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LONG-TERM VEGETATION CHANGE IN LOUISIANA TIDAL MARSHES, 1968–1992

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Abstract: The Louisiana coastal marshes form some of the most extensive wetlands within the continental United States. The problem of land loss in these coastal marshes is well-documented, but very little is known about possible changes in vegetation composition that might be associated with this loss. We analyzed vegetation data collected from 1968 to 1992 in the tidal wetlands of Terrebonne parish and described five vegetation types that occur in this region. Our data did not show the predicted change to more salt-tolerant vegetation. This is probably due to the influence of the Atchafalaya River in the study area. However, we documented a large change in the dominant vegetation of the fresh marsh. *Panicum hemitomon*-dominated marshes occupied 51% of the study area in 1968 and only 14% in 1992. This vegetation type was replaced with *Eleocharis baldwinii*-dominated marshes (3% in 1968 to 41% in 1992). This change occurred adjacent to an area of significant conversion to open water. Based on limited available data from the literature, we evaluated three potential driving factors in this change–grazing, water-level increase, and water quality–but could not determine the cause of change definitively.

Key Words: vegetation change, Louisiana, tidal marsh

INTRODUCTION

The Louisiana coastal marshes form some of the most extensive wetlands within the continental United States. Louisiana contains 27% of coastal marshes of the continental US (Louisiana 6×10^6 ha and continental US 22×10^6 ha of fresh and saline marsh) (Field et al. 1991). The problem of land loss in these coastal marshes is well-documented (Craig et al. 1979, Gagliano et al. 1981, Scaife et al. 1983, Sasser et al. 1986, Walker et al. 1987, Evers et al. 1992, Boesch et al. 1994). Louisiana is undergoing the most severe wetland loss in the United States, with loss rates estimated from 6,000 to 12,450 ha yr-1 (Britsch and Kemp 1990, Turner 1990, Britsch and Dunbar 1993). However, little is known about changes in vegetation composition associated with this severe wetland loss.

The Mississippi delta cycle, the geological process affecting most of Louisiana's coastal marshes, shows that historically fresh marshes revert to saline marshes when the marsh is no longer nourished by the river (Coleman and Gagliano 1964). At present, most of the marshes in the Louisiana delta plain are no longer nourished by the river because river flow is confined by large levees. Combined with high subsidence rates in the region and global sea-level rise (Baumann et al. 1984), this is expected to lead to salt-water intrusion into the fresh marshes. Chabreck and Linscombe (1982) showed that in the Louisiana coastal zone, saline marshes expanded at the cost of fresh marsh between 1968 and 1978. However, Wiseman et al. (1990) showed that, overall, there was no indication of increasing salinities along the Louisiana coast in the period 1960 to 1985 and that average yearly coastal salinities were negatively correlated with the yearly discharge of the Mississippi River.

The objective of this study is to determine whether vegetation community changes occurred in the tidal marshes in the Mississippi River Deltaic Plain during the period that major land loss occurred in the region.

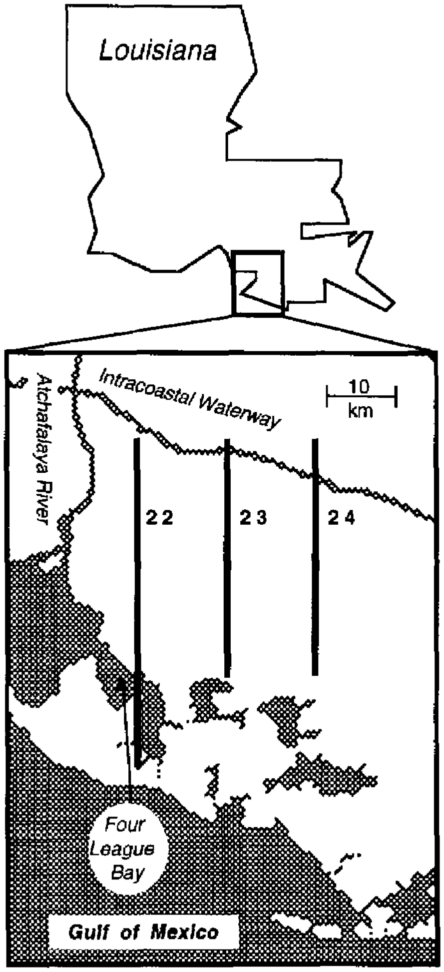

Figure 1. Location of the transects in Terrebonne parish, Louisiana. Transect numbers were assigned by Chabreck (1972).

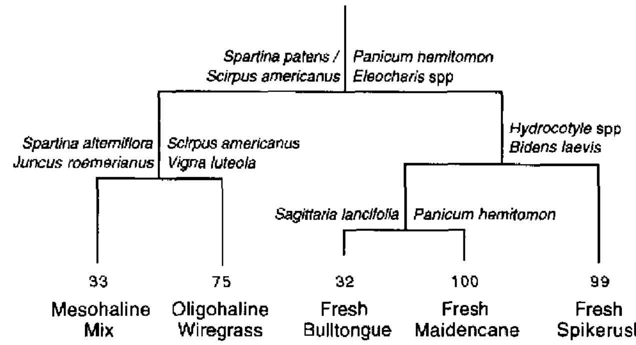

Figure 2. Summary of the TWINSPAN analysis of abundance data from all years. Numbers indicate the number of stations classified into each vegetation type.

To meet this objective, we analyzed vegetation data collected by R. Chabreck and others from 1968 to 1992 in the tidal wetlands of Terrebonne estuary to determine naturally occurring vegetation associations and how these changed between 1968 and 1992.

METHODS

Four data sets were used in this study (1968, 1978, 1984, and 1992). The 1968 data were part of a coastwide survey of vegetation funded by the Louisiana Wildlife and Fisheries Commission and the U.S. Army Corps of Engineers (USACOE). This survey consisted of 39 north-south transects spaced at 13 km (7.5' longitude) intervals, covering coastal Louisiana from the Texas to the Mississippi border (Chabreck 1972). Sample stations were spaced along these transects at 0.4 km intervals. Our analysis included the stations from transects 22, 23, and 24 that were located in fresh to brackish marshes (Figure 1), were repeatedly sampled (1968, 1978, 1984, 1992), and consisted of emergent marsh (see below). The 1968 data used in our analysis consisted of 88 stations with emergent vegetation.

In 1978, only the fresh-to-oligohaline stations (Chabreck 1972) from three transects in western Terrebonne basin were revisited for complete vegetation surveys. Sixty-seven stations with emergent vegetation from the 1978 survey were used in our analysis. These surveys were funded by the USACOE as part of a study on the possible impacts from levees on the Lower Atchafalaya River. In 1984 and 1992, the USACOE study was continued. Every other station in the fresh through brackish zones from the three transects surveyed in 1978 were revisited for complete vegetation surveys. The data set used in our analysis contained 96 stations in 1984 and 92 stations in 1992. Although stations were sampled in a similar location each year, the location in each year might not have been exactly the same, since permanent markers were not used.

Vegetation was surveyed by one observer from a helicopter hovering above the station. Each species occurring in a 30 m radius from the station was recorded and assigned an abundance value. Abundance values used were 3 = abundant, 2 = common, and 1 = rare. Species identification was improved by landing at every eighth station and collecting unknown specimens. Specimens were returned to the laboratory and identified. Unknown specimens were collected primarily in 1968, but the observer remained the same (R. Chabreck) for all surveys. The stations were also classified as natural marsh, forested (shrubs or trees), dewatered marsh, open water, beach, natural levee or ridge, spoil bank, or developed. Plant species nomenclature follows Godfrey and Wooten (1981), except for ferns which follow Correll and Correll (1975).

For the analyses in this study, only those stations classified as natural marsh were used (343 stations). Before the start of the analyses, all submerged, free-floating, and tree species were removed from the data set. Due to the relatively large size of the study plots,

Table 1. Percentage of the plots in each vegetation type that a species occurred in. Order of species is the order from the TWINSPAN table. Nomenclature follows Godfrey and Wooten (1981).

Species	Vegetation type[1]				
	FS	FM	FB	OW	MM
Aeschynomene indica L.	24			1	
Bidens laevis (L.) BSP.	45		3		
Triadenum virginicum (L.) Raf.	6	2			
Decodon verticillatus (L.) Ell.	3	2			
Hydrocotyle spp	85	39	16	9	
Sagittaria latifolia Willd.	12	2	3		
Dichromena colorata (L.) Hitchc.	7	6			
Ludwigia spp	24	17	9		
Phyla spp	2		6		
Sacciolepis striata (L.) Nash.	9	1	9	1	
Andropogon virginicus L.	3	1	3	1	
Colocasia esculenta (L.) Schott	5	1	6		
Polygonum punctatum Ell.	4	4	16	3	
Eleocharis spp	75	59	66	24	
Cephalanthus occidentalis L.	2	14			
Euphorbia spp		7		1	
Panicum hemitomon Schult.	12	94	6	1	
Zizaniopsis miliacea (Michx.) Doell. & Asch.	2	8			
Ferns	2	12	41		
Leersia oryzoides (L.) Sw.	3	3			
Myrica cerifera L.	6	12	3	1	
Setaria geniculata (Lam.) Beauv.		1	9	3	
Sagittaria lancifolia L.	12	27	94	14	
Typha spp	14	50	47	14	
Cladium jamaicense Crantz		2	3	1	
Eupatorium capillifolium (Lam.) Small	1	9	3	4	
Hibiscus lasciocarpus Cav.	1	4	6	3	
Solidago sempervirens L.		3		1	
Echinochloa spp	2	6	9	4	
Kosteletzkya virginica (L.) Presl.		7	3	7	
Phragmites australis (Cav.) Trin. ex Steud.	1	12	9	7	
Cyperus spp	14	13	13	18	5
Eleocharis parvula (R. & S.) Link	1	3		4	
Panicum spp		1	13	4	
Ipomoea sagittata Poir. in Lam.		5	19	11	
Amaranthus australis (Gray) Sauer			6	3	5
Spartina cynosuroides (L.) Roth.			6	3	5
Iva frutescens L.		1	3	7	
Paspalum vaginatum Sw.			3	4	
Bacopa monnieri (L.) Pennell	3	2		18	
Aster spp			3	5	
Sesbania spp	1		9	16	
Vigna luteola (Jacq.) Benth.	2	1	16	54	
Pluchea spp	1	6	3	24	5
Scirpus americanus Pers.	6		19	80	3
Lythrum lineare L.				7	8
Spartina patens (Ait.) Muhl.	1	3	19	88	85
Distichlis spicata (L.) Greene		1		4	74
Scirpus robustus Pursh.				1	21
Spartina alterniflora Loisel.				3	72
Juncus spp		2	22	12	44
Species not used in TWINSPAN					
Alternanthera philoxeroides (Mart.) Griseb.	1	1		1	

Table 1. Continued.

Species	FS	FM	FB	OW	MM
	Vegetation type[1]				
Ammania coccinea Rottb.				1	
Baccharis halimifolia L.			9	4	
Batis maritima L.					5
Boehmeria cylindrica (L.) Sw.		1			
Centella asiatica (L.) Urban		1			
Cuscuta indecora Choisy				1	
Habenaria repens Nutt.		1			
Pontederia cordata L.		2			
Saururus cernuus L.	1	1	3		
Scirpus spp			3		
Sonchus spp		1			
Xyris iridifolia Chapm.	1	1			
Average number of species	2.5	4.5	5.4	4.7	3.2

[1] FS = Fresh Spikerush, FM = Fresh Maidencane, FB = Fresh Bulltongue, OW = Oligohaline Wiregrass, MM = Mesohaline Mix.

parts of ponds and/or levees were included in areas that were classified as natural marsh. Some species that are difficult to identify correctly using helicopter surveys, such as some grasses and sedges, were grouped by genus. After this, species occurring in less than 5 stations (approximately 1% of the stations) were removed to reduce the influence of these rare species in the analysis and increase stability of the analysis (Tausch et al. 1995). The abundance data from all years (343 stations) were clustered using two-way indicator species analysis (TWINSPAN) with the standard setting of the program (Hill 1979) to determine vegetation types. The names of the resulting vegetation types follow Visser et al. (1998). Species diversity (average number of species per plot) calculations for each vegetation type were based on the same data reductions used for TWINSPAN, except that species occurring in less than 5 stations were retained. This diversity estimate is conservative because several species could only be identified to the genus level.

Change in vegetation types over time in the fresh and oligohaline marsh was determined using only those stations that had been surveyed in all four years. Change in the whole study area could only be shown for three years (1968, 1984, and 1992) due to the limited number of stations surveyed in 1978.

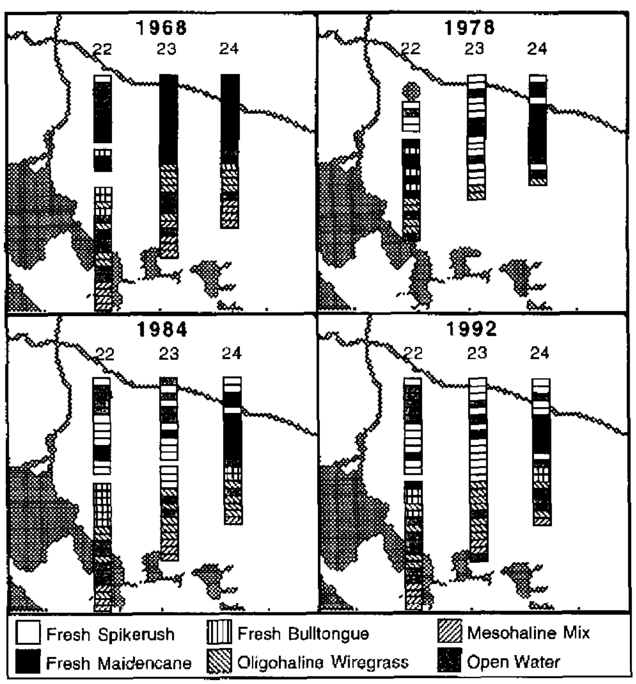

Figure 3. Approximate distribution of the vegetation types in the period 1968–92.

RESULTS

Vegetation Classification

TWINSPAN revealed five vegetation types in the study area over the period of the study (Figure 2). The mesohaline mix vegetation type is without a clear dominant and can be described as dominated by a mixture of *Spartina patens, Distichlis spicata,* and/or *Spartina alterniflora,* with *Juncus roemerianus* and *Scirpus robustus* frequently present. This vegetation type's species richness of 3.2 species per station (with a total of 12 species observed) is lower than all the other vegetation types found in this study. A complete list of species found in each vegetation type is provided in Table 1. The mesohaline mix was found on the southern end of all transects (Figure 3).

The oligohaline wiregrass vegetation type is dominated by *Spartina patens,* with *Scirpus americanus* as a frequent co-dominant. Other taxa that were frequent-

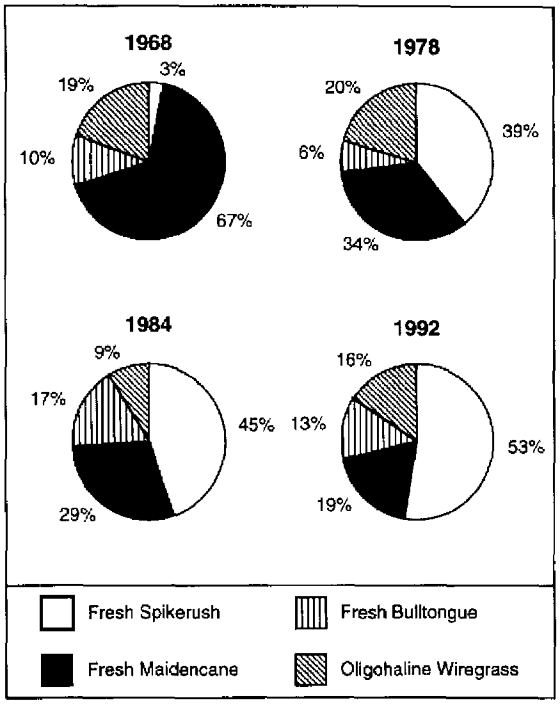

Figure 4. Change in fresh to oligohaline vegetation types in the period 1968–92.

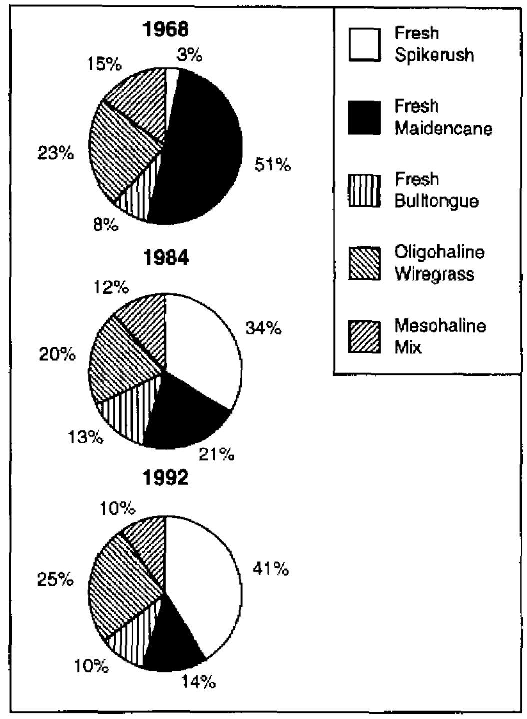

Figure 5. Change in vegetation types in the whole study area. Data from 1978 are omitted, because only fresh and oligohaline marsh were sampled in this year.

ly observed are *Vigna luteola, Eleocharis* spp, and *Pluchea* spp (*P. camphorata* (L.) DC. and *P. foetida* (L.) DC.). This vegetation type had a species richness of 4.7 species per station (total 43 species observed). This vegetation type is generally found just north of the mesohaline mix.

The fresh bulltongue vegetation type is dominated by *Sagittaria lancifolia,* with *Eleocharis* spp as frequent co-dominants. Other taxa that were frequently observed are *Typha* spp *(T. domingensis* Pers. and *T. latifolia* L.) and ferns (*Thelypteris palustris* Schott and *Osmunda regalis* L.). This vegetation type is the most diverse of the vegetation types, with a species richness of 5.4 species per station (total 39 species observed).

The fresh maidencane vegetation type is dominated by *Panicum hemitomon.* Other taxa that were frequently observed are *Eleocharis* spp, *Typha* spp, *Hydrocotyle* spp (*H. umbellata* L., *H. verticillata* Thumb., and *H. ranunculoides* L. f.), and *Sagittaria lancifolia.* This vegetation type has a species richness of 4.5 species per station (total 47 species observed).

The fresh spikerush vegetation type is co-dominated by *Eleocharis* spp (mostly *E. baldwinii* (Torr.) Chapm.) and *Hydrocotyle* spp. Other taxa that were frequently observed are *Bidens laevis, Ludwigia* spp (*L. leptocarpa* (Nutt.) Hara and *L. peploides* (HBK) Raven), and *Aeschynomene indica.* This vegetation type has a species richness of 3.9 species per station (total 37 species observed).

Vegetation Changes

Since 1968, the most striking change in vegetation in northwestern Terrebonne estuary is the rapid increase of fresh spikerush type. Most of this change (3% to 39%) occurred in the period between 1968 and 1978 (Figure 4). However, this vegetation type is still expanding (45% in 1984, 53% in 1992). This increase has mostly come at the expanse of the fresh maidencane type (Figure 3).

The oligohaline wiregrass type decreased from 20% in 1978 to 9% in 1984 (Figure 4). This change from oligohaline wiregrass to fresh bulltongue occurred mainly on transect 22 in the marshes north of Four League Bay (Figure 3). In 1992, oligohaline wiregrass increased again to 16%. During the period of study, the mesohaline mix marsh decreased slightly (Figure 5).

DISCUSSION

The relatively small change in proportions of the fresh bulltongue and oligohaline wiregrass vegetation types (Figure 4) might reflect temporal changes in salinity. The freshening of the system in 1984 might be related to the large 1983 flood of the Mississippi River system that brought large amounts of fresh water into the study area through the Atchafalaya River. Salinity does negatively affect the growth of *Sagittaria lancifolia* (McKee and Mendelssohn 1989, Grace and Ford 1996), and *Spartina patens* is generally found in mesohaline to oligohaline marshes (Mitsch and Gosselink 1993). David (1996) showed that increasing water levels lead to increasing abundance of *Sagittaria lancifolia* in the Florida Everglades without changes in salinity. However, since the expansion of *Sagittaria lancifolia* in our study area is replacing *Spartina patens*, we assume that salinity is the driving factor. The expected change toward more saline vegetation types due to salinity intrusion (see introduction) is not shown by our data. This might be due to the relative proximity of the study area to the Atchafalaya River and the fact that salinity has not shown the predicted increase (Wiseman et al. 1990, Swenson and Swarzinski 1995).

The expansion of the fresh spikerush vegetation type in northwestern Terrebonne Basin is the most dramatic change documented by this study. This conversion occurred in an area considered a "hot spot" of land loss (Leibowitz 1989). Leibowitz (1989) showed that 54% of the marsh surrounding the northern half of transect 23 converted to open water between 1956 and 1978. However, most of the marsh that converted to open water is not sampled by transect 23. Whether the fresh spikerush vegetation type is a transitional type in a degradation toward open water or a stage in succession from open water to floating maidencane marsh as proposed by Sasser et al. (1995a) cannot be answered with these data due to the large time periods between sampling.

Potential causes of this dramatic change in fresh marsh vegetation include grazing by nutria, increased water levels, and eutrophication. Nutria (*Myocastor coypus*) is a rodent introduced to Louisiana in 1937 (Evans 1970). Since its introduction, the nutria population has increased rapidly, becoming the dominant grazer in fresh and oligohaline marshes (Lowery 1974, Condrey et al. 1995). The change in species composition due to nutria grazing has been shown in Louisiana for the nearby Atchafalaya delta (Shaffer et al 1992, Evers et al. 1998), oligohaline wiregrass marshes (Taylor et al. 1994), and mesohaline wiregrass marshes (Nyman et al. 1993). Nutria grazing has also been implicated in the decline of reed swamps (*Phragmites australis*) in England (Boorman and Fuller 1981). However, the effect of nutria grazing on maidencane marshes has not yet been documented.

Kinler et al. (1980) attribute the die-back of maidencane marsh and the replacement with marshes where *Bidens laevis* is dominant (based on their species list, these are very likely to be the fresh spikerush vegetation type described in this paper) to the 1973 record flood and above average rainfall in following years. Water-level stages in NW Terrebonne system have generally increased in the last 20 years due to the decreasing efficiency of the Lower Atchafalaya River (Paul Kemp, personal communication). However, 92 percent of the maidencane marshes in Terrebonne estuary are floating (Evers et al. 1996). Although attached *Panicum hemitomon* is negatively affected by increased water levels (McKee and Mendelssohn 1989), floating *Panicum hemitomon* biomass is positively correlated with higher water levels (Sasser et al. 1995b). The positive effect of increased water level on floating *Panicum hemitomon* is presumably due to higher nutrient levels associated with increased runoff (Sasser et al. 1995b). In addition, based on 1990 imagery, fresh spikerush marsh has also started to appear in Barataria estuary (Sasser et al. 1994, Evers et al. 1996). Upper Barataria estuary is not directly affected by the Atchafalaya or Mississippi River and has not seen the same increase in water levels as NW Terrebonne (Swenson and Swarzenski 1995). Therefore, it seems unlikely that increased water levels are responsible for the change from maidencane to spikerush marsh.

Klötzli (1971) was the first to implicate eutrophication in the demise of reedswamps (*Phragmites australis* marshes) in Europe. Both nitrogen and phosphorus concentrations have significantly increased in the waters of the Mississippi River since the 1960s (Rabalais et al. 1996) and might be a factor in the change in vegetation type in our study area through its distributary, the Atchafalaya River. An extensive review of the literature on reed die-back showed that Klötzli's assumption that eutrophication is responsible for the decline of reedswamps in Europe contrasts with many field observations and experimental data (Ostendorp 1989). However, Ostendorp's (1989) review did not distinguish results from floating reed mats from those of attached reeds. An increase in the nitrogen-to-potassium ratio in the environment results in less sclerenchymatous tissue in the *Phragmites australis* rhizomes as well as a decrease in belowground biomass of floating reed (Boar et al. 1989). Therefore, floating reedswamps are more prone to breakup and are lost from eutrophic waters, while attached marshes are unaffected (Boar et al. 1981). However, the only water-quality station near our study area (Bayou Black at Gibson) showed no significant trends in water quality (turbidity, dissolved oxygen, total nitrogen, nitrate and nitrite, total phosphorus, and total carbon) between 1958 and 1991 (Rabalais et al. 1995). Therefore, it seems unlikely that eutrophication is the driving factor in the observed demise of maidencane marsh.

Our long-term vegetation data set did not show the predicted change to more salt-tolerant vegetation. This is probably due to the influence of the Atchafalaya River in the study area. We documented a large change in the dominant vegetation of the fresh marsh, where *Panicum hemitomon*-dominated marshes were replaced with *Eleocharis baldwinii*-dominated marshes. This change occurred adjacent to an area of significant conversion to open water. Using the limited available data from the literature, we evaluated to the extent possible three potential driving factors for this change grazing, water level increase, and water quality. However, the currently available data are insufficient to definitively determine the cause of change. Therefore, additional study of all potential factors in the demise of *Panicum hemitomon* is recommended.

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