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Vertical Accretion Rates in Coastal Louisiana: A Review of the Scientific Literature

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PURPOSE: Coastal marshes in Louisiana continue to deteriorate despite large-scale restoration efforts due, in part, to processes such as shallow and deep subsidence, sediment depletion, and sea level rise. The purpose of this technical note is to review and synthesize the available scientific literature concerning vertical accretion rates in Louisiana coastal marshes in response to a changing coastal environment.

BACKGROUND: Coastal marsh stability is governed by many complex processes. In particular, marsh stability is a balance between sediment accretion, marsh subsidence, and sea-level rise (Mitsch and Gosselink 2007). In coastal Louisiana there are an estimated 36,000 km^2 of freshwater and saltwater wetlands (Day et al. 2005; Mitsch and Gosselink 2007, Figure 1); however, this represents a decline of approximately 4,800 km^2 of coastal habitat over the past century (Britsch and Dunbar 1993; Barras et al. 2003; Day et al. 2000, 2005). Although the rate of decline has decreased since its peak of 109 km^2 yr^{-1} in the 1970's (Britsch and Dunbar 1993), Louisiana coastal wetlands are still being lost at an average rate of 88 km^2 per year (Day et al. 2005).

Loss of wetlands in this region has been attributed to both anthropogenic and natural processes (Day et al. 2000; Kennish 2001; Thompson et al. 2002). The majority of anthropogenic impacts such as the building of levees, canals, and spoil bank networks (DeLaune et al. 1989; Bass and Turner 1997; Turner 1997), agricultural impoundments (Sasser et al. 1986; Reed 1992), and below-ground fluid withdrawal (Pezeshki et al. 2000; Ko and Day 2004) have significantly altered wetland hydrology (Day et al. 2000; Kennish 2001). These alterations have resulted in increased submergence rates attributed to reduced sediment availability and freshwater flow (Day et al. 2000), salinity intrusions (Turner 1990), and decreased drainage in coastal wetlands (Sasser et al. 1986; Turner 1990). In addition to the anthropogenic impacts listed above, natural phenomena such as subsidence which varies by delta lobe (Britsch and Dunbar 1993; Callaway et al. 1997), and hurricanes (Barras 2006) have also negatively affected Louisiana coastal marshes leading to wetland loss.

The potential effects of eustatic sea-level rise (Turner 1991; Intergovernmental Panel on Climate Change (IPCC) 2007) are of particular concern relative to the future of Louisiana wetlands (Boesch et al. 1994; Day et al. 2005; González and Törnqvist 2009). The effects of sea level rise are compounded in the Louisiana coastal area by high subsidence rates and the micro-tidal nature of the Deltaic and Chenier Plains (Penland and Ramsey 1990; Britsch and Dunbar 1993; Day

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et al. 1995; Callaway et al. 1997). Current rates of sea-level rise in this region are estimated to be approximately 1.0 cm yr^{-1} (Penland and Ramsey 1990) and are predicted to increase to 1.3 to 1.7 cm yr^{-1} by 2100 (Day et al. 2005). Whether marsh substrate accretion can keep pace with sea level rise depends on processes involving sediment deposition on the marsh surface and below ground production of organic matter (DeLaune et al. 1983; Turner 1990; Reed 1995; Day et al. 2000). These processes vary both spatially and temporally and are not well understood in many Louisiana marsh systems.

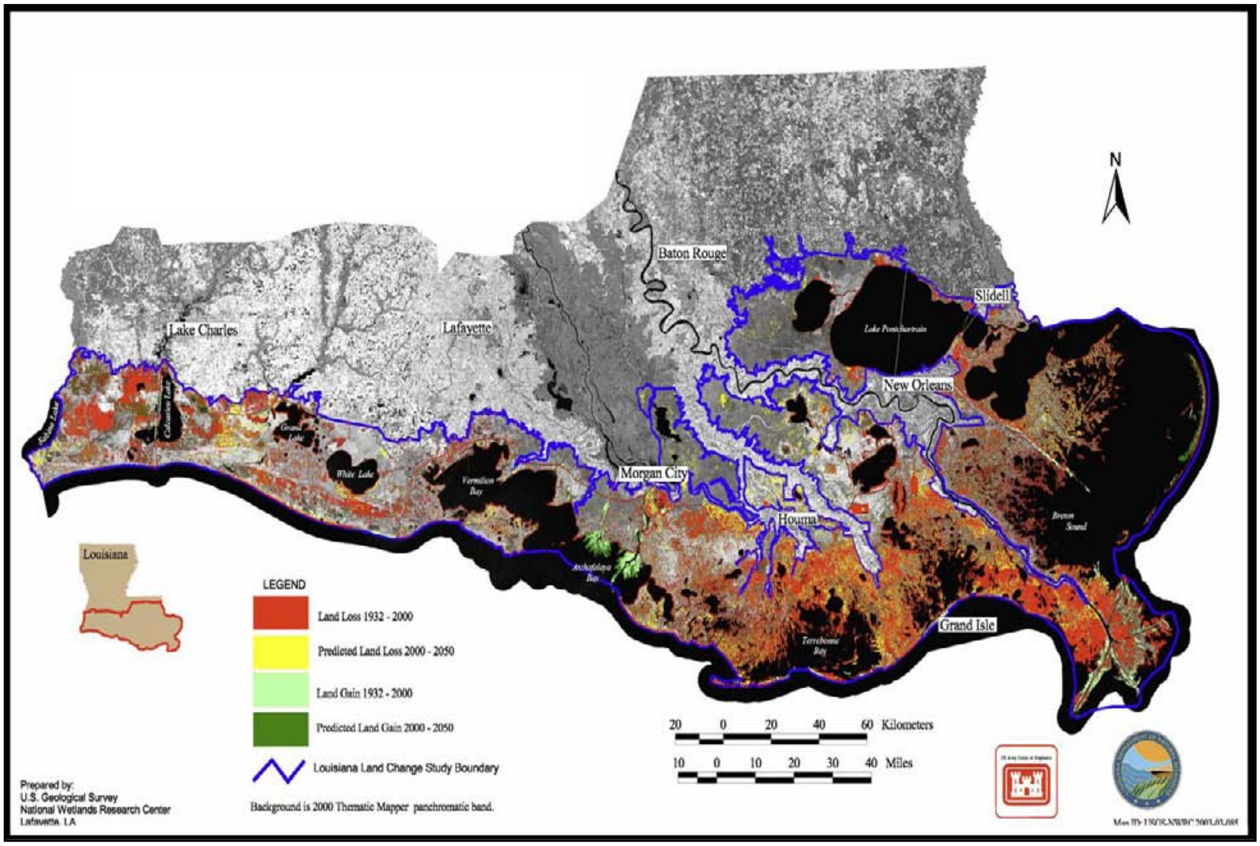

Figure 1. Map of Louisiana land loss over the last 100 years. Modified from U.S.G.S. (*http://www.nwrc.usgs.gov/upload/landloss11X17.pdf*).

REVIEW AND DISCUSSION

Measurements of accretion. Studies of marsh substrate accretion have relied on three basic approaches, which vary in the time scales they address and their sensitivity to the wide range of environments and processes that occur in the intertidal zone. Feldspar markers placed at known depths are used to monitor current accumulation rates (months to years; Cahoon and Turner 1989). Feldspar cores are simple to collect and process, are generally less expensive than other methods involving measurements of isotopes such as ^{137}Cs or ^{210}Pb, and sampling success is known at collection time (Cahoon and Turner 1989). However, markers can be lost to bioturbation and removal by tides, resuspension, or erosion, making sampling in certain environments unreliable (DeLaune et al. 1989). Long-term measurements of accumulation can be made using ^{137}Cs or ^{210}Pb tracer methods (DeLaune et al. 1978, 1989; Milan et al. 1995). By looking at elemental peaks related to nuclear testing in sediment cores, accretion rates over the past several decades (^{137}Cs from 1963 to present) up to the past 80 to 100 years (^{210}Pb) can be quantified providing a long-term accretion record (DeLaune et al. 1989). Problems with the ^{137}Cs record arise in areas with high erosion rates, large amounts of sediment wash-in from other areas, or little to no sediment accumulation (Milan et al. 1995). Milan et al. (1995) recommend that both ^{137}Cs and ^{210}Pb methods be used in coastal environments due to these types of variability.

Effects of sediment source and vegetation on accretion. Net vertical accretion is defined as the combination of deposition of sediment on the wetland surface and below-ground plant production minus erosion from the wetland surface and below-ground decomposition (Reed 1995). While both mineral and organic sediments are important in marsh accretion (DeLaune and Pezeshki 2003; Neubauer 2008), the relative importance of sediment type on accretion, and ultimately the wetland response to relative sea-level rise, varies due to region, hydrology, and wetland type (Nyman et al. 1990; Reed 1995).

Recent studies have shown that organic matter accumulation has driven sediment accretion across the salinity gradient in Louisiana marshes over the last century (Nyman et al. 1990; Turner et al. 2001, 2006; DeLaune and Pezeshki 2003; Craft 2007; Neubauer 2008). In a synthesis study of coastal freshwater accretion rates along the east and gulf coasts of the United States, organic sediment accounted for 62% of vertical accretion and was four times greater than mineral sediment on a per mass basis in Louisiana freshwater marshes compared to all other locations investigated (Neubauer 2008). DeLaune and Pezeshki (2003) found (while monitoring the effects of the Caernarvon freshwater project on Breton Sound estuarine marshes) that sediment accretion in marshes near the diversion project was also primarily related to organic sediment accumulation; however, accretion of mineral sediments supplied by the diversion was necessary to stimulate plant growth. In a comparison study of accretion rates across saline, intermediate, and freshwater wetlands in the Louisiana deltaic plain, organic sediment accumulation was also found to be important in salt marsh accumulation despite the high mineral content (> 80%; Nyman et al. 2006) and the greater contribution of mineral sediments to total sediment volume (4.0 – 6.9%) in saline compared to freshwater and intermediate wetlands (< 2.0%; Nyman et al. 1990; Turner et al. 2006). These studies concluded that in wetland sediments, organic matter forms interlocking networks providing structure to the sediment that inorganic particulate matter cannot form alone resulting in greater overall accretion (Nyman et al. 1990).

Structure provided by vegetation both above and below the sediment surface serves an important role in facilitating sediment (both organic and mineral) accumulation (Stumpf 1983; Turner 1990; DeLaune et al. 1994; Nyman et al. 2006). Wetland plants promote sediment accretion through trapping of suspended sediments from the water column, accumulation of allochthonous and autochthonous organic material, and the production of below-ground root material (Turner 1990). The ability of plants to aid in sediment accretion is significantly hampered by environmental stressors such as excessive waterlogging due to submergence and salinity intrusion (DeLaune et al. 1992, 1994; Webb et al. 1995; Thompson et al. 2002).

Hydrologic alteration. Alteration of the natural hydrology by a large complex of canals and levees throughout the Mississippi Deltaic and Chenier Plains regions has introduced higher salinity water into historically freshwater and intermediate wetland environments (Boesch et al. 1994; Day et al. 2000; Kennish 2001). Gradual increases in salinity can promote colonization by more salt-tolerant plant species and convert freshwater marshes into brackish marshes (Boesch et al. 1994). However, sudden pulses of salinity into freshwater wetlands after short-term events like hurricanes can result in wetland loss (Turner 1990). This trend is exacerbated under drought conditions where the lack of freshwater from precipitation and reduced river flows can result in hypersaline conditions (Thompson et al. 2002). Persistence of hypersaline conditions can cause loss of all wetland types, including salt marshes (Thompson et al. 2002).

Alteration of hydrology has also resulted in reduced sediment supply to many marsh systems (Boesch et al. 1994; Day et al. 2000; Kennish 2001). It is estimated that as much as 50% of the Mississippi River sediment supply is no longer available to the Louisiana deltaic plain due to damming of tributaries (Blume and Roberts 2009). The construction of canals and levees has also reduced the amount of overbank deposition, resulting in even greater sediment deficits for many backmarsh areas (Turner 1997; Day et al. 2000, 2005; Kennish 2001). Lower sediment availability leads to reduced accretion rates that are further compounded by natural subsidence rates in the Mississippi River deltaic plain, which are the greatest in the Gulf of Mexico due to the compaction of Holocene sediments (Callaway et al. 1997). A field experiment designed to quantify the effects of submergence on *Spartina alterniflora* and *Spartina patens* survival found that elevating marsh sediments 20 cm above ambient levels resulted in a significant increase in *S. patens* growth and survival compared to non-elevated cores (Webb et al. 1995). The sediment profile of the elevated cores was less reduced, had lower sulfide concentrations, and showed greater NH_4-N uptake by plants, all leading to greater plant production. If plants remain in water-logged sediments for a period of time, reduced sediment conditions can lead to reduced production and eventually death (Turner 1990; DeLaune et al. 1994; Webb et al. 1995). Once plants die, accretion rates slow and subsidence increases in response to the collapse of the living root network (DeLaune et al. 1994). This can lead to greater plant loss, continuing the cycle of degradation (DeLaune et al. 1994).

Pulses of sediment can alleviate sediment shortages and create “elevation capital” (Reed 2002). Rapid accumulation of sediment from winter storms and hurricanes is especially important for sedimentation in backmarsh areas where sediment supply may be limited (Stumpf 1983). Short-term events such as winter storms and hurricanes can provide pulses of sediment several centimeters thick on the wetland surface (Stumpf 1983; Baumann et al. 1984; Reed 1989, 2002; Cahoon et al. 1995; McKee and Cherry 2009). In the Big Branch and Pearl River estuaries located on the northern edge of the Mississippi River deltaic plain, the addition of sediment following Hurricane Katrina to subsiding marshes was still evident two years following the storm (McKee and Cherry 2009). Similar results were observed in Bayou Chitigue where 3 cm of sediment were deposited on the marsh surface after Hurricane Andrew (Cahoon et al. 1995). For measurable accretion to occur following a pulsed sediment event, deposition on a marsh surface requires a large enough quantity of available sediment, time for newly deposited sediment to become consolidated, and low sediment compaction rates (Reed 1989; Cahoon et al. 1995; McKee and Cherry 2009).

Estimates of accretion rates in Louisiana. Due to the complex factors and processes that affect sediment accretion rates, general trends in sediment accretion vary on a variety of temporal and spatial scales (Boesch et al. 1994; Day et al. 2000; Kennish 2001). Accretion trends depend on wetland type (defined by salinity), the health of the vegetative community, proximity to a water and sediment source, the time of the year, the amount of hydrologic alteration in the area (i.e. number of canals and levees), the frequency and duration of storm events, subsidence rates, and management activities in an area. For example, published accretion rates using ^{137}Cs methods for Louisiana coastal marshes show greater variability in saline (0.55 to 2.26 ± 0.09 cm yr^{-1}) back-marsh (>15 from streamside) areas compared to freshwater back-marsh (0.65 ± 0.18 to 0.90 ± 0.10 cm yr^{-1}) habitats (Table 1). Feldspar methods showed similar results with greater variation in sediment accretion as salinity increased (Table 1). However, streamside accretion measurements using feldspar methods show greater variation in freshwater wetlands with accretion ranging from 0.21 ± 0.03 to 1.18 ± 0.03 cm yr^{-1} while saline marshes only vary from 0.66 ± 0.25 to 1.52 cm yr^{-1} (Table 2). Streamside measurements using ^{137}Cs methods showed greater variation in saline (0.63 ± 0.07 to 1.35 cm yr^{-1}) compared to freshwater marshes (0.99 ± 0.17 to >1.59 cm yr^{-1}; Table 2). Accretion rates are not as varied in studies that compare accretion rates across back-marsh and streamside sites. For example, using ^{137}Cs methods, accretion in saline marshes only varied from 0.50 ± 0.13 to 0.90 ± 0.18 cm yr^{-1} and from 0.67 ± 0.02 to 1.07 ± 0.20 cm yr^{-1} in freshwater marshes (Table 3). Feldspar methods showed similar variability in accretion rates with saline marshes varying from 0.34 ± 0.05 to 0.50 ± 0.08 cm yr^{-1} (Table 3). However, freshwater marshes showed greater variation ranging from 0.27 ± 0.01 to 1.01 ± 0.02 cm yr^{-1} (Table 3). Measurements made across the Louisiana coastal plain vary significantly, highlighting the need for local measurements of accretion necessary for management and conservation in any particular area.

In addition to local variation, recent studies have highlighted the need to concurrently measure changes in marsh elevation along with sediment accretion rates (Cahoon et al. 1995; Rybczyk and Cahoon 2002; Lane et al. 2006). In a study measuring accretion rates and surface elevation changes in Bayou Chitigue and Old Oyster Bayou, Cahoon et al. (1995) found that vertical accretion measurements alone could overestimate increases in surface elevation by a factor as great as 1.5 to an entire order of magnitude. For example, in Bayou Chitigue vertical accretion rates predicted an elevation change of 5.19 ± 0.12 cm; however, due to shallow subsidence, total elevation change was only 0.29 ± 0.15 cm. Similar patterns of small elevation changes despite relatively large sediment accretion rates were observed at Bayou Chitigue in 2001 where elevation increased only 0.22 ± 0.06 cm yr^{-1} despite accretion rates of 2.26 ± 0.09 cm yr^{-1} (Rybczyk and Cahoon 2002). Changes in marsh elevation due to shallow subsidence can be significant and should be considered along with sediment vertical accretion rates when determining the ability of coastal Louisiana marshes to compensate for increased rates of sea level rise.

Table 1. Estimates of sediment accretion rates for coastal Louisiana backmarsh (> 15 m away from streamside) sites.

Site	Date	Wetland Type	Method	Accretion rate (cm yr^{-1})	Source
BACKMARSH					
Barataria Basin	--	All	^{137}Cs	0.65 - 0.75	Hatton 1981
Barataria Basin	1983	Freshwater	^{137}Cs	0.65	Hatton et al. 1983
Brenton Sound	1963-1999	Freshwater	^{137}Cs	0.65 ± 0.18	DeLaune and Pezeshki 2003
Palmetto Bayou	1963-1986	Freshwater	^{137}Cs	0.90 ± 0.10	DeLaune et al. 1989
Stable fresh	1989-1994	Freshwater	^{137}Cs	0.82 ± 0.15	Nyman et al. 2006
Tchefuncte River	1989-1994	Freshwater	^{137}Cs	0.75	Nyman et al. 2006
Barataria Basin	1983	Intermediate	^{137}Cs	0.64	Hatton et al. 1983
Calcasieu Lake	1954-1978	Intermediate	^{137}Cs	0.78	DeLaune et al. 1983
Barataria Basin	1983	Brackish	^{137}Cs	0.59	Hatton et al. 1983

Cameron Parish	1963-1986	Brackish	^{137}Cs	0.56 ± 0.11	DeLaune et al. 1989
Deteriorating brackish	1989-1994	Brackish	^{137}Cs	0.96 ± 0.32	Nyman et al. 2006
Stable brackish	1989-1994	Brackish	^{137}Cs	0.88 ± 0.14	Nyman et al. 2006
N Billy Goat Bay	1963-1990	Brackish/saline	^{137}Cs	1.06	Nyman et al. 1993
N Madison Bay	1963-1990	Brackish/saline	^{137}Cs	1.33	Nyman et al. 1993
SE Madison Bay	1963-1990	Brackish/saline	^{137}Cs	0.67	Nyman et al. 1993
W Madison Bay	1963-1990	Brackish/saline	^{137}Cs	0.78	Nyman et al. 1993
Barataria Basin	1983	Saline	^{137}Cs	0.75	Hatton et al. 1983
Barataria Basin	1963-1978	Saline	^{137}Cs	0.75	DeLaune et al. 1978
Barataria Bay	1975-1979	Saline	^{137}Cs	0.75	Baumann et al. 1984
Bay la Peur	1963-1990	Saline	^{137}Cs	0.78	Nyman et al. 1993
Bayou Chitigue	1992-2000	Saline	^{137}Cs	2.26 ± 0.09	Rybczyk and Cahoon 2002
Charles Theriot	1963-1990	Saline	^{137}Cs	0.98	Nyman et al. 1993
Chitigue (upstream)	1963-1990	Saline	^{137}Cs	1.22	Nyman et al. 1993
Chitigue (midstream)	1963-1990	Saline	^{137}Cs	0.75	Nyman et al. 1993
Chitigue (downstream)	1963-1990	Saline	^{137}Cs	0.98	Nyman et al. 1993
deMangue (upstream)	1963-1990	Saline	^{137}Cs	0.94	Nyman et al. 1993
deMangue (midstream)	1963-1990	Saline	^{137}Cs	1.28	Nyman et al. 1993
deMangue (downstream)	1963-1990	Saline	^{137}Cs	0.56	Nyman et al. 1993
Deteriorating saline	1989-1994	Saline	^{137}Cs	0.98 ± 0.36	Nyman et al. 2006
DuFrene	1963-1990	Saline	^{137}Cs	0.55	Nyman et al. 1993
Fourleauge Bay	1975-1979	Saline	^{137}Cs	0.66	Baumann et al. 1984
Grand Bayou	1963-1990	Saline	^{137}Cs	1.04	Nyman et al. 1993
Lafourche Parish	1963-1986	Saline	^{137}Cs	0.47 ± 0.09	DeLaune et al. 1989
Lake Barre	1963-1990	Saline	^{137}Cs	1.78	Nyman et al. 1993
Old Oyster Bayou	1992-2000	Saline	^{137}Cs	0.48 ± 0.09	Rybczyk and Cahoon 2002
Stable saline	1989-1994	Saline	^{137}Cs	0.59 ± 0.14	Nyman et al. 2006
Palmetto Bayou	1986	Freshwater	^{210}Pb	0.73 ± 0.22	DeLaune et al. 1989
Cameron Parish	1986	Brackish	^{210}Pb	0.55	DeLaune et al. 1989
Lafourche Parish	1986	Saline	^{210}Pb	0.42 ± 0.01	DeLaune et al. 1989
Castine Ridge	1999-2004	Forested	feldspar	0.79 ± 0.04	Brantley et al. 2008
Chinchuba Ridge	1999-2004	Forested	feldspar	0.93 ± 0.01	Brantley et al. 2008
Pointe au Chene	1988-1991	Forested	feldspar	0.52 ± 0.16	Rybczyk et al. 2002
Pointe au Chene	1988-1991	Forested	feldspar	0.34 ± 0.06	Rybczyk et al. 2002
Caernarvon diversion	1999	Freshwater	feldspar	1.57 ± 0.05*	Lane et al. 2006
Calcasieu Lake	1978-1980	Intermediate	feldspar	0.66 ± 0.22	DeLaune et al. 1983
Barataria Basin	1975-1979	Saline	feldspar	0.91	Baumann 1980
Bayou Chitigue	1992-1994	Saline	feldspar	5.19 ± 0.32	Cahoon et al. 1995
Old Oyster Bayou	1992-1994	Saline	feldspar	2.07 ± 0.10	Cahoon et al. 1995
Violet diversion	1999	Saline	feldspar	0.44 ± 0.01*	Lane et al. 2006
Barataria Bay	1975-1979	Saline	feldspar	0.90 ± 0.20	Baumann et al. 1984
Fourleauge Bay	1975-1979	Saline	feldspar	0.60 ± 0.20	Baumann et al. 1984

Table 2. Sediment accretion rates for coastal Louisiana streamside sites.					
Site	**Date**	**Wetland Type**	**Method**	**Accretion rate (cm yr^{-1})**	**Source**
STREAMSIDE					
Barataria Basin		All	^{137}Cs	1.50	Hatton 1981
Barataria Basin	1983	Freshwater	^{137}Cs	1.06	Hatton et al. 1983
DNWR	1963-2008	Freshwater	^{137}Cs	>1.59	Wilson and Allison 2008
Palmetto Bayou	1963-1986	Freshwater	^{137}Cs	0.99 ± 0.17	DeLaune et al. 1989
Barataria Basin	1983	Intermediate	^{137}Cs	1.35	Hatton et al. 1983
Barataria Basin	1983	Brackish	^{137}Cs	1.40	Hatton et al. 1983
Cameron Parish	1963-1986	Brackish	^{137}Cs	0.57 ± 0.10	DeLaune et al. 1989
Barataria Basin	1963-1978	Saline	^{137}Cs	1.10	DeLaune et al. 1978
Barataria Basin	1963-1978	Saline	^{137}Cs	1.35	DeLaune et al. 1978
Barataria Basin	1983	Saline	^{137}Cs	1.35	Hatton et al. 1983
Barataria Bay	1963-2008	Saline	^{137}Cs	0.67 ± 0.49	Wilson and Allison 2008
Barataria Bay	1975-1979	Saline	^{137}Cs	1.35	Baumann et al. 1984
Brenton Sound	1963-2008	Saline	^{137}Cs	0.80 ± 0.17	Wilson and Allison 2008
Lafourche Parish	1963-1986	Saline	^{137}Cs	0.68 ± 0.17	DeLaune et al. 1989
Lafourche Parish	1963-1986	Saline	^{137}Cs	0.63 ± 0.07	DeLaune et al. 1989
Palmetto Bayou	1986	Freshwater	^{210}Pb	0.75 ± 0.04	DeLaune et al. 1989
Cameron Parish	1986	Brackish	^{210}Pb	0.52	DeLaune et al. 1989
Lafourche Parish	1986	Saline	^{210}Pb	0.54 ± 0.16	DeLaune et al. 1989
Castine Bayou	1999-2004	Forested	feldspar	0.21 ± 0.03	Brantley et al. 2008
Chinchuba Bayou	1999-2004	Forested	feldspar	1.18 ± 0.03	Brantley et al. 2008
Caernarvon diversion	1999	Freshwater	feldspar	1.11 ± 0.07	Lane et al. 2006
Cameron Parish	1986-1987	Brackish	feldspar	1.13 ± 0.22	Cahoon and Turner 1989
Cameron Parish	1986-1987	Brackish	feldspar	0.35 ± 0.12	Cahoon and Turner 1989
Cameron Parish	1986-1987	Brackish	feldspar	0.43 ± 0.09	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Brackish	feldspar	0.66 ± 0.25	Cahoon and Turner 1989
Castine Bayou	1999-2004	Forested	feldspar	0.21 ± 0.03	Brantley et al. 2008
Chinchuba Bayou	1999-2004	Forested	feldspar	1.18 ± 0.03	Brantley et al. 2008
Caernarvon diversion	1999	Freshwater	feldspar	1.11 ± 0.07*	Lane et al. 2006
Cameron Parish	1986-1987	Brackish	feldspar	1.13 ± 0.22	Cahoon and Turner 1989
Cameron Parish*	1986-1987	Brackish	feldspar	0.35 ± 0.12*	Cahoon and Turner 1989
Cameron Parish*	1986-1987	Brackish	feldspar	0.43 ± 0.09*	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Brackish	feldspar	0.66 ± 0.25	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Brackish	feldspar	0.60 ± 0.12	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Brackish	feldspar	0.99 ± 0.20	Cahoon and Turner 1989
Barataria Basin	1975-1979	Saline	feldspar	1.52	Baumann 1980
Barataria Bay	1975-1979	Saline	feldspar	1.50 ± 0.40	Baumann et al. 1984
Fourleague Bay	1975-1979	Saline	feldspar	1.30 ± 0.60	Baumann et al. 1984
Lafourche Parish	1986-1987	Saline	feldspar	0.60 ± 0.12	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Saline	feldspar	0.66 ± 0.25	Cahoon and Turner 1989
Lafourche Parish	1986-1987	Saline	feldspar	0.99 ± 0.20	Cahoon and Turner 1989
Violet diversion	1999	Saline	feldspar	0.44 ± 0.10*	Lane et al. 2006

Table 3. Sediment accretion rates measured across both coastal Louisiana backmarsh and streamside sites.

Site	Date	Wetland Type	Method	Accretion rate (cm yr^{-1})	Source
BOTH					
Atchafalyaya	1963-1990	Freshwater	^{137}Cs	0.86 ± 0.11	Nyman et al. 1990
Atchafalyaya	1963-1990	Freshwater	^{137}Cs	0.67 ± 0.02	Nyman et al. 1990
Barataria Basin (Little Lake)	1963-1988	Freshwater	^{137}Cs	0.85	Feijtel et al. 1988
Brenton Sound (upper)	1998	Freshwater	^{137}Cs	0.83 ± 0.33	DeLaune and Pezeshki 2003
Caernarvon	1963-1992	Freshwater	^{137}Cs	0.75 ± 0.12	DeLaune et al. 1992
Manchac	1963-1992	Freshwater	^{137}Cs	1.07 ± 0.20	DeLaune et al. 1992
Pointe La Hache	1963-1992	Freshwater	^{137}Cs	0.73 ± 0.13	DeLaune et al. 1992
Brenton Sound (lower)	1998	Intermediate	^{137}Cs	0.66 ± 0.10	DeLaune and Pezeshki 2003
Atchafalyaya	1963-1990	Intermediate	^{137}Cs	0.64 ± 0.38	Nyman et al. 1990
Barataria Basin (Little Lake)	1963-1988	Brackish	^{137}Cs	0.95	Feijtel et al. 1988
Atchafalyaya	1963-1990	Brackish	^{137}Cs	0.72 ± 0.08	Nyman et al. 1990
Barataria Basin	1963-1995	Saline	^{137}Cs	0.52 ± 0.22	Milan et al. 1995
Barataria Basin	1963-1995	Saline	^{137}Cs	0.80 ± 0.25	Milan et al. 1995
Barataria Basin (Little Lake)	1963-1988	Saline	^{137}Cs	1.05	Feijtel et al. 1988
Louisiana (coastal)	1963-1992	Saline	^{137}Cs	0.70 to 0.80	DeLaune et al. 1992
Rigolets	1963-1992	Saline	^{137}Cs	0.77 ± 0.09	DeLaune et al. 1992
St. Bernard Basin	1963-1995	Saline	^{137}Cs	0.50 ± 0.13	Milan et al. 1995
St. Bernard Basin	1963-1995	Saline	^{137}Cs	0.69 ± 0.16	Milan et al. 1995
St. Bernard Parish (Shell Beach)	1963-1992	Saline	^{137}Cs	0.54 ± 0.13	DeLaune et al. 1992
Terrebonne Basin	1963-1995	Saline	^{137}Cs	0.68 ± 0.20	Milan et al. 1995
Terrebonne Basin	1963-1995	Saline	^{137}Cs	0.90 ± 0.18	Milan et al. 1995
Atchafalyaya	1963-1990	Saline	^{137}Cs	0.72 ± 0.14	Nyman et al. 1990
Castine Mid	1999-2004	Forested	feldspar	0.27 ± 0.01	Brantley et al. 2008
Chinchuba Mid	1999-2004	Forested	feldspar	1.01 ± 0.02	Brantley et al. 2008
Brenton Sound (upper)	1998	Freshwater	feldspar	1.72 ± 0.98	DeLaune and Pezeshki 2003
Caernarvon diversion	1999	Freshwater	feldspar	0.75 ± 0.04	Lane et al. 2006
Louisiana (coastal)	--	Freshwater	feldspar	0.66 ± 0.07	Craft 2007
Brenton Sound (lower)	1998	Intermediate	feldspar	1.34 ± 0.32	DeLaune and Pezeshki 2003
Louisiana (coastal)	--	Brackish	feldspar	0.60 ± 0.07	Craft 2007
Louisiana (coastal)	--	Saline	feldspar	0.50 ± 0.08	Craft 2007
Violet diversion	1999	Saline	feldspar	0.34 ± 0.05	Lane et al. 2006

Management practices. Due to the complex factors that contribute to coastal wetland loss in Louisiana, a diverse array of management activities have been implemented to conserve and restore wetlands. In areas where levees, spoil banks, and canals reduce the amount of freshwater flow and sediment available to coastal wetlands, reintroduction of freshwater, sediments, and nutrients can be made by controlled diversions or through the breeching of levees and canals (Bass and Turner 1997; DeLaune and Pezeshki 2003; Lane et al. 2006). In forested wetland systems in Bayou Lafourche, Bayou Chinchuba, and Bayou Castine the introduction of nutrient-rich treated wastewater increased accretion rates high enough to maintain marsh elevations despite increases in relative sea-level rise and high subsidence rates (Rybczyk et al. 2002; Brantley et al. 2008). In salt marshes material from channel dredging projects in the Mississippi River is also successfully being used to create and maintain salt marsh elevation in areas where coastal erosion has converted wetland habitat to open water (Streever 2000; Shafer and Streever 2000; Ray 2007). In addition sedimentation rates were increased on a small scale in coastal marshes in the La Branche wetland through the installation of sediment fences (Boumans et al. 1997). The sediment fences reduced wave shear stress and sediment resuspension, enhanced sediment deposition, and resulted in sediment consolidation and an increase in marsh elevation. While many large-scale projects are underway, and smaller restoration efforts and conservation practices have slowed wetland loss in the coastal Louisiana area, wetlands are still being lost at a greater rate in Louisiana than anywhere else in the United States (Mitsch and Gosselink 2007).

CONCLUSIONS: Average measurements of accretion across the Louisiana coastal region indicate that current accretion rates of 0.7 to 0.8 cm yr^{-1} are not adequate to keep up with current rates of relative sea level rise quantified to be 1.0 cm yr^{-1} in most regions (DeLaune et al. 1992). In addition, marsh substrate elevations are negatively affected by changes in hydrology, local subsidence, compaction, and oxidation of surface sediments, which vary over small spatial scales. Therefore, net accretion varies significantly on a local level and over time (Tables 1–3). Accurate measurements of accretion on a site-by-site basis are necessary to understand how the site is currently responding to sea level rise, to determine what management and conservation practices best address the site, and to predict how the site will respond to projected increases in sea level. Accretion should be measured using multiple methods to quantify changes in both short-term and long-term accretion rates. Finally, accretion rates should be paired with concurrent measurements of elevation change to determine if accretion rates are keeping pace with changes in rates of subsidence.

Further research is required to better understand how Louisiana wetlands are currently compensating for changes in sea level rise. Emerging research highlights the need to further investigate the accuracy of sediment accretion data and to incorporate elevation data into net accretion measurements (Kirwan et al. 2009; Roman et al. 2009), to quantify the potential impacts of eutrophication on below-ground biomass production potentially resulting in greater sediment compaction and even faster rates of marsh elevation loss (Turner et al. 2009), and the need to include spatial biological-physical feedbacks related to the development of patterns in tidal channels and their relationship to the development of vegetation platforms in wetland restoration (Temmerman et al. 2009). A better understanding of how complex costal processes can affect wetland sediment accumulation rates and ultimately changes in marsh elevation is necessary before accurate predictions on how sea level rise may impact coastal Louisiana wetlands can be generated.

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