp landings have increased steadily from the 1960s until the 1990s and have remained stable since then, while continued loss of estuarine emergent wetland area has occurred over the same time period (Fig. 2; Browder et al. 1989; Chesney et al. 2000; Zimmerman et al. 2000; O'Connor and Matlock 2005). The increase in shrimp landings over time was accompanied by a significant increase in recruitment, demonstrating that the trend in landings was not due solely to increased fishing effort (Browder et al. 1989; Zimmerman et al. 2000). This apparent contradiction—increasing shrimp yields coinciding with high rates of estuarine emergent wetland loss—may be related to the process of marsh loss due to subsidence and the utilization of marsh edge habitat by juvenile shrimp (Browder et al. 1989; Zimmerman et al. 2000). Higher densities of brown and white shrimp have been observed along marsh edge than within inner marsh habitat (Peterson

Table 2 Estimated production of brown and white shrimp from GOM estuarine emergent wetlands calculated by dividing mean shrimp landings[a] as reported by NMFS (2009) by wetland area estimates for the same years (MT = metric ton = 10^6 g)

Geographic Area	Year(s)	Estuarine Emergent Wetland Area (ha)	Mean Shrimp Landings (MT y^{-1})	Estimated Shrimp Production (kg ha^{-1} yr^{-1})	Reference for Wetland Area Estimate
FL (Gulf)	1972–1985	104,086	11,952	114.8	Field et al. 1991
FL (Gulf)	1972–1976	174,542	12,019	68.9	Alexander et al. 1986
FL (Gulf)	1985	116,437	12,451	106.9	Frayer and Hefner 1991
FL (Gulf)	1996	115,771	12,123	104.7	Dahl Dahl 2005[b]
AL	1976–1978	5,908	9,806	1659.7	Alexander et al. 1986
AL	1979–1985	10,320	8,192	793.8	Field et al. 1991
MS	1972–1985	23,836	3,876	162.6	Field et al. 1991
MS	1979–1980	25,900	3,223	124.4	Alexander et al. 1986
LA	1976–1978	707,636	43,416	61.4	Alexander et al. 1986
LA	1975–1985	697,236	39,948	57.3	Field et al. 1991
TX	1950–1951	157,990	24,984	158.1	Alexander et al. 1986
TX	1955	156,699	32,440	207.0	Moulton et al. 1997
TX	1979–1985	174,865	35,386	202.4	Field et al. 1991
TX	1992	143,920	34,061	236.7	Moulton et al. 1997

[a] Shrimp landings represent the sum of brown, white, pink, dendrobranchiata, and marine, other categories of shrimp from NMFS (2009); [b] assume 91% of estuarine emergent wetlands in Florida are on the GOM coast

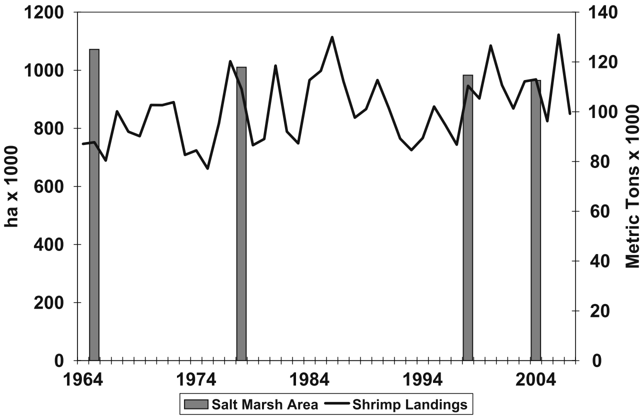

Fig. 2 Shrimp landings (metric tons) for the Gulf of Mexico from 1964–2007 (NMFS 2009) versus salt marsh area estimates (1965: Alexander et al. 1986; 1978: Field et al. 1991; 1998 and 2004: Stedman and Dahl 2008)

and Turner 1994; Minello and Webb 1997; Minello 1999). If subsiding marshes and increasing marsh edge habitat provide enhanced nursery functions for juvenile shrimp then the loss of marsh area via fragmentation may increase shrimp production over the short term (Zimmerman et al. 2000). Negative effects on GOM shrimp landings may not be observed until sufficient marsh area is converted to open water that the quantity of marsh edge habitat begins to decline (Chesney et al. 2000; Haas et al. 2004).

Another possible explanation of why continued loss of marsh habitat has not coincided with a decline in the fishery is that shrimp may be able to compensate for the loss of marsh habitat by utilizing other available habitats like submerged aquatic vegetation (SAV) and open bays (Zimmerman et al. 2000; Clark et al. 2004; Fry 2008). Although much evidence exists to demonstrate that marsh and marsh edge are important nursery habitats for juvenile shrimp, other estuarine habitats are also utilized by shrimp in the GOM (e.g., open bays, SAV, mangroves, oyster beds; Rozas and Minello 1998; Minello 1999; Zimmerman et al. 2000; Clark et al. 2004; Fry 2008; Shervette and Gelwick 2008). Studies that found higher densities of shrimp in vegetated than in nonvegetated habitats often showed either no difference in densities between SAV and marsh (Rozas and Minello 1998; Minello 1999) or that shrimp selectively inhabited SAV over marsh edge habitats when those habitats co-occurred (Clark et al. 2004).

While GOM estuarine shrub wetlands are also known to provide habitat for several recreationally important fish species, there is little empirical evidence to support a quantitative link between mangrove area and secondary production (Sheridan and Hays 2003; Manson et al. 2005a). Some evidence to support the contribution of GOM mangroves to commercial shrimp harvest has been provided by Turner (1977) and reviewed by Blaber (2007). Loss of mangrove habitat, however, has not been shown to affect fish communities or to lead to a decrease in commercial catch (Manson et al. 2005a; Greenwood et al. 2007). Linking fishery production to mangrove area (or changes in fishery production to loss of mangroves) is difficult for several reasons: 1) loss of mangroves worldwide has coincided with increased fishing effort, 2) reliable fisheries data are not available at the same spatial scale as the data on mangrove area or change, and 3) the characteristics of mangroves that may support fisheries (i.e., appropriate depth, structure, and turbidity) are also provided by other habitats in estuaries (Manson et al. 2005b; Blaber 2007).

Studies that have demonstrated the economic value of GOM coastal wetlands to fisheries have used production functions to estimate annual fishery yields from wetland area and fishing effort (e.g., Lynne et al. 1981; Bell 1997). Our objective was to quantify the fishery support service provided by GOM coastal wetlands in order to improve valuation of this ecosystem service. This effort was complicated, however, by highly variable estimates of wetland area, the contribution of other estuarine habitats to shrimp production, and the impact of correlative factors such as annual climatic variability, natural mortality, and overfishing (Minello et al. 2008). If protection of coastal wetlands is one option to optimize fishery yields, fishery managers need to know not only how much coastal wetland area is needed to sustain the GOM fisheries but also how these other factors impact production and harvest.

Storm Surge Protection

Emergent vegetation and shallow water depths associated with coastal wetlands can protect coastal communities from storm surge by providing frictional resistance that absorbs wave energy, reduces wave amplitude, and slows the forward motion of surge (Farber 1987; Möller and Spencer 2002; Shafer et al. 2002; Costanza et al. 2006b; Day et al. 2007; Resio and Westerink 2008; Krauss et al. 2009; Shaffer et al. 2009a; van Heerden et al. 2009). Storm surge attenuation was most often reported as the reduction in wave height (cm) that occurs with distance as a wave traverses a coastal wetland (km^{-1}). In addition Costanza et al. (2008) found a significant relationship between wetland area and relative damages from hurricanes along the U.S. Gulf and Atlantic coasts. The provision of this service for GOM coastal wetlands could not be estimated using wetland area on a regional scale because storm surge attenuation was reported as a function of wetland length rather than area.

GOM hurricanes of 2005, Katrina and Rita, renewed public attention on the potential value of coastal wetlands to protect coastal communities from the damaging effects of hurricane storm surge (Costanza et al. 2008; Augustin et al. 2009; Doyle 2009; Shaffer et al. 2009a). Coastal wetlands can reduce the height of storm surge waves and the distance inland that storm surge travels. Wetlands with large areas of contiguous vegetation dampen storm surge waves more effectively than narrow fringing wetlands (Shafer et al. 2002). In addition, coastal forested wetlands (e.g., baldcypress-tupelo swamp, mangrove) buffer storm winds, and therefore wind-generated waves and storm surge, more effectively than coastal marshes (Kemp 2008; Shaffer et al. 2009a). The capacity of coastal wetlands to attenuate storm surge is dependent on both the structural characteristics of the wetland (bathymetry, topography, local surface roughness, presence of levees and barrier islands, channelization, water depth, vegetation type and density) and the characteristics of the storm itself (storm track, forward speed, duration, size) (Möller and Spencer 2002; Resio and Westerink 2008; Augustin et al. 2009; Doyle 2009; Krauss et al. 2009; Shaffer et al. 2009a; Wamsley et al. 2009).

While many studies have cited storm surge attenuation rates (Table 3), the majority were based on just two primary sources (Krauss et al. 2009). The original and most cited report was a 1963 study by the US Army Corps of Engineers (USACOE) on storm surge elevations during seven hurricanes that struck Louisiana between 1907 and 1957 (USACOE 1963). Storm surge was reduced by an average of 7.0 cm km^{-1} (range 4.9–14.6 cm km^{-1}) of estuarine emergent wetland (USACOE 1963; Krauss et al. 2009; Masters 2009). Studies of tide gage levels and water marks in coastal Louisiana following Hurricane Andrew in 1992 estimated that storm surge was reduced by 4.4 to 4.9 cm km^{-1} of estuarine emergent wetland (Lovelace 1994; Swenson 1994; LDNR 1998). More recent studies using water level recorders to monitor changes in storm surge elevation in coastal wetlands demonstrated maximum water level reductions of 4.2 to 9.4 cm km^{-1} by estuarine emergent and shrub wetlands in Florida (Krauss et al. 2009) and 13.5 cm km^{-1} by estuarine emergent wetlands in Louisiana (Kemp 2008).

Models have incorporated storm surge attenuation estimates to predict the overall reduction in storm surge that would be provided by coastal wetlands based on the linear distance of wetlands in the hurricane path and the characteristics of the storm (Wamsley et al. 2009). In addition, simulation models have been used by the Federal Emergency Management Agency (FEMA) and by local governments to predict the potential surge from hurricanes and the associated risk to communities to plan evacuations (Shaffer et al. 1984). Models like Sea, Land, and Overland Surge from Hurricanes (SLOSH; Jelesnianski et al. 1992) and ADvanced CIRCulation (ADCIRC; Luettich et al. 1992) can adjust for the effects of topography, including the presence of coastal wetlands, within the predicted hurricane warning zone. ADCIRC models of Hurricane Rita's storm surge showed that extensive coastal wetlands in western Louisiana reduced the storm surge from Hurricane Rita by 5.3 to 9.1 cm km^{-1}, whereas water levels increased by 2.2 cm km^{-1} over open water and degraded coastal wetland in eastern Louisiana (Resio and Westerink 2008). SLOSH models of storm surge from Hurricane Katrina showed that the coastal wetlands of the Mississippi delta reduced water levels by 12.6 cm km^{-1} of wetland (Fitzpatrick 2008). Similar models demonstrated that the substantial loss of coastal forested wetlands in Breton Sound, Louisiana following construction of the Mississippi River Gulf Outlet (MRGO) likely exacerbated flooding during Hurricane Katrina (Kemp 2008; Shaffer et al. 2009a; van Heerden et al. 2009).

Using data on hurricane history (landfall and area of impact, wind speed, damages), spatially explicit estimates of economic activity, and wetland area within the hurricane path, Costanza et al. (2008) estimated that the average annual storm protection value (2004 US Dollar ha^{-1} yr^{-1}) of coastal wetlands in the GOM states was $7,970 for Alabama, $7,880 for Florida, $1,749 for Louisiana, $2,316 for Mississippi, and $12,365 for Texas. Coastal wetlands have the highest storm protection values in south Florida, coastal Louisiana, and parts of Texas where high storm

Table 3 Estimates of hurricane storm surge reduction by Gulf of Mexico coastal wetlands

Hurricane	Location	Wetland Class[a]	Surge Reduction (cm km^{-1})	Reference
Multiple (1907–1957)	Louisiana	E2EM	4.9–14.6	USACOE 1963
Andrew (1992)	Louisiana	E2EM	4.4–4.9	Lovelace 1994
				Swenson 1994
Charley (2004)	Florida	E2EM & E2SS	7.1–9.4	Krauss et al. 2009
Wilma (2005)	Florida	E2SS	4.2–6.9	Krauss et al. 2009
Rita (2005)	Louisiana	E2EM	5.3–9.1	Resio and Westerink 2008
Rita (2005)	Louisiana	E2EM	13.5	Kemp 2008
Katrina (2005)	Louisiana	E2EM	12.6	Fitzpatrick 2008

[a] *E2EM* Estuarine Emergent, *E2SS* Estuarine Shrub

probability, high coastal economic activity, and high wetland area overlap (Costanza et al. 2008; Doyle 2009). Costanza et al. (2006b) and Day et al. (2007) concluded that restoration of coastal wetlands would be a cost-effective and sustainable option to protect Louisiana's coastal communities from future hurricanes.

Natural coastal features such as barrier islands, shoals, marshes, and forested wetlands can provide a significant and potentially sustainable buffer against storm surge generated by tropical storms and hurricanes (Boesch et al. 2006). Quantifying the actual capacity of coastal wetlands to attenuate storm surge from hurricanes has been difficult until recently, however, due to lack of quantitative data and model validations (Boesch 2007; Masters 2009). For example, although it was widely reported that Hurricane Katrina's storm surge would have been reduced if Louisiana's coastal wetlands had been restored as originally planned (Boesch et al. 2006; Costanza et al. 2006a), additional evidence such as that provided by Kemp 2008, Shaffer et al. (2009a), and van Heerden et al. (2009) was needed to determine the degree to which coastal wetlands can reduce hurricane storm surge (Resio and Westerink 2008; Krauss et al. 2009; Masters 2009). The existing evidence supported the hypothesis that coastal wetlands reduced wave heights and the distance and speed of storm surge penetration inland. However, many experts contend that coastal wetlands would have little or no dampening effect on extreme storm surges like those produced by Hurricane Katrina (Masters 2009).

Estimates from the literature of storm surge reduction by GOM coastal wetlands ranged from 4 to 15 cm km^{-1} for estuarine emergent wetlands and 4 to 9 cm km^{-1} for estuarine shrub wetlands, although these estimates are based on limited data (Table 3). The level of storm surge attenuation provided by wetlands depends on many factors including the location, type, extent, and condition of the wetlands and the properties of the storm itself. While these and other factors hamper estimates of the value of wetlands for storm surge protection, it is clear that the loss of coastal wetlands increases the risk of property damage and loss of welfare from hurricane storm surge (Farber 1996; Costanza et al. 2006a). The impacts associated with the hurricanes of 2005 (esp. Katrina and Rita) have led to the inclusion of storm damage reduction benefits as a major consideration for future coastal wetland restoration planning efforts in Louisiana (Day et al. 2007).

Nitrogen Removal

Wetlands are widely recognized for their capacity to remove nitrogen from surface waters, thereby improving water quality by reducing the potential for eutrophication in estuaries and nearshore coastal waters. To quantify this ecosystem service for GOM coastal wetlands, the literature was reviewed for studies that reported nitrogen removal rates as a function of wetland area (g N m^{-2} yr^{-1}) or nitrogen removal efficiency (% N load removed) by wetland class. The reviewed studies primarily focused on nitrogen removal by palustrine forested and emergent wetlands that were used to treat municipal wastewater or diverted Mississippi River water (Table 4). While a wealth of knowledge exists about nutrient cycles in estuarine wetlands, few studies quantified nitrogen removal as an ecosystem service provided by estuarine wetlands in the GOM. For example, in a review of estuarine marsh flux studies, Childers et al. (2000) found only ten studies that reported net annual flux of nitrogen (g N m^{-2} yr^{-1}), but none were in the GOM. Of the reviewed studies on estuarine wetlands, Davis et al. (2001) reported N removal

Table 4 Total nitrogen removal rates for Gulf of Mexico coastal palustrine wetlands

Wetland Class[a]	Wetland Area & Type[b]	Location	N removal rate (kg N ha^{-1} yr^{-1})	Reference
PFO	1,475 ha WWT	Breaux Bridge, LA	294	Breaux and Day 1994
PFO	231 ha WWT	Thibodaux, LA	109–1,640	Boustany et al. 1997
PFO	231 ha WWT	Thibodaux, LA	2,108	Breaux and Day 1994
PFO	2.5 ha WWT	Gramercy, LA	1,003	Breaux and Day 1994
PFO PEM	467,800 ha Miss. R. diversion	Atchafalaya Basin, LA	108–116	Xu 2006
PEM	3,700 ha Miss. R. diversion	St. Charles Parish, LA (Davis Pond)	113–296	DeLaune et al. 2005
PEM	26,000 ha Miss. R. diversion	Breton Sound, LA (Caernarvon)	54–900	Mitsch et al. 2005
PEM	3,800 ha Miss. R. diversion	Barataria Basin, LA (Davis Pond)	147–1,100	Yu et al. 2006

[a] *PFO* Palustrine Forested, *PEM* Palustrine Emergent; [b] *WWT* Wastewater Treatment, *Miss. R.* Mississippi River

rates for estuarine shrub wetlands in Florida (0.015–2.19 g N m^{-2} yr^{-1}) while estimates of 4–24% (Dodla et al. 2008) and 90% (Lane et al. 2002) N removal were reported for estuarine emergent wetlands in Louisiana.

Several studies from Louisiana concluded that palustrine forested and emergent wetlands successfully removed nitrogen from wastewater effluent, thereby providing sufficient tertiary treatment for municipal release (Breaux and Day 1994; Breaux et al. 1995; Boustany et al. 1997; Blahnik and Day 2000; Cardoch et al. 2000; Day et al. 2004; Hyfield et al. 2007; Brantley et al. 2008). Many studies only reported nitrogen removal efficiency or removal rates for nitrate only; of those studies that cited total nitrogen removal rates from palustrine wetlands used for wastewater treatment, nitrogen removal rates ranged from 54 to 2,108 kg N ha^{-1} yr^{-1} (Table 4). The primary mechanism of nitrogen removal in these wetlands was denitrification although wetland nitrogen removal rates were dependent on the nitrogen loading rate, the form of nitrogen (NO_3 vs. NH_4) and the residence time of water in the wetland (Boustany et al. 1997; Mitsch et al. 2001; Lane et al. 2003; Yu et al. 2006; Brantley et al. 2008).

The diversion of Mississippi River water through coastal wetlands in Louisiana was originally intended to reduce salinity and enhance oyster production in adjacent coastal waters (Lane et al. 1999). More recently these diversions have become sources of essential sediments and nutrients that may help rebuild Louisiana's deteriorating coastal wetlands (Breaux and Day 1994; Lane et al. 1999; DeLaune et al. 2003a; DeLaune et al. 2005; Shaffer et al. 2009b). In the GOM, nitrogen loads from the Mississippi River contribute to the annual development of the "dead zone," a large area of hypoxic water offshore of Louisiana (Turner et al. 2007). Diverting Mississippi River water through coastal wetlands is one option being implemented to reduce nitrogen loads to the GOM (Mitsch et al. 2001; DeLaune et al. 2005; Mitsch et al. 2005). Some controversy exists, however, over whether these wetlands can remove enough nitrogen to have a significant impact on the GOM hypoxic zone or whether the diversions may have unintended consequences (i.e., adversely enhancing eutrophication in estuarine waters) (Lane et al. 1999; Mitsch et al. 2001; Lane et al. 2003 DeLaune et al. 2005; Mitsch et al. 2005; Turner et al. 2007; Hyfield et al. 2008; Day et al. 2009).

Several studies have been conducted at the major Mississippi River diversions (Davis Pond, Caernarvon, Atchafalaya) to determine the nitrogen removal potential of these coastal wetlands (Table 4). The Caernarvon freshwater diversion delivers Mississippi River water into Breton Sound, an estuarine complex containing 110,000 ha of fresh, brackish, and saline wetlands (Lane et al. 1999). Using data from Lane et al. (1999) and additional data from 2001, Mitsch et al. (2005) estimated that the Breton Sound wetlands removed 50 to 900 kg NO_3-N ha^{-1} yr^{-1}. The Davis Pond freshwater diversion delivers Mississippi River water to a 3,700 ha freshwater marsh-pond system that drains into Barataria Bay (DeLaune et al. 2005; Yu et al. 2006). Nitrogen removal rates in this wetland ranged from 113 to 296 kg NO_3 ha^{-1} yr^{-1} (DeLaune et al. 2005). At concentrations of 1 mg N l^{-1}, NO_3 would be removed at a rate of 147 kg N ha^{-1} yr^{-1} but the maximum capacity for nitrogen removal at elevated nitrogen concentrations would be 1,100 kg N ha^{-1} yr^{-1} (Yu et al. 2006). Yu et al. (2006) estimated nitrogen removal efficiency for wetlands in Barataria Bay, Louisiana that ranged from 42% N removal with a residence time of one day to 95% N removal with a residence time of five days. Lane et al. (2002) estimated that the nitrogen removal efficiency of wetlands receiving diverted Mississippi River water would exceed 90% if annual loading was 100 kg N ha^{-1} yr^{-1} or less. Nitrogen removal efficiencies, however, have been found to decrease with increasing nitrogen load at the Mississippi River diversions (Lane et al. 1999; Lane et al. 2002; Lane et al. 2003; Mitsch et al. 2005).

The average nitrogen removal rate was calculated as the mean of literature values for each wetland class and multiplied by the area of each wetland class to estimate the annual removal of nitrogen by GOM palustrine wetlands (Gg N yr^{-1}). Nitrogen removal rates were not available for estuarine emergent or shrub wetlands in the GOM. Average N removal rates calculated from the literature for palustrine forested wetlands are more than double those for emergent wetlands (Table 5). The

Table 5 Estimates of nitrogen (N) removal by GOM coastal wetlands (mean with range in parentheses). Wetland area estimates from Table 1. Average N removal rates from Table 4. (Gg = 10^9 g)

Wetland Class	Year	Area (ha)	Average N removal rate (kg N ha^{-1} yr^{-1})	GOM N removal (Gg N yr^{-1})
Palustrine Forested	1998	3,744,884	1,031 (109–2,108)	3,860
	2004	3,604,418		3,715
Palustrine Emergent	1998	1,124,913	435 (54–1,100)	489
	2004	1,104,812		481

estimated area of GOM palustrine forested wetlands is much greater than the area of palustrine emergent wetlands (Stedman and Dahl 2008). When the average N removal rates were multiplied by wetland areas, therefore, regional estimates of nitrogen removal by these GOM wetland classes differed substantially (Table 5). Nitrogen removal rates ranged from 109–2,108 kg N ha^{-1} yr^{-1} in palustrine forested wetlands and 54–1,100 kg N ha^{-1} yr^{-1} in palustrine emergent wetlands (Table 4). On average, GOM palustrine forested wetlands accounted for almost three times more nitrogen removed per year than GOM palustrine emergent wetlands (Table 5). The estimated loss of 160,567 ha of palustrine forested and emergent wetlands from 1998 to 2004 (Table 1) would have resulted in 154 Gg less nitrogen removed from surface waters (Gg = 10^9 g). Estimation of this ecosystem service based solely on wetland area has a high uncertainty, however, due to the variability in nitrogen removal rates associated with wetland type, condition, and location as well as nitrogen load and water residence time.

The removal of nitrogen by wetlands from land-based sources can improve water quality in receiving waters which, in turn, may have positive impacts on commercial and recreational fisheries, opportunities for recreation, and aesthetics. The benefits of this ecosystem service, however, need to be balanced by the potential negative impact of greenhouse gas emissions (i.e., nitrous oxide [N_2O]). Coastal wetlands, especially palustrine emergent wetlands, tend to release significant amounts of N_2O as an intermediate product of denitrification (Yu et al. 2006; Dodla et al. 2008). To ensure the optimum ecosystem service benefit from wetlands that are used to treat wastewater or Mississippi River water, therefore, these wetlands should demonstrate efficient nitrate removal with minimum N_2O production (Yu et al. 2006; Dodla et al. 2008).

The capacity of specific wetlands to remove nitrogen can be valued directly by comparing the cost of using or constructing wetlands to treat wastewater to the construction and operating costs of wastewater treatment plants (Breaux et al. 1995; Cardoch et al. 2000). The diversion of Mississippi River water through wetlands to decrease nitrogen loads to the GOM cannot be valued directly, however, because the benefits of nitrogen reduction would be reflected in the impact of a reduced area of hypoxia in the GOM on commercial fisheries and the value of wetland treatment would need to be compared to both the cost to construct and maintain these diversions and the concomitant value of reductions in agricultural nitrogen load in the upper Mississippi River basin. In addition, using coastal wetlands to treat nitrogen-laden water may increase accretion, productivity, and function in coastal wetlands which may enhance other ecosystem services provided by these wetlands (i.e., storm surge protection and flood retention, carbon sequestration, provision of wildlife habitat). Nitrogen removal, therefore, should be considered as a supporting ecosystem service in the valuation of GOM coastal wetlands.

Carbon Sequestration

Wetlands play an important but complex role in the global carbon cycle, contributing to the ecosystem service of greenhouse gas regulation through carbon sequestration. Wetlands serve as carbon sinks because they store large amounts of carbon in plant biomass and soils and continue to sequester carbon through photosynthetic processes, assimilation, and burial. Wetlands, however, may also act as sources of carbon dioxide (CO_2) and methane (CH_4) to the atmosphere. Estimating the total carbon sequestered by wetlands requires knowledge of soil and plant carbon stocks as well as rates of carbon fixation by plants, accumulation of carbon in soils, and emission of CO_2 and CH_4 by plants and soils. Because 98% of the total carbon pool in North American wetlands exists in wetland soils (Bridgham et al. 2006), the literature was reviewed to obtain quantitative estimates of the soil components of carbon sequestration for GOM coastal wetlands. The soil carbon pool and annual soil carbon accumulation for GOM coastal wetlands was estimated by multiplying the average stock and rate estimates from the literature for each wetland class by the GOM wetland area estimates.

Estimates of soil carbon pools (i.e., standing stock) in GOM estuarine emergent wetlands ranged from ~100–250 Mg C ha^{-1} in Florida (Mg = 10^6 g; DeLaune and Pezeshki 2003; Choi and Wang 2001, 2004) to ~100–628 Mg C ha^{-1} in Louisiana (Rabenhorst 1995). The average soil carbon pool in GOM estuarine emergent wetlands, calculated as the mean of these estimates from the literature, was 275 Mg C ha^{-1}. The soil carbon pool in mangroves could be estimated for the conterminous U.S. from data presented by Bridgham et al. (2006; i.e., divide 61,000,000 Mg C by 300,000 ha, resulting in 203 Mg C ha^{-1}). Because 85% of the mangrove wetlands in the U.S. are located in the GOM (Stedman and Dahl 2008), 203 Mg C ha^{-1} is a reasonable estimate of the soil carbon pool for estuarine shrub wetlands in the GOM.

On an areal basis, estuarine wetlands (tidal marshes, swamps, and mangroves) in the conterminous U.S. sequestered carbon at a much higher rate (2.13 Mg C ha^{-1} yr^{-1}) than other wetland types (i.e., peatlands at 0.71 Mg C ha^{-1} yr^{-1} and freshwater mineral soils at 0.17 Mg C ha^{-1} yr^{-1}) due to high sedimentation rates, high soil carbon content, and burial due to sea level rise (Bridgham et al. 2006). Soil carbon accumulation rates in GOM estuarine emergent wetlands ranged from 0.18 to 17.1 Mg C ha^{-1} yr^{-1} (Chmura et al. 2003; Table 6). Chmura et al. (2003) converted soil organic matter accumulation rates to soil carbon accumulation rates for impounded and natural salt marshes in Louisiana. The lowest rates of soil carbon accumulation (0.18–0.27 Mg C ha^{-1} yr^{-1}) were calculated for impounded brackish marshes in Louisiana with active water level management; these marshes had lower vertical accretion and organic matter accumulation rates than unmanaged, reference marshes (Cahoon 1994; Chmura et al. 2003). Natural marshes accreted soil at twice the rate of impounded marshes and, therefore, had soil carbon accumulation rates (average = 6.2 Mg C ha^{-1} yr^{-1}) that were almost double those of impounded marshes (average = 3.3 Mg C ha^{-1} yr^{-1}) (Cahoon 1994; Bryant and Chabreck 1998). Impoundment of marshes in Louisiana apparently inhibited the normal delivery of floodwaters and associated mineral sediments and organic matter accumulation that contribute to soil accretion (Nyman et al. 1993; Bryant and Chabreck 1998).

Soil carbon accumulation rates in GOM estuarine shrub wetlands ranged from 1.0 to 3.8 Mg C ha^{-1} yr^{-1} (Chmura et al. 2003; Table 6). Soil carbon accumulation was highest in a hydrologically isolated basin mangrove wetland in Rookery Bay, Florida (Cahoon and Lynch 1997); this type of wetland had highly organic sediments and low tidal flushing which led to accumulation of leaf litter. Other mangrove wetlands from the same study had lower soil carbon accumulation rates due to allochthonous input of mineral matter and daily removal of leaf litter via tidal flushing (Cahoon and Lynch 1997). Mangroves in the Florida Keys had the lowest soil carbon accumulation rates (Chmura et al. 2003); these wetlands had lower vertical accretion rates (0.19–0.42 cm yr^{-1}) than mangrove wetlands (0.46–0.78 cm yr^{-1}) in Rookery Bay, Florida (Cahoon and Lynch 1997; Callaway et al. 1997).

Because most of the wetland soil carbon pools and accumulation rates were derived from either national or local scale studies, average rates were multiplied by the area of estuarine wetlands in the GOM (Table 1) to derive regional estimates of soil carbon sequestration for GOM coastal wetlands. Rates for wetlands that were impounded or managed were excluded from the average. The average soil carbon accumulation rate was 2.6 Mg C ha^{-1} yr^{-1} for estuarine emergent wetlands and 2.1 Mg C ha^{-1} yr^{-1} for estuarine shrub wetlands (Table 7). The loss of 18,385 ha of estuarine emergent and shrub wetlands in the GOM from 1998 to 2004 would have resulted in a 47 Gg C reduction in soil carbon accumulation.

Coastal wetlands affect the global carbon cycle by sequestering carbon in soils and plant biomass and by releasing CO_2 and CH_4 to the atmosphere. The capacity of coastal wetlands to provide the ecosystem service of net greenhouse gas reduction therefore requires that the rate of carbon sequestration exceeds the rate of carbon released to the atmosphere (Whiting and Chanton 2001). Because wetlands in their natural state provide baseline estimates of carbon sinks and sources, it is the change in carbon fluxes that result from disturbance to wetlands that have the potential to impact climate change (Bridgham et al. 2006). On an areal basis, estuarine wetlands may be more valuable than other ecosystems as carbon sinks due to high carbon sequestration rates and negligible methane emissions (Choi and Wang 2004). Estuarine wetlands in the GOM, however, are being lost at an alarming rate (3,000 ha yr^{-1}; Stedman

Table 6 Soil carbon accumulation rates (Mg C ha^{-1} yr^{-1}) for Gulf of Mexico coastal wetlands. (Mg = metric ton = 10^6 g)

Location	Wetland Class[b]	Soil Carbon Accumulation (Mg C ha^{-1} yr^{-1})	References
Aransas NWR[c], TX	E2EM	1.78	Callaway et al. 1997[a]
San Bernard, TX	E2EM	2.03	Callaway et al. 1997[a]
Fina la Terre, LA	E2EM	0.18–1.36	Cahoon 1994[a]
Breton Sound, LA	E2EM	1.83	DeLaune and Pezeshki 2003
Rockefeller NWR, LA	E2EM	0.27–6.57	Cahoon 1994; Bryant and Chabreck 1998[a]
Lafourche Parish, LA	E2EM	1.86	Cahoon and Turner 1989[a]
Cameron Parish, LA	E2EM	0.41–1.15	Cahoon and Turner 1989[a]
Marsh Island NWR, LA	E2EM	3.18–7.63	Bryant and Chabreck 1998[a]
Sabine NWR, LA	E2EM	7.14–17.13	Bryant and Chabreck 1998[a]
St. Bernard Parish, LA	E2EM	1.40	Markewich et al. 1998
Biloxi Bay, MS	E2EM	1.53	Callaway et al. 1997[a]
St. Marks NWR, FL	E2EM	0.18–1.93	Choi and Wang 2004
Florida Keys, FL	E2SS	1.00–1.43	Callaway et al. 1997[a]
Rookery Bay, FL	E2SS	1.42–3.81	Lynch 1989; Cahoon and Lynch 1997[a]

[a] cited by Chmura et al. (2003); Soil C accumulation rates calculated from soil carbon densities and vertical accretion rates; [b] *E2EM* Estuarine Emergent, *E2SS* Estuarine Shrub; [c] *NWR* National Wildlife Refuge

Table 7 Estimates of soil carbon accumulation by non-impounded GOM coastal wetlands (mean with range in parentheses). Wetland area estimates from Table 1. Average soil C accumulation rates calculated from non-impounded wetlands in Table 6. (Mg = 10^6 g; Gg = 10^9 g)

Wetland Class	Year	Area (ha)	Average soil C accumulation rate (Mg C ha^{-1} yr^{-1})	GOM soil C accumulation (Gg C)
Estuarine Emergent	1998	982,969	2.55 (0.18–7.63)	2,507
	2004	965,127		2,461
Estuarine Shrub	1998	266,603	2.07 (1.00–3.81)	552
	2004	266,077		551

and Dahl 2008), consequently reducing their capacity for carbon sequestration. Coastal wetland restoration that reduces or reverses this loss, therefore, has the potential to reduce carbon emissions and enhance carbon sequestration. The value of GOM estuarine wetlands as a sink for greenhouse gases may be reflected in future carbon markets as carbon credits for restoration or creation of wetlands (Whiting and Chanton 2001).

Summary

The purpose of this paper was to review and attempt to quantify the ecosystem services provided by GOM coastal wetlands. Understanding the effect of GOM coastal wetland loss on valued ecosystem services would enable regional and state environmental managers to prioritize wetland restoration and conservation decisions. The Gulf of Mexico Alliance, for example, has a long-term goal to "*develop and implement an accurate tracking system to document gains and losses of Gulf habitats and ecosystem services*" (GOMA 2009). Ideally, every ecosystem service would be quantified by wetland class and regional estimates of service provision would be extrapolated for all GOM coastal wetlands. The data to accomplish this, however, was not readily available from the literature, which points out the need for additional research on the specific services provided by certain wetland classes and models to extrapolate those estimates to regional and national scales.

The ecological and economic links between estuarine emergent wetlands and shrimp production have been established but the contribution of other coastal wetlands to production of other commercial or recreational fisheries is less well-known. Because storm surge protection by estuarine wetlands was estimated as the reduction in wave height across a linear distance of wetlands, rather than area, this service could not be estimated on a regional scale. Nitrogen removal has been reported extensively for palustrine emergent and palustrine forested wetlands that are used to treat wastewater or diverted Mississippi River water in coastal Louisiana, but the literature is sparse for GOM estuarine wetlands or for palustrine wetlands in other GOM states. While estimates of carbon sequestration have been quantified for GOM estuarine wetlands, the values are based on few studies and almost no information is available to quantify this service for GOM palustrine wetlands.

The assessment and valuation of ecosystem services will be necessary for future wetland management decisions in the GOM. The most recent amendments to the Compensatory Mitigation Rule (Section 404 of the Clean Water Act; 33 CFR §§ 325, 332 [2008]) integrate ecosystem services into the decision-making process for wetlands mitigation, however, they do not provide guidance on how ecosystem services should be defined or assessed beyond using best professional judgment (Ruhl et al. 2009). The National Oceanic and Atmospheric Administration (NOAA) coordinates natural resource damage assessments (NRDAs) following marine oil spill incidents to "determine the restoration actions needed to bring injured natural resources and *services* back to baseline" (15 CFR §990.30); however, current NRDA methods rely primarily on replacement or restoration costs as the measure of damages rather than the total value of ecosystem services (Boyd 2010).

The Deepwater Horizon Incident (20 April 2010) and resulting oil spill in the Gulf of Mexico have refocused attention on the value of GOM coastal wetlands and the services they provide. While reports by the media, British Petroleum, and federal agencies have varied, it has been estimated that more than 4.9 million barrels of oil were released into GOM waters and that 10% of the Gulf coast shoreline has been exposed to oil (Corn and Copeland 2010). As the Deepwater Horizon oil spill occurred after this review was completed, an assessment of the long-term impacts of this oil spill on GOM coastal wetlands and their services is beyond the scope of this paper and can only be conjectured at this time. The responses of GOM coastal marshes to oil exposure are highly complex and variable, ranging from rapid recovery to complete mortality and wetland loss (Pezeshki et al. 2000). After initial damage to aboveground biomass following exposure to oil, many GOM coastal marsh plants recover completely, although different species recover at different rates (Hester and

Mendelssohn 2000; Pezeshki et al. 2000; DeLaune et al. 2003b). Indirect effects of oil on soil microbial processes such as nutrient cycling (Pezeshki et al. 2000) may impact nitrogen removal capacity while loss of wetland vegetation would likely impact carbon sequestration, fishery support, and storm surge protection. Assessing the acute and chronic effects of the Deepwater Horizon oil spill on GOM coastal wetland ecosystem services will be complicated by issues such as establishing a baseline of wetland condition and service provision prior to the incident and definitively linking oil exposure to adverse impacts on coastal wetlands that are already deteriorating as a result of subsidence and sea-level rise (Boyd 2010; Corn and Copeland 2010).

Clearly, there is a need to define wetland ecosystem services in terms of quantitative ecological and economic units to improve wetland mitigation and natural resource damage assessments. This review and assessment of the ecosystem services provided by Gulf of Mexico coastal wetlands is only a preliminary step to fill this need. Although much is known conceptually or qualitatively about wetland services, this review highlights the need for additional research to quantify wetland services at multiple geospatial and socioeconomic scales and to demonstrate the impact of future management decisions on the ability of GOM coastal wetlands to continue to provide services that affect human well-being.

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