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THE IMPACT OF WASTEWATER EFFLUENT ON ACCRETION AND DECOMPOSITION IN A SUBSIDING FORESTED WETLAND

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Abstract: Insufficient sedimentation, coupled with high rates of relative sea-level rise (subsidence plus eustatic sea-level rise), are two important factors contributing to wetland loss in coastal Louisiana, USA. We hypothesized that adding nutrient-rich, secondarily treated wastewater effluent to subsiding wetlands in Louisiana could promote vertical accretion in these systems through increased organic matter production and subsequent deposition and allow accretion to keep pace with estimated rates of relative sea-level rise (RSLR). However, we also hypothesized that nutrient enrichment could stimulate the decomposition of organic matter, thus negating any increase in accretion due to increased organic matter accumulation. To test these hypotheses, we measured leaf-litter decomposition, litter nutrient dynamics, and sediment accretion in a permanently flooded and subsiding forested wetland receiving wastewater effluent and in an adjacent control site, both before and after effluent applications began. We also measured organic and mineral matter accumulation in the treatment site before and after effluent applications began. A Before-After-Control-Impact (BACI) statistical analysis revealed that neither leaf-litter decomposition rates nor initial leaf-litter N and P concentration were affected by wastewater effluent. A similar analysis revealed that final N and P leaf-litter concentrations did significantly increase in the treatment site relative to the control after effluent was applied. Total pre-effluent accretion, measured 34 months after feldspar horizon markers were laid down, averaged ($\pm$ SE) 22.3 ± 3.2 mm and 14.9 ± 4.6 mm in the treatment and control sites, respectively, and were not significantly different. However, total accretion measured 68 months after the markers were installed and 29 months after effluent additions began in the treatment site averaged 54.6 ± 1.5 mm in the treatment site and 19.0 ± 3.2 mm in the control site and were significantly different. Additionally, after wastewater applications began, the estimated rate of accretion in the treatment site (11.4 mm yr^{-1}) approached the estimated rate of RSLR (12.3 mm yr^{-1}). Most of this increase in accretion was attributed to organic matter inputs, as organic matter accumulation increased significantly in the treatment site after effluent application began, while mineral accumulation rates remained constant. These findings indicate that there is a potential for using wastewater to balance accretion deficits in subsiding wetland systems.

Key Words: BACI, delta, nitrogen, phosphorus, restoration, sea-level rise, sediment

INTRODUCTION

Relative sea-level rise (RSLR) in the Mississippi River deltaic plain is a natural consequence of subsidence and eustatic sea-level rise (ESLR) (Cahoon et al. 1995). Subsidence, due to the compaction, consolidation, and downwarping associated with the rapid deposition of alluvial sediments accounts for approximately 90% of the estimated 1.1 to 1.3 cm yr^{-1} RSLR measured in the Louisiana delta region of the United States (Boesch et al. 1994). The ESLR component, however, is predicted to increase steadily over the next century due to the impacts of global warming (Gornitz 1995).

Wetlands can persist in the face of RSLR when vertical accretion equals or exceeds the rate of subsidence (DeLaune et al. 1983, Baumann et al. 1984, Stevenson et al. 1986). Historically, seasonal overbank flooding of the Mississippi River deposited sediments and nutrients into the interdistributary wetlands of the delta plain. These seasonal floods provided an allochthonous source of mineral sediments that contributed directly to vertical accretion. The nutrients associated with these sediments also promoted vertical accretion through organic matter production and deposition (Patrick and Khalid 1974). This sediment and nutrient source has decreased dramatically since the 1930s with the completion of levees along the entire course of the lower Mississippi (Kesel 1988) resulting in vertical accretion deficits (accretion–RSLR) and widespread wetland loss throughout the modern delta region (Day and Templet 1989, Boesch et al. 1994). Additionally, many wetlands in the deltaic region have become hydrologically isolated because of the dense network of canals and associated spoil banks constructed during the past century (Boesch et al. 1994). Spoil banks impede drainage and often physically impound wetlands, thus preventing the overland flow of any remaining sediments and nutrients into coastal wetlands. In forested wetlands, high rates of RSLR, coupled with sediment deficits and restricted water and sediment exchange, lead to nutrient limitations and increasingly long periods of inundation that are associated with decreased productivity, reduced regeneration, and tree mortality (Conner and Day 1988, 1989, Dicke and Toliver 1990, Rybczyk 1997).

Recent wetland restoration efforts in the subsiding delta region have attempted to balance vertical accretion deficits by either physically adding mineral sediments or sediment-rich water to wetlands or by constructing sediment trapping mechanisms or landforms (Boesch et al. 1994). In line with restoration efforts specifically designed to balance accretion deficits, but focusing on organic matter accumulation enhancement, Breaux and Day (1994) hypothesized that the addition of nutrient-rich, secondarily treated wastewater to hydrologically isolated, nutrient-limited, and subsiding wetlands could promote vertical accretion through increased organic matter production and deposition.

Whether or not nutrient enrichment ultimately increases wetland elevation is dependent upon the interactions between organic matter production, accretion, and subsequent decomposition and compaction. Although nutrient amendments could increase relative wetland elevation directly by stimulating productivity and subsequent organic matter accretion, amendments could also tend to decrease relative elevation by increasing the rate of decomposition of organic matter. This increase in rates of decomposition could occur by either improving initial litter nutrient quality (Coulson and Butterfield 1978, Valiela et al. 1985, Lukumbuzya et al. 1994, Rybczyk et al. 1996) or by externally increasing the nutrients available to decomposer communities (Howarth and Fisher 1976, Haines and Hanson 1979, Farchild et al. 1984).

In March 1992, the city of Thibodaux, LA, as part of its tertiary treatment program, began applying secondarily treated municipal wastewater to the Pointe au Chene Swamp, a hydrologically isolated, permanently flooded, and subsiding forested wetland in the deltaic region. This provided an opportunity to test two contradicting hypotheses; 1) that, through increased organic matter production and subsequent deposition, nutrient-rich effluent would stimulate accretion to the extent necessary to offset the rate of RSLR, and 2) that the effluent discharge would increase the rates of organic matter decomposition (thus negating any accretion due to increased production). Specifically, the objectives of the decomposition portion of this study were to determine if leaf-litter decomposition rates and initial and final litter N and P concentrations changed in response to wastewater effluent with respect to an adjacent control site. With regard to sediment accretion, the specific objectives of this study were to 1) measure mineral and organic matter accumulation rates in the treatment site before and after wastewater application began, 2) estimate long-term (30 year), pre-effluent accretion rates in the treatment site, and, 3) measure and compare short-term accretion rates before and after wastewater application in a control site, the effluent treatment site, and on a hardwood ridge separating the two.

SITE DESCRIPTION

The Pointe au Chene swamp lies on the backslope of Bayou Lafourche, an abandoned Mississippi River distributary, approximately 10 km southwest of Thibodaux, Louisiana, USA (Figure 1). The study site consists of two continuously flooded forested wetlands, separated by an exposed bottomland hardwood ridge, within a 1425-ha hydrologically restricted basin. An analysis of tidal gauges located within the vicinity of the Pointe au Chene swamp revealed that the mean annual rate of RSLR was 1.23 ± 0.34 cm between 1962 and 1982 (Penland et al. 1988, Rybczyk 1997).

The ridge site (mean elevation = 1.16 m above mean sea-level (MSL)) is approximately 300 m wide and is vegetated primarily with oaks (*Quercus nigra* L. and *Quercus laurifolia* Michx.), sweetgum (*Liquidambar styraciflua* L.), American elm (*Ulmus americana* L.), palmetto (*Sabal minor* Jacq.), and boxelder (*Acer negundo* L.). The two forested wetlands on ei-

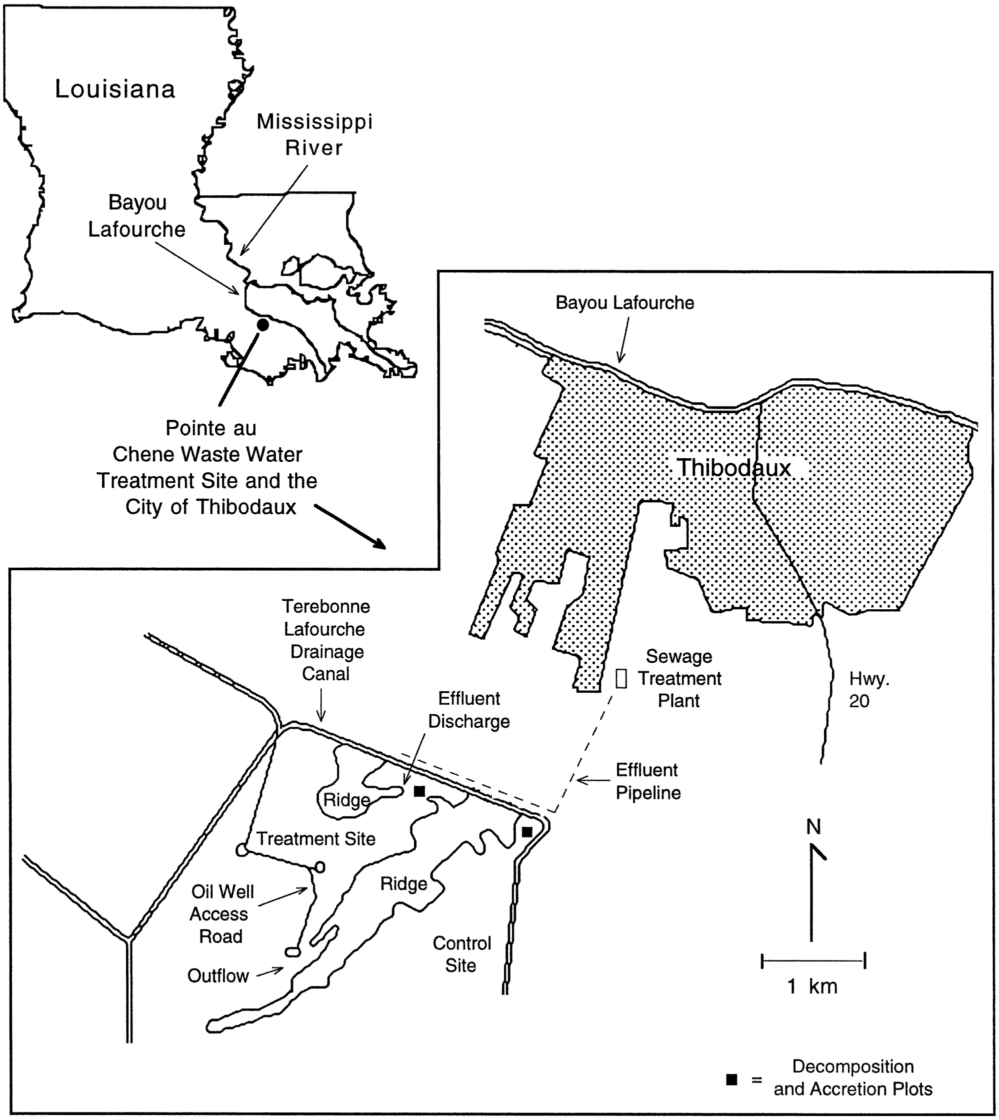

Figure 1. Map of the Pointe au Chene Swamp, located adjacent to the city of Thibodaux, Louisiana. An oil access road, a bottomland hardwood ridge, and the spoil banks associated with the Terrebonne-Lafourche drainage canal hydrologically isolate the treatment site from the surrounding wetland.

ther side of the ridge (mean elevation = 0.76 meters above MSL) are dominated by ash (*Fraxinus pennsylvanica* Marsh.), black willow (*Salix nigra* Marsh.), baldcypress (*Taxodium distichum* (L.) Rich.), water tupelo (*Nyssa aquatica* L.), red maple (*Acer rubrum* L.), and palmetto. In the two forested wetland sites, floating aquatic vegetation, consisting primarily of duckweed (*Lemna* sp.) and *Salvinia* sp., cover the water surface for most of the year (Zhang 1995). Soils are classified as Fausse clay (very-fine, montmorillonitic, nonacid, thermic Typic Fluvaquents) and Sharkey clay (very-fine, montmorillonitic, nonacid, thermic Vertic Haplaquepts). The region has a mild climate determined largely by the subtropical location (latitude 29°) and proximity to the Gulf of Mexico. The mean annual air temperature is 20.6°C, ranging from 13.0°C in January to 27.5°C in July. Mean annual precipitation is approximately 167 cm yr^{-1}.

Since March 1992, the 231-ha forested wetland on the west side of the ridge has received secondarily treated municipal wastewater at an average rate of 7.5×10^6 L day^{-1}. Wastewater is discharged from 40 pipes located on the spoil bank that serves as the northern boundary of the site (Figure 1). The effluent then flows southward between the ridge and an oil access road and exits at a point where these two features nearly meet. The combination of ridge, spoilbank, and access road hydrologically isolates the treatment swamp from the rest of the 1425-ha basin. In this paper, the forested wetland receiving wastewater effluent is referred to as

Table 1. Experimental design and results for the Pointe au Chene Swamp decomposition study. See Figure 3 for the statistical analysis of this data.

Experimental Set	Beginning and Ending Date of Experimental Set	Decay Rate k yr^{-1} and (r^2) Control Site	Treatment Site	Collection Schedule (weeks)
1) Pre-effluent	11/17/88–11/10/89	No Effluent 0.82 (0.43)	No Effluent 0.86 (0.63)	0, 1, 3, 8, 14, 18, 23, 27, 32, 36, 42, 48, 52
2) Pre-effluent	12/1/89–12/8/90	No Effluent 1.14 (0.31)	No Effluent 1.16 (0.31)	0, 7, 11, 15, 20, 24, 28, 32, 36, 40, 45, 50, 53
3) Post-effluent	2/26/92–12/10/93	No Effluent 1.71 (0.35)	Effluent 1.70 (0.31)	0, 4, 11, 15, 19, 24, 32, 36, 41
4) Post-effluent	1/22/93–1/18/94	No Effluent 0.49 (0.75)	Effluent 0.55 (0.65)	0, 3, 7, 11, 20, 29, 37, 51

the ''treatment site'' and the swamp on the eastern side of the bottomland hardwood ridge as the ''control site'' (Figure 1).

During the four year period 1988–1992 (before effluent applications began), water levels averaged +18.9 ± 0.5 cm and +19.2 ± 0.4 cm (mean ± se) above the sediment surface in the control and treatment sites, respectively. From 1992 to 1996 (after wastewater applications began in the treatment site), water levels averaged +29.3 ± 0.6 and +31.5 ± 0.3 cm in the control and treatment sites, respectively. Water levels in both sites showed similar patterns and changes in response to monthly precipitation and evapotranspiration (Day et al. 1998).

Before wastewater effluent applications began, surface water-nitrogen (N) and phosphorus (P) concentrations were similar in both the control and treatment sites (Conner et al. 1989). The annual mean concentrations of NH_4-N, NO_3-N + NO_2-N, and TKN in the Pointe au Chene swamp during the 1988–1989 pre-effluent period were 0.05 mg L^{-1}, 0.012 mg L^{-1}, and 1.34 mg L^{-1}, respectively. During the same period, mean annual concentrations of PO_4-P and Total P were 0.24 mg L^{-1}, and 0.43 mg L^{-1}, respectively. Analyses of post-effluent surface water have shown that N and P concentrations remained the same in the control site but increased in the treatment site (Zhang et al. 2000). For example, 25 m from the effluent outfall zone in the treatment site, post-effluent (1992–1994) mean annual concentrations of NO_3-N + NO_2-N increased to 5.0 mg L^{-1} and PO_4 increased to 1.6 mg L^{-1}, although these concentrations returned to background levels at the swamp outflow (Day et al. 1994).

METHODS

Leaf-litter Decomposition

Using standard litterbag methods (Conner and Day 1991), the decay of leaf-litter was followed over the course of one year for four separate years (1989, 1990, 1992, 1993) in both the control and the treatment sites (Table 1). For the remainder of this paper, each set of two (control and treatment site) litterbag studies conducted within each year will be referred to as an ''experimental set.'' Two of these year-long experimental sets were conducted before and two were conducted after wastewater effluent applications began in the treatment site in 1992 (Table 1).

Leaves were collected during the fall months immediately preceding the start of each experimental set from leaf-litter traps placed in the control and treatment sites. Thirty-six 20 cm × 20 cm nylon bags (1 mm mesh), each filled with ten grams (dry weight) of leaves, were placed randomly in each of the two sites at the beginning of each experimental set. Litterbags intended for each site were filled only with litter collected from that specific site during the previous fall. Three nylon bags were collected from each site according to the schedule shown in Table 1 for approximately one year for each experimental set.

Decayed litter samples collected in the field were rinsed in the laboratory to remove foreign matter and organisms, dried at 60° C for 48 hours, and weighed to determine loss of mass. Concentrations of C and N in the initial and final litter samples were determined by direct combustion with a C-H-N analyzer. Initial and final P concentrations were determined by inductively coupled plasma spectrometry (ICP) after a nitric acid digest. To calculate annual rates of decomposition, linear regressions were performed on the percent mass remaining versus time to calculate the decomposition coefficient (k) using a natural logarithm transformation of the exponential decay model (Olson 1963); $\ln (X/X_o) = -kt$, where X = final mass, X_o = initial mass, and t = time. The resultant k values were used in statistical analysis, described below, to detect an effluent effect.

Studies such as this one, which attempt to detect the

Table 2. BACI ANOVA design, from Underwood (1991), used to detect if wastewater effluent had an effect on decomposition rates (k) and initial and final litter nutrient concentrations. This is a replicated, before/after sampling at two locations, one control and one potential impacted sites. Samples (i.e., k rates) were measured at *t* times before and *t* times after the impact.

Source of Variation		Degrees of Freedom	Comments
Before versus After	= B	1	
Location: Control versus Treatment	= L	1	
Interaction	B × L	1	Test for impact
Year (Before versus After)	Y(B)	$2(t^* - 1) = 2$	Random effect
Interaction	L × Y(B)	$2(t - 1) = 2$	Residual error term

* In this study, $t = 2$.

effect of some ecosystem perturbation or impact (wastewater effluent in this case) on one or more response variables (k rates, or litter nutrient concentrations, for this experiment), are often difficult to analyze statistically because of problems associated with inadequate or non-existent replication. Stewart Oaten et al. (1986) and later Underwood (1991) suggested that, if the concern is with a particular impact in a particular place, large-scale environmental impact effects could be detected statistically if simultaneous observations were made at multiple times both *B*efore and *A*fter the *I*mpact, in both a *C*ontrol and *I*mpact, or treatment, site (a BACI design). To test for an impact, the differences in some parameter of interest between the control and impact site before the impact are compared to the differences between the two sites after the impact. The null hypothesis, that no impact has occurred, is rejected if the differences between the control and impact site, before the impact, are not equal to the differences between the two sites after the impact.

Since annual leaf-litter-decay data were collected in both the treatment and control site at multiple times before and after effluent was applied to the treatment site, we used a BACI analysis within an ANOVA framework (Underwood 1991) to detect changes in decomposition rates, initial litter N and P concentrations, and final litter N and P concentrations in response to wastewater effluent (Table 2). Specifically, the BACI analyses tested the null hypotheses that the differences in each of these three leaf-litter parameters between the control and treatment sites before wastewater effluent was applied to the treatment site were equal to the differences in these same parameters between the control and treatment sites after wastewater applications began in the treatment site. Each experimental set is a statistical replicate, or experimental unit, and each experimental unit is replicated twice in the before, or pre-effluent, period and twice in the after, or post-effluent, period.

Accretion

Two different horizon marker techniques were used to measure accretion. To compare accretion between the control, treatment, and ridge sites, before and after the application of wastewater to the treatment site, we used feldspar marker horizons. To estimate long-term background (30 years) rates of accretion in the treatment site, we used ^{137}Cs, a fallout by-product from above-ground nuclear weapons testing, as a marker.

Feldspar Markers. Ten 0.25 m^2 feldspar marker horizon plots were randomly placed in each of the three sites (control, ridge, and treatment) on 1 December 1988. In the treatment and control sites, the thickness of the sediment layer above the feldspar was measured in October 1991, 34 months after the marker horizons were installed and five months before wastewater application began in the treatment site, (pre-effluent period) and measured again in August 1994, 68 months after the marker horizons were installed and 29 months after effluent application began in the treatment site. During each of the two sampling periods, five cores were taken from each plot by twisting thin-walled aluminum coring tubes 10 to 20 cm deep into the swamp sediment. Coring tubes were sealed in the field and stored in a vertical position until they were returned to the laboratory. Cores were then sliced along the vertical axis, and the depth of the accumulated sediment on top of the feldspar marker was randomly sampled three times in each core and averaged to obtain a final mean for each core.

Measurements of accretion over the feldspar marker on the unflooded ridge site were taken in December 1991 and again in August 1994. For both sampling periods, measurements were made at five random locations within each plot by inserting a wide-bladed knife 15 to 20 cm into the soil and then prying the soil back on one side to expose the upper soil horizon and the feldspar marker on the other side. The distance from the feldspar layer to the soil surface could then be measured easily with a small ruler.

Since there was only one sampling time period before and one after effluent applications began, we could not use a BACI design to detect an effluent impact. Instead, a completely randomized design analysis of variance was used to test for differences in total accretion between all sites (control, treatment, ridge) in October 1991 and again in August 1994. Each plot was considered an experimental unit. If significant differences were identified, individual sites were compared by linear contrasts using a Bonferoni correction for multiple comparisons (Neter et al. 1990).

^{137}Cs Analyses. Four cores were collected randomly from the treatment site in October 1993 for ^{137}Cs analyses. Two of the cores (# 3 and 4) were collected from feldspar horizon plots and two were collected from an adjacent unmarked area. Cores were collected by pushing thin-walled aluminum coring tubes (15 cm diameter by 50 cm long) 35 to 40 cm into the swamp sediments. The four tubes were capped in the field to hold the cores in place and stored in a vertical position until they were returned to the laboratory. Cores were sectioned into 2-cm increments, dried at 60° C for at least 96 hours, and ground to a fine powder. ^{137}Cs activity in the core profile was determined by counting the gamma emissions from each ground section using a lithium-drifted germanium detector and a multi-channel analyzer (DeLaune et al. 1978). Because the sediment bulk density ranged from 0.084 g cm^{-3} to over 1.0 g cm^{-3} from the top to bottom of the cores, ^{137}Cs activity was calculated on a volume basis and also normalized by weight. The ^{137}Cs activity peak identified in each core profile corresponds with the soil surface during the peak of aboveground nuclear weapons testing and associated ^{137}Cs fallout in 1963.

Sediment Accumulation

Since the measurement of ^{137}Cs is non-destructive, each 2-cm increment was also subjected to bulk density, percent organic matter by weight, percent mineral matter by weight, and volume distribution (mineral, organic, and pore space) analyses. Bulk density (g cm^{-3}) was calculated as the ratio of the oven-dried weight of each 2-cm core section to the known wet volume of that section. Percent organic matter (by weight) in each 2-cm oven-dried section was determined by loss on ignition as described by Allen (1974). Percent mineral matter (by weight) was then calculated as the remainder. Using an approximation of the particle density of mineral matter (2.62 g cm^{-3}) and organic matter (1.14 g cm^{-3}), volume distributions of organic and mineral matter were calculated as

$$\begin{aligned}\%\ \text{volume} = {} & (\text{bulk density} \\ & \times \%\ \text{weight min. or org}) \\ & \div \text{particle density}\end{aligned}$$

The remainder of the volume (space not occupied by mineral or organic matter) was assumed to be pore space occupied by either water or gas. We used a linear regression analysis procedure (test for a slope significantly different than zero) to detect any significant change in bulk density, percent mineral matter by weight, percent mineral matter by volume, percent organic matter by weight, percent organic matter by volume, and percent pore space with depth.

Using data obtained from the two cores that contained both a feldspar and a ^{137}Cs marker, mean annual mineral and organic matter accumulation rates for the 25-year period bounded by the ^{137}Cs marker and the feldspar marker (1963 to 1988) and the 4.83-year period bounded by the feldspar marker and the core surface (Dec. 1988 to Oct. 1993) were calculated as

$$A_y = \frac{\sum_{i=1}^{n} [(D_i)(M_i)]}{T_y} (CF)$$

where

D_i = dry weight (g) for core section i
M_i = percent mineral or organic matter by weight for core section i
T_y = number of years between marker horizon layers of interest
n = number of core sections bounded by the upper and lower horizon markers of interest
CF = conversion factor to convert g accumulation/core area to g accumulation/m^2
A_y = mineral or organic matter accumulation rate (g m^{-2} yr^{-1}) for period y (1963 to 1988 or 1988 to 1993)

A completely randomized design analysis of variance was used to test for differences in both organic and mineral matter accumulation rates between the two periods.

RESULTS

Leaf-litter Decomposition

Loss of Mass. Qualitatively, patterns of weight loss were similar between the treatment and control site within years (Figure 2). Among all years, in both the control and treatment site, an initial period of relatively rapid weight loss was observed, lasting from three to twenty weeks, during which 19% to 85% of the litter mass was lost. This was followed by a period of slower

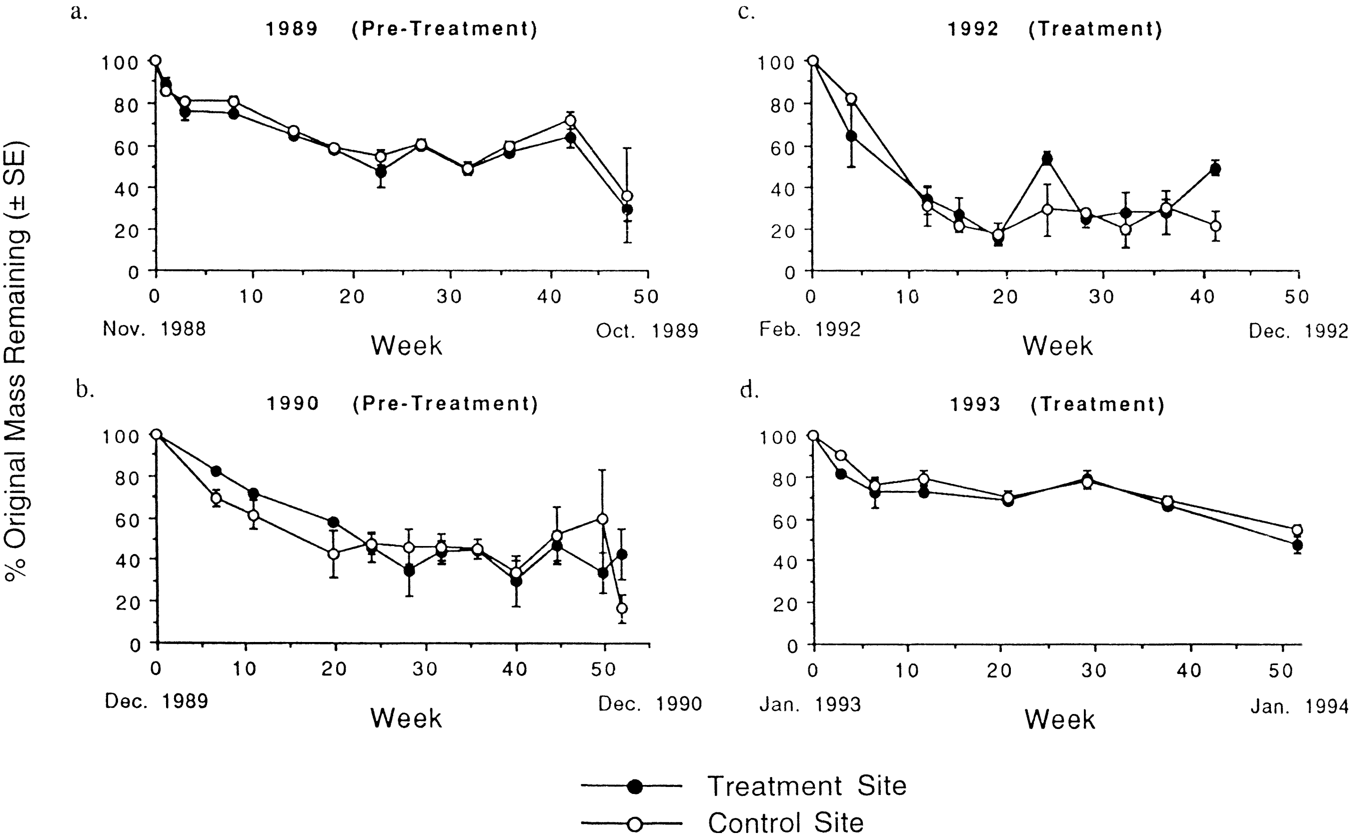

Figure 2. Percent of original dry mass remaining through time in the two Pointe au Chene sites during two pre-effluent years [1989 (a) and 1990 (b)] and two post-effluent years [1992 (c) and 1993 (d)].

decomposition that lasted for the remainder of each year-long experiment.

BACI analyses revealed that the differences in annual decomposition rates between the control and treatment site during the two pre-effluent years were not significantly different from the differences between the two sites after effluent application began in the treatment site ($F = 0.02$, $P > F = 0.90$). This indicates that effluent impacts did not affect annual rates of decomposition. The power of this test was extremely low (0.056) because there were only two true before-impact replications and only two after-impact replications. However, the least significant difference in k rates that would have been detected by this test was only 0.18 yr^{-1}, which is a reasonably small difference to detect considering that decomposition rates from year to year ranged from 0.49 yr^{-1} to 1.71 yr^{-1} in the control and treatment site (Table 1).

All the data sets used for the BACI analyses met the assumptions of additivity (Tukey 1949) and normality, but were too small to test for independence or serial correlation (Durbin and Watson 1951). However, we assumed that these data were not serially correlated because of the length of time between measurements (one year) and because new and different sets of litter were used for each experiment.

Nutrient Dynamics of the Decomposing Litter. Initial litter N concentrations (gN/g dry weight × 100 ± SE) ranged from 1.1 ± 0.0% to 1.3 ± 0.1%, and final N concentrations ranged from 1.6 ± 0.2% to 2.2 ± 0.1% among all sites and times (Table 3). Similarly, initial litter P concentrations ranged from 724.0 ± 64.2 mg/kg to 999.8 ± 21.7 mg/kg, and final concentrations ranged from 922.5 ± 46.1 mg/kg to 1823.3 ± 190.6 mg/kg among all sites and times (Table 3). Since carbon concentrations were relatively constant through time (ranging from 42.1% to 50.5% over all sites and times), C:N and C:P ratios decreased with time among all years and sites and were primarily a function of the N and P concentrations, respectively, in the litter. Initial C:N ratios ranged from 35.2 ± 0.1 to 43.7 ± 3.1, and final C:N ratios ranged from 21.8 ± 0.9 to 28.5 ± 2.9 among all sites and times (Table 3). Initial C/P ratios ranged from 479.1 ± 12.9 to 704.2 ± 63.4, while final ratios ranged from 254.7 ± 40.5 to 483.8 ± 30.7 among all sites and times (Table 3).

BACI analyses indicated that initial N and P litter

Table 3. Initial and final N (% dry weight) and P (mg/kg) concentrations (± SE), and, C:N and C:P ratios (± SE) for the treatment and control site leaf litter, measured during two pre-effluent years (1989 and 1990) and two effluent application years (1992 and 1993) in the Pointe au Chene Swamp.

Beginning–Ending Date of Experimental Set	Nutrient Concentrations[1]		C:Nutrient Ratios	
	Control Site[2]	Treatment Site[3]	Control Site[2]	Treatment Site[3]
11/17/88–11/10/89				
Initial N	1.14 (0.00)	1.14 (0.00)	39.9 (0.1)	39.9 (0.1)
Final N	1.76 (0.00)	1.64 (0.17)	23.3 (1.8)	28.5 (2.9)
Initial P	869.8 (35.4)	869.8 (35.4)	514.8 (30.8)	525.1 (20.1)
Final P	1405.0 (291.0)	1207.0 (97.5)	310.4 (87.9)	386.5 (29.5)
12/1/89–12/8/90				
Initial N	1.16 (0.08)	1.18 (0.03)	43.7 (3.1)	42.7 (1.4)
Final N	2.08 (0.01)	1.83 (0.16)	22.2 (0.2)	24.3 (1.1)
Initial P	837.9 (62.8)	817.2 (60.9)	610.4 (43.9)	624.7 (43.3)
Final P	1071.8 (36.1)	922.5 (46.1)	446.2 (12.7)	483.8 (30.7)
2/26/92–12/10/92				
Initial N	1.27 (0.05)	1.30 (0.01)	38.8 (1.9)	37.3 (0.2)
Final N	1.87 (0.03)	1.99 (0.05)	24.0 (0.2)	22.4 (0.6)
Initial P	733.1 (11.1)	706.7 (28.4)	671.7 (5.9)	690.0 (26.0)
Final P	1530.0 (107.0)	1823.3 (190.6)	297.2 (21.5)	254.7 (40.5)
1/22/93–1/18/94				
Initial N	1.36 (0.01)	1.27 (0.01)	35.2 (0.1)	39.3 (0.9)
Final N	2.16 (0.09)	2.24 (0.12)	21.8 (0.9)	20.9 (1.0)
Initial P	999.8 (21.7)	724.0 (64.2)	479.1 (12.9)	704.2 (63.4)
Final P	1149.6 (26.4)	1362.4 (110.4)	410.9 (13.2)	349.1 (25.0)

[1] Concentrations of N are reported as % dry weight, concentrations of P are reported as mg/kg dry weight.
[2] Control site never received wastewater effluent.
[3] Treatment site received wastewater effluent in 1992 and 1993.

concentrations were not affected by wastewater effluent. The differences in initial litter N and P concentrations between the control and treatment site during the pre-effluent period were not significantly different from the differences between sites during the post-effluent period (for litter N: F = 0.36, P > F = 0.55, for litter P: F = 3.79, P > F = 0.067).

However, differences in final litter N and P concentrations between the control and treatment site during the pre-effluent period were significantly different from the differences between sites during the post-effluent period, indicating an effluent effect (for litter N: F = 5.89, P > F = 0.025, for litter P: F = 9.52, P > F = 0.006). Specifically, final litter N and P concentrations were higher in the control site during the pre-effluent period and higher in the treatment site during the post-effluent period (Table 3).

Accretion

Feldspar Markers. Total pre-effluent accretion (Table 4), measured 34 months after the horizon markers were installed, averaged (± SE) 22.3 ± 3.2 mm in the treatment site (n = 5) and was not significantly different (F = 1.35, P > F = 0.27) from the pre-effluent accretion measured in the control site (14.9 ± 4.6 mm, n = 7) but was significantly greater (F = 18.5, P > F = 0.0008) than the accretion measured on the ridge (9.5 ± 1.4 mm, n = 10) (Table 4). Control and ridge site accretion were not significantly different (F = 1.7, P > F = 0.20) (Table 4). Mean accretion rates for the pre-effluent period were 7.8 ± 1.1, 5.2 ± 1.6, and 3.2 ± 0.4 mm yr^{-1} in the treatment, control, and ridge sites, respectively (Table 4).

Total accretion measured in August 1994 (Table 4), 68 months after the horizon markers were installed and 29 months after effluent application began in the treatment site, averaged 54.6 ± 1.5 mm in the treatment site (n = 3) and was significantly greater (F = 66.08, P > F = 0.0002) than total accretion after 68 months in the control site (19.0 ± 3.2 mm, n = 5). Additionally, after 68 months, total accretion in the ridge site (14.9 ± 2.8 mm, n = 7) was significantly less than accretion in the treatment site (F = 77.8, P > F = 0.0000). Accretion in the control and ridge sites was not significantly different (F = 0.91, P > F = 0.361). Accretion rates for the total 68-month period averaged 9.7 ± 0.3 mm yr^{-1} in the treatment site, 3.4 ± 0.6 mm yr^{-1} in the control site, and 2.7 ± 0.5 mm yr^{-1} on the ridge.

Total accretion and accretion rates measured after 68

Table 4. Total accretion and accretion rates (mean ± SE) measured in the Pointe au Chene swamp using feldspar and ^{137}Cs markers. Underlines connect values that are not significantly different at the 0.05 level.

Time Period	Method Marker Horizon	Total Accretion (mm) Treatment Site[1]	Total Accretion (mm) Control Site	Total Accretion (mm) Ridge Site	Accretion Rate (mm yr⁻¹) Treatment Site	Accretion Rate (mm yr⁻¹) Control Site	Accretion Rate (mm yr⁻¹) Ridge Site
12/88–10/91[2]	Feldspar	22.2 ± 3.2	14.9 ± 4.6	9.5 ± 1.4	7.8 ± 1.1	5.2 ± 1.6	3.2 ± 0.4
12/88–8/94	Feldspar	54.6 ± 1.5	19.0 ± 3.2	14.9 ± 2.8	9.7 ± 0.3	3.4 ± 0.6	2.7 ± 0.5
10/91–8/94	Calculation[3]	32.4	4.1	5.4	11.4	1.41	1.8
1963–1988	^{137}Cs to Feldspar	111.2 ± 9.7			4.4 ± 0.4		
1963–1993	^{137}Cs to Surface	166.7 ± 8.3			5.5 ± 0.3		

[1] Treatment site started receiving wastewater effluent in March 1992.
[2] Core samples on the ridge were collected two months after those in the ridge and control sites, therefore the 10/91 date in the table is actually 12/91 for the ridge site.
[3] Total accretion and accretion rates were calculated as the difference between the 8/94 to 10/91 values and the 10/91 to 8/94 values. No statistical analyses were performed on these estimates.

months integrate both pre- and post-effluent conditions. Accretion during the October 1991 to August 1994 post-effluent period was estimated by subtracting the mean December 1988 to October 1991 accretion in each site from the mean December 1988 to August 1994 accretion. Total accretion during this post-effluent period ranged from a high of 32.4 mm in the treatment site to 4.1 and 5.4 mm in the control and ridge sites, respectively (Table 4). Accretion rates during the post-effluent time period were estimated at 11.4, 1.4, and 1.8 mm yr^{-1} in the treatment, control, and ridge sites, respectively.

Accretion rates may have been overestimated on the ridge because, in several places, the feldspar layer was either exposed, even after 68 months, or missing (possibly eroded away). In places where the marker was exposed, total accretion was recorded as zero, but there was no way to account for negative accretion (erosion) using the feldspar markers.

^{137}Cs Markers. One of the four cores extracted from the treatment site for ^{137}Cs analysis showed evidence of bioturbation and was not used for analyses. Partially decomposed organic matter and live roots were abundant in the top 10 to 12 cm of the remaining cores, and the sediment was relatively unconsolidated compared to the deeper sections. Below this organic rich layer, there was a fairly abrupt transition (within 4 to 8 cm) to a consolidated clay layer, which extended to the limits of the cores (30–36 cm deep). Organic fragments and larger roots were occasionally observed in this clay layer.

Bulk density, organic and mineral matter, and volume distribution profiles were similar in the three remaining cores (Figure 3). Linear regression analyses revealed that there were significant ($P < 0.05$) increases in bulk density ($r^2 = 0.69$), percent mineral matter by weight ($r^2 = 0.69$), and percent mineral matter by volume ($r^2 = 0.68$) with depth. Bulk densities ranged from 0.11g cm^{-3} at the surface to over 1.0 g cm^{-3} at the 30-cm depth. Percent mineral matter by dry weight ranged from 36.9% near the surface to over 93.0% at the 30-cm depth, while percent mineral matter, by volume, ranged from a low of 2.0% at the surface to over 30.0% at 30 cm. In contrast, percent organic matter, by weight and percent pore space, decreased significantly with depth ($r^2 = 0.56$ and 0.69, respectively). Percent organic matter by weight ranged from 63.1% near the surface to less than 7.0% at 30 cm. Pore space, either occupied by water or air, ranged from a high of 92.2% near the surface to 54.4% at depth of 25 cm and below. Percent organic matter, by volume, remained fairly constant throughout the depth of each core, varying from 2.3% to 12.2%, and was not significantly correlated with depth ($r^2 = 0.04$).

All three undisturbed cores had an obvious ^{137}Cs peak (Figure 3). Even though bulk density changed dramatically from the top to the bottom of the cores, the depth of peak ^{137}Cs activity was the same whether activity was calculated on a core section (volume) basis or normalized by section weight. The ^{137}Cs distribution profiles shown in Figure 3 are normalized by weight. The mean thickness of the sediment layer between the 1963 ^{137}Cs marker and the 1993 surface ($n = 3$) was 166.7 ± 8.3 mm for the 30-year period, and accretion rates averaged 5.5 ± 0.3 mm yr^{-1}. This rate integrates both 28 to 29 years of pre-effluent and 1.6 years of post-effluent sediment accumulation.

To factor out effluent-influenced sediment accumulation, accretion rates were calculated using the two cores that contained both a 1988 feldspar marker and a 1963 ^{137}Cs marker. The average sediment thickness between these two layers was 111.2 ± 10.0 mm for the 25-year period, resulting in a mean ''background'' accretion rate of 4.4 ± 0.4 mm yr^{-1}.

Accretion Balance Deficits. To determine if sedimentation was keeping pace with RSLR, accretion balance deficits for the three sites were calculated by subtract-

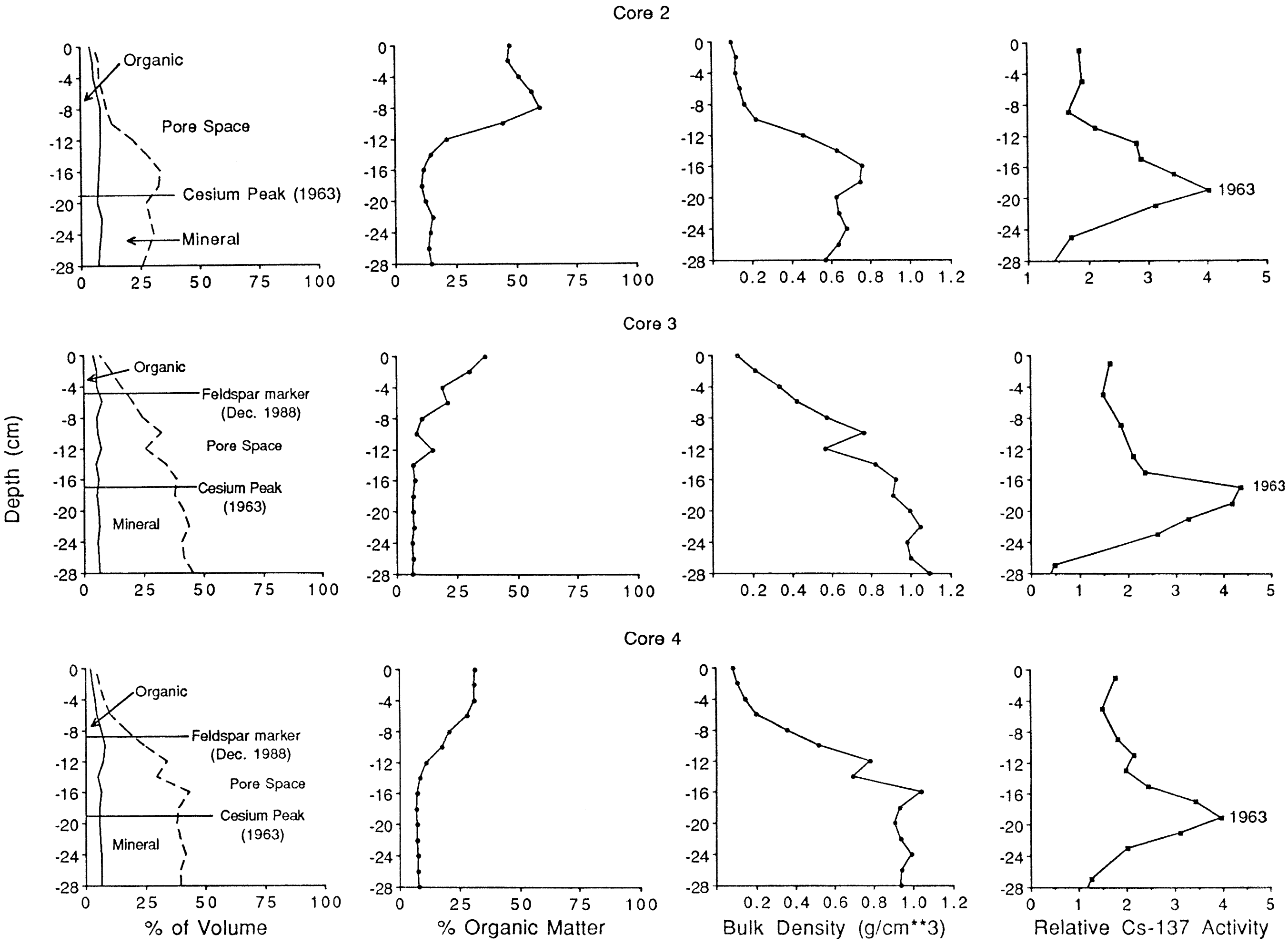

Figure 3. Volume distributions, percent organic matter (by weight), bulk density, and relative ^{137}Cs activity with depth in cores taken from the treatment site in October 1993. Cores 3 and 4 also contained a feldspar layer that marked the surface in December 1988.

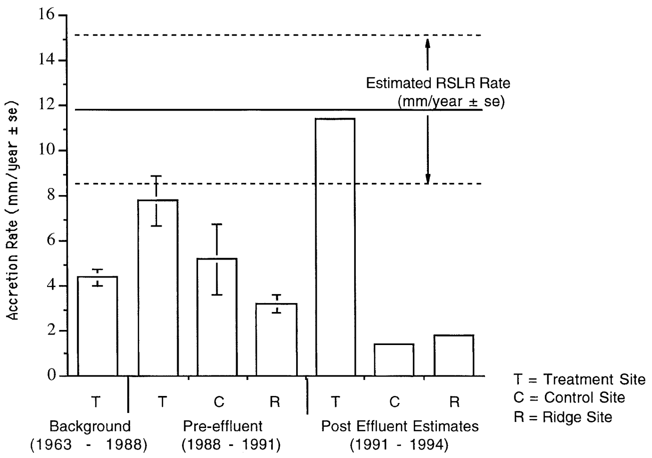

Figure 4. Background, pre-effluent, and post-effluent accretion rates (mean ± SE), relative sea-level-rise rates, and accretion balance deficits in the Pointe au Chene swamp.

ing the RSLR, estimated from tidal gauge analyses (1.23 cm yr^{-1}) from background (1963–1988), pre-effluent (1988–1991), and post-effluent (1991–1994) accretion rates. The background accretion balance deficit in the treatment site (the only site at which ^{137}Cs cores were collected) was -7.9 mm yr^{-1}. All sites had negative accretion balances during the baseline, pre-effluent period (1988–1991), with sediment deficits ranging from -9.1 mm yr^{-1} on the ridge to -4.5 mm yr^{-1} in the treatment site (Figure 4). Accretion balances remained negative during the post-effluent period in all three sites; however, in the treatment site, the deficit was only -0.9 mm yr^{-1} and fell well within one standard error of estimated RSLR rate (Figure 4).

Mineral and Organic Matter Accumulation Rates

Mean annual mineral accumulation rates between 1963 and 1988 were not significantly different than

Table 5. Average mineral and organic matter accumulation rates in the Pointe au Chene Swamp from 1963–1988 and from 1988–1993. Only organic matter accumulation rates were significantly different between time periods ($P < 0.05$).

	Accumulation Rates (g dry weight m^{-2} yr^{-1} ± SE)	
Time Period	Mineral Matter	Organic Matter
1963–1988[1]	2302.0 ± 29.4	275.9 ± 3.3
1988–1993[2]	2004.6 ± 67.0	736.7 ± 58.3

[1] Soil horizon bounded by a ^{137}Cs marker on the bottom and a feldspar marker on the top.
[2] Soil horizon bounded by a feldspar marker on the bottom and the soil surface in 1993 at the top.

mineral accumulation rates during the 1988–1993 time period ($F = 16.5$, $P > F = 0.6$) (Table 5). Organic matter accumulation rates, however, were significantly greater ($F = 62.4$, $P > F = 0.01$) during 1988–1993 than in the earlier period (Table 5).

DISCUSSION

Decomposition

Overall, wastewater effluent had no effect on rates of decomposition but did affect nutrient concentrations within the decomposing litter matrix. Although all of the annual rates of decomposition measured during this study fell within the range reported by Brinson (1990) for riverine forested wetlands, there was a large and significant variation in annual rates within sites and between years. This may be due to several sources of annual variation, including the starting date of the experiment and the initial condition of litter collected from the traps. However, this fluctuation in decay rates from year to year illustrates the strength of the BACI design because the comparison of differences between the control and treatment before the impact to comparisons of the difference between sites after the impact factors out these sources of variation.

Nitrogen and phosphorus availability limits the biological decomposition of leaf-litter because of the disparity between the high demand for N and P by decomposer organisms, with low C:N and C:P ratios, and the typically high C:N and C:P ratios found in leaf-litter (Swift et al. 1979, Enriquez et al. 1993). Final leaf-litter C:N and C:P ratios of 16:1 and 200:1, respectively, are indicative of complete microbial decay (Brinson 1977). Because the initial litter C:N and C:P ratios found during this study were typically more than twice as high as these values, it would be expected that microbial decomposers would use exogenous sources of N and P, if available, to satisfy their demand. Therefore, it is possible that the addition of supplementary N and P to a system would stimulate the rates of litter decomposition.

While numerous studies have shown this to be the case in flooded systems, others (including this one) have shown no nutrient amendment effect on the rates of decomposition (Rybczyk et al. 1996). This disparity among studies, and the reason no impact effect was detected during this study, may be due to two factors: 1) the duration of the experiment and 2) the lack of an observed effluent-effect on initial litter nutrient concentrations (Rybczyk et al. 1996).

Experimental Duration. In a previous review of 29 wetland studies that examined the effect of nutrient amendments on decomposition in flooded systems, we found that experimental duration was a good predictor of experimental outcome (Rybczyk et al. 1996). Specifically, 12 of 14 experiments that lasted less than 100 days showed that nutrient amendments positively affected decomposition rates. Conversely, only 6 of 15 studies lasting 200 days or longer showed any nutrient-amendment effect on decomposition rates. Other studies have also shown that nutrient amendments affect the initial phases of decomposition but have little or no effect on the latter stages (Valiela et al. 1985, Webster and Benfield 1986). This is due, in part, to the shift in the dominant form of substrate C as the leaf-litter decomposes from labile to more resistant ligninous C forms, which are less affected by exogenous nutrient supplies (Melillo et al. 1984). This suggests that long-term experiments, such as this one, which describe decomposition with simple, one-compartment exponential decay models, are not sensitive to processes that affect only the early stages of decomposition. However, since this study was concerned primarily with issues and processes that are best measured in time scales of years (accretion, production, relative sea-level rise, long-term decomposition), only simple exponential models were used to summarize annual decomposition rates.

Initial Leaf-Litter Concentrations. Typically, nutrient enrichment/decomposition experiments fall into two general categories: 1) those that enrich the plant tissue (and consequently the litter) before it is decomposed (internal treatments) and 2) those that fertilize the litter incubation site, but use a common litter source in both the control and treatment sites (external treatments). With external experiments, the treatment effect is ''nutrient applied to the wetland,'' whereas with internal experiments, the treatment effect is ''initial litter nutrient concentration.'' In general, internal fertilization experiments show a fertilizer effect on rates of decomposition, while results from external experiments are less conclusive (Rybczyk et al. 1996).

There are two reasons why it would be expected that enriched litter would have a faster rate of decomposition than un-enriched litter. First, enriched litter has a greater supply of limiting N and P for microbial decomposition, and second, there is some evidence that, proportionally, the refractory litter components decrease and the soluble and labile fractions of the litter increase as wetland plant tissue is enriched (Debusk and Dierberg 1984, Valiela et al. 1984). This may also explain why internal type experiments were more likely to show an amendment effect than external experiments because, while both treatments supply N and P to nutrient limited decomposers, only the internally enriched litter is structurally affected by the treatments. Originally, this study was designed to allow for an internal nutrient effect on decomposition rates because we hypothesized that litter taken from the treatment site after effluent additions began would have higher initial N and P concentrations relative to the control. Instead, we found that wastewater effluent did not affect initial litter nutrient concentration, resulting in a *de-facto* ''external treatment'' type experiment. In contrast to this study, several researchers have demonstrated that wastewater effluent additions to wetlands increased initial litter nutrient concentrations (Brown 1981, Bayley et al. 1985, Aschmann et al. 1990).

That we measured no effluent effect on initial litter nutrient concentrations is most likely a function of two factors. First, the translocation of nutrients, or the leaching of labile N and P, prior to leaf abscission (Chapin 1980) may have negated any effluent associated increase in N and P concentrations in live leaf material. Second, both sites were continually flooded. Flood-related physiological stress, such as root growth inhibition (Pezeshki 1991) or the limitation of active uptake of nitrogen under hypoxic conditions (Bandopadhyay et al. 1993), may have limited nutrient uptake in general, regardless of the nutrient amendment regime.

Sediment Accretion

Since the primary source of mineral sediments to the Pointe au Chene swamp was eliminated when levees along the Mississippi were completed in the 1930s, we initially suspected that we would find little or no mineral sediments above the 1963 ^{137}Cs horizon. We found instead that the post-1963 mineral accumulation rates in the treatment site were higher than or within the range of published rates for coastal salt marshes in this region (Callaway 1994) and comparable to accumulation rates for other bottomland hardwood forests (Johnston 1991). The bottomland hardwood ridge may be the source of these mineral sediments to the treatment swamp because measured accretion rates on the ridge site were so low and possibly overestimated.

In contrast to mineral-matter accumulation, organic matter accumulation rates increased significantly during the 1988–1993 period. This is probably due to effluent-stimulated organic matter accretion after March 1992. Productivity measurements at this site, taken concomitantly with this study (Day et al. 1993, 1994), showed that the production of floating aquatic vegetation (predominantly *Lemna* sp.) increased in the treatment site after the introduction of wastewater effluent and increased in relationship to the control site. Additionally, other studies have shown that floating aquatic vegetation can be a significant source of organic matter in wetland wastewater treatment systems (Harvey and Fox 1973, Odum et al. 1975). However, it is critical to note that the more recently accumulated organic sediments have had less time to decompose than has the 1963–1988 cohort and that the apparent increase in accumulation rates in the newer sediments may be an artifact of this.

Unlike the decomposition experiments, the statistical models used to analyze the accretion data did not test whether wastewater effluent affected accretion rates in the Pointe au Chene swamp. However, the weight of the evidence, both statistical and biological, strongly suggests that there was indeed a ''treatment'' effect in this specific forested wetland. First, we found that accretion rates in the treatment site increased in relation to rates in the control after wastewater was applied to the treatment site. Second, productivity measurements at this site, taken concomitantly with this study (Day et al. 1993, 1994), showed that the production of floating aquatic vegetation (predominantly *Lemna* sp.) increased in the treatment site after the introduction of wastewater effluent and increased in relation to the control site. Third, soil profile analyses (Table 5) in the treatment site indicated that organic matter accumulation had increased since wastewater effluent additions began. Fourth, hydroperiods were similar in both sites over the 68 months of this experiment (Day et al. 1998). Finally, this study showed that decomposition rates were not affected by wastewater effluent. Therefore, we cannot propose any mechanism, other than effluent-stimulated organic matter production, that would have caused the observed increase in accretion rates in the treatment site relative to the control.

Accretion Balance Deficits and the Restoration Potential of Wastewater Effluent

The ultimate success of this type of restoration effort depends upon whether or not wastewater effluent can stimulate primary production to the point that or-

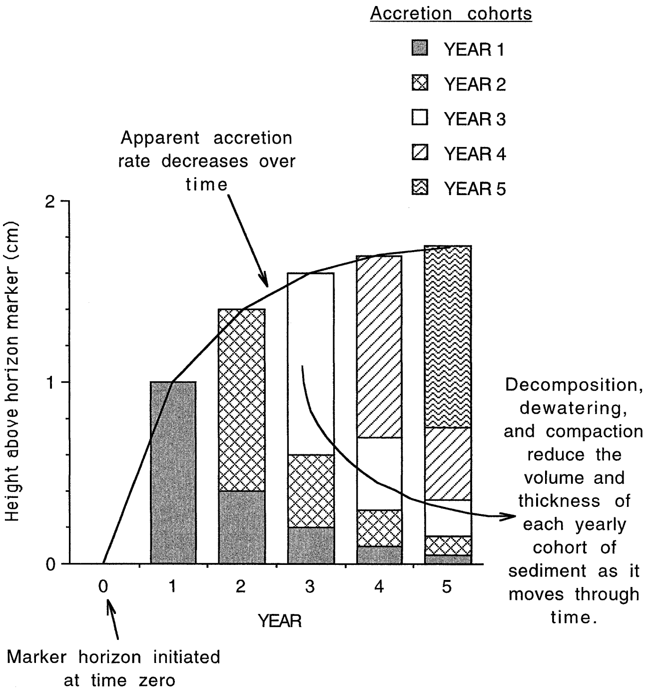

Figure 5. Conceptual diagram of sediment accretion dynamics above a horizon marker over time. Given that the yearly cohort of sediment is constant, the apparent rate of accretion (solid line) decreases with time because of decomposition and compaction.

ganic matter accretion can balance the rate of RSLR. Estimates of accretion balance should be viewed with caution, however, because short-term accretion measurement methods (feldspar horizon markers for example) fail to fully integrate long-term but significant decomposition, dewatering, and compaction processes (DeLaune et al. 1989, Reed and Cahoon 1993). As a result, accretion balance deficits often are underestimated because accretion rates are overestimated. A conceptual diagram of this process is shown in Figure 5; although yearly inputs of sediments over the feldspar marker remain constant, the volume of each yearly cohort is reduced over time, and thus, integrated accretion rates decrease with time. Other factors also contribute to the uncertainty involved with estimating accretion balance deficits. For example, even within a single basin, subsidence rates are highly variable (Turner 1991), and the RSLR measured in a waterway, where gauges are usually located, may not represent RSLR in shallow intradistributary wetlands of interest.

The best estimates of accretion balance can be made when the measurement technique not only spans a long enough period to integrate decomposition and compaction processes, but also spans the same time period as the tidal gauge record used to estimate RSLR (Turner 1991). Therefore, the baseline accretion balance deficit of -7.9 mm yr^{-1}, calculated using accretion rates obtained from ^{137}Cs horizons (1963–1988) and RSLR rates obtained from 1962–1982 tidal gauge analysis (Penland et al. 1988), is probably the most accurate background or baseline deficit estimate for the Point au Chene swamp. The time spanned by this estimate is also long enough to integrate most decomposition and some compaction and dewatering processes.

Even though the post-effluent accretion balance deficit of -0.9 mm yr^{-1} in the treatment site is probably an underestimate of the long-term deficit rate, it is an order of magnitude less than background deficits, indicating that there is a potential for using effluent to balance accretion deficits. However, because this wetland is already permanently flooded, rates of accretion would have to exceed rates of RSLR for several years to relieve flooding stress.

The question remains as to how the overall accretion balance would change and how the entire wetland system would respond if, and when, the wetland surface ever became sub-aerial. It is probable that decomposition rates would increase as the soil became increasingly oxygenated (Conner and Day 1991). Rates of mineral deposition likely would decrease as well, as elevation increased (French 1993). Both of these processes would tend to decrease overall rates of accretion. Conversely, both above- and below-ground primary production would likely increase if the wetland was submerged periodically, rather than permanently (Day and Megonigal 1993). This, in turn, would tend to decrease the accretion balance deficit and possibly allow for seedling establishment and forest regeneration (Keeland and Conner 1999).

These questions can best be answered by long term-monitoring at this site (which is occurring) and by developing ecosystem models that incorporate the various non-linear feedback mechanisms that affect wetland elevation. For example, a relative elevation model ecosystem developed for this system revealed that changes in wetland elevation were much more responsive, or sensitive, to changes in primary production than to changes in rates of decomposition (Rybczyk et al. 1998). This would suggest that, as flooding decreased, increased organic matter production and accretion would offset any increase in rates of decomposition.

CONCLUSION

We originally hypothesized that the addition of nutrient-rich, secondarily treated wastewater to hydrologically isolated, nutrient-limited, and subsiding wetlands could promote vertical accretion through increased organic matter production and deposition. However, we also recognized that nutrient enrichment could increase the rates of organic matter decomposition, thus negating any affect of increased productivity and accretion.

This study found that the nutrients associated with wastewater effluent did not affect annual rates of leaf-litter decomposition. We also found that the rates of sediment accretion in general, and organic matter accumulation, in particular, increased in the treatment site after effluent applications began. More significantly, after effluent applications began, the rates of accretion in the treatment site approached the estimated rates of RSLR.

While this study and the issue of wetland loss due to high rates of RSLR may presently apply only to the extensive wetlands associated with subsiding delta regions, the worldwide eustatic sea-level-rise component of RSLR is expected to increase steadily over the next century due to the impacts of global warming (Gornitz 1995). Therefore, this region can serve as a model for other coastal wetlands that also may face problems associated with rising water levels in the near future.

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