

Marsh vertical accretion via vegetative growth

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Abstract

Coastal marshes accrete vertically in response to sea-level rise and subsidence. Inadequate accretion and subsequent conversion of coastal marshes to open water generally is attributed to inadequate mineral sedimentation because mineral sedimentation is widely assumed to control accretion. Using ¹³⁷Cs dating to determine vertical accretion, mineral sedimentation, and organic matter accumulation, we found that accretion varied with organic accumulation rather than mineral sedimentation across a wide range of conditions in coastal Louisiana, including stable marshes where soil was 80% mineral matter. These results agreed with previous research, but no mechanism had been proposed to explain accretion via vegetative growth. In an exploratory greenhouse experiment, we found that flooding stimulated root growth above the marsh surface. These results indicated the need for additional work to determine if flooding controls accretion in some marshes by stimulating root growth on the marsh surface, rather than by mineral accumulation on the marsh surface. Restoration or management that focus on mineral sedimentation may be ineffective where a relationship between accretion and mineral sedimentation is assumed rather than tested.

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1. Introduction

Coastal wetland soils are studied to estimate sea-level rise because wetland surfaces vertically accrete as sea level rises (Redfield and Rubin, 1962; Orson et al., 1998). Coastal wetland soils also are studied because shallow open water replaces wetlands if vertical accretion is slower than submergence. Many studies have predicted the fate of specific coastal wetlands by comparing vertical accretion rates to submergence rates (e.g., DeLaune et al., 1983; Thom, 1992; Roman et al., 1997; Orson et al., 1998).

Often, factors limiting vertical accretion are not explored, and vertical accretion is described as depending on a combination of mineral sedimentation and organic matter accumulation

(e.g., Redfield, 1972; Warren and Niering, 1993; Neubauer et al., 2002; Morris et al., 2002; Rooth et al., 2003). When limiting factors are addressed, the fate of coastal wetlands generally is reported to depend primarily on mineral sedimentation (Hatton et al., 1983; Stevenson et al., 1986; Nyman et al., 1990; Reed, 1990; Thom, 1992; Temmerman et al., 2004). Vertical accretion measurements sometimes are used to estimate the amount of mineral sediment contained in marsh soils (Hutchinson and Prandle, 1994), and the mass of mineral sedimentation often is used to compare vertical accretion among sites (Stevenson et al., 1988; Wolaver et al., 1988; Reed, 1989; Childers and Day, 1990; Kuhn et al., 1999; Temmerman et al., 2003). Using the term sediment accumulation to refer to changes in elevation (e.g., Roman et al., 1997) also illustrates the widely held assumption that vertical accretion primarily depends on mineral sedimentation. Tidal, wave, and storm energy delivering sediments are believed to indirectly influence accretion by governing sedimentation (Baumann et al., 1984; Mitsch and Gosselink, 1984; Stoddart et al., 1989), and research

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designed to understand sedimentation in coastal marshes is common.

Despite the widespread assumption that vertical accretion depends on mineral sedimentation or a combination of mineral sedimentation and organic matter accumulation, we are aware of eight reports of vertical accretion depending upon organic matter accumulation (McCaffrey and Thomson, 1980; Hatton et al., 1983; Bricker-Urso et al., 1989; Nyman et al., 1993; Callaway et al., 1997; Anisfeld et al., 1999; Turner et al., 2000; Chmura and Hung, 2004). McCaffrey and Thomson (1980) named this type of accretion “accretion via vegetative growth.” Whereas vertical accretion via sedimentation is widely assumed, vertical accretion via vegetative growth seldom is assumed perhaps because quantitative analyses supporting, and explanatory mechanisms describing, vertical accretion via organic matter accumulation are lacking.

We examined coastal marsh soils in coastal Louisiana to determine if vertical accretion was limited by mineral sedimentation or organic matter accumulation; i.e., to determine if accretion varied with mineral sedimentation or with organic matter accumulation. As noted, some reports use the term “sedimentation” to refer to a mass of mineral sediments or to a mass of mineral sediments and organic matter, and others use it to refer to elevation. Hereafter, “accretion” refers to a vertical distance, “sedimentation” refers to a mass of mineral material, and “organic accumulation” refers to a mass of organic material.

We also studied plant growth characteristics because a mechanism for accretion via vegetative growth was not proposed in previous studies. We contend that accretion via vegetative growth must be activated by flooding and be inactivated by draining, as is accretion via sedimentation. Our vegetative studies were designed to rule out the possibility that aquatic root production could control accretion via vegetative growth. This was an exploratory experiment designed to determine if more complex studies are warranted. We use the term “aquatic root” as Koncalova (1990) did to indicate roots that grow into flood water rather than into soil. We studied aquatic roots of *Spartina patens* Ait Muhl., which occurs throughout the Atlantic coast and Gulf of Mexico coast of North America, and is the most common emergent plant in Louisiana’s 16,000 km² coastal wetlands (Chabreck, 1970).

2. Methods

2.1. Accretion

2.1.1. Core collection sites

We analyzed 68 cores from 31 sites in coastal Louisiana (Fig. 1). We classified these 31 sites as one of three conditions: (1) non-fresh stable, (2) non-fresh deteriorating, or (3) fresh stable. All non-fresh stable sites were *Spartina patens*- or *Spartina alterniflora*-dominated wetlands with relatively slow wetland loss rates, adjacent to Four League Bay (Fig. 1, site 5). All non-fresh deteriorating sites were *Spartina patens*- or *Spartina alterniflora*-dominated wetlands with relatively rapid wetland loss rates north of Lake Barre (Fig. 1, site

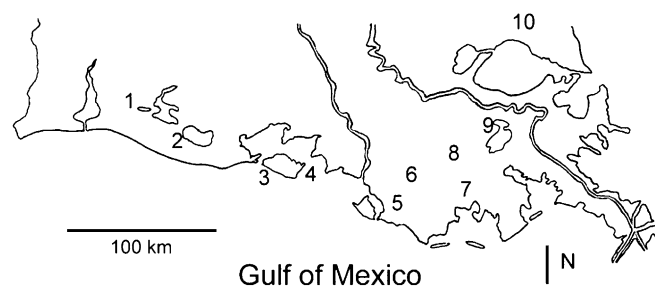


Fig. 1. Map of coastal Louisiana. Cores were collected at Lake Misere (1), a non-fresh stable region (5), a non-fresh deteriorating region (7), Bayou Blue (8), Lake Salvador (9), and Tchefuncte River (10) between 1989 and 1994. Intact plugs of living *Spartina patens* Aiton Muhl were collected from stable marsh at Rockefeller Refuge (2) and Marsh Island (3), and from deteriorating marsh at Marsh Island (4) and Jug Lake (6).

7). All fresh stable sites were *Panicum hemitomon* Schultes- or *Sagittaria lancifolia* Willd.-dominated wetlands with relatively slow wetland loss rates.

The non-fresh stable and non-fresh deteriorating sites have been described regarding wetland loss rates (May and Britsch, 1987; Britsch and Dunbar, 1993) wetland loss maps (Britsch and Dunbar, 1993; Barras et al., 1994), wetland loss mechanisms (Nyman et al., 1993; DeLaune et al., 1994), flood frequency and duration (Wang et al., 1993, 1994, 1995), submergence potential (Rybczyk and Cahoon, 2002), plant stress and productivity (Nyman et al., 1995b; Webb et al., 1995), sediment availability (Baumann et al., 1984; Moeller et al., 1993; Wang et al., 1994; Nyman et al., 1995a; Wang, 1995; Adams et al., 1997; Walker and Hammack, 2000), and subsurface geological features (Penland et al., 1988). Floodwaters from the Atchafalaya River supply abundant freshwater and sediment to the non-fresh stable region each spring, but the non-fresh deteriorating region depends on local rainfall and local sediment reworking for freshwater and sediment. The non-fresh stable region contained similar amounts of brackish marsh, dominated by *Spartina patens*, and saline marsh, dominated by *Spartina alterniflora*, whereas the non-fresh deteriorating region contained more saline marsh than brackish marsh. Those plants are common in temperate North America, where they form extensive, almost monospecific stands. Marsh loss averaged 37 ha yr⁻¹ between 1974 and 1983 in the non-fresh stable region but 515 ha yr⁻¹ in the non-fresh deteriorating region during the same time period (Britsch and Dunbar, 1993). In the non-fresh stable region during 1993, water levels ranged from 15 to 25 cm relative to the marsh surface, and were characterized by long draining periods interspersed by sporadic flooding periods (Wang, 1995). In the non-fresh deteriorating region, water levels during 1993 were lower and ranged from

10 to 50 cm relative to the marsh surface, and were characterized by sporadic draining interspersed by long flooding periods (Wang, 1995). The long periods of flooding in the non-fresh deteriorating region were attributed to a persistent accretion deficit (Rybczyk and Cahoon, 2002) caused by faster shallow subsidence than at the non-fresh stable region (Cahoon et al., 1995). The persistent flooding in the non-fresh deteriorating region stressed emergent vegetation (Nyman et al., 1995b), which

can eventually cause marsh vegetation to drown and lead to peat collapse (DeLaune et al., 1994) and marsh loss. Many of the studies of the non-fresh deteriorating region refer to their study sites as Bayou Chitigue because they focused sampling adjacent to that bayou, whereas many of the studies of the non-fresh stable region refer to their study sites as Old Oyster Bayou because they focused sampling adjacent to that bayou. We too had three sites on each of those bayous, but our other 21 sites were on nearby bayous in those regions. Data from the non-fresh deteriorating region (15 sites, 30 cores) were discussed previously (Nyman et al., 1993) but are re-analyzed here with new data from 12 sites (24 cores) in the non-fresh stable region and with a more robust statistical technique than before.

Cores ($n = 14$) also were collected from four fresh marshes. These cores were not analyzed as intensively as those from the non-fresh regions but were instead used to characterize accretion in mineral-starved conditions. Vegetation at all fresh sites (Fig. 1, sites 1, 8, 9, and 10) was characterized by stands of *Panicum hemitomon* Schultes and *Sagittaria lancifolia* Willd. Both species are common and form extensive, almost monospecific stands throughout the southeastern United States and in South America. All fresh sites were tidal except Lake Misere (Fig. 1, site 1). Wetland loss rates for these sites have not been quantified as they have for the non-fresh sites, but they can be classified as having relatively slow wetland loss rates similar to our non-fresh stable region, and slower wetland loss rates than our non-fresh deteriorating region, based on wetland loss maps of coastal Louisiana (Barras et al., 1994).

2.1.2. Core collection and processing

Between August 1989 and August 1992, a pair of 45- to 55-cm long, 15-cm diameter cores was collected from each of 15 sites in the non-fresh deteriorating region and 12 sites in the non-fresh stable region. The fresh marsh sites were sampled between August and October 1994, except for Bayou Blue, which was sampled September 1990. Streamside marsh was not sampled; all sites were located in inland marsh, i.e., beyond 5 m from the edge of bayous and lakes. Cores were cut into 3-cm or 2-cm layers. Accretion (cm) of the marsh surface since 1963 was estimated from the depth of the soil layer containing the most ^{137}Cs and the number of years between core collection and 1963 (DeLaune et al., 1978). ^{137}Cs dating has been used to measure vertical accretion in tidal freshwater (e.g. Neubauer et al., 2002) as well as tidal saline wetlands (e.g. DeLaune et al., 1978). ^{137}Cs mobility generally is not a problem because illites bind ^{137}Cs internally within days so that only destruction of the clay lattice (e.g., digestion with 5 N nitric acid) causes desorption (Lomenick and Tamura, 1965; Pardue et al., 1989). Hover et al. (2002) found illitic clays near one of the tidal freshwater sites that we sampled (Bayou Blue). See Ritchie and McHenry (1990) for a review of ^{137}Cs dating applications, and Nyman et al. (1993) for more information on ^{137}Cs mobility.

Net accretion was not quantified, but as noted, the long flooding duration (Wang et al., 1994) indicated that net accretion was inadequate to counter submergence in the non-fresh

deteriorating region. The amounts of bulk accumulation (g m^{-2}), sediment accumulation (g m^{-2}), and organic matter accumulation (g m^{-2}) in each core since 1963 also were determined from bulk density (g cm^{-3}), percent sediment (%), and percent organic matter (%) of the soil layer containing the 1963 surface and all overlying soil layers, as described in Nyman et al. (1993). Briefly, bulk density was determined from the weight of oven-dried soil layers, and the calculated volume of the layers prior to drying. The mass of soil that accumulated since 1963 in each core, hereafter called bulk accumulation, was estimated from the soil bulk density of the layer containing the 1963 marsh surface, and all overlying core layers, with eq. (1):

$$\text{Bulk accumulation (g m}^{-2}\text{)} = \left(\sum_{1963 \text{ depth}}^{\text{surface}} \left(\text{bulk density (g cm}^{-3}\text{)} \times \text{layer height (cm)} \right) \right) \times 10,000 \text{ cm}^2 \text{ m}^{-2} \quad (1)$$

The sediment density and organic density of soil in each core layer were estimated from the soil bulk density, percent sediment, and percent organic matter measured individually for each core layer. The mass of sediment that accumulated since 1963 in each core, hereafter called sedimentation, was estimated from the sediment density of the layer containing the 1963 marsh surface, and all overlying core layers, with eq. (2):

$$\text{Sedimentation (g m}^{-2}\text{)} = \left(\sum_{1963 \text{ depth}}^{\text{surface}} \left(\text{sediment density (g cm}^{-3}\text{)} \times \text{layer height (cm)} \right) \right) \times 10,000 \text{ cm}^2 \text{ m}^{-2} \quad (2)$$

Likewise, the mass of organic matter that accumulated since 1963 in each core, hereafter called organic accumulation, was estimated from the organic density of the layer containing the 1963 marsh surface, and all overlying core layers, with eq. (3):

$$\text{Organic accumulation (g m}^{-2}\text{)} = \left(\sum_{1963 \text{ depth}}^{\text{surface}} \left(\text{organic density (g cm}^{-3}\text{)} \times \text{layer height (cm)} \right) \right) \times 10,000 \text{ cm}^2 \text{ m}^{-2} \quad (3)$$

These analyses were made on only 55 of the 68 cores because estimates of bulk density or percent mineral matter were not obtained from 13 cores (one from the non-fresh deteriorating region; two from the non-fresh stable region, and 10 from the fresh, stable region).

2.1.3. Statistical analyses of core data

Statistics characterizing soil bulk density (g cm^{-3}), mineral matter content (%), accretion rates (cm yr^{-1}), sedimentation rates ($\text{g m}^{-2} \text{ yr}^{-1}$), and organic accumulation rates ($\text{g m}^{-2} \text{ yr}^{-1}$) were calculated from the soil containing the 1963 marsh surface, and

all overlying soil, in each core. Those descriptive characteristics were compared among the three regions as a completely randomized design. These tests were unrelated to our hypotheses but were conducted to characterize accretion and soil properties and to document differences among the three regions. Tukey's studentized range test (HSD) was used to determine which regions differed from the others for each characteristic that differed significantly among the regions. Statistics presented in tables and text were mean and standard deviation; the format used was mean (standard deviation).

Our primary goal was to test the hypotheses that variation among cores in accretion (cm) was unrelated to variation among cores in sedimentation (g m^{-2}) and organic matter accumulation (g m^{-2}). If accretion was unrelated to sedimentation, then sedimentation could not have controlled accretion; if accretion was unrelated to organic matter accumulation, then organic matter accumulation could not have controlled accretion. We omitted the fresh stable region from this analyses because sedimentation and organic matter accumulation were determined only in four of those 14 cores. We analyzed the remaining two regions separately because differences in sediment availability probably caused them to differ in sedimentation, and differences in flooding probably caused them to differ in organic accumulation. These data were analyzed using Type II multiple regression analysis (Sokal and Rohlf, 1969); the residuals were normally distributed and unrelated to the predicted values. Our statistical conclusions were the same if we instead used an equivalent nonparametric correlation analysis (i.e., Spearman partial correlation coefficients).

2.2. Aquatic root study

The effects of water depth and salinity on aquatic root initiation were studied in greenhouse experiments using intact plugs of *Spartina patens*. These are the first experiments of which we are aware to determine the effects of flooding and salinity on aquatic root production in a common wetland plant species. We use the term "aquatic root" as Koncalova (1990) did to describe roots that grow above the soil surface into flood water rather than below the soil surface into soil. Koncalova (1990) noted that aquatic roots differ from soil roots in that aquatic roots lack aerenchyma, and they uptake nutrients rather than anchor plants. These exploratory experiments were designed to test the hypothesis that production of aquatic roots is unaffected by flooding and salinity. If true, then we contend that aquatic roots could not cause accretion via vegetative growth, and future work would not be justified to determine how aquatic roots decompose, compact, and ultimately affect marsh surface elevation, or to determine the relative role of aquatic roots, soil roots, above ground tissue, and mineral sediments in elevation change.

2.2.1. Plug collection sites

Plugs were collected from Rockefeller State Wildlife Refuge and Game Preserve, Marsh Island State Wildlife Refuge (two sites), and near Jug Lake Louisiana (sites 2, 3, and 6 respectively in Fig. 1). Plugs were collected by pushing

sharpened PVC pipe (15 cm diameter, 30 cm length) into the marsh surface, removing the pipe and enclosed soil and live plants from the marsh, and placing a PVC cap over the bottom. After returning the plugs to the greenhouse, the plugs were stored in a 890-L tank of fresh water and flooded to the soil surface. The plugs were never removed from the PVC containers.

2.2.2. Greenhouse experiment

We monitored the number of stems with aquatic roots in plugs that we exposed to different water salinities (0.3, 7, and 15) and water levels (-5 cm, 0 cm, and $+5$ cm relative to the soil surface in the plugs). Those values are within the range of water salinity and level in *Spartina patens* dominated marshes in Louisiana. Salinity in bayous of marshes dominated by *S. patens* in coastal Louisiana averages 8.6, whereas water levels average 9 cm below the marsh surface (J.A. Nyman, unpublished data). Measurements of soil pore water are rare, but soils generally are saturated to the surface (Nyman et al., 1994) and salinity is less variable than in adjacent bayous (J.A. Nyman, unpublished data). Hereafter, we refer to the water level treatments as drained, saturated, and flooded. The water salinity in six 890-L tanks was manipulated with a commercial aquarium salt (Forty Fathoms marine mix). There were two tanks for each water salinity treatment. Water depth within plugs was manipulated by elevating plugs assigned to 0 and -5 cm treatments.

Each of the 54 plugs were randomly assigned to the treatments. Just prior to applying treatments, we marked 20 randomly selected stems in each plug. In a few plugs that contained fewer than 20 stems, all stems were marked. The number of stems that we monitored ranged from 12 to 20 (average = 19.5). We then counted the number of marked stems with aquatic roots for 25 or 26 days after we initiated treatments. We calculated the percent of stems with aquatic roots at the beginning of the experiment, the percent of stems with aquatic roots at the end of the experiment, and the change in that percentage.

2.2.3. Statistical analyses of greenhouse data

The data were analyzed to test for differences in the change in percent of stems with aquatic roots as a three-by-three factorial (water level by salinity) with blocking on site. If flooding failed to stimulate aquatic root production then aquatic roots could not cause accretion via vegetative growth. We used a randomization technique to determine statistical significance (Edington, 1995) rather than an *F*-test, because the data were not normally distributed and because the sites were not randomly selected from the population of all sites in Louisiana dominated by *Spartina patens*. Randomization tests require neither random sampling nor normally distributed data (Edington, 1995). Randomization tests are extremely accurate; they are the benchmark against which parametric tests are compared when deciding if a parametric test is robust or not (Edington, 1995). We selected the Type III sums of squares of the observed data as our test statistic. To test a term in the model, we randomly re-assigned that term to the observed data 4999

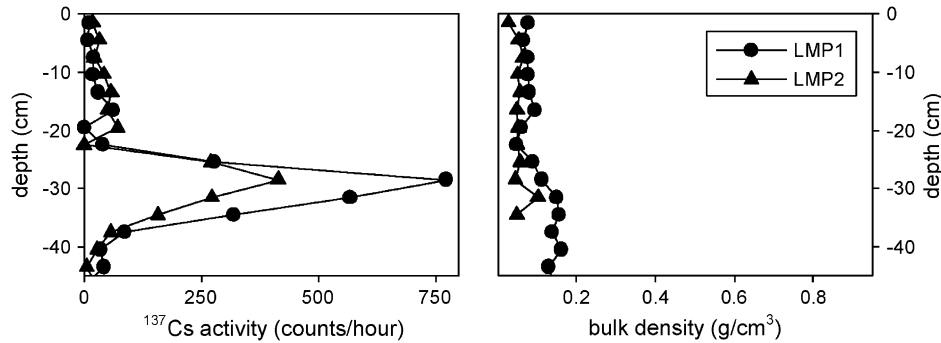


Fig. 2. Example of ^{137}Cs and bulk density profiles with depth in a pair of cores collected from a fresh stable site. This pair of cores was one of two collected near Lake Misere, Louisiana. This pair was from *Panicum hemitomon* dominated marsh (#1 on Fig. 1).

times and then re-calculated the Type III sums of squares each time. For instance, to test the salinity effect, only salinity was randomly reassigned to the observed data and the test statistic was re-calculated 4999 times. If there was no salinity effect, then the Type III sums of squares calculated for salinity from the observed data would be exceeded by many of the Type III sums of squares calculated when the salinity was randomly assigned to the observations. Each term in the model and each interaction tested required a separate set of 4999 randomizations and calculations of the Type III sums of squares. The probability of a term in the model having no effect was calculated from the portion of the 5000 Type III sums of squares (1 observed and 4999 generated from randomly arranged data) that equaled or exceeded the observed Type III sums of squares. We used an alpha level of 0.05. SAS software was used for all calculations and randomizations.

3. Results

The 1963 marsh surface was almost always in identical or adjacent layers in replicate cores from the same site (e.g., Figs. 2, 3). Profiles of ^{137}Cs profiles with depth reflected gradual changes in atmospheric deposition of fallout with maximum rates in 1963 (e.g., Figs. 2, 3). Bulk density profiles reflected the more dynamic process of mineral sedimentation (e.g., Figs. 2, 3).

Accretion differed among the regions ($P = 0.0063$); Tukey's test indicated that accretion in the non-fresh stable region was less than that in the non-fresh deteriorating region

(Table 1). Accretion in the fresh stable region did not differ from that in the other two regions.

Bulk density differed among the regions ($P < 0.0001$); Tukey's test indicated that bulk density differed among all three regions with bulk density greatest in the non-fresh stable region and least in the fresh stable region (Table 1). Mineral matter content ($P < 0.001$), bulk accumulation ($P < 0.0001$), and sedimentation ($P < 0.0001$) all differed among the regions such that values were lower in the fresh stable region than in the two non-fresh regions, which did not differ significantly from one another (Tables 1 and 2). Accretion rates were not correlated to bulk density ($P = 0.2288$).

Organic accumulation did not differ among the regions ($P = 0.2751$). Accretion rates were not correlated to organic matter content ($P = 0.5767$). Thus, the two non-fresh regions differed in accretion but not in sedimentation or organic accumulation, which indicated that similar masses of sediment and organic matter were incorporated into different volumes of soil.

Our primary hypothesis was tested with multiple regression analyses of cores from the two non-fresh regions. Those analyses indicated that variation in accretion was unrelated to sedimentation in either the stable ($t = 0.8120$) or deteriorating region ($t = 0.9542$) but was related to organic matter accumulation in both regions ($t < 0.0001$ and $t < 0.0001$, respectively). Visual examination of the data illustrates the lack of a relationship between accretion and sedimentation (Fig. 4) and a strong relationship between accretion and organic accumulation (Fig. 5). The strength of the relationship between

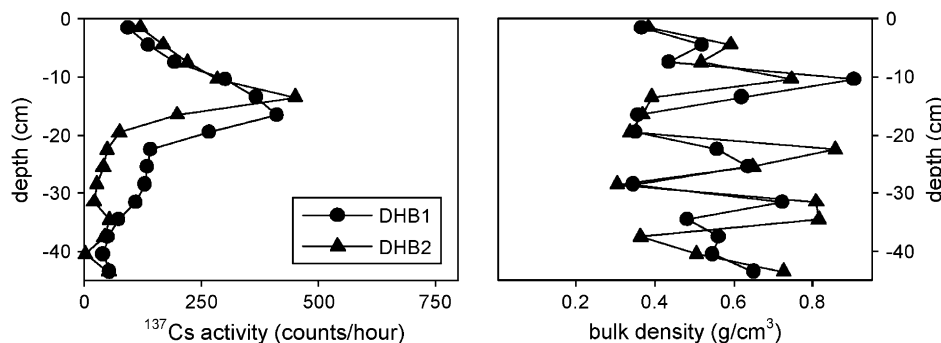


Fig. 3. Example of ^{137}Cs and bulk density profiles with depth in a pair of cores collected from a non-fresh stable site. This pair of cores was one of 12 collected from near Four League Bay, Louisiana (#5 on Fig. 1). This pair was from *Spartina alterniflora* dominated marsh.

Table 1

Mean and standard deviation of some soil characteristics in stable and non-fresh deteriorating marshes in coastal Louisiana. Bulk density was measured in the upper 15 cm of soil; all other parameters were measured in post-1963 soil

	Mineral content (%)	Bulk density (g cm ⁻³)	Accretion (cm yr ⁻¹)	Bulk accumulation (g m ⁻² yr ⁻¹)	Sedimentation (g m ⁻² yr ⁻¹)	Organic accumulation (g m ⁻² yr ⁻¹)
Non-fresh stable						
Saline (n = 12)	79 (7)	0.38 (0.11)	0.59 (0.14)	2383 (972)	1928 (898)	424 (82)
Brackish (n = 12)	66 (4) ^a	0.23 (0.04)	0.88 (0.14)	2012 (351)	1270 (255) ^a	604 (83) ^a
Average	73 (8)	0.31 (0.11)	0.74 (0.20)	2198 (739)	1629 (750)	506 (122)
Non-fresh deteriorating						
Saline (n = 22)	74 (6)	0.24 (0.06) ^b	0.98 (0.36)	2588 (1170) ^b	1970 (941) ^b	618 (261) ^b
Brackish (n = 8)	56 (7)	0.12 (0.02)	0.96 (0.32)	1266 (529)	724 (326)	542 (217)
Average	70 (10)	0.21 (0.07)	0.98 (0.35)	2224 (1188)	1626 (990)	597 (248)
Fresh stable						
Average (n = 14)	38 (18) ^c	0.10 (0.03)	0.82 (0.15)	878 (226)	374 (202) ^c	538 (94) ^c

^a n = 10.

^b n = 21.

^c n = 4.

accretion and organic accumulation also was evident in the R^2 of simple regression models relating accretion to organic accumulation, which were 0.6957 in the stable region and 0.8764 in the deteriorating region.

A relationship between organic accumulation and accretion does not require that organic accumulation controlled accretion, but the lack of relationship between sedimentation and

accretion requires that sedimentation could not have caused accretion. We therefore concluded that organic accumulation controlled accretion in these marshes.

Analyses of 14 cores collected from the fresh stable region confirm that some marshes can accrete rapidly and sufficiently to counter submergence even when sedimentation, tidal energy, and wave energy are minute. Accretion at these sites

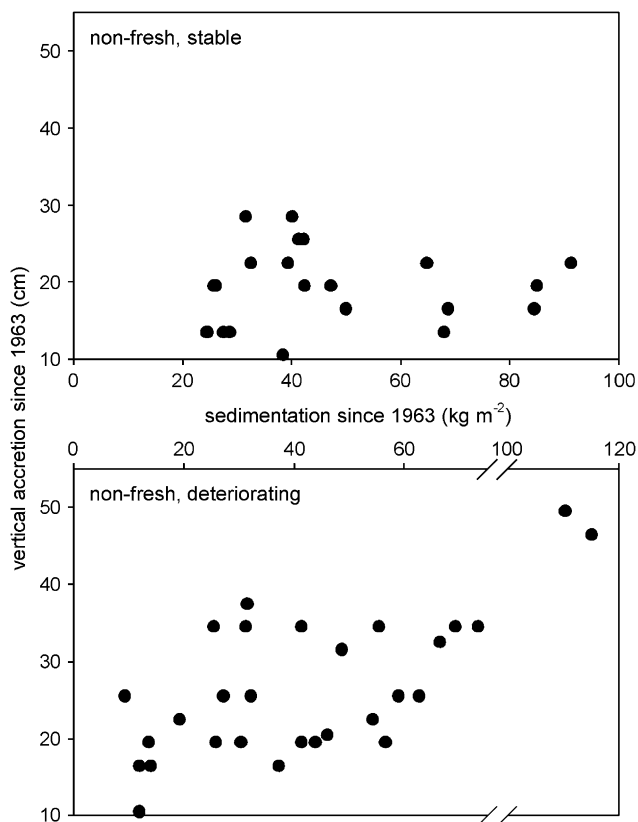


Fig. 4. Relationship between vertical accretion and sedimentation in a non-fresh stable region and a non-fresh deteriorating region of coastal Louisiana.

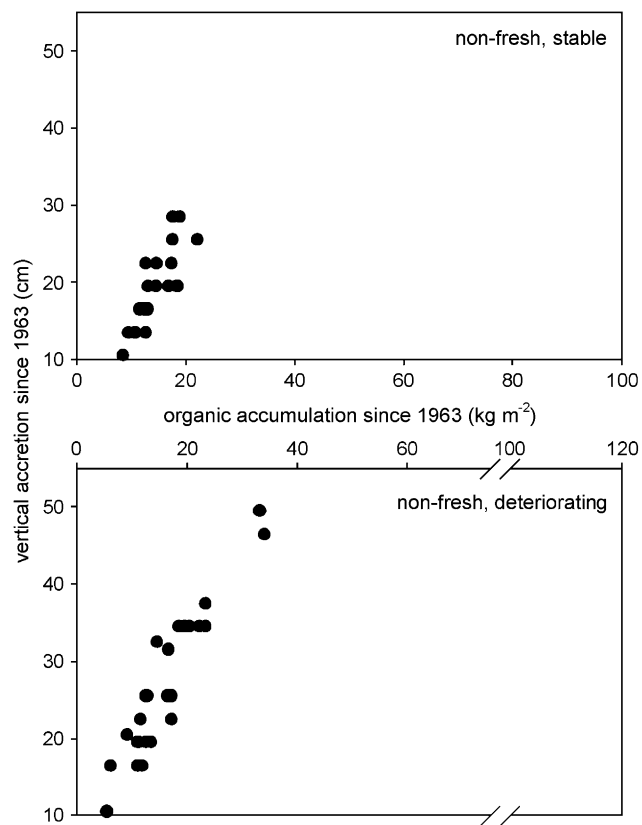


Fig. 5. Relationship between vertical accretion and organic accumulation in a non-fresh stable region and a non-fresh deteriorating region of coastal Louisiana.

Table 2

Some soil characteristics at four freshwater marshes in coastal Louisiana. A pair of cores was collected at Bayou Blue, four cores were collected at other sites. Mineral sediments are virtually unavailable at all except the Tchefuncte River site

Site	Accretion (cm yr ⁻¹)	Bulk accumulation (g m ⁻² yr ⁻¹)	Sedimentation (g m ⁻² yr ⁻¹)	Organic accumulation (g m ⁻² yr ⁻¹)
Bayou Blue	0.83	917	461	456
Lake Misere	1.02	936	488 ^a	621 ^a
Lake Salvador	0.72	621	85 ^a	617 ^a
Tchefuncte River	0.72	1056	No data	No data

^a Data from one core.

was similar to that in the two non-fresh regions even though sedimentation rates were roughly 50% slower (Table 2).

In the greenhouse experiment, we observed new aquatic roots within 3 days of initiating treatments. Analyses of those data indicated that flooding promoted aquatic-root production ($P = 0.0002$). Flooding caused approximately 10% of the stems to develop aquatic roots; drainage and saturation did increase the percentage of stems with aquatic roots (Fig. 6). Graphical examination of the data suggest that aquatic root production was greatest when flood water was 1 ppt (Fig. 6) but statistical analyses indicated that there was no interaction between salinity and flooding ($P = 0.0904$) and no salinity effect ($P = 0.1154$). We also observed that aquatic roots became difficult to distinguish from soil roots within 25 days of initiating treatments, which may have led us to under-estimate the percentage of stems with water roots as the experiment continued.

4. Discussion

4.1. Relationships among accretion, sedimentation, and organic accumulation

The most important difference between the non-fresh stable region and the non-fresh deteriorating region is that accretion

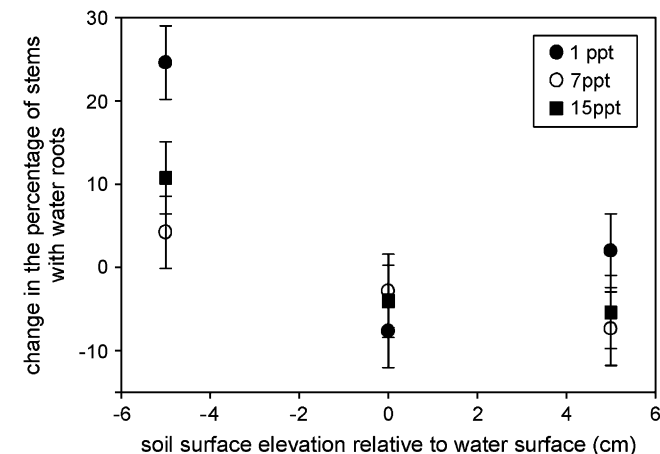


Fig. 6. Changes during the 25 or 26 days after applying salinity and flooding treatments in the percent of *Spartina patens* Aiton Muhl stems having aquatic roots.

was adequate in the non-fresh stable region but inadequate in the non-fresh deteriorating region. Wetland loss maps (May and Britsch, 1987) indicate that inadequate accretion, rather than shoreline erosion, cause most marsh loss in the non-fresh deteriorating region. Accurate identification of the factor limiting accretion is essential to efficiently manage coastal wetlands threatened by inadequate accretion, such as the non-fresh deteriorating region.

It would be incorrect to assume that accretion depends on sediments where and when there is a greater mass of sediments than organic matter. Limiting factors are identifiable because as they vary in abundance, their product also varies. For instance, P sometimes limits plant growth but is always less abundant than N. If sedimentation limited accretion at our sites, then accretion would be greater where sedimentation was greater. To the contrary, differences in sedimentation did not cause differences in accretion among our sites. Instead, our statistical analyses agree with previous researchers who used logic (McCaffrey and Thomson, 1980; Hatton et al., 1983; Bricker-Urso et al., 1989; Anisfeld et al., 1999), simple correlations (Chmura and Hung, 2004), or multiple regression analyses (Nyman et al., 1993; Turner et al., 2000) to conclude that some coastal marshes accrete via a vegetative growth mechanism rather than by sedimentation. It appears that some coastal marshes are analogous to coral reefs in that both result from biologic rather than physical processes.

In the non-fresh deteriorating region, accretion via vegetative growth was insufficient to maintain elevation. The combination of accretion that depends on organic matter accumulation, with inadequate accretion to maintain elevation, apparently results in a positive feedback loop of inadequate vertical accretion and increasing flooding stress on vegetation (Nyman et al., 1993). In the non-fresh stable region, however, accretion via vegetative growth was adequate to prevent rapid marsh loss. In the fresh marshes as well, where sedimentation was approximately 50% slower, accretion via vegetative growth was adequate to prevent rapid marsh loss.

Although accretion in all three regions was controlled by organic accumulation, the relationship between the mass of organic accumulation and elevation varied among the regions and apparently among the marsh types. The amount of organic matter required to cause one centimeter of accretion varied from 565 g m⁻² in brackish portions of the non-fresh deteriorating brackish marsh to 758 g m⁻² in saline portions of the non-fresh stable marsh. In the fresh marshes, an intermediate amount of organic matter, 656 g m⁻², was required to accrete 1 cm. Different relationships between organic accumulation and accretion result when different masses of material are incorporated into similar volumes of soil, which agrees with previous research showing that variability in pore space explains much variability in accretion (Callaway et al., 1996; Anisfeld et al., 1999).

Ward et al. (1998) and Allen (2000) described coastal marshes where mineral sedimentation was greater where flooding was greater. In contrast, our non-saline regions differed in flooding (Wang, 1995) but not in sedimentation. One possible reason that our non-saline regions had similar

sedimentation despite differing in flooding is that they also differed in suspended sediment availability. Suspended sediment concentrations were higher in the stable region (Wang et al., 1994) than in the deteriorating region (Wang et al., 1993), because only the stable region was flooded annually by the Atchafalaya River (Baumann et al., 1984). Thus, more flooding at the deteriorating region may have offset the less abundant suspended sediments there, whereas less flooding at the stable region may have offset the more abundant suspended sediments there.

The two non-fresh regions differed in bulk density but not sedimentation, which resulted from similar masses of mineral sediments being incorporated into different volumes of soil. This situation leads to the counterintuitive result that differences in bulk density need not be an indication of differences in sedimentation as some have previously concluded (e.g., Kearney et al., 1994). Our two non-fresh regions had similar sedimentation rates, but different bulk density, because the regions accreted at different rates.

4.2. Conditions under which accretion occurs via vegetative growth

There has been little examination of the conditions under which accretion occurs via vegetative growth. Previous workers speculated that accretion via organic accumulation was associated with marsh loss, sediment starvation, or inadequate tidal action (Hatton et al., 1983; Stevenson et al., 1985; Craft et al., 1993). However, our data and analyses demonstrate that accretion via organic accumulation also occurred where sediment availability was high and marsh was stable. It is possible that where a dependence of accretion on sedimentation has been assumed rather than tested, a re-examination of relationships among accretion, sedimentation, and organic matter accumulation may disclose that accretion via organic accumulation is more widespread than reported.

4.3. Mechanism of accretion via vegetative growth

Managing marshes that accrete via vegetative growth will require an understanding of the mechanism(s) that promote organic matter accumulation and pore space. Analyses of relationships among accretion, sedimentation, and organic matter accumulation at our sites demonstrated unequivocally that accretion was unrelated to sedimentation but provide no insight into processes that could allow organic accumulation to control accretion. Some researchers speculated that detritus originating from aboveground plant parts (Craft et al., 1993; Rybczyk et al., 1998), root tissue inserted into the sediments (Wolaver et al., 1988), or enhanced sediment trapping by litter (Rooth et al., 2003) increased elevation. Those processes might be important at their sites, but at our sites, accretion appeared to proceed via the production of aquatic roots that develop into a fibrous network just above the existing marsh surface that in time became indistinguishable from the pre-existing root network and marsh surface (Figs. 7, 8). In time those new roots could become buried by newer, more elevated roots and would develop aerenchyma (Hook, 1984). At our



Fig. 7. The root network at the marsh surface in a typical *Spartina alterniflora* Loisel. marsh in coastal Louisiana. The root network is visible when mud is gently washed away. This photograph was made by protecting the right half of a vertically sliced 15-cm diameter core from a stream of water that washed mud away from the left half of the core. The core was collected from salt marsh adjacent to the Louisiana Universities Marine Consortium near Cocodrie, Louisiana, USA.

sites, accretion via vegetative growth was apparently similar to accretion via mineral sedimentation in that both occur above the existing marsh surface and both create new, more elevated marsh surfaces.



Fig. 8. The root network at the marsh surface in a typical *Spartina patens* Aiton Muhl marsh in coastal Louisiana. Aquatic roots occur above the marsh surface; deposits are dominated by roots and upright stems rather than detritus. Differences in oxidative–reductive status cause the color difference between the oxic surface and anoxic subsurface. This photograph was made by vertically slicing a 15-cm diameter core. The core was collected from stable, brackish marsh at the Russel Sage Marsh Island Wildlife Refuge, Louisiana, USA.

Our greenhouse experiments were designed to eliminate the possibility that flooding and desiccation cause a self-regulating system of accretion via root production in the most common species in Louisiana's 16,000 km² of coastal marshes (Chabreck, 1970). That possibility would have been eliminated if root initiation did not differ significantly among the treatments, but we found that flooding stimulated aquatic root production. However, the short duration of our experiments and narrow range of flooding and salinity preclude any conclusion other than accretion via vegetative growth might be related to root growth above the marsh surface. Nor did these experiments relate aquatic root production, growth, and survival to changes in elevation. We recommend future studies that attempt to recreate our observations and also determine the effects of currents, flooding, and salinity on aquatic root growth, litter accumulation, sediment uplift, and accretion. We also recommend that future studies be shorter and include aquatic roots that originate from the soil surface, because we observed that many new aquatic roots initiated within 3 days after flooding began, and that there were many more aquatic roots originating vertically from the soil surface than horizontally from the stems. In the following paragraph, we explore some of the possible questions raised by our observations.

The origin of aquatic roots may vary by plant species. In *Spartina alterniflora*, they apparently arise solely from negatively geotropic roots produced in flood water by existing roots just below the soil surface (Fig. 7). In *Spartina patens*, some aquatic roots apparently originated upright stems (Fig. 8). Flooding, hypoxia, and ethylene stimulate root development near the soil surface with soil roots anchoring the plant and aquatic roots taking up nutrients (Blom et al., 1990; Koncalova, 1990; Naidoo and Naidoo, 1992). Aquatic root production may be stimulated by prolonged flooding with shallow, moving fresh water, such as occurs during river floods (Hook, 1984). We hypothesize that submergence gradually causes anoxia to kill soil roots, aquatic roots to become soil roots via the development of a barrier to oxygen loss such as a lignified exodermis, and the production of new aquatic roots above the old marsh surface. Such root production in response to flooding, in conjunction with the cessation of aquatic root production upon the cessation of flooding (Blom et al., 1990; Koncalova, 1990), might form a stable, negative-feedback system whereby accretion via vegetative growth could maintain the elevation of a marsh surface within a slowly rising intertidal zone.

4.4. Roles of sediment in accretion via vegetative growth

Sphagnum bogs and floating wetlands provide ample evidence that freshwater wetlands do not require sediment to accrete. Sphagnum bogs have existed for centuries with only atmospheric deposition of sediment yet accrete continually (Mitsch and Gosselink, 1984). However, sediment probably indirectly affects accretion occurring via vegetative growth. Sediments that weakly bind PO₄, such as clays that contain free iron or aluminum oxide (Ellis and Truog, 1955; Mitsch

and Gosselink, 1984), probably increase plant growth where it is limited by P availability.

The worldwide lack of floating non-fresh marshes suggests that non-fresh wetlands require some soil mineral matter. Additional evidence that non-fresh marshes require mineral matter is found in relationships between plant biomass and soil bulk density in saline (Broome et al., 1975; DeLaune and Pezeshki, 1988) and brackish (Nyman et al., 1994) marshes. The requirement of non-fresh marshes for soil mineral matter may be related to sulfate (SO₄²⁻) delivered by sea water, which is reduced to toxic sulfides (S²⁻) produced from sulfate by soil bacteria. Reduced Fe removes sulfide via precipitation, but sulfide availability can exceed Fe availability (Griffin and Rabenhorst, 1989). Thus, Fe may improve plant growth in the presence of sulfate reduction, and the amount of S²⁻ that wetland plants can tolerate may depend partly on the amount of soil Fe. These relationships among soil Fe, soil SO₄²⁻, soil S²⁻, and plant growth were first proposed by King et al. (1982) working within a Georgia salt marsh. If the amount of soil Fe required for plant growth varies with sulfide at a larger scale than King et al. (1982) proposed, then tidal, non-saline marshes experiencing salt water intrusion (which may accompany faster global sea level rise rates) may fail to convert to a more saline type unless sedimentation increases because soil mineral matter content is generally much lower in tidal, non-saline marshes than in saline marshes (Mitsch and Gosselink, 1984; Nyman et al., 1990). Thus, tidal non-saline marshes that accrete via vegetative growth may not survive marine intrusion even when the increase in salinity is very gradual unless sufficient soil Fe develops for salt marsh conditions.

The existence of organic matter accumulation controlling accretion over the 30-year time spans that we studied does not preclude the existence of periods during which sedimentation controls accretion. Infrequent hurricanes can deposit sediments in a few days that are equivalent to 4 to 11 times the long-term annual rate of accretion (Nyman et al., 1995a). It is possible that some variation in accretion rates between sites, as well as over-time within sites (e.g., Kearney et al., 1994), results from switching between accretion via vegetative growth and accretion via sedimentation. If switching between accretion via vegetative growth and accretion via sedimentation occurred regularly on an annual cycle, then one would expect layers of fibrous material alternating with layers of almost pure sand as described by Redfield (1972).

5. Conclusions

We applied the concept of limiting factors to marsh vertical accretion and demonstrated that accretion did not increase with sedimentation at our sites. Our conclusions may not apply to streamside marshes, which we did not sample. Our analyses of relationships among accretion, sedimentation, and organic matter accumulation agreed with previous conclusions that accretion in some coastal marshes proceeds via a vegetative growth mechanism as first reported by McCaffrey and Thomson (1980). Our analyses also demonstrated that accretion via

vegetative growth is neither a consequence of inadequate mineral sediment availability nor a cause of inadequate accretion. We recommend that coastal marsh managers test, rather than assume, which factor controls accretion at their sites. Otherwise, efforts to manage coastal marshes in areas with rapid submergence or during periods of rapid sea level rise may be ineffective or inefficient.

Our exploratory greenhouse experiment failed to eliminate a role for aquatic roots in accretion via vegetative growth. However, the short duration of our experiments and narrow range of flooding and salinity preclude any conclusion other than accretion via vegetative growth might be related to root growth above the marsh surface. Future research should focus on factors controlling aquatic root growth and the relative importance of allochthonous organic matter, aboveground detritus, belowground roots and aquatic roots in vertical accretion.

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