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A relative elevation model for a subsiding coastal forested wetland receiving wastewater effluent

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Abstract

This paper describes a wetland elevation/sediment accretion model for a subsiding forested wetland located within the coastal zone of Louisiana, USA. We designed the model to determine if the addition of secondarily treated municipal wastewater to the wetland could stimulate organic matter production and deposition to the point that sediment accretion would balance relative sea level rise (deep subsidence plus eustatic sea level rise (ESLR)). We also used the model to simulate the effect of predicted increases in ESLR on wetland stability and to determine the amount of additional mineral sediment that would be required to compensate for relative sea level rise. The model utilizes a cohort approach to simulate sediment dynamics (organic and mineral matter accretion, decomposition, compaction, and below-ground productivity) and yields total sediment height as an output. Sediment height is balanced with ESLR and deep subsidence, both forcing functions, to calculate wetland elevation relative to mean water levels. The model also simulates primary production (roots, leaves, wood, and floating aquatic vegetation) and mineral matter deposition, both of which are feedback functions of elevation. Simulated wetland elevation was more sensitive to the uncertainty surrounding estimates of deep subsidence and future ESLR rates than in other processes that affect wetland elevation and could be influenced by wastewater (i.e. rates of decomposition and primary productivity). The model projected that, although the addition of wastewater effluent would increase long term accretion rates from 0.35 to 0.46 cm year^{-1}, it would not be enough to offset the current rate of relative sea level rise. A series of mineral input simulations revealed that, given no increase in ESLR rates, an additional 3000 g m^{-2} year^{-1} of mineral sediments would be required to maintain a stable elevation.

Keywords: Accretion; Below-ground production; Climate change; Decomposition; Eustatic sea level rise; Floating aquatic vegetation

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1. Introduction

1.1. Background and objectives

Coastal wetland elevation, relative to sea level, is a function of numerous processes including eustatic seal level rise (ESLR), compaction, decomposition, deep subsidence, allogenic organic matter accumulation and allogenic mineral matter deposition, all operating at different time scales. Historically, seasonal overbank flooding of the Mississippi river deposited mineral sediments into the interdistributary wetlands of its delta plain that counterbalanced relative sea level rise (subsidence plus ESLR) in the region. Furthermore, the nutrients associated with these sediments promoted vertical accretion through organic matter production and deposition (Nyman and DeLaune, 1991). This sediment and nutrient source has been eliminated since the 1930's with the completion of levees along the entire course of the lower Mississippi river, resulting in vertical accretion deficits (accretion-relative sea level rise) and widespread wetland loss throughout the modern delta region (Kesel, 1988; Day and Templet, 1989; DeLaune et al., 1991; Conner et al., 1993; Boesch et al., 1994). Contributing further to the problem, the ESLR component of relative sea level rise (RSLR) is expected to accelerate over the next 100 years (Gornitz, 1995). If coastal wetlands in general, and deltaic wetlands in particular, are to persist in the face of rising water levels, they must be able to accrete sediments at a rate such that surface elevation gain is sufficient to offset RSLR.

The goal of several recent wetland restoration projects in the subsiding delta region of Louisiana has been to balance vertical accretion deficits by adding supplemental mineral sediments to wetlands or by constructing sediment trapping mechanisms or landforms (Boesch et al., 1994). Attempts have been made to predict the fate of wetlands subject to these types of sediment management practices, or to an acceleration in RSLR, by comparing current and predicted rates of RSLR to measured rates of sediment accretion and then calculating an accretion deficit, surplus or balance (Stevenson et al., 1986; DeLaune et al., 1987; Templet and Meyer-Arendt, 1988; Bricker-Urso et al., 1989; Conner and Day, 1989; Day et al., 1995). For example, in a coastal forested wetland in Louisiana, DeLaune et al. (1987) measured RSLR rates of 1.36 cm year^{-1} and accretion rates of only 0.63 cm year^{-1}, to derive an accretion balance deficit of 0.73 cm year^{-1}. However, these types of calculations should be viewed with caution because the typically short-term field measurements of accretion (1 or 2 years) do not fully integrate other long-term processes, such as compaction and decomposition, that also affect wetland elevation. Additionally, these types of measurements do not take into account possible feedback mechanisms on the processes that affect elevation. Specifically, changes in elevation can result in changes in decomposition, allogenic sediment deposition and autogenic primary production. For this reason, ecosystem models, that incorporate feedback mechanisms and simulate critical processes over the proper time scale, can be useful for examining the response of wetland elevation to various management practices or to increasing rates of sea level rise.

We present here a wetland elevation/sediment accretion model for the Pointe au Chene swamp, a subsiding forested wetland located within the coastal zone of Louisiana (Fig. 1), that has been receiving secondarily treated municipal wastewater since 1992. Since organic as well as mineral matter has been shown to be a critical component of vertical accretion in many coastal wetlands (Hatton et al., 1983; Gosselink and Hatton, 1984; Callaway et al., 1996), we hypothesized the addition of non-toxic secondarily treated wastewater effluent to this swamp could simulate organic matter accretion to the point that wetland elevation could keep pace with RSLR. Therefore, we designed this model to examine the effects of nutrient enrichment on elevation in this swamp. In a more generic sense, however, we used the model to examine the relative influence of many of the processes that affect relative wetland elevation. Thus the results of the numerous simulations undertaken as part of this study shed some light on the possible effects of predicted increases in ESLR on all coastal wetlands.

The specific objectives of the model were to: (1) determine the relative sensitivity of wetland eleva-

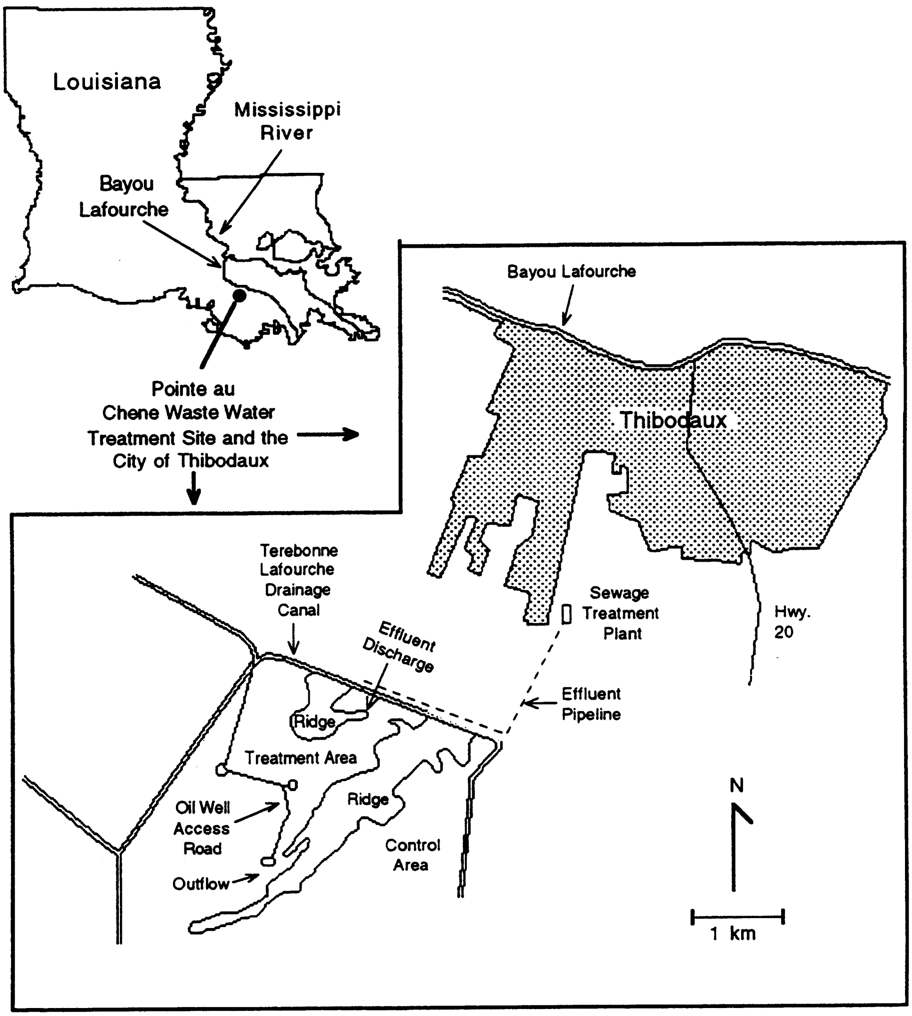

Fig. 1. Map of the Pointe au Chene swamp, located adjacent to the city of Thibodaux, Louisiana, USA. An abandoned oil access road, a bottomland hardwood ridge, and the spoil banks associated with the Terrebonne–Lafourche drainage canal, hydrologically isolate the treatment site from the surrounding wetlands.

tion to variations in rates of ESLR, subsidence, primary production, sediment compaction, decomposition and mineral accretion; (2) simulate the effect of predicted increases in ESLR on wetland elevation at the Pointe au Chene swamp; (3) examine how the uncertainty surrounding estimates of deep subsidence in the delta region affect model-generated predictions of wetland sustainability; (4) simulate the long term (50 years) effects of wastewater effluent additions on wetland elevation; and (5) determine the amount of mineral sediment additions that would be required to compensate for RSLR.

1.2. Modeling wetland elevation

Existing marsh elevation models have focused on simulating sub-sets of the processes that affect wetland elevation and have either ignored other processes or included them as forcing functions. French (1993) and Allen (1990), for example, developed detailed algorithms to simulate allo-

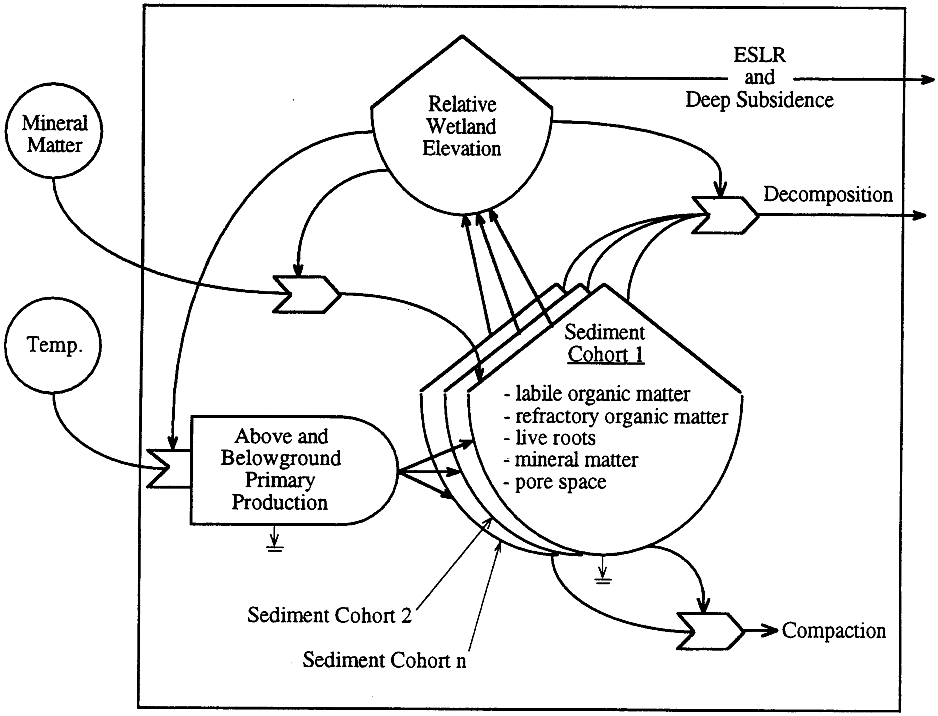

Fig. 2. An energy circuit (Odum, 1983) conceptual diagram of the relative elevation model. Deep subsidence, ESLR, temperature and mineral matter inputs are system forcing functions. Wetland elevation modifies primary production, mineral matter inputs and decomposition. There are 18 sediment cohort layers.

genic sediment deposition in Great Britain salt marshes as a feedback function of elevation. However, in both of these models, autogenic organic matter was entered as pre-compacted, predecomposed, forcing function. In contrast, Randerson (1979) constructed a salt marsh development model that focused primarily on simulations of plant community structure and productivity as a function of elevation, but relied on simple plant biomass versus accretion regressions to simulate mineral sediment accretion. Morris and Bowden (1986) developed a yearly sediment cohort model that simulated many of the below-ground processes that contribute to tidal marsh elevation, including labile and refractory organic matter decomposition and belowground production. However, the primary focus of this was model was to simulate N, C and P dynamics in a sediment column and not to simulate changes in relative marsh elevation in response to sea level rise. Chmura et al. (1992) was the first to develop a sediment cohort model that was specifically designed to simulate relative marsh elevation and stability under various sea level rise scenarios. This model, however, made no distinction between organic and mineral matter inputs and, as a consequence, assumed a homogeneous sediment composition with depth. Additionally, inputs of sediment were modelled as a constant and were not a function of elevation.

Although compaction is an active process in shallow marsh sediments (Penland et al., 1988), most marsh elevation/sediment dynamics models have either ignored this process (Randerson, 1979), included it as a forcing function (Chmura et al., 1992) or input sediments as precompacted units (Allen, 1990; French, 1993). Callaway et al. (1996) developed a cohort sediment accretion model for coastal wetlands, similar in framework to the model developed by Morris and Bowden (1986), that simulated compaction as a function of the density of sediment above a given cohort.

Table 1
Model parameters, descriptions, values and sources

Symbol	Description	Values and units	Source
A. State variables			
L	Live leaf biomass	g d.w. m^{-2}	Field data
R_t	Total live root biomass	g d.w. m^{-2}	Day and Megonigal, 1993
V	F.A.V. biomass	g d.w. m^{-2}	Sklar, 1983
W	Live tree biomass	g d.w. m^{-2}	Field data
$B(n)$	Refractory organic matter in cohort (n)	g d.w. cm^{-2}	Field data/simulation
$M(n)$	Mineral matter in cohort (n)	g d.w. cm^{-2}	Field data/simulation
$Q(n)$	Labile organic matter in cohort (n)	g d.w. cm^{-2}	Field data/simulation
$R(n)$	Live root biomass in cohort (n)	g d.w. cm^{-2}	Field data/simulation
B. Forcing functions			
D	Maximum mineral input	0.00443 g cm^{-2} week^{-1}	Estimated from field data
E_1	Rate of ESLR, initialized at current rate	15 cm in 100 years	Gornitz, 1995
E_2	IPCC 'best guess' estimate ESLR	48 cm in 100 years	Gornitz, 1995
E_3	IPCC 'business as usual' estimate ESLR	66 cm in 100 years	Gornitz, 1995
S	Local deep subsidence rate	0.0207 cm week^{-1}	Penland et al., 1988
T	Mean weekly temperature	°C	NOAA weather records
C. Rates and constants			
f_1	Labile fraction of FAV litter	0.8 unitless	Field data
f_2	Labile fraction of above-ground biomass	0.3 unitless	Estimated from field data
f_3	Labile fraction of live roots	0.2 unitless	Day et al., 1989
f_4	Rate of root litter production	0.3 year^{-1}	Day and Megonigal, 1993
f_5	Leaf litter production rate	0.015 week^{-1} if week$>$25 and $<$45 else 1.0 if week$>$45	Field data
f_6	Temperature dependent FAV litter production rate (week^{-1})	if $T<13$ then.3 else.08 (week^{-1})	Calibration
k_1	Decomposition rate of deep refractory organic matter	0.0001 week^{-1}	Calibration
k_2	Decomposition rate of labile root organic matter	0.028 week^{-1}	Day et al., 1989
k_3	Decomposition rate of surface labile organic matter	0.028 week-1	Field data
k_4	Decomposition rate of refractory organic matter	0.0008 week^{-1}	Field data
k_5	Decomposition rate of surface refractory organic matter	0.0029 week^{-1}	Field data
l_i	Initial height of sediment column	29.80 cm	Model generated
l_r	Initial relative wetland elevation	0 cm	Field data
p_k	Half saturation constant for soil compaction	2.5 cm^3 g^{-1}	Calculated from field data
p_m	Minimum fraction of pore space in soil	0.5813 unitless (0–1)	Field data
p_x	Maximum fraction of pore space in soil	0.9316 unitless (0–1)	Field data
r	Root distribution constant	0.3 cm^{-1}	Calibration
v_k	FAV crowding constant	-0.0255 g d.w.$^{-1}$ m^{-2}	Calibration
v_{max}	Max. net FAV production rate	2.66 week^{-1}	Rejmankova, 1975
w_1	Tree flood tolerance	0.75 (m)	Phipps, 1979
w_2	Tree mortality rate	0.0006 week^{-1}	Rybczyk et al., 1995
D. Functions			
$C_{func}(n)$	Pore space compaction function	unitless (0–1)	Model generated
H_{func}	Elevation function that modifies g_{max}	unitless (0–1)	Phipps (1979)
m_{func}	Mineral input as a function of elevation	unitless (0–1)	Field data
T_{func}	Temperature function that modifies v_{max}	unitless (0–1)	Estimated
V_{func}	Exponential FAV crowding function	unitless (0–1)	Rejmankova, 1982

Table 1 (Continued)

Symbol	Description	Values and units	Source
E. Output			
a_l	Above-ground labile litter inputs to surface cohort	g d.w. cm^{-2} week^{-1}	Model generated
$b(n)$	Bulk density of cohort n	g d.w. cm^{-3}	Model generated
$g(n)$	Total mass of all cohorts above cohort n	g d.w. cm^{-2}	Model generated
g_{max}	Maximum net leaf productivity	g d.w. m^{-2} week^{-1}	Derived from field data
$h(n)$	Total height of cohort n	cm	Model generated
h_d	ESLR rate + deep subsidence rate	cm week^{-1}	Model generated
h_r	Wetland elevation relative to mean water level	cm	Model generated
$h_t(n)$	$h(n)$ + height of all cohorts above	cm	Model generated
h_w	Water table depth	m	Model generated
$m_h(n)$	Height of mineral matter in cohort n	cm	Model generated
$m_v(n)$	% mineral matter by volume in cohort n	unitless	Model generated
$m_w(n)$	Weight of mineral mass in cohort n	g d.w. cm^{-2}	Model generated
$o_h(n)$	Height of organic matter in cohort n	cm	Model generated
$o_v(n)$	% organic matter by volume in cohort n	unitless (0–1)	Model generated
$o_w(n)$	Weight of organic mass in cohort n	g d.w. cm^{-2}	Model generated
$p_h(n)$	Height of pore space in cohort n	cm	Model generated
$p_s(n)$	Fraction of pore space in cohort n	unitless (0–1)	Model generated
r_g	Total net root production (modified by elevation)	g d.w. cm^{-2} week^{-1}	Model generated
$r_i(n)$	Root distribution to cohort n	g d.w. cm^{-2}	Model generated
$r_l(n)$	Root litter input for cohort n	g d.w. cm^{-2} week^{-1}	Model generated
$r_p(n)$	Weekly root production input to cohort n	g d.w. cm^{-2} week^{-1}	Model generated
s	Root production at surface (surface intercept)	g d.w. cm^{-2} week^{-1}	

The model presented here is a logical extension of the sediment cohort models developed by Morris and Bowden (1986) and later, Callaway et al. (1996). Additionally, it incorporates a simple mineral deposition feedback function that is derivative of the algorithms developed by French (1993) and Allen (1990). Finally, it mechanizes the many processes related to wetland elevation that Chmura et al. (1992) programmed conceptually into her original wetland stability/RSLR model.

2. Methods

2.1. Site description

The Pointe au Chene swamp (Fig. 1) consists of two permanently flooded forested wetlands (*Acer rubrum*, *Salix nigra*, *Taxodium distichum* associations) separated by a slightly elevated bottomland hardwood ridge (*Quercus nigra*, *Liquidambar styraciflua* and *Ulmus americana* associations). In the flooded sites, floating aquatic vegetation (FAV), consisting primarily of duckweed (*Lemna* sp.) and *Salivinia* sp., covers the water surface for most of the year (Zhang, 1995). Since 1992, the 231 ha forested wetland on the western side of the ridge has received secondarily treated municipal wastewater at a rate of 15000 m^{-3} day^{-1}. The ridge site is $\approx$40 cm higher than the flooded sites and is not inundated for most of the year. Accretion balance deficits, due primarily to a lack of allogenic sediment inputs coupled with high rates of RSLR, approach 0.79 cm year^{-1} for the forest (Rybczyk et al., 1996a). A comprehensive site description is given by Rybczyk et al. (1995).

2.2. Model framework and platform

The model utilizes a cohort approach (tracking discreet packages of sediments through depth and time) to simulate sediment dynamics (organic and mineral matter accretion, decomposition, compaction, and below-ground productivity). These

Table 2
State variables and differential equations for the sediment dynamics sub-model

Labile organic matter sediment cohorts, $Q(n)$

$$dQ(n)/dt=(a_l f_2)+(r_l(n)f_3)+(Z_l(n-1)Q(n-1))-(Q(n)k_x)-(Z_l(n)Q(n))$$

where:

$Q(n)$	Labile organic matter in cohort n (g d.w. cm^{-2})
a_l	Above-ground labile litter inputs to surface cohort (g d.w. cm^{-2} week^{-1})
f_2	Labile fraction of above ground biomass (unitless)
$r_l(n)$	Root litter inputs to cohort n (g d.w. cm^{-2} week^{-1})
f_3	Labile fraction of root litter (unitless)
$Z_l(n-1)$	Transfer rate of labile organic matter from overlying cohort (g d.w. cm^{-2} year^{-1})
$Q(n-1)$	Labile organic matter in overlying cohort (g d.w. cm^{-2})
k_x	k_2 or k_3 depending upon the position of the cohort: decomposition rate of labile organic matter (week^{-1})
$Z_l(n)$	Transfer rate of labile organic matter to underlying cohort (g d.w. cm^{-2} year^{-1})

Refractory organic matter sediment cohorts, $B(n)$

$$dB(n)/dt=(a_l\ (1-f_2))+(r_l(n)(1-f_3))+(Z_r(n-1)B(n)-1)-(B(n)k_i)-(Z_r(n)B(n))$$

where:

$B(n)$	refractory organic matter in cohort n (g d.w. cm^{-2})
$Z_r(n-1)$	Transfer rate of refractory organic matter from overlying cohort (g d.w. cm^{-2} year^{-1})
$B(n)-1$	Refractory organic matter in overlying cohort (g d.w. cm^{-2})
k_i	k_1, k_4 or k_5 depending upon the position of the cohort: decomposition rate of refractory organic matter (week^{-1})
$Z_r(n)$	Transfer rate of refractory organic matter to underlying cohort (g d.w. cm^{-2} year^{-1})

Mineral matter in sediment cohorts, $M(n)$

$$dM(n)/dt=(Dm_{func})+(Z_m(n-1)M(n-1))-(Z_m(n)M(n))$$

where:

$M(n)$	Mineral matter in cohort n (g cm^{-2})
D	Maximum potential mineral inputs (g d.w. cm^{-2} week^{-1})
m_{func}	Elevation function (unitless)
$Z_m(n-1)$	Transfer rate of mineral matter from overlying cohort (g cm^{-2} year^{-1})
$M(n-1)$	Mineral matter in overlying cohort (g cm^{-2})
$Z_m(n)$	Transfer rate of mineral matter to underlying cohort (g cm^{-2} year^{-1})

Table 2 (*continued*)

Live roots in sediment cohorts, $R(n)$

$$dR(n)/dt=r_i(n)-(f_4\ R(n))$$

where:

$R(n)$	Live root biomass in cohort n (g d.w. cm^{-2})
$r_i(n)$	root production (root) distributed to cohort n (g d.w. cm^{-2})
f_4	Rate of root litter production (week^{-1})

dynamics produce model-generated changes in sediment characteristics including; bulk density, organic matter volume and mass, mineral matter volume and mass and pore volume, and yields total sediment height as an output. Sediment height is then balanced with ESLR and deep subsidence, both forcing functions, to determine wetland elevation relative to sea level. The model also simulates primary production (roots, leaves, wood, and FAV) and mineral inputs, both of which are feedback functions of the model-generated marsh elevation (Fig. 2). The model consists of three linked sub-models: (1) primary productivity; (2) sediment dynamics; and (3) relative elevation. Each of these sectors will be described separately in the Results section.

We programmed the model using STELLA iconographic modeling software (Richmond et al., 1987). An Euler numerical method, with a $\Delta t=1$ week, was used to solve the finite difference equations generated by the STELLA software. A list and description of the state variables, forcing function, rates, constants, functions and outputs programmed into the model are shown in Table 1.

2.3. Initialization and calibration

We used field measurements collected as part of numerous research projects associated with the Pointe a Chene wetlands (Zhang, 1995; Crozier et al., 1996; Boustany et al., 1997; Rybczyk, 1997) for model initialization, calibration and validation. Field data required for model calibration include annual above-ground production, some estimate of sediment accretion and, most critically, depth profiles of sediment bulk density, % organic matter, % mineral matter and % pore space. The model was first run for 100 simulated

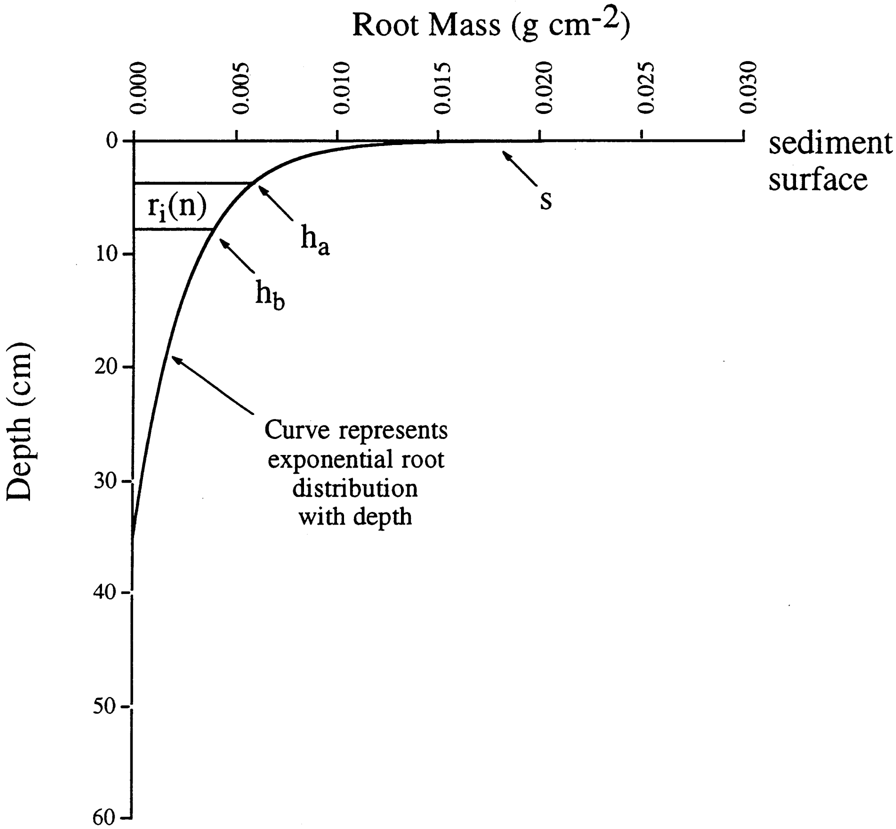

Fig. 3. Parameters describing simulated root distribution with depth. Root inputs to each sediment cohort (n) are shown as $r_i(n)$; h_a is the depth to the top of the cohort (n); h_b is the depth to the bottom of the cohort (n), s is the surface intercept of the exponential equation that describes root biomass distribution.

years, to generate a baseline simulated soil column and a 'cyber' space for roots to grow. Output from this 'pre-simulation' was then used to initialize the sediment column state variables for future simulations. For calibration, the model was run for an additional 23 simulated years with time zero representing 1970 (sediment cores, for calibration, were collected in 1993). Concurrent studies (Rybczyk, 1997) suggested that this site began experiencing periods of prolonged flooding during the early seventies. Therefore wetland elevation, relative to mean annual water level, was initialized at zero. We utilized a step-wise calibration procedure (Mitsch and Reeder, 1991). The primary production sub-model was calibrated first, as this model provided critical input to the sediment dynamics sub-model. After accurate productivity simulations were obtained, the sub-model was linked to the sediment dynamics sub-model. The sediment dynamics sub-model was calibrated with data obtained from sediment cores collected in the field and with measurement of accretion obtained by ^{137}Cs analysis (Rybczyk, 1997). A comparison of simulated and actual water levels after 20 years provided an additional point for calibration.

2.4. Validation

We performed a validation exercise using an independent data set collected from the bottomland hardwood ridge located adjacent to the treatment swamp. To run the validation simulation, only the forcing functions and input parameters were modified from the original treatment-site model (the model was re-initialized, but not re-calibrated). After re-initialization the model was run for 100

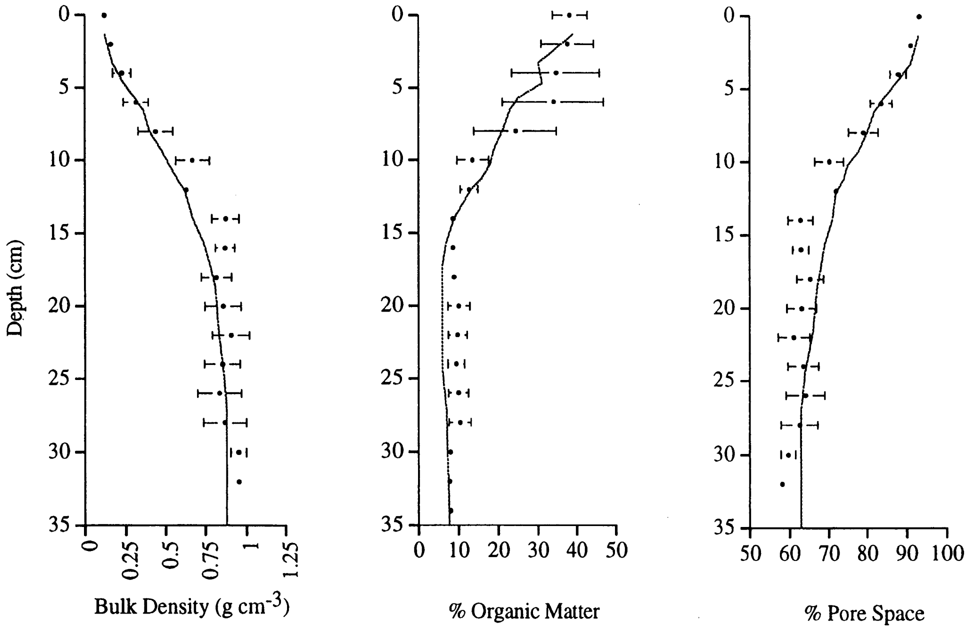

Fig. 4. Pointe au Chene swamp sediment profiles. Field measurements are shown as dots with standard error bars. Solid lines represent simulated results.

pre-simulation years, to build a soil column, and then from 1970 to 1993, in a manner similar to that described in the calibration section. Model-generated and real world results were compared visually using predicted versus observed plots for bulk density, pore space and % organic matter. This technique represents goodness of fit as vertical deviations from the 'perfect fit' line with a slope of one (Mayer and Butler, 1993). We also calculated the dimensionless statistic, EF (modeling efficiency) (Loague and Green, 1991), to relate the simulated to observed values at the ridge site. EF parallels the coefficient of determination (R^2) except that the lower bounds for the EF is negative infinity while the lower bounds for R^2 is zero. A perfect fit would be indicated by an EF of one and values less than zero would be indicative of a poor fit (Mayer and Butler, 1993).

Table 3
Calibration points for the Pointe au Chene simulation (for simulated and field year 1989)

Parameter	Simulated results	Field measurement ± se
^{137}Cs accretion rates	0.40 cm year^{-1}	0.44 ± 0.04 cm year^{-1}
Leaf litter	413 g m^{-2} year^{-1}	386.1 ± 18.2 g m^{-2} year^{-1}
Wood production	413 g m^{-2} year^{-1}	431.7 g m^{-2} year^{-1}
Tree standing crop	13.7 kg m^{-2}	13.6 kg m^{-2}
Relative wetland elevation	−16.8 cm	−17.4 ± 1.8 cm

2.5. Sensitivity analyses

We examined the sensitivity of relative wetland elevation to changes in the forcing functions and parameters that control wetland elevation (e.g. RSLR, primary production, soil compaction, decomposition and mineral inputs). Because changes in wetland elevation over time were not always linear, analyses were run for 5 and 50 year time periods. Parameters were varied by ± 5 and 50%. Relative sensitivity was calculated as: (% change in relative elevation/% change in parameter). Higher relative sensitivity values indicate greater sensitivity to a given parameter. It should be noted that a systematic analyses of this type may not reflect the probable or even possible range of variation for

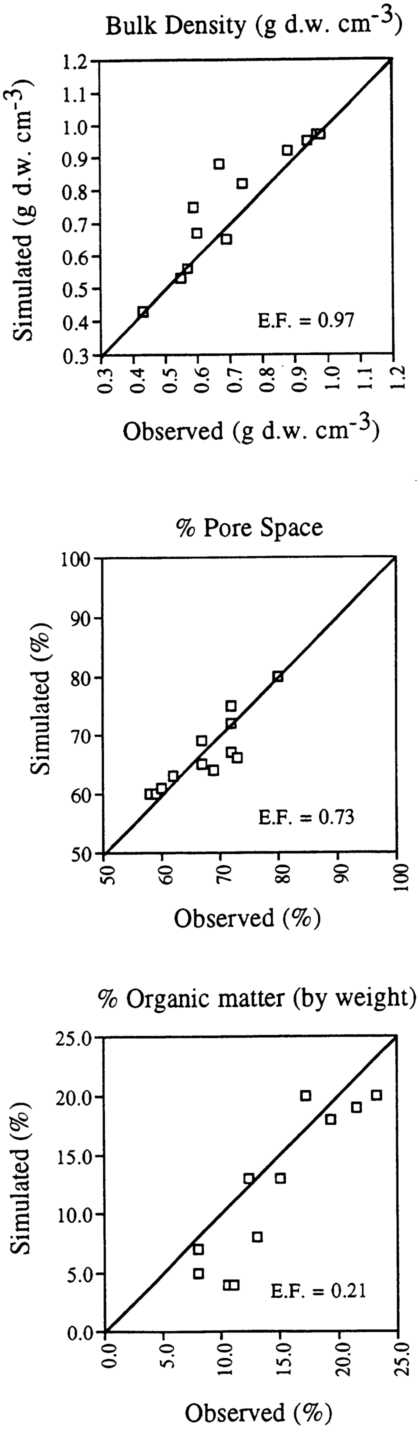

Fig. 5. Observed versus predicted plots for the Point au Chene ridge site validation exercise. E.F. = modeling efficiency (see text for explanation).

each parameter. However, once relative sensitivity is known, especially sensitive parameters can be re-examined within a known or realistic range.

2.6. Model applications

2.6.1. ESLR scenarios

To simulate the effect of various ESLR scenarios at the Pointe au Chene swamp under baseline conditions (no effluent), we used as forcing functions an estimated deep subsidence rate of 1.08 cm year^{-1} (Penland et al., 1988) and three Intergovernmental Panel on Climate Change (IPCC) predictions for ESLR: (1) 'Current trends' 15.6 cm in the next 100 years; (2) 'Best estimate' 48 cm in the next 100 years; and (3) 'Business as usual' 66 cm in the next 100 years (Gornitz, 1995). The ESLR function plus the deep subsidence represented RSLR in the wetland. For the first model run, simulations began in model-year 1970 under the 'current trends' ESLR scenario. Then in model-year 1990 the 'best estimate' ESLR forcing function was switched on for another 30 model-years (all three ESLR scenarios were assumed to be similar until 1990 (Gornitz, 1995)). The same procedure was followed for the remaining IPCC scenarios.

2.6.2. Subsidence rates

Numerous researchers have discussed the uncertainty surrounding estimations of the deep subsidence component of RSLR (Penland et al., 1988; Turner, 1991; Cahoon et al., 1995). For example, estimates for deep subsidence in the vicinity of the Pointe au Chene swamp range from 0.53 to 1.08 cm year^{-1} (Penland et al., 1988). To examine the overall effect of subsidence rates (over the reported range) on relative elevation, the model was run under baseline conditions (ESLR scenario = 'current conditions', no wastewater), only varying subsidence over its estimated range.

2.6.3. Wastewater applications

Field studies revealed that effluent additions did not increase tree production at the Pointe au Chene site (Rybczyk et al., 1996a; Rybczyk, 1997), but did increase the percent cover of the FAV (Zhang, 1995), therefore we used the model

Table 4
Relative sensitivity of wetland elevation to ±5 and 50% changes in model parameters[a]

Parameters	Description	±5%		±50%	
		5 years	50 years	5 years	50 years
A. RSLR parameters					
E_1	Current eustatic sea level rise	0.19	0.16	0.19	0.16
S	Local deep subsidence rate	1.49	1.21	1.46	1.21
B. Production parameters					
g_{max}	Maximum leaf net productivity	0.49	0.24	0.49	0.21
H_{func}	Flooding stress function	0.50	0.24	0.49	0.21
r	Root distribution constant	0.06	0.001	0.06	0.006
r_g	Net root production	0.27	0.08	0.27	0.08
v_{max}	FAV net productivity	0.02	0.007	0.03	0.07
w_2	Tree mortality rate	0.03	0.16	0.03	0.10
C. Soil compaction parameters					
p_k	Soil compaction constant	1.85	2.32	2.02	.19
p_m^b	Maximum pore space	22.42	2.62	—	—
p_x^b	Minimum pore space	9.84	1.27	—	—
D. Decomposition parameters					
f_1	% labile FAV	0.22	0.07	0.19	0.06
f_2	Fraction of labile leaf material	0.08	0.02	0.08	0.02
f_3	Fraction of labile root material	0.03	0.01	0.02	0.015
k_1	Decomposition rate of deep organics	0.18	0.02	0.02	0.02
k_2	Decomp. rate of labile roots	0.007	0.0004	0.008	0.0006
k_3	Decomposition rate of surface labile organic	0.01	0.001	0.01	0.002
k_4	Decomposition rate of refractory roots	0.05	0.05	0.05	0.05
k_5	Decomposition rate of surface refraction	0.03	0.007	0.02	0.007
E. Mineral inputs					
D	Maximum mineral inputs	0.31	0.23	0.20	0.22

[a] Because changes in wetland elevation over time are not always linear, analyses were run for five and fifty year time periods. Relative sensitivity was calculated as: (% change in relative elevation/%change in parameter)
[b] Values could not be varied ±%50 because doing so propagated logic errors in the compaction algorithm.

to examine the simulated effect of organic matter deposition from FAV on wetland elevation. In natural systems impacted by wastewater effluent, standing crops as high as 400 g d.w. m^{-2} have been observed (Ewel and Odum, 1984). To simulate the effect of wastewater effluent in the Pointe au Chene swamp, the model was run under natural condition settings from simulated year 1970–1992 (actual effluent additions began in 1992), and then FAV production values were switched so that peak standing crop reached 400 g d.w. m^{-2} for each of the remaining 28 years of the simulation (50 total years). Under these conditions simulated annual FAV production equalled 1434 g d.w. m^{-2}. The model was run under IPCC ESLR 'current conditions' scenario (a linear increase in eustatic sea level) so that the contribution from ESLR would remain constant throughout the simulation.

2.6.4. Mineral inputs

Using conservative forcing function estimates of ESLR (IPCC 'current conditions scenario') and subsidence (0.53 cm $year^{-1}$), and no wastewater effluent, we ran a series of 100 year simulations, starting in model year 1970, in which we varied only the addition of mineral sediment, to determine the amount of re-introduced sediment that would be required to keep pace with estimated rates of RSLR.

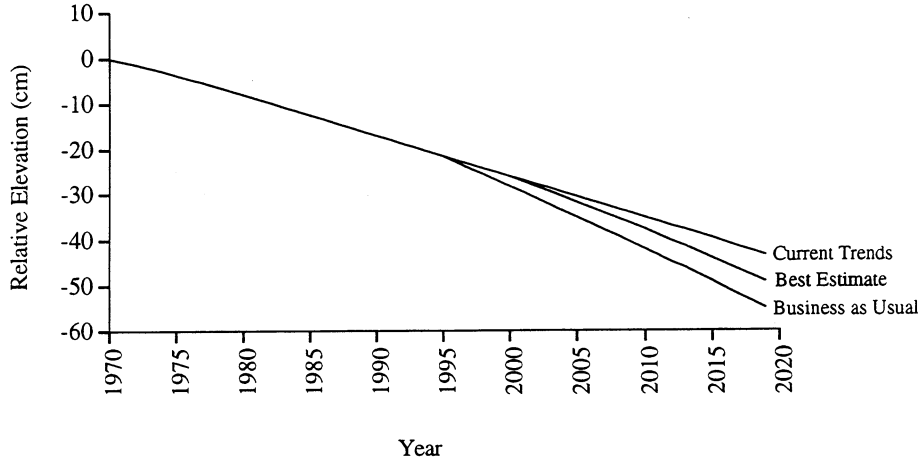

Fig. 6. Changes in relative wetland elevation under three IPCC ESLR scenarios (Gornitz, 1995): (A) 'Current trends' 15.6 cm in the next 100 years; (B) 'Best estimate' 48 cm in the next 100 years; and (C) 'Business as usual' 66 cm in the next 100 years. After a 50 year simulation, final relative wetland elevations for scenarios A, B and C are - − 42.3, − 49.18 and − 54.97 cm, respectively.

3. Results and discussion

3.1. Model description

3.1.1. Sediment dynamics sub-model

The sediment dynamics sub-model contains the following four state variables, each replicated once in each of 18 sediment cohort layers: (1) $Q(n)$, representing labile organic matter in cohort (n); (2) $B(n)$, representing refractory organic matter in cohort (n); (3) $M(n)$, representing mineral matter in cohort (n); and (4) $R(n)$, representing live root biomass in cohort (n). Sediment state variables are passed from cohort to cohort according to the simulated yearly time sequence; 1 (surface cohort), 1, 1, 1, 1, 2, 2, 2, 2, 2, 5, 5, 5, 5, 5, 10, 10 and 10 + (deepest cohort) years. Thus short term sediment processes, most of which occur near the sediment surface, are simulated within the cohorts with the shortest retention period. This allows for precise calibration and resolution of output. Deep sediment process, which for the most part occur at decades-long time scales, are simulated within the cohorts with the longest retention time. Transfer time between cohorts can be modified for systems with unusually slow or fast accretion.

The differential equations describing the changes in these state variables with time are shown in Table 2. Maximum mineral inputs are the only forcing functions in this sub-model, as other inputs are model generated. This sub-model simulates the decomposition of organic matter, the inputs of mineral matter, the distribution of root biomass and sediment compaction. These process are outlined below. Output includes the following sediment characteristics by cohort; bulk density, sediment height, organic and mineral matter mass and volume, pore space and live root mass (organic and mineral matter volumes are calculated based on the organic and mineral mass in each cohort and the specific density of organic and mineral matter (1.14 and 2.61 g cm^{-3}, respectively (DeLaune et al., 1983)).

3.1.1.1. Decomposition. The model separates all organic matter (roots, leaf litter and floating aquatic litter) into leachable and refractory pools, each with its own decay rate. Thus, the model is generic in the sense that by changing the original proportion of organic matter that is either liable or refractory, it can be used for a variety of wetland plant species. Additionally, decomposition rates for the surface cohort are separate from the decomposition rates for the rest of the cohorts (allowing for a distinction from leaf and root organic matter). Finally, there is a separate, depth-dependent decomposition rate for deep re-

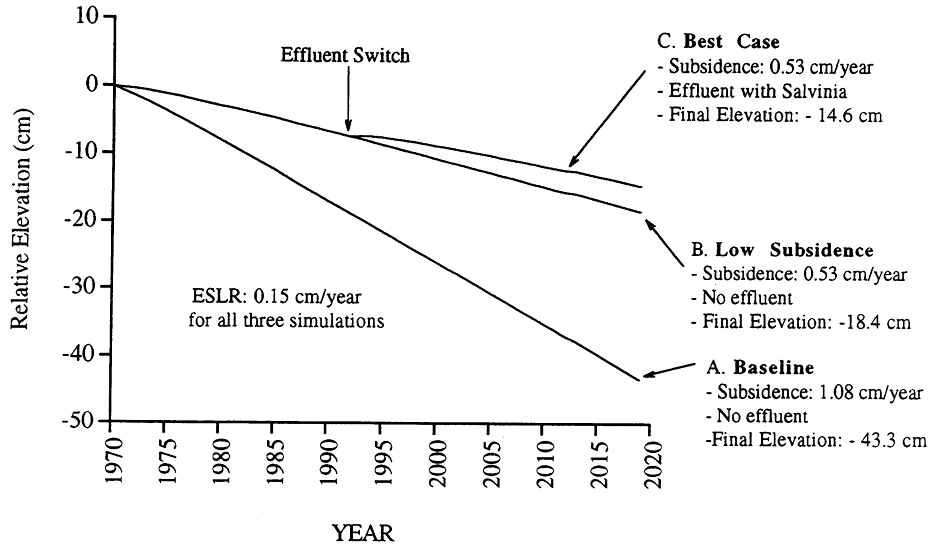

Fig. 7. Changes in relative wetland elevation under two estimates of deep subsidence (lines A and B). Also shown is a best case scenario (line C) where primary production is maximized and subsidence rates are minimized.

fractory material. Decomposition for each organic matter state variable in each cohort is described by a simple negative exponential model. For example the differential equation that describes the decomposition of labile organic matter on the sediment surface (cohort 1) is:

$$\mathrm{d}Q(n)/\mathrm{d}t = -k_3 Q(n) \tag{1}$$

where; Q, organic matter standing crop (g d.w. cm^{-2}) for cohort (n), in this case the surface cohort; and k_3, decay rate of surface labile organic matter(week^{-1}).

3.1.1.2. Mineral inputs. Previous models have simulated mineral inputs as a function of marsh elevation and tidal range (French, 1993; Callaway et al., 1996). Because there are no measurable tides at the Pointe au Chene wetland, mineral inputs are simulated as a simple linear function of wetland elevation. Mineral inputs are maximized when relative wetland elevation is below mean water level and minimized as elevation increases above mean water levels. Maximum mineral inputs, D, were estimated from accretion and soil core analysis data obtained in the field and are entered into the model as a forcing function.

3.1.1.3. Root distribution. Although root production is simulated in the primary productivity submodel, root biomass is distributed to the sediment cohorts in the sediment dynamics sub-model. We used an adaptation of the distribution algorithm, originally developed by Morris and Bowden, (1986), where root biomass is assumed to be greatest near the surface and decreases exponentially with depth (Fig. 3). The fraction of the total root biomass allocated to each cohort is calculated as:

$$r_{\mathrm{i}}(n) = s[\mathrm{e}^{(-rh_{\mathrm{b}})} - \mathrm{e}^{(-rh_{\mathrm{a}})}]/-r \tag{2}$$

where: $r_{\mathrm{i}}(n)$, root input to each sediment cohort (n) (g d.w. cm^{-2}); s, weight of roots at sediment surface (g d.w. cm^{-2}); r, root depth distribution constant (cm^{-1}); h_{b}, depth to the bottom of the cohort (n) (cm); and h_{a}, depth to the top of the cohort (n) (cm).

The variable, s, which must be known in order to use Eq. (2) to partition roots to the sediment column, is the surface intercept of the exponential equation that describes root distribution:

$$R_t/10\,000 = s(\mathrm{e}^{(-r\ \mathrm{depth})}) \tag{3}$$

where: R_t, total root biomass (g d.w. m^{-2}); and depth, depth of rooting zone (cm).

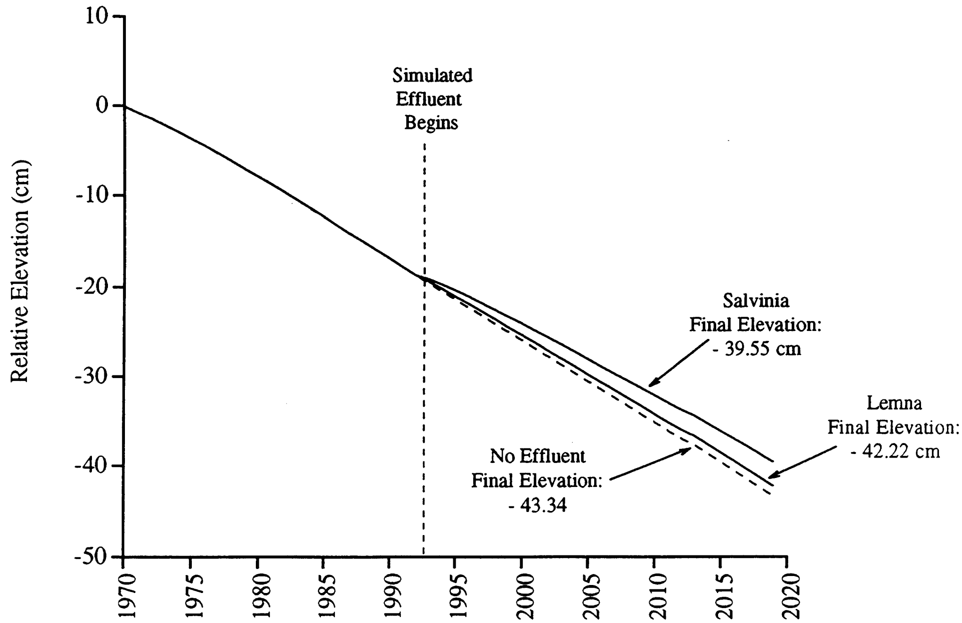

Fig. 8. The effect of wastewater effluent and floating aquatic vegetation on relative wetland elevation. ESLR is fixed at 0.15 cm year^{-1}. Effluent additions begin in simulated year 1992.

To solve for s, Eq. (2) can be re-arranged as:

$$s = r_i(n)/[[e^{(-rh_b)} - e^{(-rh_a)}]/-r] \quad (4)$$

Then, if total root biomass is known (as it is in this model), we let $r_i(n)$ represent the entire sediment column, from the surface to a depth of infinity, so that $r_i(n)$ equals total root biomass. In this special case, the expression, $e^{(-rh_b)}$, approaches 0 and, $e^{(-rh_a)}$, approaches 1 for any value of r. Therefore, Eq. (4) simplifies to:

$$s = R_t/(-1/-r) \quad (5)$$

3.1.1.4. Sediment compaction. Simulated soil compaction is a function of organic matter decomposition, simulated separately, and the reduction of sediment pore space (primary consolidation) (Penland and Ramsey, 1990). Callaway et al. (1996) simulated the compaction of pore space as an asymptotic decrease with depth, bounded by preset minimum and maximum pore space values. We use a modified version of Callaway's algorithm, where the decrease in pore space for a give cohort ($p_s(n)$) is a function of the mass of material above it:

$$p_s(n) = p_m + ((p_x - p_m)C_{func}(n)) \quad (6)$$

where: $p_s(n)$, fraction of pore space in cohort (n) (unitless from 0 TO 1); p_m, minimum pore space for the entire sediment column (unitless from 0 to 1); p_x, maximum pore space for the entire sediment column (unitless from 0 to 1); and

$$C_{func}(n) = 1 - (g(n)/(p_k + g(n)) \text{ (unitless)}. \quad (7)$$

The parameter, $C_{func}(n)$, describes a Michaelis–Menten type reduction in pore space where: $g(n)$, mass of sediment overlying cohort (n) (g cm^{-2}); and p_k, half saturation compaction constant (cm^3 g^{-1}).

The constants p_m, p_x and p_k are derived from site specific soil cores collected to a depth of ≈ 40 cm.

3.1.2. Primary productivity sub-model

This sub-model simulates the production of in-situ organic matter, which is then allocated to the sediment dynamics sub-model, either on the surface, as litter, or within the simulated sediment soil column as root biomass. Organic matter is separated into three state variables associated with tree growth: L (tree leaf biomass); W (woody stem biomass); and R_t (below-ground biomass); and one state variable, V, representing FAV biomass.

3.1.2.1. Above-ground tree biomass. Changes in leaf biomass standing crop are calculated as:

$$\mathrm{d}(L)/\mathrm{d}t = (g_{\max} H_{\mathrm{func}}) - (L f_5) \tag{8}$$

where: L, leaf biomass (g d.w. m^{-2}); $g_{\max}$, maximum net leaf productivity (g d.w. m^{-2} week^{-1}); H_{func}, wetland flooding function that modifies $g_{\max}$ (unitless from 0 to 1); and f_5, leaf litter production rate (week^{-1}).

Simulated net tree leaf production is limited by the amount of live above-ground wood biomass available for support. To estimate the maximum leaf production ($g_{\max}$) for a given amount of wood biomass, we examined 87 forest productivity and structure data sets, collected from 37 forested wetlands in coastal Louisiana and South Carolina (unpublished data), that contained measurements of both annual leaf productivity and above-ground wood biomass. These 87 data sets were grouped in into 15 size classes according to the above-ground wood biomass (0–5 kg m^{-2}, $>5- =10$ kg m^{-2}... $>70- =75$ kg m^{-2}). Then, using only the one data set within each size class with the greatest annual leaf productivity, above-ground wood biomass was regressed (second order polynomial) against leaf productivity to yield an estimate of weekly maximum net leaf production where:

$$g_{\max} = ((11.0 + (37.8W/1000)) - (0.267(W/1000)^2))/52 \tag{9}$$

where: W, standing live wood biomass (g d.w. m^{-2}).

Maximum leaf production ($g_{\max}$) is limited by the water level function (H_{func}) originally developed by for southeastern US forested wetlands (Phipps, 1979):

$$H_{\mathrm{func}} = 1 - 0.5511\,(h_{\mathrm{w}} - w_1)^2 \tag{10}$$

where: h_{w}, water table depth (m); and w_1, optimum water table depth for a given species (m).

Phipps generalized equation was intended for water levels below the surface. If simulated water levels were above the surface, then leaf productivity was held constant at 78% of $g_{\max}$ (Conner and Day, 1989). Leaf litter production was simulated a function of time and the amount of plant biomass present, and was calibrated to reflect field measurements.

More than 7 years of baseline field measurements from the Thibodaux site showed that wood biomass production was roughly equivalent to leaf biomass production, therefore changes in stem biomass standing crop are modeled as:

$$\mathrm{d}(W)/\mathrm{d}t = (g_{\max} H_{\mathrm{func}}) - (W w_2) \tag{11}$$

where: W, stem biomass (g d.w. m^{-2}); and w_2, mortality rate derived from field data (week^{-1}).

3.1.2.2. Root biomass. Mitsch and Ewel (1979) described a forested wetland stress-subsidy hypothesis that suggested that too much or too little water reduces total net primary productivity. However, most of the evidence for this hypothesis

Table 5
Relationship between mineral inputs and accretion rates given two different initial elevations

Initial conditions for both scenarios:		
Above-ground (leaf and wood)	838.0 g m^{-2} year^{-1}	
Below-ground root production	427.4 g m^{-2} year^{-1}	
F.A.V. production	467.0 g m^{-2} year^{-1}	
Total production	1732.4 g m^{-2} year^{-1}	
Subsidence rate	0.53 cm year^{-1}	
Mineral inputs	3000 g m^{-2} year^{-1}	
	Scenario A	Scenario B
Initial relative elevation:	0 cm	−10 cm
After 100 year simulation		
Above-ground (leaf and wood)	1588.4 g m^{-2} year^{-1}	791.5 g m^{-2} year^{-1}
Below-ground root production	1588.4 g m^{-2} year^{-1}	403.6 g m^{-2} year^{-1}
F.A.V. production	0 g m^{-2} year^{-1}	467.0 g m^{-2} year^{-1}
Total production	3176.8 g m^{-2} year^{-1}	1662.1 g m^{-2} year^{-1}
Final relative elevation	1.2 cm	−13.3
Accretion rate	0.72 cm year^{-1}	0.59 cm year^{-1}

was derived from observations of above-ground production only. Recent studies have suggested that less flooded sites are actually the most productive when below-ground productivity is also considered (Megonigal and Day, 1992; Day and Megonigal, 1993). To reflect these recent findings, change in root biomass is simulated as:

$$d(R_t)/dt = r_g - (R_t f_4) \tag{12}$$

where: R_t, live root biomass (g d.w. m^{-2}); $r_g = ((g_{max} H_{func})2)$ if water levels are ≤ 0 cm or $((g_{max} H_{func})0.51)$ if water levels are >0 cm; and f_4, root litter rate (week^{-1}).

3.1.2.3. Floating aquatic vegetation. Sklar (1983) developed an aquatic materials flow model of a cypress tupelo forest in Louisiana that described primary production of *Lemna* sp. as a function of its own biomass. We used a similar approach to simulate floating aquatic biomass, where:

$$d(V)/dt = ((v_{max} V_{func} T_{func})V) - (f_6 V) \tag{13}$$

where: V, floating aquatic macrophyte standing crop (g d.w. m^{-2}); v_{max}, maximum net primary production rate (week^{-1}); V_{func}, space limitation function (unitless from 0 to 1); T_{func}, temperature limitation function; and f_6 = FAV litter production rate (week^{-1}).

V_{func} describes the exponential decrease of this unitless multiplier with increasing macrophyte biomass, where:

$$V_{func} = e^{(v_k V)} \tag{14}$$

where: v_k = exponential crowding coefficient (g d.w.$^{-1}$ m^{-2}); T_{func} describes a piece wise linear temperature function (Bowie, 1985) of the form:

$$T_{func} = ((1/(t_{opt} - t_{min}))\mathrm{T})\ (t_{min}/(t_{opt} - t_{min}))$$
$$\text{if temp.} > 13, \text{else } 0 \tag{15}$$

where: T, mean weekly temperature (°C); t_{opt}, optimum temperature for growth (°C); and t_{min}, minimum temperature for growth (°C).

3.1.3. Relative elevation sub-model

Previous wetland accretion/subsidence models (Callaway et al., 1996) have focused on intertidal marshes and have modeled wetland elevation relative to mean sea level. Although the Pointe au Chene wetland is not intertidal, the hydroperiod is influenced by RSLR (Conner et al., 1989). In 1989, elevation at the treatment sites was measured at ≈ 76 cm above sea level (Conner et al., 1989). However, during the pretreatment years 1989–1990, when precipitation was near normal, mean annual water depths in the Pointe au Chene treatment site were 17.4 ± 1.8 and 19.5 ± 2.4 cm, respectively, and the site was continually flooded during this period. Therefore, wetland elevation is simulated relative to mean annual water depth and not mean sea level. This operates under the assumption that precipitation is constant from year to year and, in effect, adds a 'correction factor' to mean sea level to account for local hydrologic conditions.

Wetland elevation, relative to mean annual water level, is simulated as the balance between ESLR plus deep subsidence, both forcing functions, and the total net accretion (mineral matter accretion + organic matter accretion-shallow subsidence) of organic and inorganic material in the sediment column. Shallow subsidence is modeled explicitly with the decomposition and compaction functions described in the sediment dynamics sub model.

3.2. Calibration and initialization

The simulated soil column profile closely matched the profile obtained from cores collected at the Thibodaux site during 1993 (Fig. 4). Simulated and actual production, accretion and elevation values were in close agreement. Simulated litter and wood production were within 6% of field measurements (Table 3). Model generated long term accretion rates were within one standard error of accretion rates measured in the field using ^{137}Cs analyses (Rybczyk, 1997). Simulated water depths (16.8 cm) were also within one standard deviation of actual values.

3.3. Validation

Observed versus simulated results were in close agreement and all EF values were above zero (Fig. 5). Deviations from the 'perfect fit' line in the bulk density and % pore space plots were attributed to a lens of low bulk density material occurring between 6 and 8 cm in the soil cores that the model did not simulate. Additionally, the model consistently underestimated the % organic matter in the soil column by 2–5% (Fig. 5). This was probably due to inaccurate estimates of decomposition rates during initialization. The simulated long term accretion rate (30 years) was 0.95 mm year^{-1} compared to the mean observed accretion rate of 1.0 ± 0.3 mm ($\pm$ se) year^{-1}. Simulated above-ground production (leaf plus wood), was 823 g d.w. m^{-2} year^{-1}, compared to an observed total above-ground production of 816 g d.w. m^{-2} year^{-1} during 1990.

3.4. Sensitivity analyses

Simulated wetland elevation was most sensitive to the soil compaction functions p_x (maximum pore space), p_m (minimum pore space) and p_k (the soil compaction constant) (Table 4). Sensitivity to these functions are an artifact of the compaction algorithm. Changes in these parameters trigger an immediate re-calculation of the compaction algorithm that affects the entire simulated soil column. No other parameters have an instantaneous effect on the entire sediment column. Sensitivity to these parameters decrease with time.

The uncertainties surrounding estimated rates of ESLR in the next century, and current rates of deep subsidence in the coastal zone, have been well documented (Turner, 1991; Gornitz, 1995) and this analyses revealed that wetland elevation was relatively sensitive to the forcing functions controlling these processes (ESLR and deep subsidence, respectively) (Table 4). The results from a series of simulations where these functions are varied over their predicted ranges are shown in the Model Applications sections.

Elevation was also relatively sensitive to the functions that control organic matter production (H_{func}, g_{max} and r_g) (Table 4). These functions would tend to increase elevation given that they were stimulated by effluent additions. Alternatively, some studies have shown that effluent amendments could also stimulate organic matter decomposition (Rybczyk et al., 1996b), negating any elevation increase due to increased production. However, this analyses indicated that wetland elevation was relatively insensitive to the parameters that control the rates of organic matter decomposition (Table 4). Finally, wetland elevation was also shown to be relatively sensitive to changes in mineral inputs (D) (Table 4). In the following section, we also examine the potential for mineral supplements to counter the effects of RSLR under various scenarios.

3.5. Model applications

3.5.1. ESLR scenarios

The simulations revealed that, under all three sea level rise scenarios and without any intervention, relative wetland elevation would decrease and remain below zero for the entire 50-year simulation (Fig. 6). After 50 years the difference in relative wetland elevation between the highest elevation (current trends scenario) and the lowest (business as usual scenario) was 11.67 cm. It was unnecessary to run the model for additional years because mineral inputs reached the programmed maximum early in the simulation, autogenic organic matter production continued to decrease and no other process was programmed that would reverse the simulated trend towards decreasing elevation.

3.5.2. Subsidence rates

Starting in model year 1970 (with an initial relative elevation of 0), relative elevation decreased to -18.4 cm (18.4 cm below mean annual water levels) after 50 years when subsidence rates equalled the minimum 0.53 cm year^{-1}, and decreased to -43.3 cm at the maximum subsidence rate of 1.08 cm year^{-1} (Fig. 7). Since mean ($\pm$ se) monthly water levels at the Pointe au Chene site averaged 19.5 ± 0.24 cm in 1990, this would suggest that the actual rates of deep subsidence are closer to highest estimates (1.08 cm year^{-1}) made by Penland et al. (1988). However, under any subsidence scenario, simulated rates of accretion did not keep pace with simulated rates of RSLR.

3.5.3. Wastewater applications

Baseline (non-effluent) simulated peak standing crop and annual production for FAV were 132.7 and 467.0 g d.w. m^{-2} $year^{-1}$, respectively, similar to both Sklar's simulated production value of 475.6 g d.w. m^{-2} $year^{-1}$ and the estimate of Porath et al. (1979) of *Lemna* sp. production in natural systems (474.4 g d.w. m^{-2} $year^{-1}$). Simulated peak standing crop measurements were also in close agreement with observed values of 167.0 g d.w. m^{-2} (Sklar, 1983). Increasing the annual rate of production (to 1434 g d.w. m^{-2}) after 1992 to simulate the effect of wastewater on productivity had little effect on elevation, increasing the relative elevation only 1.12 cm over the baseline simulation (*Lemna* sp. without effluent) in the remaining 28 years (Fig. 8). This was primarily due to the rapid decomposition rate for *Lemna* sp., as 80% of the litter is distributed to labile organic pool, which disappears quickly from the simulated soil column.

It was noted in the field that, after effluent additions began in 1992, the floating aquatic fern *Salvinia* sp. replaced *Lemna* sp. as the dominant FAV in the treatment site (Zhang, 1995). Preliminary decomposition experiments conducted at the site revealed that, unlike the primarily labile *Lemna* sp., 50% of *Salvinia* sp. consisted of refractory material (unpublished data). In addition, supplemental field experiments revealed that the decomposition rate of *Salvinia* sp. was markedly slower than decomposition rates for *Lemna* sp. (unpublished data). To simulate the effect of this species shift on relative wetland elevation, we followed the same scenario described for the effluent stimulated *Lemna* sp. simulation, except that after 1992, the constant that determines the ratio of labile to refractory material in FAV leaves (f_2) was changed from 0.8 to 0.5 and the refractory decomposition rate parameter, k_5, was switched from 0.028 to 0.0007 $week^{-1}$. Under this scenario, final simulated wetland elevation after 50 years was -39.55 cm, 2.67 cm higher than the '*Lemna* with effluent' simulation and 3.79 cm higher than the '*Lemna* without effluent ' simulation (Fig. 8).

The effect of these simulated changes on long-term accretion rates (equivalent to ^{137}Cs measurements in the field) were also examined. From 1970 to 1992, the baseline simulated long term accretion rate was 0.35 cm $year^{-1}$. Under the effluent stimulated *Lemna* sp. scenario, long term accretion rates increased to 0.36 cm $year^{-1}$ over the remaining 28 years. For the equivalent time period, long-term accretion rates in the *Salvinia* sp. simulation increased to 0.46 cm $year^{-1}$.

None of the individual simulation scenarios suggested that relative wetland elevation would keep pace with RSLR rates. Therefore, we asked the question, 'Is their any reasonable combination of simulation scenarios that would lead to a stable or even increasing wetland elevation relative to water levels?' To answer this, we programmed a 'best case' scenario where subsidence rates were initialized at the minimum rate estimated by Penland et al. (1988) (0.53 cm $year^{-1}$), ESLR rates were fixed at the most conservative IPCC estimate ('current conditions'), and simulated organic matter production included effluent stimulated *Salvinia* sp. production. Even under these optimal conditions, relative elevation decreased, from an initial value of 0 cm to -14.6 cm by 2020 (Fig. 7).

3.5.4. Mineral inputs

The previous simulations revealed that biologically reasonable increases in primary productivity alone would not balance accretion deficits at the site. How much supplemental mineral sediments then, would be required to balance the observed deficits? Given an initial elevation of 0 cm, the wetland required an additional 3000 g m^{-2} $year^{-1}$ of mineral sediment to maintain a stable elevation over 100 years. However, given an initial elevation of -5 cm, the wetland required an additional 4000 g m^{-2} $year^{-1}$ to maintain that initial negative elevation, and 4500 g m^{-2} $year^{-1}$ to approach 0 cm elevation. In effect, when initial elevations were 0 cm or higher, overall accretion rates were higher than when initial elevations were below 0 cm, even though mineral inputs were equal (Table 5).

These model-based predictions parallel results from forested wetlands in the southeast. Above-ground production is low in permanently flooded wetlands relative to those that are seasonally inundated (Conner and Day, 1988). In addition, Day and Megonigal (1993) have shown that below-ground production and root standing crop biomass are dramatically reduced in permanently flooded forested wetlands. Therefore, there would be little or no autogenic response to the addition of mineral sediments until a critical elevation is reached at which there is some relief from flooding stress during the growing season. This critical point will vary by species (Phipps, 1979), and by year, depending upon local hydrologic conditions (for the purposes of this simulation we have imposed the 0 cm elevation as this critical point). However, once a critical elevation is obtained, ecosystem response can include increased above and below-ground production, seedling establishment and forest regeneration.

Management implications of these findings are important in Louisiana and other states that are considering sediment diversions or additions as a form of wetland restoration; adding 5 cm of pre-compacted mineral sediment to a wetland that is permanently flooded with 30 cm of water will only raise the elevation 5 cm. However, the same addition to a wetland of higher elevation, or one that is not permanently flooded during the growing season, can result in a greater increase in elevation. The set of simulations shown in Table 5 illustrate this point. Under Scenario A, the addition of 3000 g m^{-2} year^{-1} of sediments, in combination with increased organic matter production, maintains the wetland above the critical elevation. Long term accretion rates equal 0.72 cm year^{-1} for a return of 0.24 cm year^{-1} of accretion for every kg m^{-2} of sediment delivered. In the permanently flooded wetland, represented by scenario B, the same 3000 g resulted in a long term accretion rate of only 0.59 cm year^{-1} for a return of 0.19 cm year^{-1} of accretion for every kg of sediment delivered.

4. Conclusions

To simulate the response of wetland elevation to ESLR, deep subsidence, wastewater effluent and mineral inputs, an integrated wetland elevation model was developed that links a primary production and sediment column sub-model to an elevation sub-model. The advantages of this model over traditional accretion deficit calculations for predicting wetland sustainability are that mineral inputs and productivity are feedback functions of elevation and that the model integrates the effects of long-term processes such as compaction and decomposition.

Analyses revealed that wetland elevation was relatively sensitive to organic matter production, mineral inputs and rates of subsidence and ESLR. Of particular note, elevation was found to be more sensitive to rates of deep subsidence, a forcing function, than to organic matter production and inputs of mineral matter. This suggests that accurate estimates of deep subsidence are critical for predicting wetland sustainability given various management scenarios involving any sediment manipulation strategy.

A series of simulations suggested that at the Pointe au Chene swamp, without additional mineral sediment supplements, even under the best-case scenario (low rates of deep subsidence, IPCC 'current conditions' ESLR scenario and effluent stimulated organic matter production) wetland sediment accretion would not be enough to keep pace with current or predicted rates of RSLR. We also found that, because of an autogenic primary production response, mineral supplements were more effective at maintaining relative elevation when applied before a wetland was permanently inundated.

This model represents a first step towards integrating the many processes that affect wetland elevation; the next generation of wetland elevation models will need further refinements to increase our understanding of sea level rise impacts and various management scenarios. First, new models should incorporate landscape-level processes. Spatializing model processes will allow us to estimate landscape-level changes in sediment accumulation and to evaluate whether sufficient sediment inputs are available to accommodate RSLR (Costanza et al., 1990). Second, we need to incorporate a more mechanistic simulation of mineral sediment inputs. French (1993) has modeled mineral sediment dynamics successfully, and the combination of his model with our models of below-ground processes would substantially strengthen overall understanding of wetland sediment dynamics. A third improvement will be to develop better links between elevation or vegetation type and sediment processes (including production and decomposition). This will allow for more realistic changes in biological processes as a function of elevation. Fourth, models such of these should be linked to site specific, spatially explicit hydrologic models. Such a coupling would allow for the simulation, in time or space, of 'drawdown' windows that would allow for forest re-generation. Finally, many data gaps in our knowledge of wetland below-ground processes still exist (particularly sediment compaction, root production as a function of elevation, and long-term decomposition rates).

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