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Coastal Wetland Subsidence Arising from Local Hydrologic Manipulations

R. EUGENE TURNER*

Coastal Ecology Institute, Louisiana State University, Baton Rouge, Louisiana 70803

ABSTRACT: Twenty-three estimates of soil subsidence rates arising under the influence of local hydrologic changes from flap-gates, weirs, dikes, and culverts in tidal wetlands were compared to 75 examples of subsidence in drained agricultural wetlands. The induced subsidence rates from these hydrologic modifications in tidal wetlands can continue for more than 100 years, and range between 1.67 to 0.10 cm yr^{-1} within 1 to 155 years after the hydrologic modifications commence. These subsidence rates are lower than in freshwater wetlands drained for agricultural purposes, decline with age, and are significant in comparison to the rates of global sea level rise or the average soil accretion rates. The elevation change resulting from local hydrologic manipulations is significant with respect to the narrow range of flood tolerances of salt marsh plants, especially in microtidal environments.

Introduction

Tidal wetlands are sometimes manipulated for a variety of purposes using variable or fixed water control structures such as weirs, flap-gates, dikes, or excavated channels. These manipulations are meant to support diverse or even competing goals, including agricultural expansion, waterfowl management, mosquito control, navigation, and fisheries (Stearns et al. 1940; Turner et al. 1989; Simenstad and Warren 2002). The long-term effects of these kinds of hydrologic management on tidal wetland soil structure and accumulation are not well documented, but some consequential changes are known. The restoration of former agricultural impoundments must compensate for land subsidence (Fig. 1; also sometimes referred to as autocompaction; Simenstad and Warren 2002), and undesirable vegetation quality and soil chemical changes may occur (Roman et al. 1984; Portnoy and Giblin 1997). Some of these changes may be quite subtle. Hoar (1975) documented significantly lower soil redox potential behind marshes with weirs, compared to nearby reference marshes without weirs, but only in the summertime. Such occasional soil chemistry changes could be accompanied by higher sulfide concentrations, a stressor that can cause plant mortality in salt marshes (Mendelssohn et al. 1981). Stearns et al. (1940) provided an example of how two small ditches (about 25 cm wide and deep) lowered the water levels and the soil surface of a Delaware marsh. Within two years the plant community began to shift or become open water, the muskrat harvest declined, and new muskrat mounds were located at higher elevations. Stearns et al. (1940) measured a slight rebound in the third year after the ditches were filled.

The effects of local hydrological modification on soil accumulation in tidal wetlands are potentially onerous when considered in the context of sea level rise or acceleration, anticipated wetland restoration efforts, and wetland dredge and fill permit evaluation. The position of healthy tidal wetland plants must be maintained in the vertical plane in the face of a relative rise in sea level, whether that rise comes from global (eustatic) changes in the sea surface, soil compaction and oxidation, or geological adjustments beneath the Holocene sediments. The impacts of hydrologic change on coastal wetland sediments may accumulate with time, and be a particularly formidable obstacle to their restoration if significant soil subsidence is induced. However, compared to agricultural systems that are continually drained, tidal wetlands may experience lower subsidence rates if the tidal re-wetting of the marsh surface is an important factor.

Although comparative and experimental studies of how an altered hydrology affects soil subsidence in coastal wetlands appear lacking, there are several relevant analyses of subsidence in mires and bogs. The subject of this analysis is how the soil subsidence rates of tidal wetlands subject to hydrologic management compare to the experience in peaty substrates of freshwater wetlands. A practical objective of this paper is to quantify the amount of subsidence resulting from water control structures, how variable this rate is among wetlands, and how these rates change over time. An additional theme to address is the implications of these processes for restoration and management purposes.

* Tele: 225/578-6454; fax: 225/578-6326; e-mail: euturne@lsu.edu

Fig. 1. Upper: The Holme Post at Holme, England in July 1997. The dates are the surface elevation at that time. The post was sunk into a clay layer in 1848, and the height of the surface layer determined since. The surface of the peat was originally above sea level (marked), but the peat is now 2 m below sea level (photograph by the author). Lower: Horse grazing in the La Grand-Mare Marais-Vernier, Normandy, France. The Marais-Vernier is part of the Brotonne Regional Natural Park. The top of the tussocks was the surface of the marsh before drainage. Photograph courtesy Dr. J. Pokorny, Trebon, Czech Republic.

Subsidence in Mires and Bogs

Peaty wetlands have been drained and reclaimed for mostly agricultural uses for more than 1,900 years, but also for urban expansion (Stephens et al. 1984; Allen and Fulford 1990; Heathwaite et al. 1990). A ubiquitous result of drainage is the decrease in soil elevation (Fig. 1), notably documented for inland wetlands at Holme, England, the Bavarian Donaumoos, Germany, and the Everglades Agricultural Experiment Station (Florida, United States). At these places (and others), a post is anchored to an impenetrable subsurface layer and used to record changes in the soil surface. The post at the Bavarian mire near Karlshuld, Germany, was established in 1839 about fifty years after cultivation started, and is apparently the oldest such subsidence meter. The original fen post at Holme, England, was specifically established in 1852 to monitor subsidence after the draining of Whittlesey mere. A second post was added in the 1950s when the first one became unstable. The base of the original Holme post is now 2 m below sea level (Fig. 1, upper). Subsidence at both the Bavarian and Holme mires amounted to more than 4 m within 130 and 150 years, respectively (Heathwaite et al. 1990). A similar post at the Everglades (Florida) shows more than 3 m of subsidence since 1924 (Stephens and Steward 1976). Although soil subsidence in drained agricultural fields occurs mostly in the soil horizon above the drainage structure, significant subsidence may occur below the drain. Heathwaite et al. (1990) provide an example for a 4 m deep peat with a drain at 125 cm below the original surface layer. Subsidence below the drain was 40% of the total subsidence and due to compaction from denser de-watered soils above the drain, and generally drier soil conditions that led to organic matter oxidation throughout the soil profile.

Eggelsmann (1976), Stephens et al. (1984), and Heathwaite et al. (1990) summarized some patterns about soil subsidence for a variety of drained inland wetlands. Subsidence in mires is largely due to the loss of water from soil pore spaces, which are up to 97% of the total soil volume. This water loss leads to settlement and decreased porosity (permeability) as compaction and bulk density increases. Drying increases the oxidation rate of carbon reserves (Stephens et al. 1984). Fire, wind erosion, and leachates may be additional important factors.

The combined effects of agricultural drainage may result in subsidence rates that are >10 cm in the first year of drainage, and that subsequently decrease to between 1 and 5 cm yr^{-1} for the next 50 years. Subsidence after 50 years is even lower, and may continue for centuries. Stephens et al. (1984) reported on a study of a Dutch polder that subsided 1–2 m over 8–10 centuries (1.7 mm yr^{-1}).

Segeberg (1960) and Eggelsmann (1990) synthesized observations from a variety of agricultural reclamation projects in northern Europe to produce graphs and formulae describing the anticipated subsidence resulting from drainage. They found that dense soils (solids > 12%) compacted at about 25% the rate of almost floating soils (<3% solids), and that subsidence rates were positively related to temperature, drain density, nutrient availability, and peat thickness, but that subsidence was inversely related to soil wetness. Harris et al. (1962) described similar relationships between water level, soil subsidence, and nitrogen for wet agricultural fields. The influence of nutrient enrichment on soil subsidence is perhaps the least understood influence on wetland stability.

I compiled results from the literature on the subsidence rates in coastal wetlands that the authors determined to be the result of flap-gates, dikes, culverts, ditches, and weirs. These rates are then compared to subsidence rates in wetlands (mostly freshwater wetlands) drained for agricultural purposes.

Methods

Data on subsidence rates in wetlands converted to agricultural fields are from literature sources, as well as some newly acquired data collected by the author for the Holme Post in 1999 and the Cloverly agricultural field (south Louisiana) in 2001. These data are all under some type of forced (pumped) drainage and are listed in Table 1. Data from tidal wetlands under the influence of weirs, levees, culverts, drainage ditches, or flap-gates are also derived from the literature and are listed in Table 2. Data choices were restricted to one of the following determinations of changes in soil surface elevation: measured against a post anchored to stable substratum (e.g., the Holme Post); relative to surveyed benchmarks (e.g., the mosquito ditch study of Stearns et al. 1940); or relative to a nearby reference marsh (e.g., Okey 1918a; Roman et al. 1984). The years of exposure to the new hydrologic conditions are described as years since hydrologic modification or hydrologic change in the following discussion. The Florida wetlands investigated by Okey (1918a,b) included coastal wetlands, but not all of them were clearly tidal wetlands. The same author investigated the subsidence in wetlands of coastal Louisiana, all of which appear to be wetlands subject to tides (Turner and Neill 1984).

A simple linear regression was made of the age under hydrologic modification (dependent variable) and average subsidence rate (independent value). Subsets of the data were examined to estimate the predicted subsidence rate after 10 years of hydrologic modification began. These data subsets were from Okey (1918a,b) for south Florida and coastal Louisiana, Skertchly (1877) for a variety of English fens in the 1800s, the Holme post, 30 agricultural developments in freshwater wetlands that excluded the previous three data sets, all freshwater wetlands, and tidal wetlands exclusive of ditched wetlands.

Data on 207 salt marsh accretion rate estimates are from Turner et al. (2001). These estimates are for sediment cores dated using the accumulation of ^{137}Cs (Milan et al. 1995), whose peak activity represents the 1963–1964 horizon, which is usually from sediments in the upper 50 cm of the core.

Results

The range of the subsidence rates observed in freshwater wetlands drained for agricultural purposes was between 0.17 to 55 cm yr^{-1} within 4 months to 950 years after drainage (Table 1 and Fig. 2). The highest rates were observed soon after drainage commenced, which is consistent with that reported in the literature described above. The range of subsidence rates observed in tidal wetlands under the influence of flap-gates, culverts, dikes, ditches, etc. was between 0.10 to 1.67 cm yr^{-1} within 1 to 155 years after drainage (Table 2 and Fig. 2). These latter subsidence rates also declined with years since hydrologic change, but the subsidence rates were, in general, lower than those observed in drained freshwater wetlands. The difference between the subsidence rates in freshwater and tidal wetlands increased with time (Fig. 2).

Table 3 is a comparison of the estimated cumulative subsidence after 10 years based on a linear regression model where $\log_{10}$ (subsidence) = intercept + $\log_{10}$ (years). The number of samples used in the model, the adjusted coefficient of determination, slope, and intercept are indicated. All equations are significant at $p < 0.01$. The highest subsidence (cm) after the first 10 years were for the English fens examined by Skertchly (1877) and for the Holme Post, and were nearly identical (160 and 156 cm, respectively; Table 3). The lowest subsidence values after the first 10 years were for the tidal wetlands (22 cm).

Figure 1 also includes an indication of the global sea level rise (0.23 cm yr^{-1}), which is seen to be several times lower than the subsidence rates of these wetlands after 100 years (note the log scale). Also indicated is the accumulation rate for 207 salt marshes based on the ^{137}Cs dating, which is for the post-1963/1964 depositional horizon (Turner et al. 2001). This average accumulation rate is many times lower than the loss of the marsh surface elevation in the first 100 years after the local hydrologic changes began to cause increased subsidence above background rates. It is obvious by these two comparisons that the subsidence rates induced by hydrologic modifications in tidal wetlands are significant if current estimates of either sea level rise

TABLE 1. Data sources used for estimates of subsidence in agricultural fields on former wetlands.

Location	Site	Years	cm yr^{-1}	Source	Elevation Relative To:
California		26	7.62	Stephens et al. 1984	not stated
California	Sacramento-San Joaquin Delta	78	1.50	Rojstaczer and Deverel 1995	benchmark elevation
Oregon		64	0.74	Taylor 1983	dike
Israel		7	9.86	Shoham and Levine 1968	benchmarks
Michigan		5	2.26	Stephens et al. 1984	not stated
New York		60	2.50	Stephens et al. 1984	not stated
Indiana		6	1.88	Stephens et al. 1984	not stated
Florida		54	2.72	Stephens et al. 1984	not stated
Florida		7	3.43	Stephens et al. 1984	not stated
Netherlands		100	0.70	Stephens et al. 1984	not stated
Netherlands		6	1.33	Stephens et al. 1984	not stated
Norway	Coos Bay	65	2.34	Stephens et al. 1984	not stated
USSR	Hula marsh	47	2.13	Stephens et al. 1984	not stated
Netherlands	Utrecht	950	0.17	Stephens et al. 1984	not stated
Netherlands	western Netherlands	900	0.17	Stephens et al. 1984	not stated
Germany	Holstein	0.33	30.30	Stephens et al. 1984	not stated
Germany	Holstein	0.66	21.21	Stephens et al. 1984	not stated
Germany	Kehdinger moor	12	24.17	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Ostenholzer moor a	74	3.38	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Ostenholzer moor b	74	3.11	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Esterweger Dose (VM) a	89	1.85	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Esterweger Dose (VM) c	74	2.16	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Esterweger Dose (VM) b	74	0.95	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Kongingsmoor a	63	1.75	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Kongingsmoor b	47	1.70	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Kongingsmoor c	63	1.43	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Kongingsmoor d	63	1.03	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Butzflether moor	11	3.64	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Ritscher Moor a	8	4.38	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Ritscher Moor b	8	3.13	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
Germany	Ritscher Moor c	8	2.50	Heathwaite et al. 1990 presentation of Illnick and Eggelsmann 1977	in reference to underlying soil elevation?
English fens	9 sites	9 to 60	1 to 21	Skertchly 1877	
Holme post	9 intervals	12 to 145	3 to 12	Richardson and Smith 1977; Turner personal observations	fixed post
Florida coastal wetlands	7 sites	1 to 3	11 to 55	Okey 1918a,b	reference marsh
Louisiana coastal wetlands	20 sites	3 to 69	1 to 18	Okey 1918a,b; Turner personal observations	reference marsh

or accretion are considered noteworthy influences on coastal wetland stability.

Discussion

The subsidence rates of coastal wetlands modified with flap-gates, weirs, drainage ditches, culverts, etc. decline with time in a pattern similar to that observed for freshwater peaty wetlands drained for agricultural purposes. When normalized for time under hydrologic modification, the subsidence rates caused by hydrologic modification are lower, but not much lower, than observed in the agricultural lands formed within wetlands. Two explanations for this lower subsidence rate in tidal wetlands are offered: coastal soils have a higher inorganic content, and therefore compress less; and tidally-flooded soils will be wetted more frequently than drained inland soils, and therefore

TABLE 2. Literature sources for data on tidal marsh subsidence following local hydrologic changes (e.g., weirs, flap-gates, dikes, ditches, and levees). The changes are based on before-and-after measurements or comparison with nearby reference sites.

Site	Years	cm yr^{-1}	Source	Restriction	Measured Relative To:
Drake's Island	148	0.39	Burdick et al. 1997	dike/tide gate	reference marsh
Mill Brook	24	1.17	Boumans et al. 2002	road/culvert/tide gate	reference marsh
Oak Knoll	65	0.25	Boumans et al. 2002	road/culvert	reference marsh
NS Meadow (DF)	155	0.10	Portnoy and Giblin 1997	dike/weir	reference marsh
Herring River (DD)	90	0.17	Portnoy and Giblin 1997	dike	reference marsh
Pamet River (DF3)	60	0.25	Portnoy and Giblin 1997	railroad embankment	reference marsh
NS Meadow (DF2)	124	0.48	Portnoy and Giblin 1997	dike	reference marsh
Ash Creek	21	0.95	Roman et al. 1984	dike/tide gate	reference marsh
Turney Creek	21	1.67	Roman et al. 1984	tide gate	reference marsh
Hammock River	87	0.46	Roman et al. 1984	tide gate/road	reference marsh
Leetes	90	1.11	Anisfeld et al. 1999	tide gate	not stated
Great Harbor	110	0.55	Rozsa 1997	tide gate	reference marsh
Muskrat marsh	1	1.30	Stearns et al. 1940	mosquito ditch	fixed elevation
Muskrat marsh	2	1.10	Stearns et al. 1940	mosquito ditch	fixed elevation
Marsh Island 1	38	0.77	Bryant and Chabreck 1998	impoundment	reference marsh
Marsh Island 3	43	0.72	Bryant and Chabreck 1998	impoundment	reference marsh
Marsh Island 14	43	0.46	Bryant and Chabreck 1988	impoundment	reference marsh
Marsh Island 15	43	0.46	Bryant and Chabreck 1998	impoundment	reference marsh
Great Harbor	40	1.50	Rozsa 1997	tide gate	not stated
San Pablo Bay	120	0.83	Krone and Hu 2001	dike	benchmark
Bolsa Bay	80	1.00	Eilers 1980	dam	benchmarks
Coos Bay	64	1.50	Taylor 1983	dike	reference marsh
Elk River	90	0.83	Thom et al. 2002	dike	not stated
Mean	67.78	0.78			
±1 SD	43.89	0.45			
Maximum	155.00	1.67			
Minimum	1.00	0.10			
Number	23	23			

have a water table closer to the surface, resulting in lower organic oxidation rates.

An analysis of 207 dated sediment cores from salt marshes from the western Atlantic and the Gulf of Mexico indicates that the vertical accretion rate in salt marsh surface layers (since 1963) is mostly controlled by organic material, and not by inorganic material (Turner et al. 2001). The total volume of inorganic and organic material combined is typically less than 10% in these salt marshes (Turner et al. 2001) and even lower in brackish marshes. The inorganic matter embedded in the soil profile of a tidal wetland will remain with subsidence, and the bulk density increase. The increased subsidence under drainage is due to the loss of water and organic material, not mineral matter. If soil wetness is inversely related to subsidence rates in tidal wetlands, as it is for freshwater wetland drained for agriculture, then tidal wetlands should have lower subsidence rates than freshwater wetlands. This is because tides re-wet the wetland surface on a daily basis, whereas the freshwater wetlands may have longer periods of drying whose frequency and duration are dependent on climatic cycles. There is some sediment deposition in tidal wetlands that may compensate, somewhat, for the subsidence from organic losses.

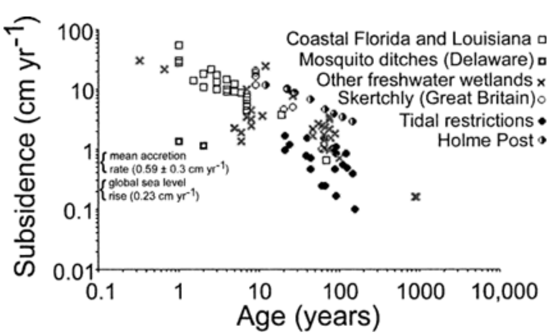

Fig. 2. A log transformed plot of the average subsidence rate (cm yr^{-1}) and years of hydrologic modification for different wetlands. Data sources are discussed in the text and the categories are the same as those in Table 3. The mean accretion rate (since 1963–1964; mean ± 1 SE) for the 207 salt marshes described in Turner et al. (2001) is indicated. The global rise in mean sea level is also shown (Gornitz et al. 1982). The data are in Tables 1 and 2.

One might consider the following hypothetical and illustrative example to scale the significance of these subsidence rates. Assume that there is 100 cm of subsidence in a coastal wetland that has a fully-functional flapgate for 100 years. This amount of relative change in the wetland surface is 4 times more than anticipated from present-day sea level rise (0.23 cm yr^{-1}), and twice the rate anticipated after the effects of global climate change accumulate by the end of this century. If restoration re-

TABLE 3. A comparison of the estimated cumulative subsidence after 10 years based on a linear regression model where $\log_{10}$ (subsidence) = intercept + $\log_{10}$ (years). The number of samples used in the model, 1 standard error of the slope (SE), the adjusted coefficient of determination, slope, and intercept are indicated (n, Adj. R^2, slope, and intercept, respectively). The data are in Tables 1 and 2. All equations are significant at $p < 0.01$.

Wetland	n	Adj. R^2	$\log_{10}$ Subsidence ($cm\ yr^{-1}$)	1 SE $\log_{10}$ Slope	Intercept	Estimated Subsidence (cm) After 10 Years
Freshwater wetlands						
A. Coastal Louisiana and Florida drained freshwater wetlands	27	0.88	−0.83	0.06	1.51	49
B. Skertchly's (1877) data on English fens	9	0.78	−1.25	0.23	2.46	160
C. Holme Post	9	0.98	−0.62	0.035	1.8	156
D. Other freshwater wetlands (exclusive of A, B, and C)	30	0.62	−0.53	0.76	1.15	42
E. All drained freshwater wetlands	75	0.64	−0.58	0.05	1.38	63
Tidal wetlands						
F. Tidal wetlands exclusive of ditched wetlands (Stearns et al. 1940)	23	0.27	−0.73	0.24	1.07	22

sulted in an average surface accumulation rate of 0.59 cm yr^{-1}, and disregarding soil auto-compaction, then it would take several centuries to reach equilibrium with the surface layer of the surrounding wetland maintaining equilibrium with present sea level rise (100 cm/(0.59–0.23 cm yr^{-1}) = 278 years). This is a meaningfully-long period of adjustment and is a far longer planning horizon than most management agencies accommodate. The slow and continuing subsidence in wetland modified by water control structures in coastal wetlands has the potential of bringing significant changes to these wetlands, if not their conversion to open water. These changes are almost imperceptible to visual documentation without a fixed reference point and many years of observation.

Subsidence of even a few centimeters can have a profound effect on tidal systems if the plants are found at the margins of their habitat space. Redfield (1972) described the range of tolerance to be about 70% of the tidal range for 15 Atlantic coast salt marshes whose tides range between 0.4 to 2.9 m. Salt marshes in the microtidal environments of south Louisiana occupy only a few decimeters of the tidal range (McKee and Patrick 1988). Subsidence rates sufficient to expose most Louisiana salt marshes to conditions normally outside of their natural flooding tolerance could happen within 10 years after hydrologic modification (Table 3), implying that these marshes could easily be driven to become open water habitat within a few years of higher subsidence rates of 1 cm yr^{-1}.

That soil subsidence increases when local hydrologic modifications occur helps explain results from an analysis of coastal wetland loss in south Louisiana for areas with and without weirs. Turner et al. (1989) found that there were more ponds created from 1955 to 1978 from wetland in areas with weirs, than in nearby reference areas for the Biloxi Marshes in St. Breton Sound estuary, south Louisiana. They also summarized literature results documenting how three different wetlands with a system of weirs had more ponds and less emergent wetland than in nearby reference wetlands. The general nature of these conclusions is also applicable to an analysis of the hydrologic impacts of dredged canals, which are common in south Louisiana. These canals are postulated to cause wetland to water conversion because of their indirect impacts on local hydrology (Turner 1997). Canals are typically >5 m deep and 30 m wide, and have a continuous levee parallel on both sides of the canal that is built from the dredged spoil deposits. Swenson and Turner (1987) showed how the length of both flooding and drying is higher behind these spoil banks, whereas tidal variability is reduced. One of the implications of this study is that canals act in the same manner as mosquito ditches (but with a cross-sectional area > 2,000 times larger) to dry out wetlands more frequently than in wetlands without them. The spoil bank levees reduce water moisture at some times, but increase it at others, because of the restricted flows between water and wetland aboveground and belowground.

It seems reasonable to consider that nutrient additions to tidal wetlands might enhance subsidence rates as happens in agricultural wetlands under some kind of enhanced drainage (Segeberg 1960; Harris et al. 1962; Eggelsmann 1990). Two studies in salt marshes showed that this may be the case. Both Valiela et al. (1976) and Morris and Bradley (1999) documented a decline in belowground organic accumulation when nitrogen was applied over the salt marsh. Valiela et al. (1976) extrapolated the reduction in belowground tissues to equate to a 1 cm annual decline in surface elevation. The eutrophication of coastal waters is widespread (Cloern 2001; Rabalais 2002), and may be another human-induced stressor compromising the ability of salt marsh soils to accumulate organics. There may be compensatory accretion occurring, however, as a result of nutrient-enhanced emergent plant growth that traps additional amounts of inorganic material (e.g., Morris et al. 2002). The time it takes to reach an equilibrium in response to the aboveground and belowground effects of nutrient additions may be decades long if the results from salt marsh rehabilitation or creation can be applied. Craft et al. (1999) found that, although the emergent plant cover in a restored salt marsh reached that of reference sites within a decade, the soil organic content would not reach an equilibrium position equivalent to reference marshes for at least 25 years.

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