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## The Use of Secondarily Treated Wastewater Effluent for Forested Wetland Restoration in a Subsiding Coastal Zone.

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THE USE OF SECONDARILY TREATED WASTEWATER EFFLUENT FOR  
FORESTED WETLAND RESTORATION IN A SUBSIDING COASTAL ZONE

A Dissertation

Submitted to the Graduate Faculty of the  
Louisiana State University and  
Agricultural and Mechanical College  
in partial fulfillment of the  
requirements for the degree of  
Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by

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May 1997



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## ABSTRACT

Insufficient sedimentation, coupled with high rates of relative sea-level rise (RSLR), are two important factors contributing to wetland loss in coastal Louisiana. I hypothesized that adding nutrient rich, secondarily treated, wastewater effluent to subsiding coastal 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 RSLR (subsidence plus eustatic sea-level rise). To test this hypothesis, I measured processes affecting wetland elevation including, organic matter decomposition, sediment accretion, aboveground primary production, and, plant tissue nutrient (N, P, K, Ca, Mg, Fe) concentrations, in a coastal forested wetland receiving wastewater effluent, and in an adjacent control site, both before and after effluent applications began. A Before-After-Control-Impact statistical analysis revealed that neither aboveground tree production nor annual rates of decomposition were affected by wastewater effluent. However, because of increased floating aquatic vegetation production in the treatment site, rates of sediment accretion increased significantly after wastewater applications began (from 7.8 to 11.4 mm/yr), and approached the estimated rate of RSLR (12.0 mm/yr). No corresponding increase was observed in the control site. In general, N, P and K green leaf concentrations increased in the treatment site, with respect to the control, after effluent applications began. A wetland elevation ecosystem model, that incorporated elevation feedback mechanisms and simulated above and belowground primary production, sediment dynamics (decomposition, compaction and accretion) and mineral inputs over decades, was developed to examine the long term response of wetlands to increasing rates of RSLR, and to predict the effect of effluent additions on elevation. Model-generated sediment height was balanced with eustatic sea-level rise and deep subsidence, both

forcing functions, to determine wetland elevation relative to sea-level. Data gathered as part of the field study were used for calibration and validation. Simulations revealed that wetland elevation was more sensitive to the uncertainty surrounding estimates of eustatic sea-level rise and deep subsidence than in possible effluent-related changes in autogenic processes, such as decomposition and primary production.

## **CHAPTER 1**

### **INTRODUCTION**

#### **BACKGROUND AND OBJECTIVES**

The concept of using wetlands as tertiary processors of wastewater effluent has received much attention in the last three decades and there is a sizable body of literature pertaining to wetland wastewater systems. Much of the early work on wastewater applications to forested wetland systems of the southeast (specifically cypress swamps) was conducted at the University of Florida's Center for Wetland Resources in the 1970's. Ewel and Odum (1984), documented many of the ecosystems studies and models that were developed during this period. Comprehensive case histories of other wetland wastewater treatment systems, and reviews of wetland wastewater treatment in general, have been provided by Godfrey (1985), Reddy and Smith (1987), Hammer (1989), Kadlec and Knight (1996), and Knight (1994).

The potential benefits of using wetlands to process secondarily treated, non-toxic municipal wastewater effluent are threefold. First, given proper loading rates, wetlands can be long term sinks and transformers for the inorganic nutrients that are characteristic of secondarily treated wastewater (Godfrey 1985; Kadlec and Alvord 1989). Second, the cost for the development and maintenance of natural treatment systems can be considerably less than traditional tertiary treatment systems (Breux 1992; Breux and Day 1994; Breux et al. 1995). Finally, the use of wetland wastewater treatment systems allow for the potential enhancement and restoration of the functional attributes associated with wetlands (e.g. groundwater re-charge, flood control, biological productivity) (Kadlec and Knight 1996; Rybczyk et al. 1996a).



The focus of this dissertation is on the last of these three benefits; the use of wastewater effluent for wetland enhancement and restoration.

Hammer and Bastian (1989) state that the valuable functions of natural wetlands, in terms of biological productivity, fish and wildlife habitat, flood protection and water quality improvement, are poorly understood. They reason that the possible benefits derived from these natural effluent processor systems are not worth the partial or complete loss of these functions that might be incurred by adding wastewater to wetlands. These concerns are certainly legitimate for healthy wetland systems. Additionally, in many regions of the country, natural wetlands are unavailable for use as tertiary processors. This is not the case for Louisiana, which has over 2.8 million hectares of wetlands (Louisiana Dept. of Culture, Recreation and Tourism 1988). More importantly though, most of these coastal wetlands have been hydrologically altered, and are isolated from the alluvial systems responsible for their creation (Boesch 1994). This makes these wetlands especially vulnerable to the high rates of relative sea level rise (RSLR: subsidence plus eustatic sea level rise) associated with deltaic systems (Penland 1988) and to predicted increases in eustatic sea level rise (Gornitz 1995).

Wetlands have been shown to 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). In the past, seasonal overbank flooding of the Mississippi river deposited large amounts of sediments into the interdistributary wetlands of the delta plain. Not only did these floods provide an allogenic source of mineral sediments, which contributed directly to vertical accretion, but the nutrients associated with these sediments promoted vertical accretion through increased autogenic organic matter production and deposition, and the formation of soil through increased root growth. 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, resulting in vertical accretion deficits ( $RSLR > \text{accretion}$ ) throughout the coastal region.

Contributing further to the problem of vertical accretion deficits, many wetlands in the deltaic region have been hydrologically isolated from surrounding marshes, swamps and bayous due to an exponential increase in the construction of canals and spoil banks during the past century (Turner and Cordes 1987). In addition to impeding drainage and, in many cases, physically impounding wetlands, these spoil banks also prevent the overland flow of sediments and nutrients into coastal wetland forests.

The total acreage of swamp forest in the Louisiana coastal zone has decreased by 50% from 1956 to 1990 (Barras et al. 1994). Furthermore it has been predicted that increased rates of eustatic sea level rise could eliminate most of the remaining forested wetlands (Delaune et al. 1987). In the wetland forests of southeastern Louisiana, Conner and Day (1988) estimated vertical accretion deficits ranging from 2.5 to 10.8  $\text{mm yr}^{-1}$ , which leads directly to increased flooding duration, frequency and intensity. Productivity decreases observed in these wetlands may be attributed to either the direct physio-chemical effects of flooding (i.e. anoxia or toxicity due to the reduced species of S and Fe), flood related nutrient limitations (i.e. denitrification or the inhibition of mineralization), nutrient limitations due to a reduction in allogenic nutrient supplies, lack of regeneration, or most likely, some combination of these factors (Mitsch and Gosselink 1986).

Recent efforts to restore wetlands in the subsiding delta region have focused on attempts to decrease vertical accretion deficits by either physically adding sediments to wetlands or by installing sediment trapping mechanisms (i.e. sediment fences), thus increasing elevation and relieving the physio-chemical flooding stress (Boesch et al 1994). Day et al. (1992) proposed an alternate restoration strategy by hypothesizing that adding nutrient rich secondarily treated wastewater to hydrologically isolated and

subsiding wetlands could promote vertical accretion through increased organic matter production and deposition. This hypothesis operates under the assumption that the observed decreases in productivity in these systems are due not only to extended periods of inundation, but also to decreased nutrient inputs. Many studies have shown that nutrient rich wastewater does stimulate productivity in wetlands (Odum et al. 1975; Turner et al. 1976; Mudroch and Copobianco 1979; Bayley et al. 1985; Knight 1992; Hesse 1996), and a few have documented increases in rates of sediment accumulation (Craft and Richardson 1993).

In practice, wetland elevation is the function of numerous processes other than organic matter production and deposition, including mineral matter accumulation, organic matter decomposition, sediment compaction, deep subsidence and eustatic sea level rise (Figure 1.1). While nutrient additions may stimulate productivity, they may also have a negative effect on elevation by increasing the decomposition rate of organic matter (Rybczyk et al. 1996b). The objective of this dissertation was to use an integrated field study and modeling approach to examine the effect of secondarily treated effluent on the processes that affect wetland elevation in a selected coastal forested wetland, and to determine if effluent could stimulate vertical accretion enough to offset observed rates of RSLR in the coastal zone.

Chapter two of this dissertation provides a review and analysis of the recent literature pertaining to the effect of nutrient amendments on decomposition in wetland ecosystems. Chapter three describes the results of a decomposition field study in a forested wetland receiving wastewater effluent. Chapters four and five describe the results of sediment accretion and forest productivity field experiments, respectively, in the same wetland. Finally, chapter six documents the program and results of an ecosystem model designed to examine the response of wetland elevation to effluent related changes in the processes that affect wetland elevation (Figure 1.1), determine

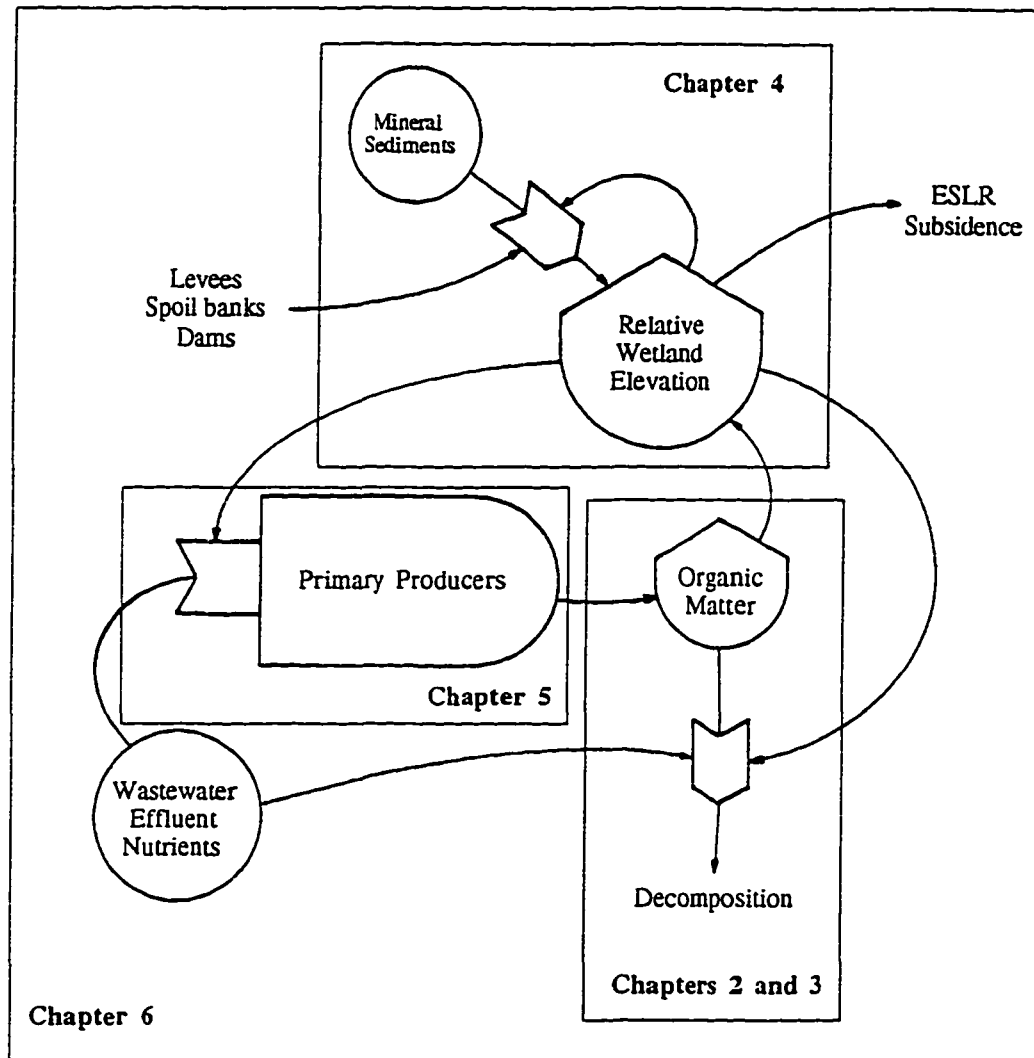


Figure 1.1 Processes affecting wetland elevation relative to sea level.

the relative sensitivity of wetland elevation to these processes and predict the long term effects of effluent addition on wetland elevation. Data collected as part of the field studies outlined in chapters three, four and five, were used to initialize, calibrate and validate the model.

## **SITE DESCRIPTION**

In 1992, as part of its tertiary treatment program, the city of Thibodaux, Louisiana began applying secondarily treated municipal wastewater to the Pointe au Chene swamp, a hydrologically isolated and permanently flooded forested wetland, located on the backslope of the abandoned Bayou Lafourche distributary in the Terrebonne Basin. The site consists of two almost continuously flooded forested wetlands, separated by a bottomland hardwood ridge, oriented northeast to southwest, all confined within a 1425 ha basin. Complete site descriptions, and maps, are provided in chapters three and four. The 231 ha forested wetland site on the west side of the ridge was designated as the treatment site and the wetland site on the east side of the ridge served as the control and never received wastewater effluent. Baseline studies of accretion, primary productivity, water quality, hydrology, and decomposition began in both sites in 1988 and 1989, before any wastewater was applied to the swamp. In March 1992, the treatment site began receiving secondarily treated municipal wastewater at a rate of  $7.5 \times 10^6 \text{ L day}^{-1}$  (and continues as of this writing). Field measurements of all parameters continued in both sites after effluent additions began. Over the 1425 ha basin, loading rates of N and P from the wastewater effluent average  $3.1 \text{ g m}^{-2} \text{ year}^{-1}$  and  $0.6 \text{ g m}^{-2} \text{ year}^{-1}$  respectively. The effluent water itself contributes approximately  $19 \text{ cm m}^{-2} \text{ year}^{-1}$  to the basin.

The sequence of events that contributed to the hydrologic isolation, and permanent flooding, in the Pointe au Chene swamp are typical of other wetlands in the Terrebonne Basin. The present wetland overlies of a sequence of deltaic Holocene

sediments over 100 meters thick. It is the weight of these historic sediments that drive the processes (e.g. compaction of tertiary, Pleistocene and Holocene deposits, geosynclinal downwarping) that account for the deep subsidence affecting the basin (Penland 1988). Mineral sediment deposition to the Pointe au Chene swamp, which served to balance the rate of subsidence, was at a maximum over 1000 years ago when Bayou Lafourche was the principal deltaic distributary (Frazier 1967). Even after the last major avulsion event, in which the principal distributary switched from Bayou Lafourche to the modern Mississippi River, Bayou Lafourche remained an active distributary to the modern Mississippi river. In 1904 the Bayou was cutoff from the river by the construction of a dam at its head in Donaldsonville (Brown 1933). Before this time, the bayou was subject to frequent overflow, and there was probably some limited sediment distribution to the backswamps (Fowler 1933). Even before 1904, landscape-wide anthropogenic alterations were occurring that restricted water and sediment exchange and hydrologically isolated the Pointe au Chene wetlands. The Shriver to Gibson spur of the Southern Pacific Railroad, which still serves as the southern boundary to the 1425 ha treatment basin, was constructed in 1852 and restricted water flow and sediment exchange to and from the south. The levees associated with Terrebonne Lafourche drainage canal, which further isolate the wetland from the north and west were constructed in the 1920's and 30's. Natural levees and ridges isolate the wetland from the east. Thus, a combination of events, both natural and anthropogenic, sealed the fate of the Pointe au Chene swamp. At present, the extent of hydrologic isolation is such that there is virtually no source for mineral sediment deposition to counter balance subsidence, other than erosion from the surrounding levees and higher ridges within the swamp itself.

## DETECTING ENVIRONMENTAL IMPACTS

Studies, such as this one, which attempt to detect the effect of some ecosystem-scale perturbation (wastewater effluent in this case) on one or more response variables (decomposition rates for example) are difficult to analyze statistically because of problems associated with either, inadequate or non-existent replication, a lack of control sites, a failure to differentiate between treatment effect variation and background variation, or some combination of these (Stewart-Oaten et al. 1986; Osenberg et al. 1994). Often, these problems arise because the perturbation was unplanned (e.g. oil spills, hurricanes) or because of the expense, impracticality or impossibility of replicating treatments (e.g. power plant outfall, wastewater effluent). Green (1979) outlined an optimal ANOVA experimental design for detecting impact effects in which multiple measurements of some response variable are made once Before and once After an impact, in both a Control and an Impact site (BACI design). A significant time by site interaction would then indicate an impact effect. Conceptually, this design is an improvement over those which have either no control site or no "before impact" observations, because it attempts to isolate background variation from treatment effect variation by testing whether or not the *differences* in the response variables between the two sites change after the impact (Stewart-Oaten et al. 1986). However, Hurlbert (1984) correctly identified Green's multiple observations in each site, at each time, as non-independent sample units, not independent experimental units. Thus, Green's design is non-replicated, or "psuedoreplicated in time", and statistical conclusions which imply impact effects or not, are invalid.

Stewart-Oaten et al. (1986) presented a modification of Green's optimal design which addressed the problem of non-replication. Stewart-Oaten first pointed out that most environmental impact type problems are concerned with a specific impact, at a specific place, and are not the general problem of determining the effects of these kinds

of impacts in these kinds of places. Therefore, impact effects can be detected statistically in the former situation if simultaneous observations are made, at multiple times both before and after the impact, in both the control and treatment site, providing the following assumptions (in addition to the standard ANOVA assumptions) are met:

1) Additivity: Time and location (site) effects are additive (i.e., in the absence of the perturbation, the expected Impact-Control difference is the same for all dates).

2) Independence: Observed differences from different dates are independent.

If these assumptions can be met, then the experiment is replicated (in time). The differences between the control and impact site before the impact (measured at multiple time periods simultaneously in each site), are compared to the differences between the two sites after the impact (again, measured at multiple time periods).

This modified BACI design is difficult to use in practice. First, biologically mediated responses to environmental perturbations are often not additive (Stewart-Oaten et al. 1986, 1992). Second, the response variable associated with the two sites may have been changing relative to each other even before the perturbation (Stewart-Oaten et al. 1986). Third, for some response variables, even observations a year or two apart may be serially correlated or non-independent (Stewart-Oaten et al. 1992). Finally, the nature of this design suggests that the power (the probability of detecting an impact effect when there actually is one) of these types of BACI tests will be low (Osenberg et al. 1994; Underwood 1994).

The problems with power are perhaps the most critical, and have been discussed at length by Underwood (1994). The BACI described by Stewart-Oaten et al (1986) is, at its simplest, a two sample *t*-test which compares the *differences* in some response variable between a control and impact site before the impact, to the *differences* after the impact. For example, assume that the population of some organism is



measured at yearly intervals for three years in both a control and a proposed impact site, before an anticipated impact, and then is measured again, at yearly intervals, for three more years in both sites after the impact. Given that all the assumptions of the BACI test, including additivity and independence, are met, then the degrees of freedom for the critical value of  $t$ , to test for an impact effect, would only be  $[(n_1 - 1) + (n_2 - 1)]$  or  $[(3 - 1) + (3 - 1)] = 4$ , not many degrees of freedom for six years of hard labor, and consequently, not much power.

Some of the problems associated with the BACI design can be solved by using various transformations (for additivity and normality), and by choosing appropriate response variables. However, in situations where there are no before-impact observations, no control sites or no replication, either some other statistical technique, such as Bayesian Inference (Reckhow 1990) or Time Series Analysis (Jassby and Powell 1990) should be used, or the case for impact effect must be argued on some non-statistical grounds. In situations where the BACI design can be used, but replication is low, it is at least useful to calculate the power of the test if possible, in order to be aware of the possibility of a type II error, and to give some indication of the least detectable difference.

Since the field studies presented here were conducted in both a control site and the proposed effluent impact site, both before and after effluent applications began in the treatment site, I utilized the modified Stewart-Oaten BACI statistical design, when appropriate, to detect the effluent-induced impacts on the parameters of interest.

## **TERMINOLOGY**

Throughout this dissertation I discuss the various processes that affect wetland elevation and the measurement of those processes. There has been some confusion in the literature concerning the exact definition of some of these processes (e.g. RSLR and deep subsidence) and the methods used to measure them. Penland (1988), and later,

Cahoon et al. (1995), provided clear definitions and discussion of the terminology relating to subsidence and RSLR. In accordance with their convention, the definitions of the terms used in this study are provided in Table 1.1.

## **RELATED STUDIES AT THE POINTE AU CHENE SWAMP**

Although the focus of this work is on the use of wastewater effluent for the purpose of restoration and enhancement, concomitant studies have focused on other aspects of wetland wastewater treatment at the Pointe au Chene site. Three documents, filed with the Louisiana Department of Environmental Quality, document pre-discharge baseline environmental conditions (Conner and Day 1990) at the site, and two years of post effluent monitoring (Day et al. 1993; Day et al. 1994).

Zhang (1995) described the effects of wastewater effluent on water quality, sediment nutrient concentrations, and the chemical composition of floating aquatic vegetation at the site. Her study assessed the long term ability of the swamp to treat secondarily treated wastewater effluent from the city of Thibodaux. In general Zhang found that, within the immediate 231 ha treatment zone, N and P concentrations in the water were reduced 100% and 66% respectively from effluent inflow to outflow. In a related review, Rybczyk et al. (1996a) concluded that the effective tertiary processing of effluent at this site could be attributed to the following; 1) the dominant species of N in the effluent was the oxidized  $\text{NO}_3$ , which rapidly denitrifies in the dystrophic waters of the swamp, 2) loading rates were low compared to other wetland systems, and 3) high rates of accretion and burial in these systems provide a permanent sink for P. Two other studies, Crozier et al. (1996) and Boustany et al. (in press) also documented the high rates of denitrification at this site.

In a series of papers, Breaux (1992), Breaux and Day (1994) and Breaux et al. (1995), conducted an economic cost benefit analysis of the wastewater treatment operation at Thibodaux. They conservatively estimated a capitalized cost savings,

**Table 1.1 The terminology of wetland elevation; definitions, methods of measurement and conceptual relationships.**

<b>Term</b>	<b>Definition</b>	<b>Method of Measurement</b>	<b>Conceptual Relationship</b>
A) Vertical Accretion	Surface accumulation of mineral and organic matter, usually over some shallow marker horizon. Also integrates processes occurring on and within the upper part of the marsh substrate (e.g. root growth and decomposition) <sup>1</sup> .	Marker horizons such as <sup>137</sup> Cs or Feldspar. Also <sup>210</sup> Pb.	
B) Surface Elevation Change	Change in wetland elevation relative to a subsurface datum (usually 3 to 5 meters deep when using the Sediment Erosion Table (S.E.T.) method for measurement of this parameter) <sup>2</sup> .	S.E.T..	= A - C
C) Shallow Subsidence	Primary compaction, decomposition and dewatering that occurs above the subsurface datum described in B above <sup>2</sup> .	Both a marker horizon and a S.E.T.. are required.	= B - A
D) Deep Subsidence	Primary compaction occurring below the subsurface datum, secondary compaction and other isostatic processes such as geosynclinal downwarping <sup>3</sup> .	Tidal gauge analyses	= G - F - C
E) Total Subsidence	Shallow plus deep subsidence.	Obtained by difference	= C + D
F) Eustatic Sea Level Rise (ESLR)	Global mean sea level rise due primarily to long term variations in climate.	Long-term tide-gauge records, mathematical models.	= G - E
G) Relative Sea Level Rise (RSLR)	Long term, absolute vertical relationship between the land water surface <sup>3</sup> . On the marsh surface, RSLR should be calculated as ESLR + deep subsidence + shallow subsidence. In practice, RSLR, measured using tidal gauge record analyses, represents only: ESRL + deep subsidence.	Tidal gauge analyses	= F + E

<sup>1</sup>Reed and Cahoon 1993

<sup>2</sup>Cahoon et al. 1995

<sup>3</sup>Penland et al. 1988

using natural wetland wastewater treatment rather than conventional tertiary treatment, of between \$448,000 to \$504,000 over a thirty year period. They further noted that capitalized savings could be as high as \$1,300,000 dollars, depending upon the disinfection system employed prior to wetland discharge.

Finally, two papers document the effects of unplanned perturbations to the system that occurred during the period of effluent discharge. Rybczyk et al. (1995) examined the effect of Hurricane Andrew, which passed through the site in August 1992, on water quality and forest structure. Hesse et al. (1996) documented the effects of nutria herbivory in the treatment site. Both of these studies are briefly reviewed in Chapter 5.

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## **CHAPTER 2**

### **NUTRIENT ENRICHMENT AND DECOMPOSITION IN WETLAND ECOSYSTEMS: MODELS, ANALYSES AND EFFECTS\***

#### **INTRODUCTION**

Decomposition refers to the breakdown of organic matter to carbon dioxide, water and inorganic mineral components (mineralization) (Dickinson and Pugh 1974). Inorganic components can also be re-incorporated into the litter matrix during decomposition (immobilization) . Generally, nutrient availability limits the rate of biological decomposition of plant organic matter because of the disparity between the high demand for nitrogen and phosphorus by decomposer organisms that use plant litter carbon as an energy source and the relatively low concentrations of nutrients found in the leaf litter (Swift et al. 1979, Neely and Davis 1985, Enriquez et al. 1993). Nutrient amendments to wetland ecosystems can potentially increase the rates of decomposition by either improving initial litter nutrient quality, via fertilization of the growing plant (Coulson and Butterfield 1978, Valiela et al. 1985, Lukumbuzya et al. 1994), or by increasing externally, the nutrients available to decomposer communities (Howarth and Fisher 1976, Haines and Hanson 1979, Fairchild et al. 1984). Nutrient amendments can also affect the mineralization and immobilization of nutrients within the decomposing litter matrix by altering the distribution and amounts of nutrients associated with the labile and refractory litter components, and by increasing the external pool of nutrients that can be re-incorporated into the decomposing litter matrix (Kaushik and Hynes 1971, Howarth and Fisher 1976, Andersen 1978, Coulson and Butterfield 1978, Elwood et al. 1981, Marinucci et al. 1983, DeBusk and Dierberg

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1984, Fairchild et al. 1984, Neely and Davis 1985, Valiela et al. 1985, Hohmann and Neely 1993).

The decomposition of plant litter has been shown to be central to the cycling and storage of nutrients and energy in many littoral aquatic and wetland ecosystems (Rich and Wetzel 1978, Goldman and Kimmel 1978, Hanson et al. 1984, Wilson et al. 1986, Harrison 1989, Jordan et al. 1989, Davis 1991, Craft and Richardson 1993). Since wetlands are impacted by both point and non-point sources of anthropogenic nutrient enrichment (Yates and Sheridan 1983, Morris 1991, Baker 1992, Kinzig and Socolow 1994), it is critical to understand the effects of nutrient enrichment on litter decomposition to quantify and predict the changes in nutrient and energy cycling and storage that can ultimately affect overall ecosystem function and structure (Davis 1991, Craft and Richardson 1993). This is especially the case for constructed and natural wetlands utilized for wastewater treatment because accurate estimates of nutrient transformations and storage are required to calculate appropriate loading rates, predict rates of sediment accretion and prevent system saturation (Helle 1983, Ewel and Odum 1984, Godfrey 1985, Hammer 1989, Day et al. 1992, Knight et al. 1992, Knight 1994). In these systems, nutrients stored as undecomposed sediment organic matter can represent the largest long term sink for both N and P (Dierberg and Brezonik 1984).

Although it has been generally recognized that nutrient amendments can stimulate the rate of litter decomposition (Webster and Benfield 1986), results from studies designed to evaluate the effects of nutrient amendments on the decomposition of obligate or facultative wetland plant tissue have been equivocal. We reviewed 24 studies (Table 2.1) that examined the effects of nutrient amendments, most commonly nitrogen and phosphorus, on the rates of wetland plant litter decomposition in either, wetland ecosystems, laboratory wetland mesocosms, streams or vegetated littoral zones of lakes. Eight of these studies reported that nutrient amendments accelerated the

Table 2.1. Studies that examined the effect of nutrient amendments on the decomposition rates of wetland and riparian plant species. Several studies conducted multiple experiments.

	Site of Experiment	Duration of Experiment	Species Decomposed	Amendment Category <sup>1</sup>	Nutrient Applied <sup>1</sup>	Effect on Decomp. Rates	Reference
1.	Laboratory	13-14 days	<i>Carex</i> sp.	External	N, P, N+P	None	Federle et al. 1982
2.	Laboratory	25 days	<i>Myriophyllum spicatum</i>	a) External b) Ext. x Int.	a) N, P, N+P b) N	a) Only N increased rates b) Increased rates	Carpenter and Adams 1979
3.	Laboratory	28 days	<i>Salicornia virginica</i> <i>Spartina alterniflora</i> <i>Juncus roemerianus</i>	External	N	Increased rates	Haines and Hanson 1979
4.	Laboratory	5 weeks	<i>Acer saccharum</i>	External	N, P, N+P	Only N+P increased rates	Howarth and Fisher 1976
5.	Laboratory	56 days	<i>Spartina alterniflora</i>	Internal	N	Increased rates	Marinucci et al. 1983
6.	Laboratory	56 days	<i>Acer saccharum</i>	External	N, P, N+P	N and N+P increased rates	Fairchild et al. 1984
7.	Laboratory	56 days	<i>Ulmus americana</i> <i>Fagus grandifolia</i> <i>Alnus rugosa</i>	External	N, N+P	Increased rates	Kaushik and Hynes 1971
8.	Lake littoral zone, Pennsylvania	8 weeks	<i>Quercus alba</i> <i>Acer rubrum</i>	External	Eutrophic lake water	None	Dachler and Majumdar 1992

(table con'd)

	<u>Site of Experiment</u>	<u>Duration of Experiment</u>	<u>Species Decomposed</u>	<u>Amendment Category</u> <sup>1</sup> Ext. x Int.	<u>Nutrient Applied</u> <sup>1</sup>	<u>Effect on Decomp. Rates</u>	<u>Reference</u>
9.	Lake and wastewater effluent pond, Florida	77 days	<i>Eichhornia crassipies</i>	Ext. x Int.	Wastewater effluent	Increased rates	Debusk and Dierberg 1984
10.	2nd order stream, Tennessee	95 days	<i>Quercus rubra</i>	External	P	Increased rates	Elwood et al. 1981
11.	a) Laboratory	a) 34 days	<i>Nymphae alba</i>	a) Internal	a) Eutrophic lake water	a) Increased rates	Brock et al. 1985a
	b) Laboratory	b) 28 days	<i>Nymphae alba</i>	b) External	b) N,P,N + P	b) N and N+P increased rates in some cases	
	c) Netherlands pool	c) 105 days	<i>Nymphae alba</i>	c) ext. x Int.	c) Eutrophic lake water	c) Increased rates	
12.	Laboratory	200 days	<i>Sparganium eurycarpum</i>	External	N	None	Hohmann and Neely 1993
13.	Stream, Washington	200 - 240 days	<i>Acer circinatum</i> <i>Acer macrophyllum</i> <i>Alnus rubra</i> <i>Psuedotsuga menziesii</i>	External	N	None	Triska and Sedell 1976
14.	Freshwater marsh, Florida	9 months	<i>Sagittaria lancifolia</i> <i>Panicum sp.</i> <i>Pontideria cordata</i> <i>Hibiscus sp.</i>	Combined Ext./int.	Wastewater effluent	None	Bayley et al. 1985

(table con'd)

	<u>Site of Experiment</u>	<u>Duration of Experiment</u>	<u>Species</u>	<u>Amendment Category<sup>1</sup></u>	<u>Nutrients Applied<sup>1</sup></u>	<u>Effect on Decomp. Rates</u>	<u>Reference</u>
15.	Denmark Lakes	10 months	<i>Phragmites communis</i>	Combined Ext./Int.	Eutrophic lake water	Increased rates	Andersen 1978
16.	Cypress dome, Florida	1 year	<i>Taxodium distichum</i>	Combined Ext./Int.	Wastewater effluent	Increased rates	Dierberg 1980
17.	Forested wetland, Louisiana	287-361 days	Mixed riparian hardwood leaves	Combined Ext./Int.	Wastewater effluent	None	Rybczyk and Day 1993
18.	Blanket bog, British Isles	1 year	Numerous of bog species	a) Internal  b) External	N, P	a) Increase rates for N, no effect for P  b) None	Coulson and Butterfield 1978
19.	Cypress domes, Florida	500 days	<i>Taxodium distichum</i>	External	Wastewater Effluent	None	Deghi et al. 1980
20.	Prairie marsh, Iowa	505 days	<i>Sparganium eurycarpum</i> <i>Typha glauca</i>	a) External  b) Internal	a) N+P  b) Wastewater Effluent	a) None  b) Increased rates	Neely and Davis 1985
21.	Peatland, Michigan	590 days	<i>Carex sp.</i> <i>Chamaedaphne sp.</i> <i>Salix sp.</i> <i>Betula sp.</i>	Combined Ext./Int.	Simulated Secondary Wastewater	None	Chamie 1976

(table con'd)

	<u>Site of Experiment</u>	<u>Duration of Experiment</u>	<u>Species</u>	<u>Amendment Category<sup>1</sup></u>	<u>Nutrients Applied<sup>1</sup></u>	<u>Effect on Decomp. Rates</u>	<u>Reference</u>
22.	Salt marsh, New England	600 - 700 days	<i>Spartina alterniflora</i> <i>Spartina patens</i> <i>Distichlis spicata</i>	Ext. x Int.	N+P+K	Internal and external treatments increased rates during first year, internal effect was greater	Valiela et al. 1985
23.	Everglades, Florida	2 years	<i>Cladium jamaicense</i> <i>Typha domingensis</i>	Combined Ext./Int.	Natural gradient of surface water nutrient concentrations	Increased rates	Davis 1991
24.	Brackish marsh, Chesapeake Bay	860 days	<i>Typha angustifolia</i>	External	N+P	None	Jordan et al. 1989

<sup>1</sup>See Table 2.4 for category description

<sup>2</sup>Independent applications of nutrients are listed separately (separated by commas in cases where more than one experiment was conducted), applications of more than one nutrient at a time are listed as X+X. In some experiments the nutrient amendment regime was wastewater effluent.

rate of decomposition (Kaushik and Hynes 1971, Andersen 1978, Haines and Hanson 1979, Dierberg 1980, Elwood et al. 1981, Marinucci et al. 1983, DeBusk and Dierberg 1984, Davis 1991), seven studies reported varying results (Howarth and Fisher 1976, Coulson and Butterfield 1978, Carpenter and Adams 1979, Fairchild et al. 1984, Brock et al. 1985a, Neely and Davis 1985, Valiela et al. 1985), and nine studies found that amendments had no effect on rates of litter decomposition (Chamie 1976, Deghi et al. 1980, Federle et al. 1982, Bayley et al. 1985, Jordan et al. 1989, Daehler and Majumdar 1992, Hohmann and Neely 1993, Triska and Sedell 1976, Rybczyk and Day 1993).

The overall objective of this review was to identify the factors that could account for the apparently equivocal results among these studies, examine the biological, chemical or physical mechanisms that explain the controlling influence of these factors on decomposition and, where applicable, place these findings in ecological perspective. Experimental factors affecting the outcome of these studies included; experiment duration, the nutrient amendment regime, the initial condition of the litter, whether or not the experiment was conducted in the laboratory or in the field and, the type of nutrient that was applied. We also reviewed the effects of enrichment on patterns of nutrient mineralization and immobilization in decomposing litter, as these do not necessarily reflect patterns of mass loss. It is important to note that other processes, which can also be affected by nutrient amendments, such as primary productivity (Rybczyk et al. 1996) and denitrification (Nowicki 1994) can also affect system structure and storage. However, this review focused on the effects of nutrient amendments on the litter compartment only.

In the course of this review we also determined that several elements related to the statistical analysis of decomposition data sets, including the type of model used, the method of analysis and common analyses errors, critically affected the interpretation of

experimental results. Therefore we included a section on the statistical analysis of decomposition/enrichment studies. We begin with this section as it defines much of the terminology used in the remainder of paper.

## **STATISTICAL MODELS AND ANALYSES**

### **Definitions**

Statistical models have two components: 1) the mathematical or *deterministic* component, which describes the average or expected value that would be measured over all experiments under the same conditions and, 2) the *random* component which describes the random deviations from the deterministic component. The *population* refers to the statistical population or set of all possible measurements and experiments that the investigator would like to make conclusions about. *Parameters* refer to statistical parameters or the characteristics of the population such as the mean, slope or intercept of a statistical model for the population.

The null hypothesis that nutrients have no effect on decomposition is tested by determining whether the parameters of the statistical model have been altered by the nutrient amendments. Various statistical models have been used to describe decomposition. Some statistical models assume that the material undergoing decomposition is homogeneous, other models assume that the material is heterogeneous and that each component has a different rate of decay. Both the type of model used, and the statistical analyses of experimental results, are shown in subsequent sections to affect the conclusions that are reached with regard to the effect of nutrient amendments on decomposition.

### **Statistical Populations of Nutrient Amendment Experiments**

The spatial population for decomposition and nutrient amendment studies have included wetland plant litter located in nutrient enriched streams and wetlands (Howarth and Fisher 1976, Elwood et al. 1981), ponds and wetlands used for wastewater



treatment (Deghi et al. 1980, Bayley et al. 1985) and managed ponds and lakes (Coulson and Butterfield 1978, Almazan and Boyd 1978, DeBusk and Dierberg 1984, Brock et al. 1985a) (Table 2.1). Additionally, some experiments were conducted in the laboratory under controlled homogenous conditions and inferences were made to a natural population exposed to heterogeneous conditions (Howarth and Fisher 1976, Almazan and Boyd 1978, Carpenter and Adams 1979, Hanson 1979, Fairchild et al. 1984, Hohmann and Neely 1993).

Inferences are generally made to two types of temporal populations. Some investigators focus on the *mean* mass loss at a specific time after decomposition has begun and are interested in testing the affects of nutrient amendments on one or more times (Kaushik and Hynes 1971, Chamie 1976, Howarth and Fisher 1976, Coulson and Butterfield 1978, Haines and Hanson 1979, Federle et al. 1982, Debusk and Dierberg 1984, Fairchild et al. 1984, Jordan et al. 1989, Hohmann and Neely 1993, Lukumbuzya 1994). Other investigators are interested in the effect of nutrient amendments over the entire course of decomposition rather than at a specific time, and test the effect of nutrients on the decomposition *rates* for one, several or all of the components of the litter (Triska and Sedell 1976, Carpenter and Adams 1979, Deghi et al. 1980, Elwood et al. 1981, Marinucci et al. 1983, Brock et al. 1985a).

### **Deterministic and Random Components of Statistical Models**

The proportion of original mass remaining, ( $y_p$ ), is a function of time:  $y_p = f(t)$ , where  $f(t)$  is generally a non-linear deterministic function of time. Most decomposition studies record the mass of the litter, ( $y_l$ ), as it changes with time, ( $t$ ). The proportion of original mass remaining at time  $t$ , ( $y_p$ ), is the ratio of the mass at time  $t$  divided by the initial mass, ( $y_o$ ), where  $0 \leq y_p \leq 1$ ; the units of  $y_p$  are percent (of original mass). The *absolute* decomposition rate is the first derivative of  $f(t)$  with respect to time [ $f'(t)$ ] and the absolute decomposition rate divided by the proportion of original mass remaining at

time  $t$  [ $f'(t) \cdot y_p^{-1}$ ]] is the *relative* decomposition rate (Wieder and Lang 1982), both of which are deterministic functions of time. The units are percent $\cdot$ time $^{-1}$  and time $^{-1}$  for the absolute and relative decomposition rates respectively. The decay constants or decomposition rates ( $k$ ) reported in the literature are the parameters of the absolute or relative decomposition rate functions; the units of  $k$  are time $^{-1}$  for the simple exponential decay model.

Carpenter (1981) presented a general model of decomposition which encompasses most mass loss decomposition models (MLD) used for the statistical test of decomposition rates. His model was based on the premise that “detritus is composed of a large number of chemical compounds and complexes that vary in their susceptibility to decay.” Empirical evidence for such models was provided by Minderman (1968) in forest systems. Carpenter showed how various assumptions lead to all of the common MLD models. The most common MLD model assumes that there is only one constituent, i.e. the litter is homogenous and has a constant rate of decay. This model of the proportion of mass remaining at time  $t$  is referred to as the simple exponential decay model (SED):

$$y_p(t) = e^{(-k_1 \cdot t)} \quad (1)$$

Where  $k_1$  is the constant proportion of the log of the remaining mass that decays in each time unit (Carpenter 1982, Wieder and Lang 1982). This model has one population parameter that must be estimated:  $k_1$ . This model was used to examine the effect of nutrient amendments by Brock et al. (1985b), Carpenter and Adams (1979), Chamie (1976), Elwood et al. (1981), Hohmann and Neely (1993), Marinucci et al. (1983), and Triska and Sedell (1976).

Other investigators have used a two compartment exponential decay model (TCED) which assumes 1) that the litter can be partitioned into two proportions: the

recalcitrant (A) and labile (1-A) fractions and 2) that the rate of decomposition for each component is constant:

$$y_p(t) = \left[ A e^{(-k_1 t)} \right] + \left[ (1 - A) e^{(-k_2 t)} \right] \quad (2)$$

Where  $k_1$  and  $k_2$  are the decay constants for the recalcitrant and labile proportions respectively (Carpenter 1981). This model has three population parameters that must be estimated: A,  $k_1$  and  $k_2$ . The TCED model has been used to examine the effect of nutrient amendments on decomposition by Brock et al. (1985b) and was fit to data by Debusk and Dierberg (1984) but the parameters were not used to examine the effect of nutrient amendments on decomposition.

A third model was tested in aquatic systems by Brock et al. (1985b) and was discussed by Godshalk and Wetzel (1978). This model assumes that the leaf litter has an exponentially declining rate of decay, i.e. the rate parameter,  $k$ , of the SED model is assumed to vary as a function of time,  $k(t)$ :

$$k(t) = -k_1 * e^{-k_2 * t}.$$

This results in an MLD with an exponentially declining decay rate referred to as the decaying coefficient decomposition model (DCD):

$$y_p(t) = \exp \left[ -\frac{k_1}{k_2} e^{(-k_2 t)} \right] \quad (3)$$

This model has two population parameters that must be estimated:  $k_1$  and  $k_2$ .

The TCED model has been shown by various investigators to fit decomposition data better than the SED model (Carpenter 1982, De Lyon et al. 1983, Brock et al. 1985b) and the DCD model (Brock et al. 1985b). The DCD model has also been shown to fit better than the SED model (Godshalk and Wetzel 1978, Carpenter 1982).

In general, the random component is assumed to be normally distributed with a mean of zero and a homogenous variance. Very few investigators have tested or reported the results for the distribution of the residuals from decomposition

experiments. Carpenter (1982) noted that the residuals were normally distributed. Lukumbuzya et al. (1994) used a log transformation of percent mass loss data in an ANOVA to correct for heterogeneous variances but noted that the problem persisted even after transformation. The arcsine root transformation and log transformation have been used by Jordan et al. (1989) and Lukumbuzya et al. (1994) respectively, on percent mass remaining data, to meet the assumptions of homoscedasticity required for the analysis of variance.

### **Statistical Analysis of Nutrient Amendment Experiments**

Two types of statistical analysis have been used to determine whether nutrient amendments affect decomposition. The first approach tests the null hypothesis that the remaining mass at one or more times are equal with and without nutrient amendments. This approach generally takes the form of an ANOVA or a set of t-tests and will be referred to here as a *test of means*. The second approach tests the null hypothesis that the rate parameters for the MLD models are equal with and without nutrient amendments. The second approach generally requires a regression analysis, either linear or non-linear, or it may involve a two point estimate (final minus initial mass) of the rate of change in mass with time. The second approach will be referred to as a *test of rates*. The deterministic or mathematical models described previously are used to describe the average behavior of the population over time for a test of rates. Various tests exist to test the null hypothesis that the parameters for a particular mass loss decomposition model are equal with and without nutrient amendments. Cerrato (1990) and Kimura (1980) compare and illustrate test statistics for the test of rates ( i.e. parameters of nonlinear regression equations).

A test of means has the advantage that no MLD model must be assumed in order to test the null hypothesis that the mass loss on any particular date is the same with and without nutrient amendments, however, there are disadvantages. First, inferences are

limited to the times for which data is available. It is not possible to extrapolate to some future date nor interpolate to some date between the sample dates without assuming an MLD model. Second, in ANOVA, the interaction must be interpreted before main effects. Interpretation of interactions for designs with two or more treatments in addition to time are complicated and difficult to convey to the readers (e.g. Lukumbuza et al. 1994). Third, mass loss decomposition curves for treatments with and without nutrient amendments may cross each other. Fogg (1988) documented several cases where a treatment that initially had a lower proportion of mass remaining at time  $t$ ,  $[y_p(t)]$ , compared to a control, eventually had proportionately more mass remaining than the control. Short term experiments conducted during a limited time period in which one treatment experiences greater mass loss may be mistakenly extrapolated to another time period when the situation is, in fact, reversed.

Tests of rates have several advantages. First, using common statistical computer software (e.g. SAS, PROC NLIN) or approximation formulae (Venus and Causton 1979), confidence intervals for the mass loss,  $y(t)$ , can be calculated for any time (under the assumption that the model holds at all times). Second, the parameters of MLD models are biologically interpretable and succinct whereas factorial designs or repeated measures designs, with several factors in the main plot that are typical of a test of means (e.g. Lukumbuza 1994), result in a plethora of parameters that are difficult or impossible to interpret.

Tests of rates also have some disadvantages. First, the use of an inappropriate model may have unpredictable effects on type I and type II statistical error rates (Gerrodette 1987, Peterman 1990, Fairweather 1990) and may lead to misleading predictions of mass loss at dates outside the range of the data. Second, MLD models either fail to accurately describe decomposition associated with changes in the

environment, such as seasonal changes in temperature, or become complicated in order to incorporate the additional parameters.

### **Statistical Errors**

Inappropriate assumptions about the distribution of the data from decomposition experiments, failure to use appropriate statistical models or neglecting to draw random samples from the statistical population of interest results in studies that may falsely conclude that nutrient amendments have affected decomposition (a false alarm or type I statistical error) or that fail to detect a change in decomposition due to nutrient amendments (false complacency or type II statistical error). This section examines some common errors in the statistical analysis used by investigators to test the null hypothesis that nutrient amendments do not affect decomposition.

### **Inappropriate Models**

Model discrimination involves the use of an appropriate criterion to rank two or more hypothetical models (Dastoor 1990). The objective is to choose the 'best' model rather than determine the 'true' model. The second section of Dastoor's paper gives a good review of the statistical terminology for model discrimination. Cox (1960, 1962) presented a general method based on the Neyman-Pearson likelihood ratio for discriminating between models. Gallant (1987:139-145) illustrated how to use the error sum of squares to discriminate between full and reduced non-linear regression models such as the TCED and SED models respectively.

Failure to use the appropriate statistical model will result in estimates for population parameters that are nonsensical, misleading and will have unpredictable effects on type I error rates. Jordan et al. (1989) noted that their data did not fit a simple exponential decomposition model but did not mention how this was determined. Carpenter (1982) used a ratio of mean squares as suggested by Draper and Smith (1966) to compare models. Brock et al. (1985b) compared the fit of one, two and three

parameter models (SED, DCD and TCED models respectively) using the coefficient of determination ( $r^2$ ) and found that the three parameter model “showed a slightly better fit” than the two parameter model and the two and three parameter models “described the organic weight loss of the [plant] material during its breakdown in the field better” than the one parameter model. However, the coefficient of determination has several potentially misleading characteristics for multiple linear regression (Kleinbaum et al. 1988). The coefficient of multiple determination ( $R^2$ ) is “not a meaningful descriptive statistic for nonlinear regression” because the sum of squares for the regression model (SSR) and the sum of squares error (SSE) do not necessarily sum to the total sum of squares (Neter et al. 1989). For non-linear regression, the coefficient of determination is not used for comparing or discriminating between models.

### **Replication Errors**

In studies of nutrient amendment the smallest unit which receives an application of fertilizer may be a pond or a lake or a plot. An experimental unit is the smallest unit that receives an *independent* application of the treatment. Generally the litterbags placed in each plot, lake or pond and then withdrawn over the course of the experiment are sample units rather than experimental units because they did not receive *independent* applications of the nutrients. Without replication (i.e. without multiple ponds, lakes or plots) no hypothesis tests can be conducted, no inferences can be made to the population of interest and the results are merely hypothesis.

Pseudoreplication is defined “as the testing for treatment effects with an error term inappropriate to the hypothesis being considered” (Hurlbert 1984). Generally pseudoreplication results in an underestimate of the standard error of the treatment means and an inflated type I error rate; this means that there will be a tendency to conclude that nutrient amendments have affected decomposition when in fact that is not the case. Investigations which used only one stream or site per treatment suffered from

pseudoreplication (e.g. Howarth and Fisher 1976, Triska and Sedell 1976, Elwood et al. 1981, Debusk and Dierberg 1984). Other investigators tested the equality of the decay constants for a SED model between treatments with only one experimental unit (e.g. Triska and Sedell 1976, Carpenter and Adams 1979, Elwood et al. 1981). They did this by using the variance among the sample units (i.e. the litter bags) to test the null hypothesis for the treatments (i.e. nutrients); this too is an example of pseudoreplication. True replication rather than pseudoreplication should be used in the future. We suggest that when several pseudoreplicated studies have addressed the same issue, a consensus among the studies be used as a guide to answer the questions involved and that the tendency for false significant results be kept in mind by the reader.

### **Other Design and Analyses Errors**

Repeated measurement designs are distinguished from factorial designs by their distinct restrictions on randomization (Neter et al. 1990). Repeated measures designs are distinguished by the repeated measurement from the same experimental unit or subject where much of the variability is due to differences between subjects existing prior to the experiment (Winer 1971). Failure to account for the restriction on randomization of a repeated measures design by mistakenly analyzing the experiment as a factorial design with time as a cross-classified factor results in an underestimate of the variance due to time. Underestimation of the error term for the time effect and the interaction of time with the other factors results in inflation of the type I error rate for the hypothesis involved in those terms. A number of investigators analyzed repeated measures designs as factorial designs (e.g. Chamie 1976, Haines and Hanson 1979). Several investigators interpreted the main effects in factorial designs even though they had significant higher order interactions which were not discussed (e.g. Haines and Hanson 1979, Neely and Davis 1985). Some investigators did not correct their type I error rates for multiple comparisons and ran the risk of making more statements of



statistical significance than is generally acceptable (e.g. Deghi et al. 1980). Results from those studies should be interpreted cautiously.

## **EXPERIMENTAL FACTORS**

### **Experiment Duration**

The breakdown of plant material in wetland systems has been described as a three phase process (Hanson et al. 1984, Melillo et al. 1984, Valiela et al. 1985, Webster and Benfield 1986, Moran et al. 1989) and there is some evidence that nutrient amendments do not affect each stage equally (Valiela et al. 1985). The first phase is characterized by the rapid loss of mass due to the leaching of the extremely labile organic fractions and soluble compounds and usually lasts for a month or less in aqueous conditions (Valiela et al. 1985). Over 50% of the total litter mass can be lost during this phase (Kulshreshtha and Gopal 1982). The second phase is characterized by the microbial decomposition of labile, but less soluble, organic matter, and can last for up to a year, and finally, the last phase is dominated by the slow decomposition of the remaining refractory compounds. Although these processes overlap, it is first leaching, then the decomposition of labile carbon and, finally, the slow decomposition of refractory compounds that, in turn, dominate and control the overall rate of decomposition (Melillo et al. 1989).

The duration of the 29 individual experimental sets listed in Table 2.1 ranged from 13 to 860 days. Fourteen of these experiments lasted 105 days or less and the remaining 15 experiments lasted for 200 days or longer. Short term experiments (105 days or shorter) were more likely than long term experiments to show a nutrient amendment effect on rates of decomposition. Twelve of the 14 short term studies found that, at least for some treatments, nutrient amendments increased the rates of decomposition. In contrast, only six of the 15 long term experiments (200 days or longer) found any nutrient amendment effect. To further illustrate this, we identified 59

separate experiments, within the 29 experimental sets, that reported percent remaining mass at the end of the experiment. For each of these, the percent remaining litter mass for the nutrient enriched treatment was subtracted from the percent remaining litter mass for the control, at the end of each experiment, to obtain a difference. This difference was then plotted against the duration of the experiment (Figure 2.1). All significant differences between treatments were positive and all non-significant differences between treatments were recorded as zero. Differences ranged from zero (22 of the 59 data points) to 52.7 %. The seven largest differences were all recorded for experiments that lasted for 28 days or less, while 16 of the 22 non-significant differences were reported for experiments that lasted 200 days or longer.

Short term experiment may be more likely to reveal a significant treatment effect because nutrient amendments have been shown to affect the initial phases of decomposition, but have little or no effect on the latter stages (Valiela et al. 1985, Webster and Benfield 1986). For example, Valiela et al. (1985) examined the effects of nitrogen enrichment on the decomposition of the salt marsh grasses *Spartina alterniflora* and *Spartina patens*. Decomposition rates were greater in both the initial leaching phase and the intermediate phase for litter decomposed in nutrient amended plots, but amendments had no effect on the final refractory phase. This lack of an observed amendment effect in the later stages of decomposition may be due, in part, to the shift in the dominant form of substrate carbon as the leaf litter decomposes, from labile carbon to more resistant ligninous carbon forms. These resistant forms of carbon may be relatively less affected by exogenous nutrient supplies (Melillo et al. 1984).

Short term experiments (two weeks to three months) may reveal statistical differences in decomposition rates due to nutrient amendments, but these differences may ultimately have no effect on total loss of mass over one or more years. Long term experiments, especially ones which describe decomposition with simple one

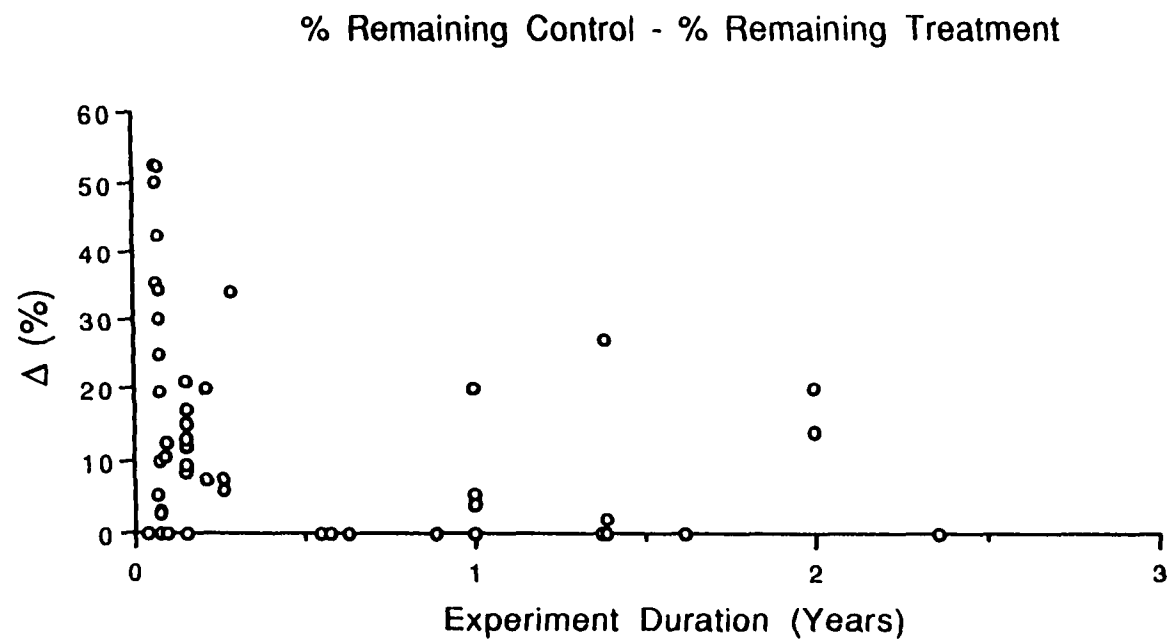


Figure 2.1. The range of effects of nutrient amendments on decomposition rates, determined by examining 59 separate experiments from 24 studies.  $\Delta$  = % remaining control - % remaining treatment.

compartment exponential decay models, or ones which simply compare total loss of mass at the end of the experiment, may not be sensitive to processes which affect only the early stages of decomposition. Generally all mass loss decomposition (MLD) models describe a decomposition process that decelerates (exponentially) as labile fractions of the original litter are decomposed and absolute decomposition rates approach zero, resulting in humic acid end products (Tate 1987) that may last for thousands of years (Satchell 1974). In large part, the reason that differences in decomposition are often found in short term experiments but less often in long term experiments may be a reflection of the negative exponential decline of mass to a minute asymptotic recalcitrant fraction.

As an example, we will use a simple exponential decay model (SED) to show that; 1) the *difference* in mass remaining between treatments with and without nutrient amendments will increase to a maximum and then decrease with time, 2) the difference between treatments may be statistically significant for only limited periods of time and the difference in remaining mass between treatments becomes statistically non-significant as time increases and, 3) the time from the beginning of the experiment until the time when the maximum difference occurs is less than about one year when the annual rate constants ( $k$ ) for the SED model are greater than 1.0 (a comprehensive review by Petersen and Cummins (1974) indicates that the annual leaf decomposition rates ( $k$ ) for the majority of wetland plant families fall between  $0.7 \text{ year}^{-1}$  and  $18.0 \text{ year}^{-1}$ ). To demonstrate these points, typical parameter estimates for the decay constant,  $k$ , (Table 2.2) and a range of variance estimates (Table 2.3) were chosen from two reviewed articles. High and low estimates for the standard deviation of the proportion of mass remaining (Table 2.3) were chosen to reflect a 'typical' range of sampling variability, and were used to calculate confidence intervals for the mean proportion of mass remaining. A sample size of five was used to calculate standard errors from

Table 2.2. Decomposition rate estimates for simple exponential decay models used to simulate litter mass remaining as a function of time.

Species	Treatment	$k_1$ (day <sup>-1</sup> )	$k_1$ (year <sup>-1</sup> )
<i>Nuphar lutea</i> <sup>1</sup>	1) rain water	0.0264	9.63
	2) eutrophic water	0.0599	21.8
<hr/>			
<i>Quercus rubra</i> <sup>2</sup>	3) control stream	0.015	5.47
	4) phosphate enriched stream	0.019	6.93

<sup>1</sup>Brock et al. 1985b

<sup>2</sup>Elwood et al. 1981

Table 2.3. Confidence interval estimates for simple exponential decay models used to simulate litter mass remaining as a function of time.

Variance estimate	Standard deviation (%) <sup>1</sup>	Standard error (%) of difference between 2 means <sup>2</sup>	Confidence interval of difference between 2 means <sup>3</sup>
a) low	± 4	2.52	5.83
b) high	± 20	12.64	29.16

<sup>1</sup>Estimates of standard deviations from Figure 3b of Brock et al. 1985b.

<sup>2</sup>Standard error of difference based on 5 replicates per treatment (Sokal and Rohlf 1981).

<sup>3</sup>Confidence intervals are based on assumption of normality for the % mass loss data.

standard deviations reported in the literature. Statistical power would increase by increasing the number of replicates. However, the total number of decomposition bags required for an experiment is a product of three things 1) the number of sampling times, 2) the number of cross-classified factors and treatment levels and, 3) the number of replicates for each time-treatment combination. Therefore, based on a review of the experimental methods reported for all 24 studies examined here, and practical considerations, the statistical power associated with five replicates per time-treatment combination is considered by the authors to be appropriate for heuristic purposes.

In the first example, estimates for the decay constants,  $k_1$  (SED model), for *Nuphar lutea* incubated in rainwater (treatment 1, Table 2.2), and a eutrophic site (treatment 2, Table 2.2) were used to plot the percent mass remaining versus time (Figure 2.2a) for each treatment. In addition, the difference in percent mass remaining between the two treatments was calculated and plotted (Figure 2.2a). The difference in percent mass remaining from Figure 2.2a was plotted with the confidence intervals from Table 2.3 (Figures 2.2b) to illustrate that the test of means is likely to detect differences between treatments only during a limited time period, i.e. when the difference exceeds the confidence interval (for the null hypothesis that the difference between the treatments is zero). For a situation with high sampling variability (i.e. large CI) this would only occur for the short period between 0.05 to 0.15 years after decomposition begins (Figure 2.2b). Low sampling variability (i.e. small CI) increases the opportunity for the investigator to detect a significant difference to the period between approximately 0.01 to 0.35 years after decomposition begins. The maximum difference in percent mass remaining ( $D_{\max}$ ) between the two treatments was 33% (Figure 2.2b) and occurred at about 0.07 years after the experiment began. The time at which  $D_{\max}$  occurs will be referred to as  $T_{\max}$ .

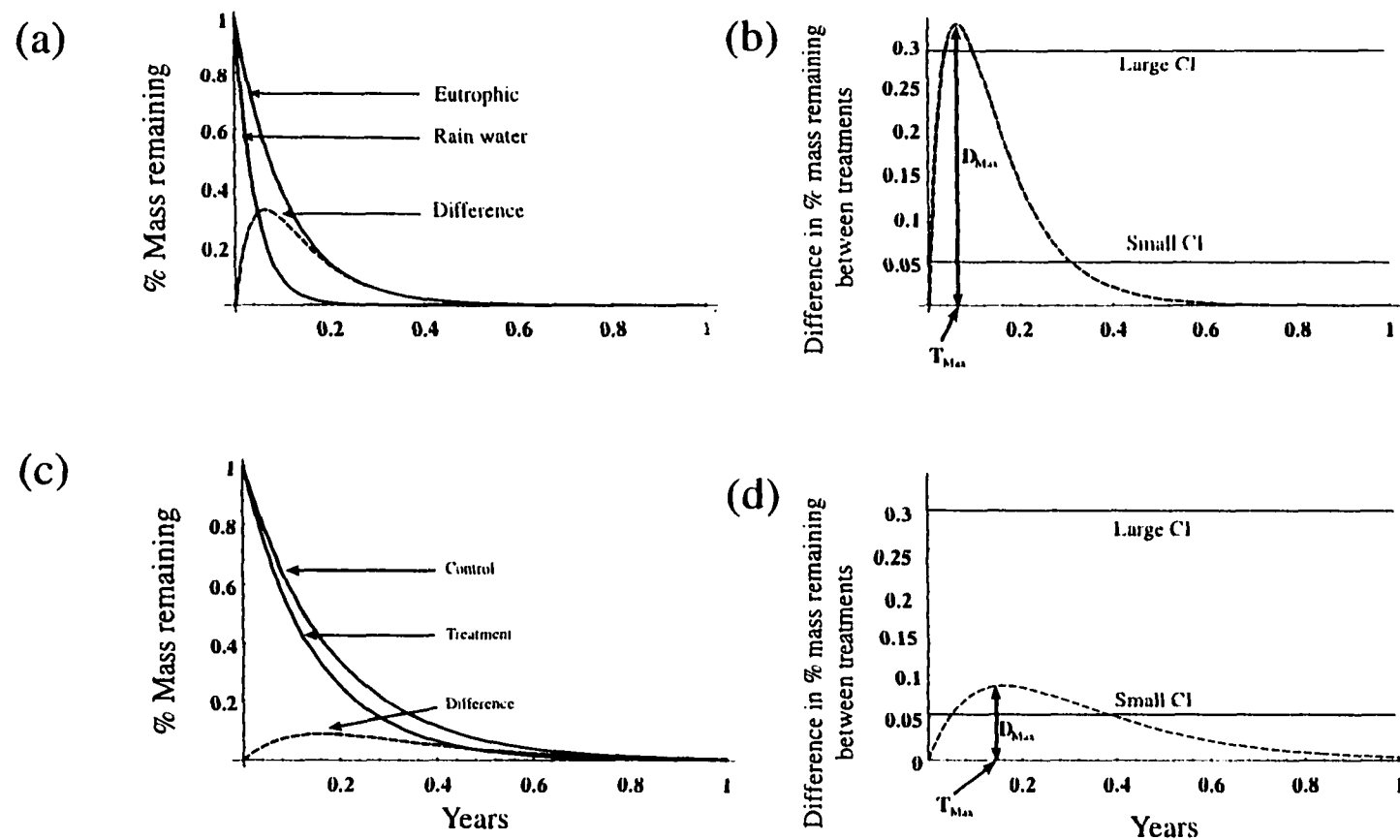


Figure 2.2. a) Percent mass remaining versus time for SED model based on parameter estimates for the two treatments reported by Brock et. al. (1985b), and differences between the two treatments over time. b) The difference between the two treatments, from Brock et. al. 1985b, plotted with large and small confidence intervals from Table 4.3. c) Percent mass remaining versus time for SED model based on parameter estimates for the two treatments reported by Elwood et. al. (1981), and the differences between the two treatments over time. d) The difference between the two treatments, from Elwood et. al. (1981), plotted with large and small confidence intervals from Table 4.3.

The second example uses the estimated decay rates,  $k_1$ , for *Quercus rubra* leaves, as reported by Elwood et al. (1981), in a control and phosphate amended stream (treatments 3 and 4 respectively in Table 2.2) (Figure 2.2c). The difference in percent mass remaining between the control and treatment were plotted in Figure 2.2d. In that case, the expected difference between the two treatments failed to exceed the large confidence intervals but was larger than the small confidence interval from about 0.1 to 0.35 years after initiating the experiment.  $D_{\max}$  was approximately 9% and  $T_{\max}$  was approximately 0.17 years (Figure 2.2.d)

Figure 2.2 illustrates that under a range of parameter estimates for the decomposition constant ( $k_1$ ), the difference in the mean percent of remaining mass between two treatments rapidly approaches a maximum and then declines. The period when the observed differences exceeds the confidence intervals (of the differences) represents a window in time during which differences in percent mass loss are most likely to be detected. Figure 2.2 also shows that large within treatment variability reduces or narrows the window during which difference are likely to be statistically significant. Large within treatment variance results in large confidence intervals and the observed differences may be smaller than the confidence interval; in that situation significant differences are unlikely.

The difference in percent mass remaining,  $\Delta_{\text{pmr}}$ , between treatments with and without nutrient amendments for the SED model can be expressed as a function of the decay constants ( $k_1$ ) for each treatment and time ( $t$ ):

$$\Delta_{\text{pmr}} = \left( e^{-k_{1,\text{with}} * t} \right) - \left( e^{-k_{1,\text{without}} * t} \right). \quad (4)$$

The maximum difference in percent mass loss between the two treatments occurs when the derivative of  $\Delta_{\text{pmr}}$  with respect to time is zero (i.e. at  $t = T_{\max}$ ). Solving the function  $\Delta_{\text{pmr}}$  at time equal to  $T_{\max}$  results in the maximum difference in percent mass loss between the two treatments (i.e.  $D_{\max}$ ).



To illustrate the relation between  $T_{\max}$  and the decomposition rates for litter, with and without nutrient amendments ( $k_{1,\text{with}}$  and  $k_{1,\text{without}}$  respectively), the derivative of equation 4 with respect to time was set equal to zero and solved for the variable  $t$ . Figures 2.3a and 2.3b show contour plots of  $T_{\max}$  for decomposition rates ranging from zero to six, for treatments with and without nutrient amendments. Note that for most realistic combinations of decay constants from nutrient amendment experiments, the maximum difference in percent mass remaining will occur between 0.05 and one year after the experiment is initiated.

The general relationship between decomposition rates for litter, with and without nutrient amendments, and the maximum difference in the proportion of mass loss between the two treatments is shown in Figure 2.4. The  $T_{\max}$  values in Figure 2.3b were used to calculate the corresponding  $D_{\max}$  using equation 4 (Figure 2.4). The regions coinciding with the large confidence intervals (vertical stripes) and small confidence intervals (gray) from Table 2.3 were plotted to indicate the combinations of decomposition rates that are likely to be statistically significant for the level of statistical power presumed in the previous paragraphs (Figure 2.4).

### **Amendment Regimes**

Experiments designed to determine the effect of nutrient amendments on decomposition either amend the environment surrounding the decomposing litter with nutrients (external treatment), fertilize the growing plant so that the subsequent litter itself has higher internal nutrient concentrations than the control litter (internal treatment), or design an experiment that, in some manner, combines both external and internal treatments. We have organized these amendment regimes into four application classifications; (1) external, (2) internal, (3) internal x external and (4) combined internal/external. Table 2.4 outlines the experimental methods for each of these treatments. For the purposes of this review, litter derived from plants grown initially in

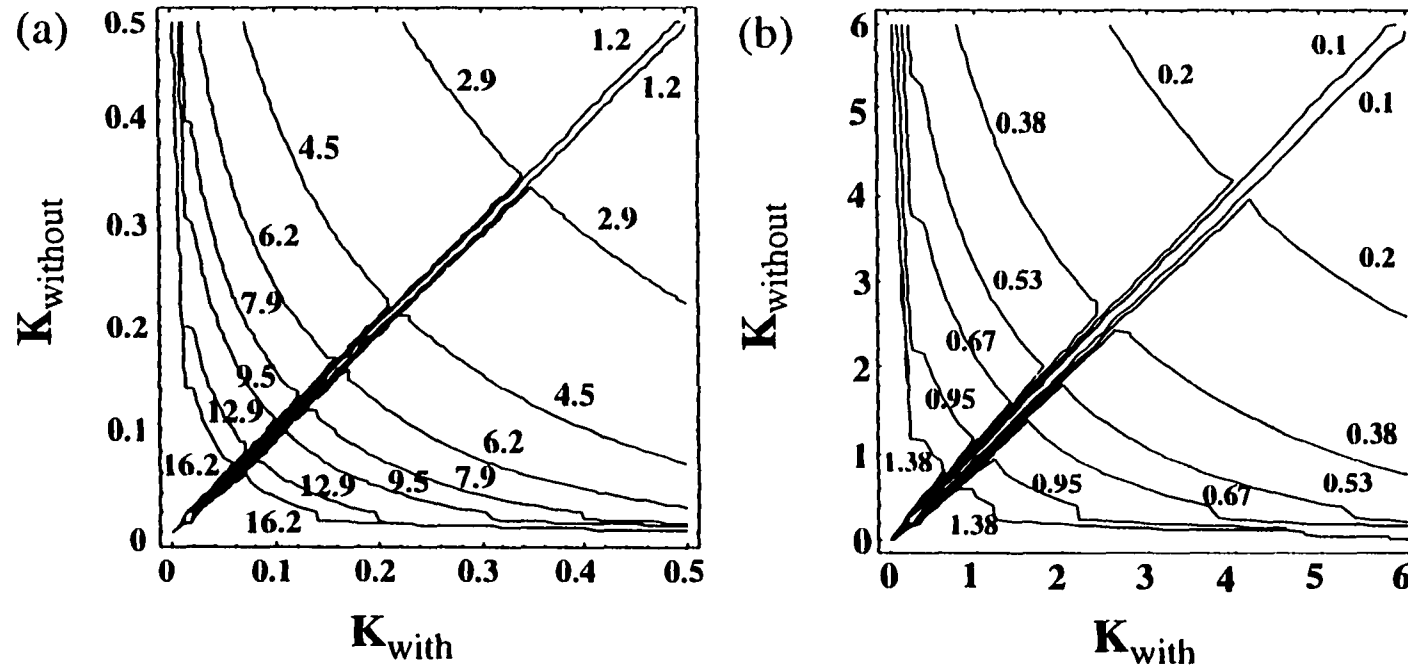


Figure 2.3. Contour plot of the time in years,  $T_{\max}$ , at which the derivative of equation 4, with respect to time, is equal to zero.  $T_{\max}$  is the time at which the maximum difference in percent mass remaining occurs between the two treatments (with and without nutrient amendment) under the SED model. The y and x axis are the rate constant,  $k_1$ , for the treatment without and with nutrient amendment, respectively. a)  $k_1$  ranging from 0 to 0.5, b)  $k_1$  ranging from 0 to 6.0.

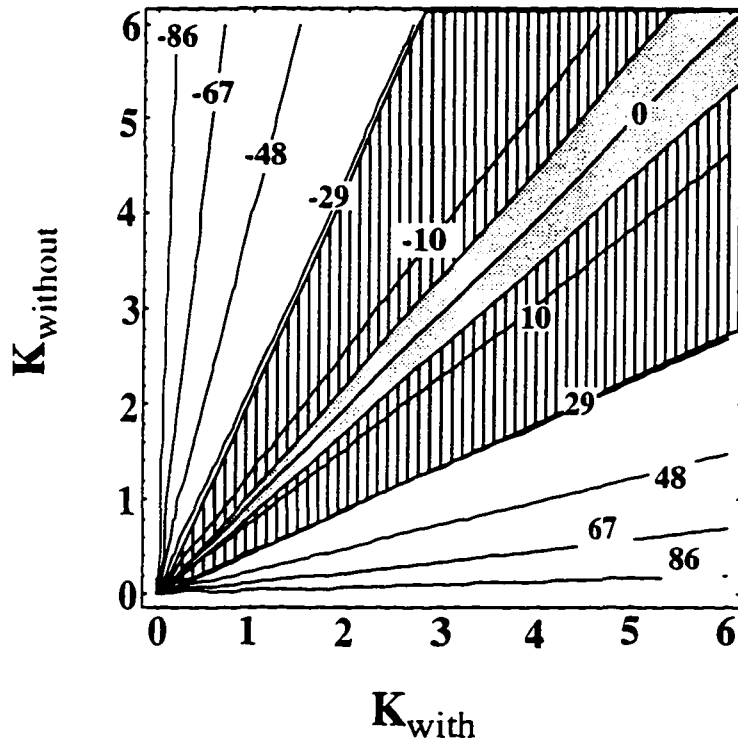


Figure 2.4. Contour plot of the maximum difference in the proportion of mass lost,  $D_{\max}$ , between two SED models. The y and x axis are the rate constant,  $k_1$ , for the treatment without and with nutrient amendment, respectively. When  $K_{\text{with}}$  is greater than, equal to or less than  $K_{\text{without}}$  then  $D_{\max}$  is positive, equal or negative respectively. The units of  $D_{\max}$  are % mass remaining.

**Table 2.4. Categories of nutrient amendment regimes used as part of studies designed to evaluate the effects of nutrient enrichment on decomposition.**

<u>Amendment Regime</u>	<u>Litter Incubation Site Amended with Nutrients<sup>1</sup></u>	<u>Litter Enriched Internally Prior to Decomposition<sup>2</sup></u>	<u>Treatment</u>
1. External	Yes	No	The same type of plant litter is distributed between plots or tanks which then receive different nutrient application treatments.
2. Internal	No	Yes	Plants are grown in fertilized sites and in control sites, thus affecting internal carbon:nutrient ratios. Enriched and control litter are decomposed at a common site.
3. Internal x External	Yes	Yes	Plants are grown in both a high nutrient site and in a control or low nutrient site. Enriched litter from the high nutrient site is decomposed in both the high nutrient site and the control site, and the litter from the low nutrient or control site is also decomposed at both sites.
4. Combined Internal/External	Yes	Yes	Enriched litter collected from plants growing in a fertilized or high nutrient site is decomposed in that site only, and litter collected from similar plants growing in a control or reference site is decomposed in the control or reference site only.

<sup>1</sup>Site may be a tank, mesocosm or field site.

<sup>2</sup>Plants are grown in either high and low nutrient environments thus affecting initial litter carbon:nutrient ratios.

a nutrient amended site, as part of an internal-type experiment, will be referred to as "enriched" litter.

Of the 24 papers reviewed, twenty employed only one of the experimental categories outlined in Table 2.4, three utilized two types of categories and one utilized three different categories, for a total of 29 sets of experiments (Table 2.1). Fourteen of these experiments used some form of internal treatment (either internal, internal x external, or, combined internal/external). Of these fourteen, eleven found faster rates of decay for the enriched litter, compared to the control. The remaining 15 experiments were classified as external, and in contrast to the internal experiments, only seven of the 15 found increased rates of decomposition in the amended sites.

There are two reasons why it would be expected that enriched litter would have a faster rate of decomposition than non-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. For example, Debusk and Dierberg (1984) found that water hyacinth (*Eichhornia crassipes*) aerial tissue collected from a mesotrophic site in south Florida contained 57.7% decay resistant fiber components (cellulose, hemicellulose and lignin), compared to a fiber content of only 48.9% for water hyacinth collected from a nearby eutrophic site. Similarly, Valiela et al. (1984) found that initial soluble phenolic concentrations were higher in fertilized *Spartina alterniflora* litter than in control litter. 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.

The interaction between the amendment regime and the duration of the experiment (discussed in the previous section) is especially noteworthy. All seven of the external experiments which did report a nutrient amendment effect were short term experiments, and all three of the internal experiments that found no effect were long term experiments (Table 2.5).

Table 2.5. Proportion of experiments concluding that nutrient amendments had a significant (positive) effect on rates of decomposition (summarized from Table 2.1).

		<u>Amendment Regime</u>	
		<u>External</u>	<u>Internal</u>
Duration of Experiment	< 100 days	8 out of 10	4 out of 4
	> 200 days	0 out of 6	6 out of 9

Researchers planning long term studies should carefully consider the type of nutrient amendment regime employed. Long term enrichment might reasonably be expected to increase plant nutrient concentrations in a natural system and, subsequently, the initial nutrient concentration of the litter. Therefore, an experimental design involving an internal nutrient amendment regime would be recommended if the objective of the study were to predict or estimate long term sediment and nutrient accumulation.

**Condition of Litter**

The initial condition of litter may also affect the outcome of the experiments designed to determine the effect of nutrient amendments on decomposition. Old or partially decayed litter, dominated by refractory carbon, may not be as responsive to fertilization as fresh litter (Hohmann and Neely 1993). Additionally, as discussed in the "Amendment Regime" section, enriched litter has been shown to have a higher fraction of soluble components than similar non-enriched litter. Enriched litter that has been leached prior to its utilization in a nutrient amendment study has, in effect, already

undergone the first phase of decomposition. This is precisely the phase that investigators have found to be mostly likely to show some nutrient amendment affect (Melillo et al. 1984, Valiela et al. 1985, Webster and Benfield 1986). Some investigators intentionally pre-leach litter prior to use in amendment studies (Kaushik and Hynes 1971, Howarth and Fisher 1976, Elwood et al. 1981, Federle et al. 1982, Fairchild et al. 1984). In these situations, at least the condition of the litter is known and is homogeneous among treatments. More commonly, however, litter is collected from traps or clip plots. In these situations, the amount of leaching prior to collection is unknown since significant amount of leaching can occur immediately after senescence and abscission, especially if the litter is exposed to moisture or precipitation.

### **The Use of Laboratory Vs. In Situ Litter Incubations.**

Factors in the field can interact with decomposition to either mask nutrient amendment effects (e.g. hydroperiod, temperature, redox potential) or enhance them (e.g. increased microbial and macroinvertebrate activity). This, in turn, can lead to conclusions concerning the outcome of a particular nutrient amendment study that may not have been made with an analogous laboratory experiment.

Eutrophication in the field can promote anaerobic conditions, which is often controlled for in the laboratory (Harrison 1989), that have been shown to decrease rates of decomposition (Godshalk and Wetzel 1978, Conner and Day 1991). Additionally, single nutrient amendment studies (e.g. N enrichment only) can cause another nutrient to limit the rate of decomposition, thus giving the appearance of no amendment affect. This may especially be a problem in the field, as opposed to the laboratory, where the concentrations of other nutrients are not controlled and often not measured (Swift et al. 1979).

Carpenter and Adams (1979) pointed out that the establishment of microbial decomposers in the litter matrix may be more rapid in the field because of the large

supply of potential colonists, as opposed to the laboratory. Additionally, nutrient amendments have been shown to increase the density and activity of the macroinvertebrate decomposers that are present in the field but often absent from laboratory incubation studies (Coulson and Butterfield 1978, Tenore et al. 1979, Elwood et al. 1981, Fairchild et al. 1984, Valiela et al. 1984).

### **Type of Nutrient Applied**

Although numerous amendment studies examined the effects of either nitrogen or phosphorus, or the effects of a multi-nutrient wastewater effluent application, on decompositional processes, we identified five studies (Table 2.1) that examined the effects of N and P applied separately, and in combination (Kaushik and Hynes 1971, Howarth and Fisher 1976, Carpenter and Adams 1979, Federle et al. 1982, Brock et al. 1985a). None of these studies found that P amendments alone had a significant affect on decomposition rates. In contrast, four of the five found that N amendments significantly increased decomposition rates (Kaushik and Hynes 1971, Howarth and Fisher 1976, Carpenter and Adams 1979, Brock et al. 1985a). Of the four studies that did show a N amendment effect, three (Kaushik and Hynes 1971, Howarth and Fisher 1976, Brock et al. 1985a) found even higher rates of decomposition rates for the combined N and P treatments. This suggests that when nitrogen is supplied in excess, phosphorus may become limiting (Howarth and Fisher 1976).

### **NUTRIENT DYNAMICS**

While numerous factors influence whether or not nutrient amendments affect the loss of total litter mass, the effect of nutrient amendments on the loss of mass can be independent of the effect on nutrient mineralization and immobilization. As is the case with loss of mass, the nutrient dynamics of decomposing litter has also been described as three phase process, characterized by a period of leaching, followed by a period of net immobilization, and finally, a period of net mineralization (Melillo et al. 1984).



Numerous studies have documented a pattern of increasing N and P concentrations in decaying wetland plant litter over time (Kaushik and Hynes 1971, Triska and Sedell 1976, Brinson 1977, Dierberg 1980, Neely and Davis 1985, Valiela et al. 1985, Moran et al. 1989, Conner and Day 1991, Hietz 1992). This is due, in part, to the loss of carbon, via microbial respiration, as N and P is conserved (Brinson 1977, Swift et al. 1979). However, in many instances, the *absolute* amounts of N and P in decomposing wetland plant litter has been shown to exceed the original amount in the litter (Boyd 1970, Kulshreshtha and Gopal 1982, Yates and Day 1983, Benner et al. 1991, Conner and Day 1991). This indicates that other mechanisms, that accumulate as well as conserve nutrients, including the microbially mediated immobilization of exogenous N and P also contribute to the observed patterns of increasing nutrient concentrations through the decay continuum (Brinson 1977, Melillo et al. 1984, Webster and Benfield 1986). Nutrient amendments, therefore, would increase the pool of exogenous N and P available for potential incorporation into the decomposing litter matrix (nutrient immobilization).

Comparisons of final N and P concentrations measured in litter incubated in nutrient amended wetland systems, versus the final concentrations in the same type of litter decomposed in control or reference sites, generally show higher final N and P litter concentrations in the amended sites (Kaushik and Hynes 1971, Howarth and Fisher 1976, Elwood et al. 1981, Fairchild et al. 1984, Hohmann and Neely 1993, Rybczyk and Day 1993). Some authors, however, found no amendment effect (Triska and Sedell 1976, Jordan et al. 1989), or, increased final P concentrations only (Federle et al. 1982, Neely and Davis 1985). Studies that examined the effects of separate N, P and N + P applications on final litter nutrient concentrations found the greatest effect for combined N + P applications (Howarth and Fisher 1976, Federle et al. 1982, Fairchild et al. 1984). Among the studies that detected significantly higher

final litter nutrient concentrations in the nutrient amended sites, differences between the control and treatment litter ranged from 3.9%, for final N concentrations in decomposing *Sparganium eurycarpum* litter (Hohmann and Neely 1993), to a 3-fold difference in final P concentrations in decomposing *Carex sp.* litter (Ferdele et al. 1982).

Internal amendment studies (Table 2.4) enrich plant tissue, and subsequent litter, before it is decomposed, thereby increasing initial litter N and P concentrations and lowering the C:nutrient ratios. For example, Carpenter and Adams (1979) found that the initial N concentration (2.43% dry weight) of enriched *Myriophyllum spicatum* litter was 1.8 times higher than the concentration (1.33% dry weight) in the control litter. Others have reported a 1.1 to 3-fold increase in initial litter N concentrations between control and enriched litter (Andersen 1978, Coulson and Butterfield 1978, Marinucci et al. 1983, DeBusk and Dierberg 1984, Neely and Davis 1985, Valiela et al. 1985). Similar increases in initial litter P concentrations have also been reported in response to P enrichment (Coulson and Butterfield 1978, Marinucci et al. 1983, Davis 1991). The effect of internal enrichment on initial litter concentrations has also been shown to vary by species. Neely and Davis (1985) found that the initial N concentrations of enriched *Sparganium eurycarpum* litter was 2.4 time higher than the control litter. In contrast, the N concentrations in *Typha glauca* litter, subject to the same enrichment regime, were only 1.1 times higher than the control *Typha* litter. In some cases it has been shown that while nutrient amendments may enrich live plant tissue, the actual effect on litter may be negligible due to the translocation of nutrients prior to senescence and leaf abscission (Davis 1991, Vitousek 1982).

There is some evidence that decomposing litter with initially high N and P concentrations accumulate relatively less N and P than non-enriched litter (Neely and Davis 1985). This has been attributed to; 1) a more rapid loss of nutrients and carbon

in enriched litter, compared to control litter, during the initial "leaching" phase of decomposition (Valiela et al. 1985, Neely and Davis 1985) and, 2) decreased immobilization of exogenous N and P because internal sources are able to meet the microbial demand (Neely and Davis 1985).

The majority of amendment studies that examined net N or P immobilization in decomposing litter report that litter, either enriched or not, was a net sink for N or P when placed in nutrient rich or amended sites (Kaushik and Hynes 1971, Howarth and Fisher 1976, Triska and Sedell 1976, Andersen 1978, Carpenter and Adams 1979, Dierberg 1980, Elwood et al. 1981, Federle et al. 1982, Fairchild et al. 1984, Neely and Davis 1985, Valiela et al. 1985, Jordan et al. 1989, Davis 1991). However, Carpenter and Adams (1979), in a 25 day experiment, found a net immobilization of N, but not P, and Ferdele et al. (1982) found a net immobilization of P, but not N after 14 days. Additionally, Marinucci et al. (1983) and Hohmann and Neely (1993) found that wetland litter decomposed in nutrient amended laboratory mesocosms was a net source for both N and P after 56 and 200 days respectively, although for both studies, significantly more litter N and P was mobilized in the nutrient amended sites than in the respective control sites. Several authors reported an initial net loss of N and P (Triska and Sedell 1976, Andersen 1978, Dierberg 1980, Neely and Davis 1985, Valiela et al. 1985) corresponding with the rapid leaching phase of decomposition, followed by a period of immobilization in which absolute amounts of N or P eventually exceeded original amounts. This may explain, in part, the results (net mineralization) obtained by Ferdele et al (1982), Carpenter and Adams (1979) and Marinucci et al. (1983) discussed above, as all three of these studies were relatively short (13, 25 and 56 days respectively), and probably did not last beyond the initial leaching phase.

## CONCLUSIONS

Nutrient enrichment can affect both the rates of decomposition, and, the mineralization and immobilization of nutrients within the decomposing litter. Several experimental factors can potentially affect measured *rates* of decomposition during amendment studies, including initial litter condition, the type of nutrient applied and whether or not the experiments were conducted in the field, or in the laboratory. However, the two primary factors that controlled the outcome of studies designed to test for the effects of enrichment on the rates of decomposition were the nutrient amendment regime and the duration of the experiment. Nutrient amendment regimes can be grouped into two general categories; 1) those that enrich the plant tissue, and subsequent 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). Internal treatment experiments were more likely to show that nutrient amendments increased rates of decomposition because, as well as increasing the supply of limiting nutrients, enriched litter was shown to have proportionally more labile material than non-enriched litter.

The duration of the nutrient amendment studies reviewed here ranged from 13 to 860 days. The majority (85%) of the short term studies (100 days or less) found that nutrient amendments increased rates of decomposition, while only 40% of the studies lasting 200 days or more found that amendments increased rates. This was due in part to the relative insensitivity of the refractory litter components to exogenous nutrient amendments, compared to labile components and, in part to the short period in time during which differences in mass between treatments are statistically significant. For  $k_1$  greater than 1.0 this period occurs within the first year after the experiment is initiated. Qualitatively, we found that the interaction between the amendment regime and the experiment duration to be an good predictor of the outcome of

enrichment/decomposition experiments. No long term external amendment study found any effect of enrichment on decomposition rates, while all short term, internal amendment experiments concluded that enrichment significantly increased the rates of decomposition (Table 2.5).

Rates of mass loss do not always reflect rates of nutrient loss. Generally, enrichment studies that have examined the nutrient dynamics of decomposing litter have found; 1) initially higher initial N and P concentrations in litter derived from plants grown in enriched sites, 2) higher final N and P concentrations in litter decomposed in enriched sites, compared to control sites, and 3) that litter decomposed in nutrient amended sites was a net sink for N and P. The accumulation of litter mass, and the nutrients associated with that litter, is a function of decomposition rates and, net nutrient mineralization and immobilization within the decomposing litter matrix. This review suggests that, in the long term, rates of organic matter accumulation in enriched sites will not be influenced by enrichment-related changes in decomposition rates, although enrichment could affect patterns of nutrient accumulation. However it should be noted that few decomposition studies have examined the long term effect on enrichment on rates of decomposition; only one study in this review lasted longer than two years, and 15 of the 24 were shorter than one year.

We reviewed the statistical models and analyses used to test the null hypotheses that nutrient enrichment does not affect rates of decomposition. A minority of the papers used a test of rates to compare treatments and fewer yet reported their decomposition rate constants. We feel that this is unfortunate because, as well as standardizing reporting methods, tests of rates allow for the calculation of confidence intervals at any time and thus serves as a better predictive tool for system managers and modelers than tests of means. We also identified several common statistical design and analyses errors including the use of inappropriate models, the use of the coefficient of

determination for non-linear regression, pseudoreplication, analyzing repeated measurement experiments as factorial designs and the failure to correct for multiple comparisons. These errors may lead to the false conclusion that nutrient amendments have affected decomposition (type 1 error), or result in a failure to detect a difference between treatments (type 2 errors).

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## **CHAPTER 3**

### **DECOMPOSITION RATES AND NITROGEN AND PHOSPHORUS DYNAMICS IN A SUBSIDING FORESTED WETLAND RECEIVING WASTEWATER EFFLUENT**

#### **INTRODUCTION**

Relative sea level rise (RSLR) in the Mississippi River deltaic plain is a natural consequence of subsidence and eustatic sea level rise. 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 RSLR measured in the Louisiana delta region (Boesch et al. 1983; Penland and Ramsey 1990). The eustatic sea level rise component, however, is predicted to steadily increase over the next century due to the impacts of global warming (Gornitz 1982; Nummendal 1982; Day and Templet 1989, Peltier and Tushingham 1989, Woodroffe 1993).

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. 1985). 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 allocthonous source of mineral sediments, which contributed directly to vertical accretion, and the nutrients associated with these sediments promoted vertical accretion through organic matter production and deposition (Patrick and Khalid 1974; 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, resulting in vertical accretion deficits

(accretion - RSLR) and widespread wetland loss throughout the modern delta region (Mendelssohn et al. 1983; Kesel 1988; DeLaune et al. 1991; Day and Templet 1989; Conner et al. 1993; Boesch et al. 1994). Additionally, many wetlands in the deltaic region have been hydrologically isolated from surrounding marshes, swamps and bayous because of an dense network of canals and spoil banks constructed during the past century (Turner and Cordes 1987). Spoil banks impede drainage and often physically impound wetlands, thus preventing the overland flow of sediments and nutrients into coastal wetlands, creating essentially ombrotrophic systems.

Recent wetland restoration projects 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). However, studies have shown that organic matter is also a critical component of vertical accretion in Louisiana coastal wetlands (Hatton et al. 1983; Gosselink and Hatton 1984; Bricker-Urso et al. 1989; Nyman and DeLaune 1991; Callaway 1994). In line with restoration efforts specifically designed to balance accretion deficits, but focusing on organic matter accumulation enhancement, rather than mineral supplements, Day et al. (1992) 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.

Although numerous studies have shown that wetlands are effective tertiary processors of wastewater effluent (Ewel and Odum 1884; Godfrey 1985; Hammer 1989; Knight 1994), nutrient enrichment could either increase or decrease wetland elevation, depending upon the interaction between organic matter production, accumulation and subsequent decomposition. As hypothesized, nutrient amendments could directly increase relative elevation by stimulating autochthonous productivity and

subsequent organic matter accretion (Day et al. 1992). Indirectly, the water and nutrients associated with wastewater could also increase relative elevation by contributing to anaerobic conditions which reduce decomposition rates and organic sediment loss (Godshalk and Wetzel 1978; Conner and Day 1991). Alternatively, nutrient amendments could decrease relative elevation by influencing litter nutrient dynamics and increasing the rate of decomposition of organic matter, either by improving initial litter nutrient quality (Coulson and Butterfield 1978; Valiela et al. 1985; Lukumbuzya et al. 1994), or by increasing, externally, the nutrients available to decomposer communities (Howarth and Fisher 1976; Haines and Hanson 1979; Farchild et al. 1984).

In the spring of 1992, the city of Thibodaux, LA began applying secondarily treated municipal wastewater to the adjacent hydrologically isolated and subsiding Pointe au Chene swamp as part of its tertiary treatment program. Subsidence in this region has been estimated at  $1.2 \pm 0.3$  cm/yr (Penland et al. 1988) and background accretion rates have been measured at  $0.4 \pm 0.1$  mm/yr (Chapter 4) resulting in vertical accretion deficits of 0.8 cm/yr. Because the effects of effluent nutrients on decomposition could substantially affect the balance between accretion and subsidence at this site, the goals of this study were to determine if the following leaf litter decomposition parameters at the Pointe au Chene wetland, 1) decomposition rates, 2) initial N and P concentrations, 3) final N and P concentrations and, 4) net N and P immobilization, change in response to wastewater effluent, with respect to an adjacent control site.

## **SITE DESCRIPTION**

The Pointe au Chene swamp lies on the backslope of Bayou Lafourche, an abandoned Mississippi River distributary, 10 km southwest of Thibodaux, Louisiana (Figure 3.1). This wetland has been isolated from its historical source of

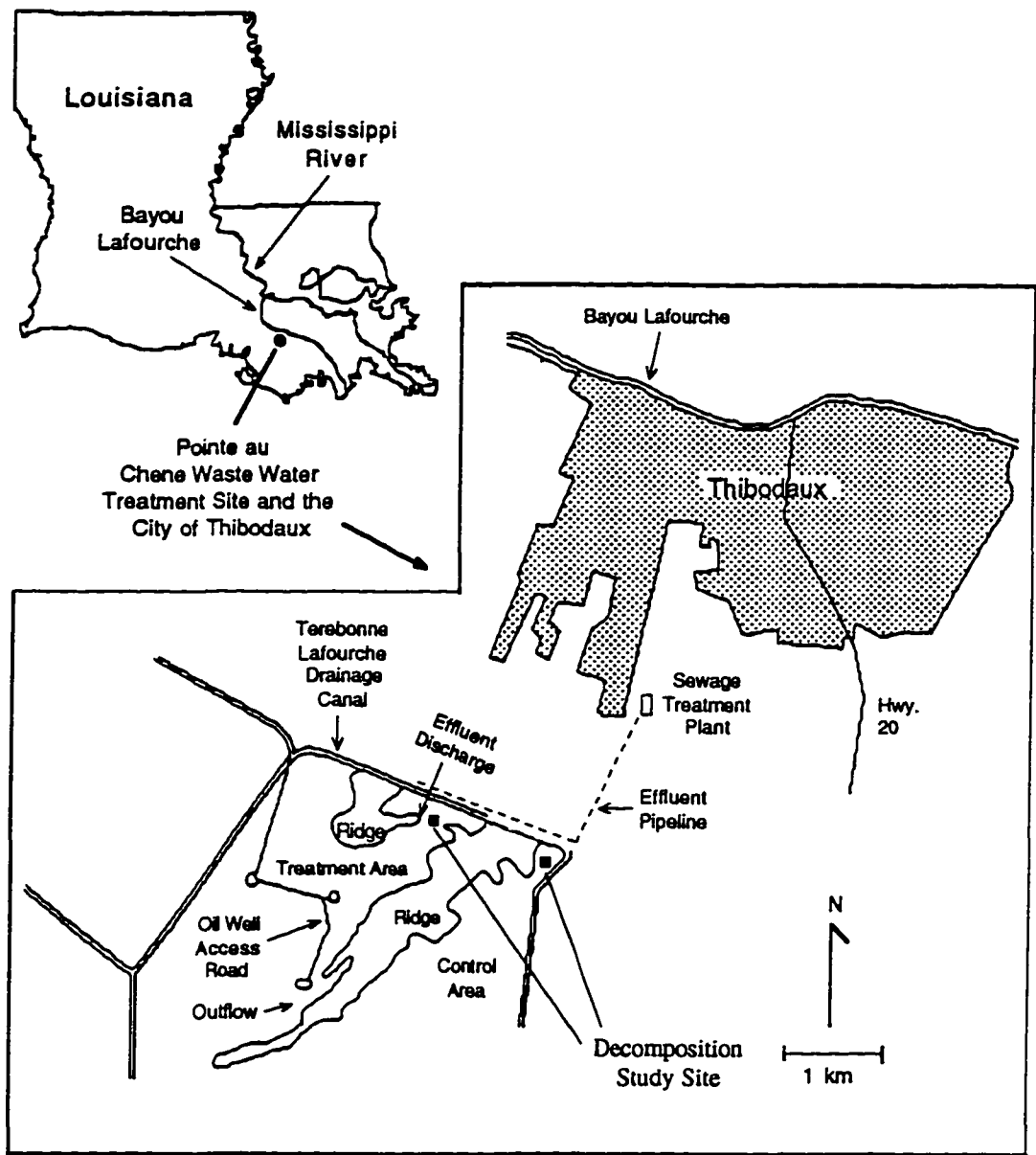


Figure 3.1. Map of the Pointe au Chene Swamp, located adjacent to the city of Thibodaux, Louisiana. An oil access road, a bottomland hardwood ridge, and the spoil banks associated with the Terrebonne-Lafourche drainage canal, hydrologically isolate the treatment site from the surrounding wetland.



sediments since at least the 1930's with the completion of flood control levees along the Mississippi. The study site consists of two almost continuously flooded forested wetlands, separated by a bottomland hardwood ridge, within a 1425 ha hydrologically restricted basin. The ridge site (mean elevation = 1.16 meters above mean sea level (MSL)) is approximately 300 m wide and is vegetated primarily with oaks (*Quercus nigra* and *Q. obtusa*), sweetgum (*Liquidambar styraciflua*), American elm (*Ulmus americana*), palmetto (*Sabal minor*), and boxelder (*Acer negundo*). The two forested wetlands on either side of the ridge (mean elevation = 0.76 meters above MSL) are dominated by ash (*Fraxinus pennsylvanica*), black willow (*Salix nigra*), baldcypress (*Taxodium distichum*), water tupelo (*Nyssa aquatica*), red maple (*Acer rubrum*), and palmetto (*Sabal minor*). Soils are classified as Fausse clay (very-fine, montmorillonitic, nonacid, thermic Typic Fluvaquents) and Sharkey clay (very-fine, montmorillonitic, nonacid, thermic Vertic Haplaquepts). They contain 60-80% clay in the B horizon, which effectively restricts groundwater flow (Conner et al., 1989). 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/year. Rainfall can be quite variable and the total annual precipitation has ranged from 79 cm in 1962 to 222 cm in 1940.

Since March 1992, the 231 ha wetland site on the west side of the ridge has received secondarily treated municipal wastewater at an average rate of  $7.5 \times 10^6$  L/day. Wastewater is discharged from 40 pipes located on the spoil bank that serves as the northern boundary of the site (Figure 3.1). 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 report, the

forested wetland receiving wastewater effluent is referred to as the "treatment site" and the swamp on the eastern side of the bottomland hardwood ridge as the "control site" (Figure 3.1).

Before wastewater effluent applications began, surface water nitrogen and phosphorus concentrations were similar in both the control and treatment sites (Conner et al. 1989). Annual mean concentrations of in the Pointe au Chene swamp during the 1988 - 1989 pre effluent period were 0.05 mg/L and 0.012 mg/L, respectively, and, mean annual concentrations of and Total P were 0.24 mg/L and 0.43 mg/L, respectively. Mean concentrations of  $\text{NH}_4\text{-N}$ ,  $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$  and  $\text{PO}_4\text{-P}$  in the wastewater effluent were 3.7 mg/L, 8.5 mg/L and 1.9 mg/L, respectively, during the two post-effluent years. Analyses of post-effluent surface water have shown that nitrogen and phosphorus concentrations have remained the same in the control site but have increased in surface waters where all the treatment site experimental plots for this study were located (between 25 meters and 50 meters from the effluent distribution pipes). For example, 25 meters from the effluent outfall zone in treatment site, post effluent (1992 - 1994) mean annual concentrations of  $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$  increased to 5.0 mg/L and  $\text{PO}_4$  increased to 1.6 mg/L, although these concentrations returned to background levels at the swamp outflow (Day et al. 1994).

## **METHODS**

### **Decomposition Dynamics**

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 3.1). 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

Table 3.1. Experimental design for the Pointe au Chene Swamp decomposition study.

Experimental Set	Beginning and Ending Date of Experimental Set	Control Site	Treatment Site	Collection schedule (weeks)
1) Pre-effluent	11/17/88 - 11/10/89	No Effluent	No Effluent	0, 1, 3, 8, 14, 18, 23, 27, 32, 36, 42, 48, 52
2) Pre-effluent	12/1/89 - 12/8/90	No Effluent	No Effluent	0, 7, 11, 15, 20, 24, 28, 32, 36, 40, 45, 50, 53
3) Post-effluent	2/26/92 - 12/10/93	No Effluent	Effluent	0, 4, 11, 15, 19, 24, 32, 36, 41
4) Post-effluent	1/22/93 - 1/18/94	No Effluent	Effluent	0, 3, 7, 11, 20, 29, 37, 51

Table 3.2 Annual decomposition rate constants 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	Control Site <sup>1</sup>	Treatment Site <sup>2</sup>
Decay Rate: $k \text{ yr}^{-1} (r^2)$		
11/17/88 - 11/10/89	-0.82 (0.43)	-0.86 (0.63)
12/1/89 - 12/8/90	-1.14 (0.31)	-1.16 (0.31)
2/26/92 - 12/10/93	-1.71 (0.35)	-1.70 (0.31)
1/22/93 - 1/18/94	-0.49 (0.75)	-0.55 (0.65)

<sup>1</sup>Control site never received wastewater effluent<sup>2</sup>Treatment site received wastewater effluent in 1992 and 1993

experimental sets were conducted before, and two were conducted after, wastewater effluent applications began in the treatment site in 1992 (Table 3.1).

Leaves used for each experimental set were collected in the control and treatment site, from leaf litter traps, during the fall months immediately preceding the start of each experimental set. Thirty six 20 cm x 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. Triplicate samples were collected from each site, according to the schedule shown in Table 3.1, for approximately one year for each experimental set. For the first experimental set (1989), leaves collected in each site were pooled, and a homogenous mix of litter was placed in both sites. For the three remaining experimental sets, litterbags intended for each site were filled only with litter collected from that specific site during the previous fall.

The triplicate samples collected in the field were rinsed in the laboratory to remove foreign matter and organisms, dried at 60<sup>o</sup> C for 48 hours, weighed to determine loss of mass, and analyzed for C and N by direct combustion with a C-H-N analyzer. Phosphorus was determined by inductively coupled plasma spectrometry (ICP) after a nitric acid digest. Annual rates of decomposition (k) were calculated for each site within each experimental set by regressing the natural logarithm of the percent remaining mass against time, according to the equation,  $\ln (X/X_0) = -kt$ , where X= weight at time t,  $X_0$  = initial weight and t = time in years (Olson 1963).

Hydroperiod can be an important factor in controlling decomposition rates in bottomland hardwood forests (Conner and Day 1991), therefore, water levels were measured monthly at 10 locations in each site during the course of the study to determine if hydroperiods (flooding frequency and duration) were similar in both the control and treatment sites. Depths were averaged by site to obtain one monthly value per site.

## Statistical Analysis

Stewart Oaten et al. (1986) 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 Before and after the Impact, in both a Control and Impact 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, provided that time and location effects are additive (i.e., in the absence of the perturbation, the expected Impact-Control difference is the same for all dates) and that the observed differences for different dates are independent. While Stewart-Oaten suggested a *t*-test to compare differences before to differences after, Underwood (1991, 1992, 1994) presented a generalized ANOVA modification of Stewart-Oaten's BACI design, with an *F*-statistic that was statistically equivalent to the *t* test proposed by Stewart-Oaten.

Since, for this study, annual leaf litter decay data were collected at multiple times (twice), both before and after effluent was applied to the treatment site, in both a control and treatment site, I used a BACI analysis, within an ANOVA framework (Underwood 1991), to detect changes in decomposition rates, initial N and P concentrations, final N and P concentrations and N and P immobilization, in response to wastewater effluent. Specifically, the BACI analyses tested the null hypotheses that the differences in each of these four leaf litter parameters between the control and treatments 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 experiment unit, and each experimental unit is replicated twice in the before, or pre-effluent period, and twice in the after, or post-effluent period.

## **RESULTS**

### **Hydrology**

Patterns of water-level fluctuations were similar in the control and treatment site during the entire course of this study (1989 - 1994), however, water levels were not measured during the period from January 1990, which marked the end of the baseline pre-effluent period, to May 92, which marked the beginning of the post-effluent period (Figure 3.2). Except for October 1990, the litterbags in both sites were continually underwater during all four year-long decomposition experimental sets. Water levels in the control site averaged ( $\pm$  SE)  $14.4 \pm 0.9$  cm,  $18.7 \pm 1.4$  cm,  $24.2 \pm 1.6$  cm and  $30.8 \pm 1.6$  above the forest floor during 1989, 1990, 1992 and 1993 respectively. In the treatment site water levels averaged ( $\pm$  SE)  $18.4 \pm 0.9$  cm,  $20.7 \pm 1.01$  cm,  $31.0 \pm 1.41$  cm and  $36.5 \pm 1.2$  cm for the same four years respectively.

### **Loss of Mass**

Qualitatively, patterns of weight loss were similar between the treatment and control site within years (Figure 3.3). 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 decomposition which lasted for the remainder of each year-long experiment.

Overall loss of mass, summarized as annual decomposition rates ( $k$ ), was significantly different ( $P < .05$ ) from year to year within sites (Table 3.2). However,

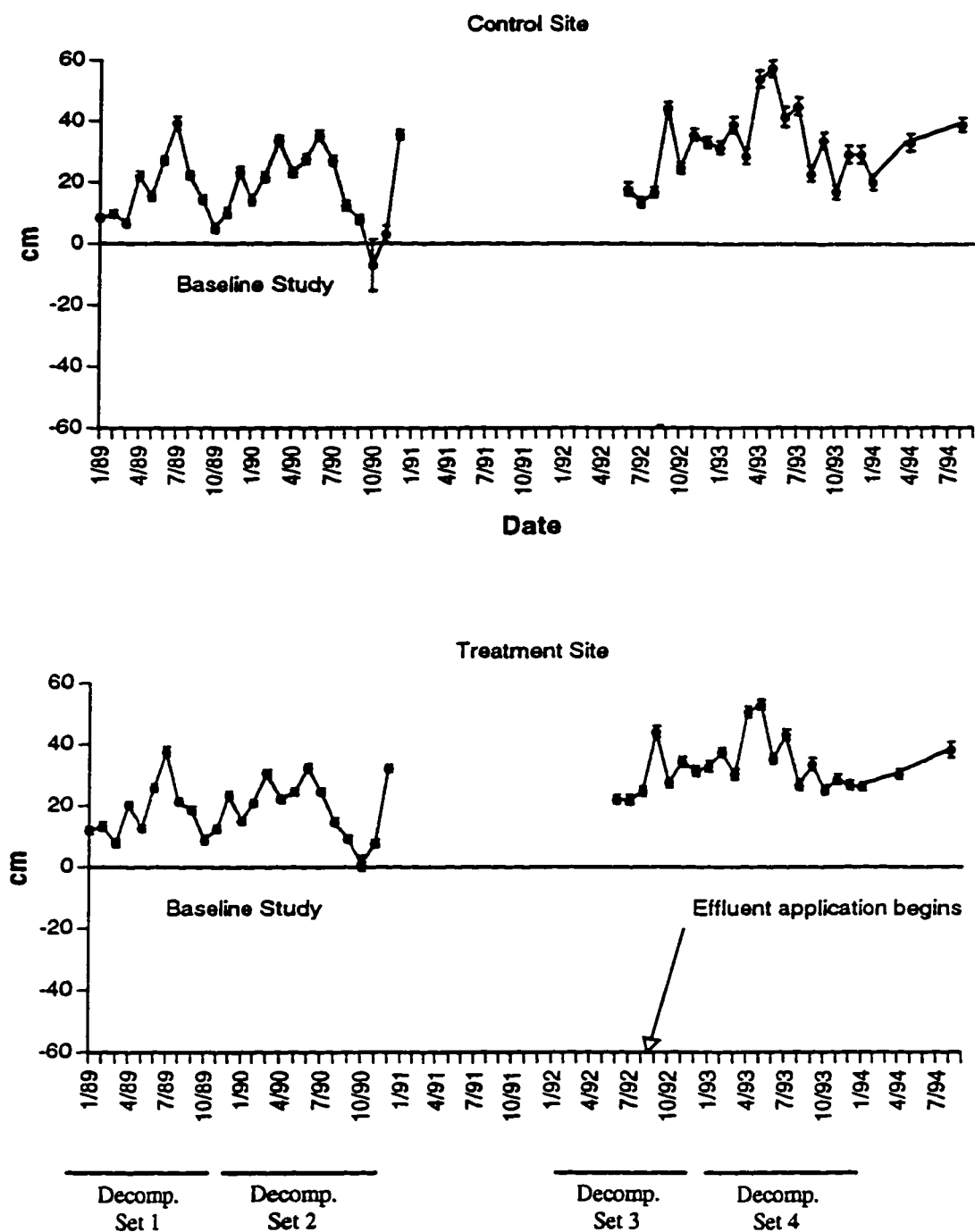


Figure 3.2. Mean monthly water levels ( $\pm$  SE) in the control and treatment sites in the Pointe au Chene Swamp during the four sets of decomposition experiments. Marsh surfaces are not of equal elevation.

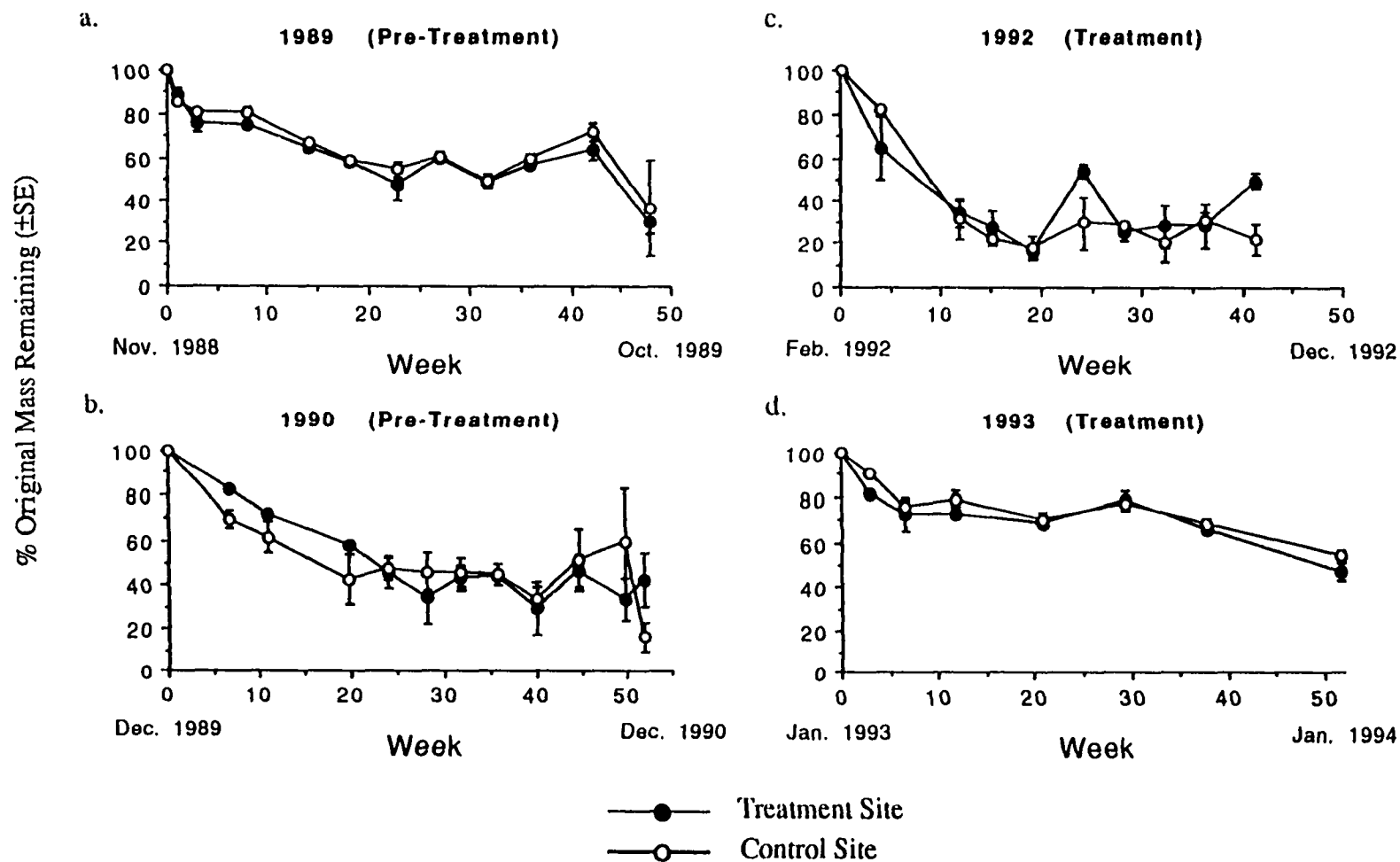


Figure 3.3. Percent of original dry mass remaining through time in the two Pointe au Chene swamp sites during two pre-effluent years, 1989 (a.) and 1990 (b.) and two post effluent years, 1992 (c.) and 1993 (d.).



BACI analysis revealed that the differences in annual decomposition rates between the control and treatment site during the two pre-effluent years were not significantly different than the differences between the two sites after effluent application began in the treatment site. A sample ANOVA table for the BACI design is shown as Appendix B. This indicates that effluent impacts did not affect annual rates of decomposition (Figure 3.4). 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, which is a reasonably small difference to detect considering that decomposition rates from year to year ranged from -0.49/yr to -1.71/yr in the control and treatment site (Table 3.2).

All the data sets used for the BACI analyses met the assumptions of additivity (Tukey 1949), but were too small to test for independence or serial correlation (Durbin and Watson 1951). However, I 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**

Carbon concentrations of the decomposing litter remained relatively constant (Figure 3.5) through time. Initial litter concentrations (g C / g dry weight) ranged from 45.5% to 50.5% over all sites and times and final concentrations ranged from 42.1% to 47.1%.

Qualitatively, litter N and P concentrations increased with time in both sites and during all years (Figure 3.6 and Figure 3.7 respectively). Initial litter N concentrations (g N/ g dry weight) ranged from  $1.1 \pm 0.0 \%$  to  $1.3 \pm 0.1 \%$  and final N concentrations ranged from  $1.6 \pm 0.2 \%$  to  $2.2 \pm 0.1 \%$  among all sites and times (Table 3.3). Similarly, initial litter P concentrations (mg P / g dry weight) ranged

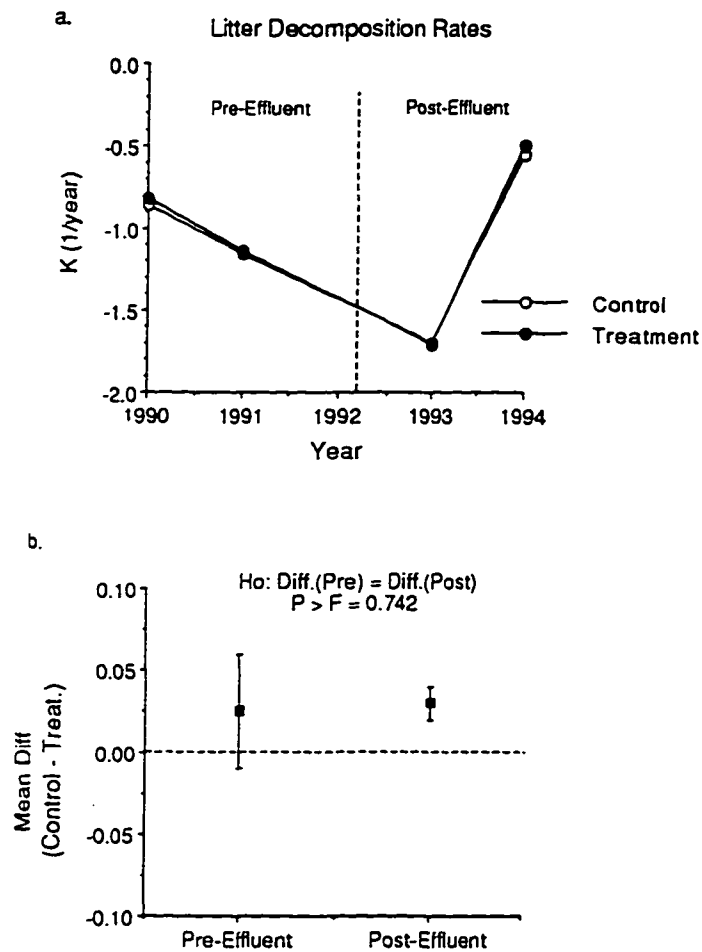


Figure 3.4. Decomposition rates (a) for the four yearly decomposition sets (two pre and two post-effluent) in the control and treatment site, and the mean ( $\pm$ SE) pre and post-effluent differences (b) in rates between sites (control - treatment). Figure 3.4.b shows that the difference between sites in the pre-effluent period was not significantly different than the difference between sites after effluent was added to the treatment site.

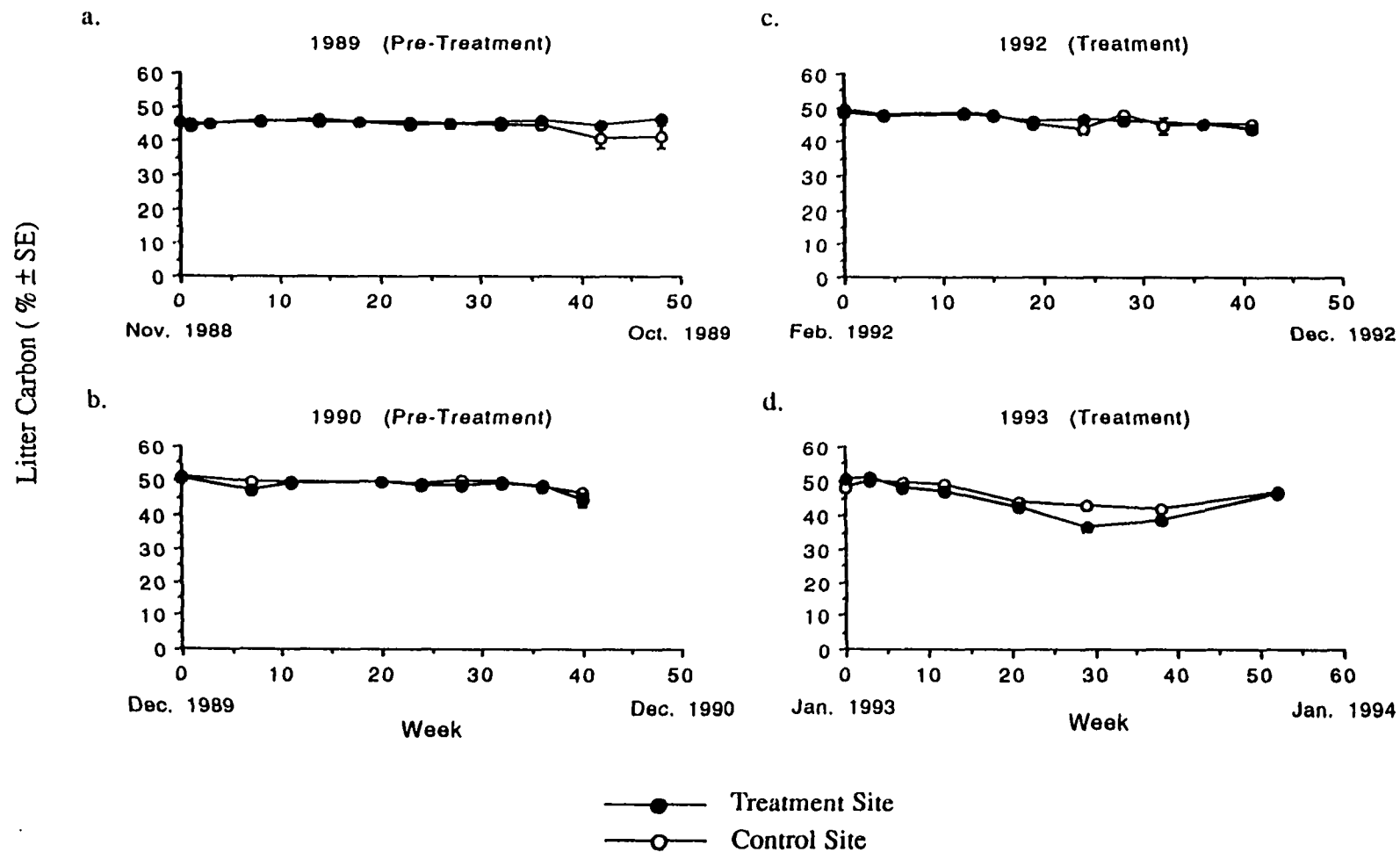


Figure 3.5. Leaf litter carbon concentrations through time in the two Pointe au Chene swamp sites during two pre-effluent years, 1989 (a.) and 1990 (b.) and two post-effluent years, 1992 (c.) and 1993 (d.).

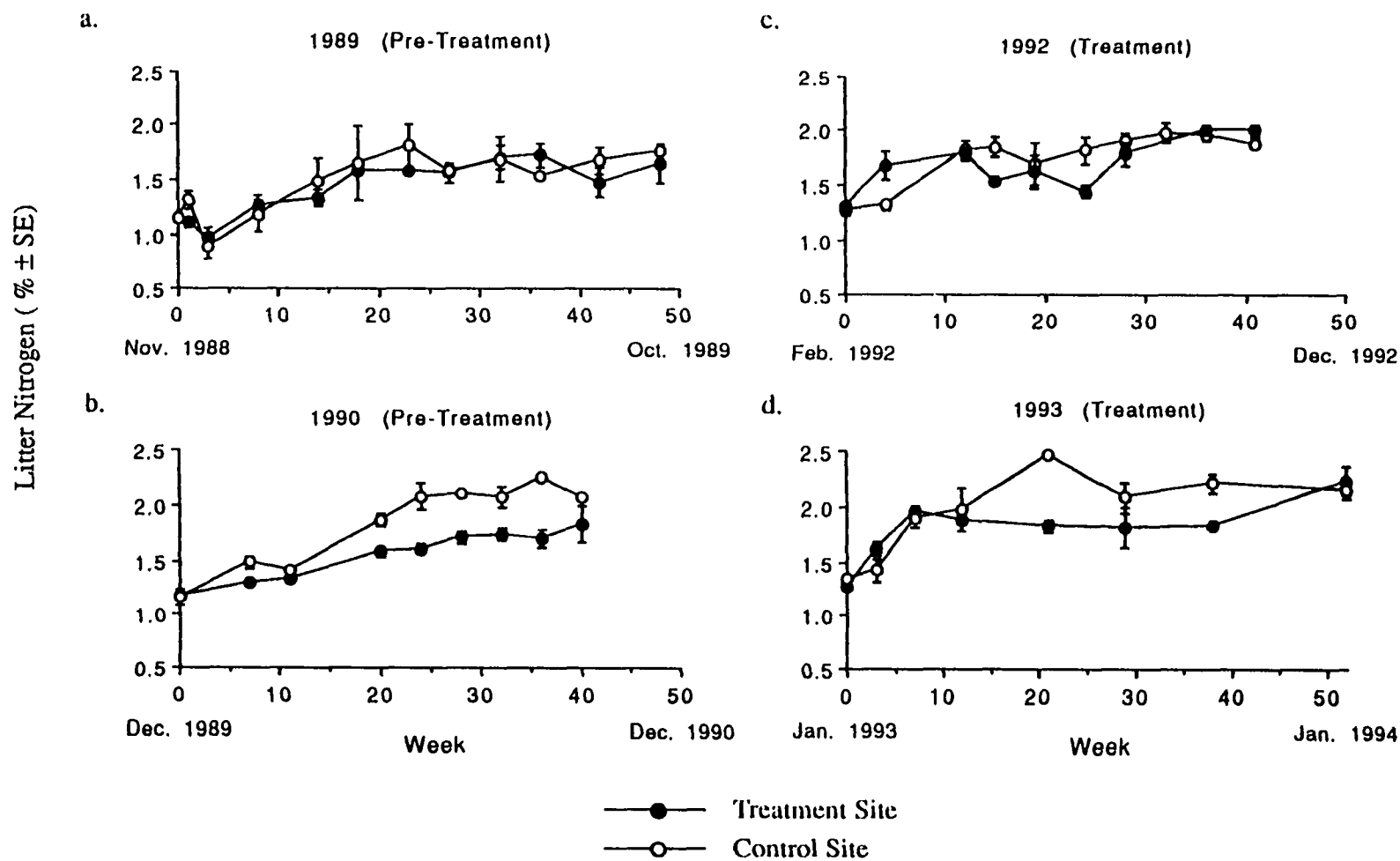


Figure 3.6. Leaf litter nitrogen concentrations through time in the two Pointe au Chene swamp sites during two pre-effluent years, 1989 (a.) and 1990 (b.) and two post-effluent years, 1992 (c.) and 1993 (d.).

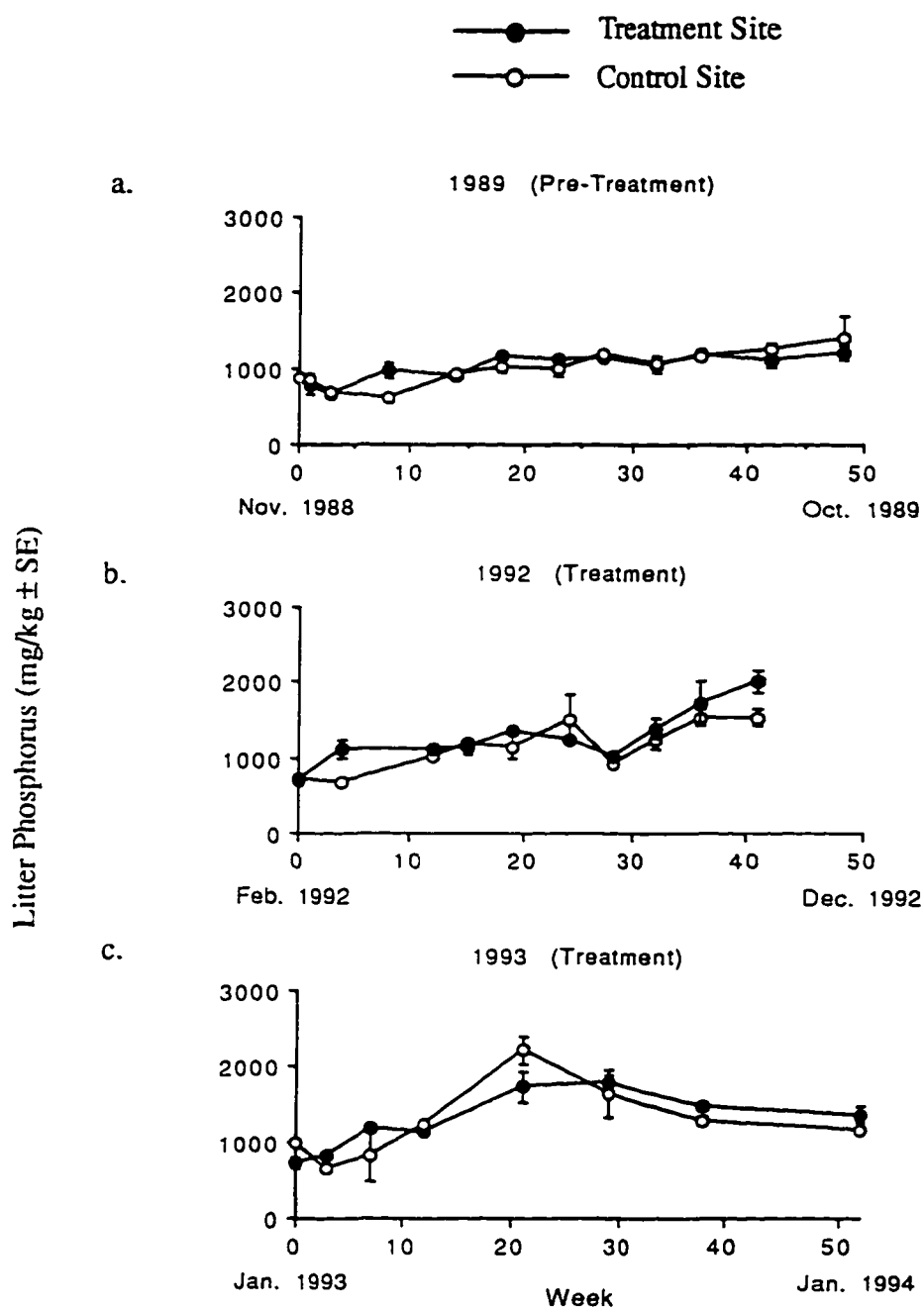


Figure 3.7. Leaf litter phosphorus concentrations through time in the two Pointe au Chene swamp sites during one pre-effluent year, 1989 (a) and two post effluent years, 1992 (b) and 1993 (c).

Table 3.3. Initial and final N (% dry weight) and P (mg/kg) concentrations ( $\pm$  SE), and, C:N and C:P ratios ( $\pm$  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 <sup>1</sup>		C:Nutrient Ratios	
	Control Site <sup>2</sup>	Treatment Site <sup>3</sup>	Control Site <sup>2</sup>	Treatment Site <sup>3</sup>
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 (.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)	466.2 (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)

<sup>1</sup>Concentrations of N are reported as % dry weight, concentrations of P are reported as mg/kg dry weight

<sup>2</sup>Control site never received wastewater effluent

<sup>3</sup> Treatment site received wastewater effluent in 1992 and 1993

from  $724.0 \pm 64.2$  mg/kg to  $999.8 \pm 21.7$  mg/kg and final concentrations ranged from  $922.5 \pm 46.1$  mg/kg to  $1823.3 \pm 190.6$  mg/kg among all sites and times (Table 3.3). Because carbon concentrations were relatively constant through time, C/N and C/P ratios also decreases 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  $35.2 \pm 0.1$  to  $43.7 \pm 3.1$ , and final C:N ratios ranged from  $21.8 \pm 0.9$  to  $28.5 \pm 2.9$  among all sites and times (Table 3.3). Initial C/P ratios ranged from  $479.1 \pm 12.9$  to  $704.2 \pm 63.4$  while final ratios ranged from  $254.7 \pm 40.5$  to  $483.8 \pm 30.7$  among all sites and times (Table 3.3).

BACI analysis indicated that initial N and P litter 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 than the differences between sites during the post-effluent period (Figure 3.8). Differences in final litter N and P concentrations during the pre-effluent site were significantly different ( $P < .05$ ) than the differences between sites during the post-effluent period (Figure 3.9), indicating an effluent effect. Specifically, final litter nitrogen and phosphorus concentrations were higher in the control site during the pre-effluent period and higher in the treatment site during the post-effluent period.

The percent of the original N and P remaining in the litter (Figure 3.10 and Figure 3.11 respectively) was plotted against time to examine overall patterns of net nitrogen and phosphorus immobilization (net gain of N or P in the decaying litter matrix) among sites and years. In general, changes in absolute N and P amounts over time exhibited a three phase pattern; an initial phase of net mineralization (net loss of N and P), which reflected the initial rapid loss of litter mass due to leaching, second, a net immobilization phase, which peaked between months 21 and 42 (Table 3.4),

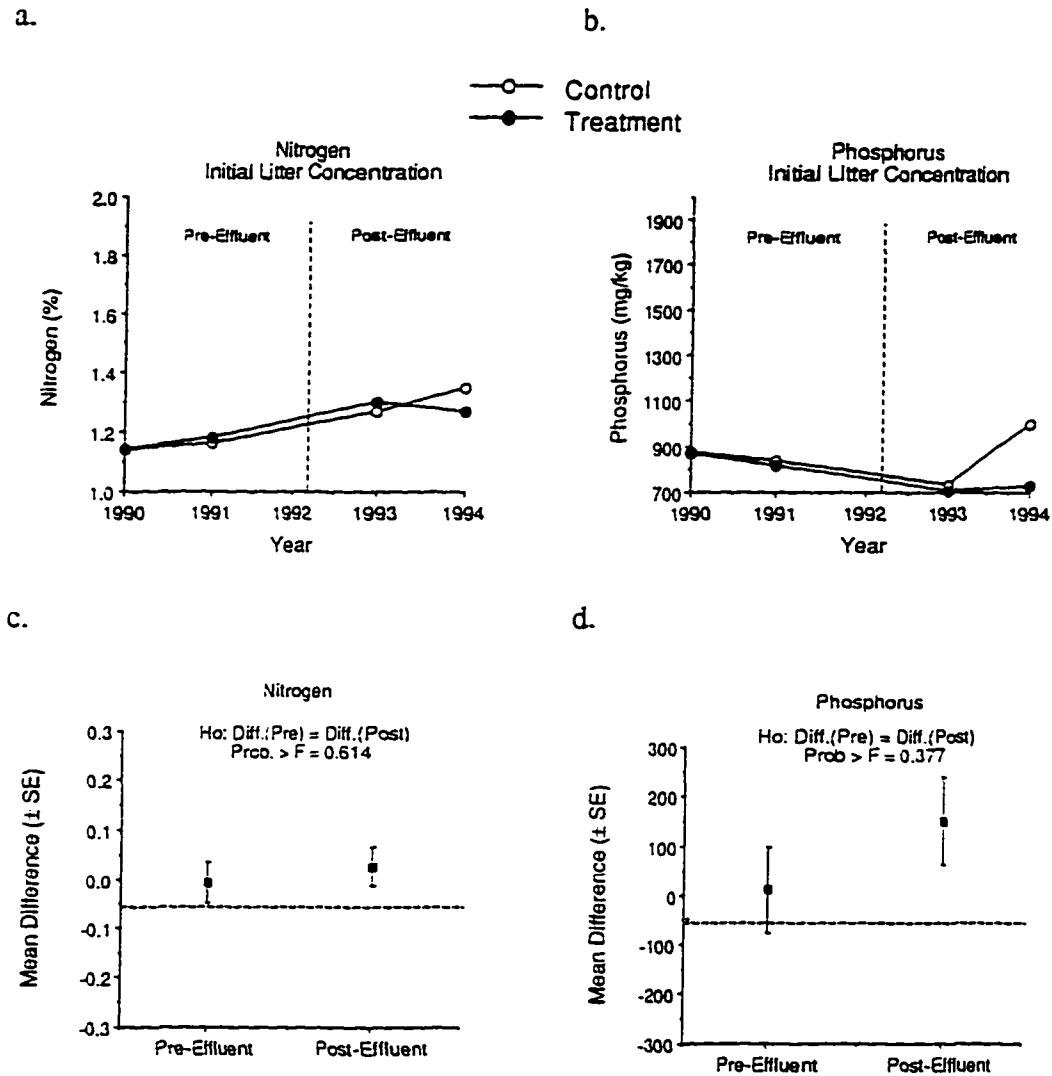


Figure 3.8. Initial mean concentrations of litter nitrogen (a) and phosphorus (b) for the four yearly decomposition sets (two pre and two post effluent years) in the control and treatment sites, and the mean ( $\pm$  SE) pre and post-effluent differences, between the control and treatment sites (control - treatment), for initial nitrogen (c) and phosphorus (d) litter concentrations. Figures 3.4.c and 4.4.d show that, for N and P respectively, the differences in initial litter concentrations between sites in the pre-effluent period were not significantly different than the differences between sites after effluent was added to the treatment site.



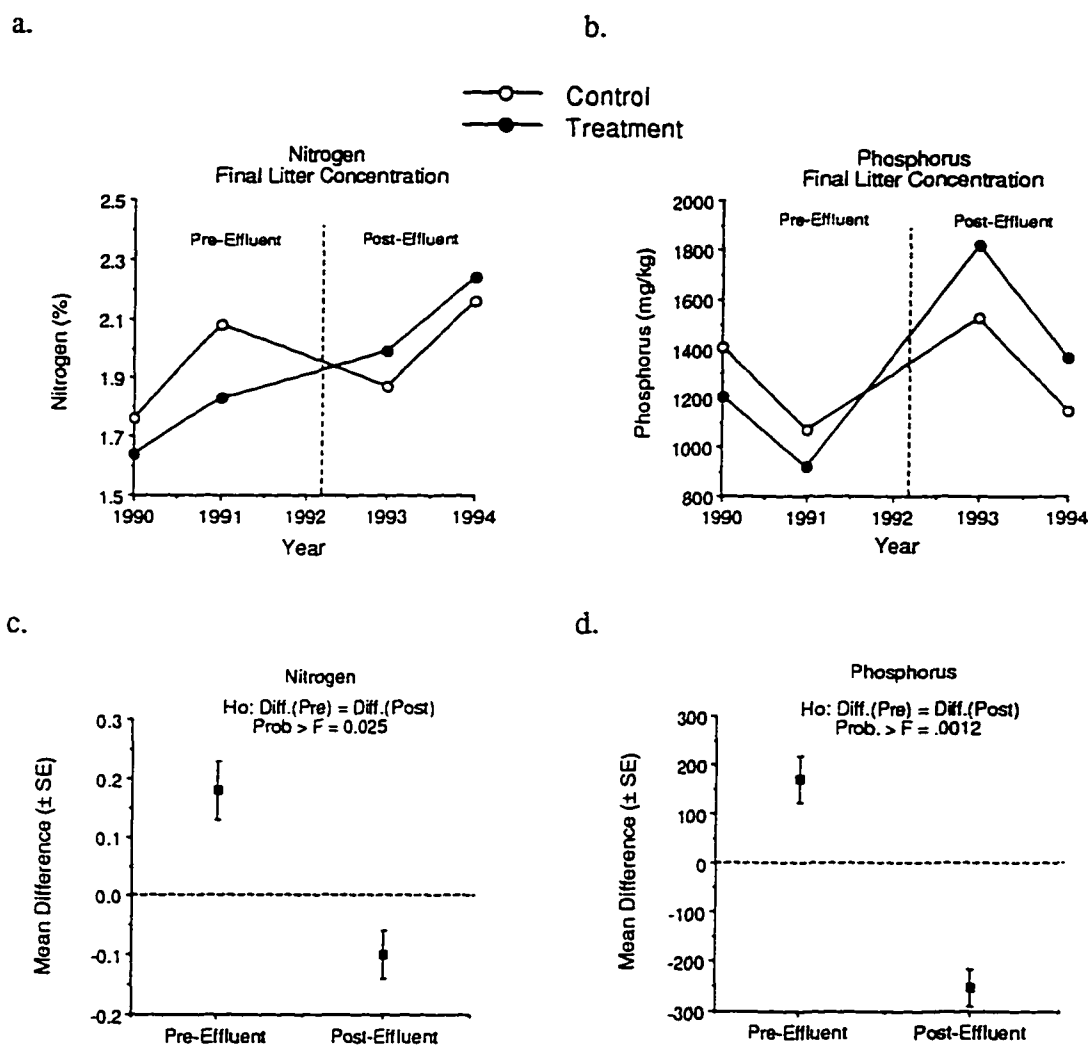


Figure 3.9. Final mean concentrations of litter nitrogen (a) and phosphorus (b) for the four yearly decomposition sets (two pre and two post effluent years) in the control and treatment sites, and the mean ( $\pm$  SE) pre and post-effluent differences, between the control and treatment sites (control - treatment), for final nitrogen (c) and phosphorus (d) litter concentrations. Figures 3.4.c and 4.4.d show that, for N and P respectively, the differences between sites in the pre-effluent period were significantly different than the differences between sites after effluent was added to the treatment site.

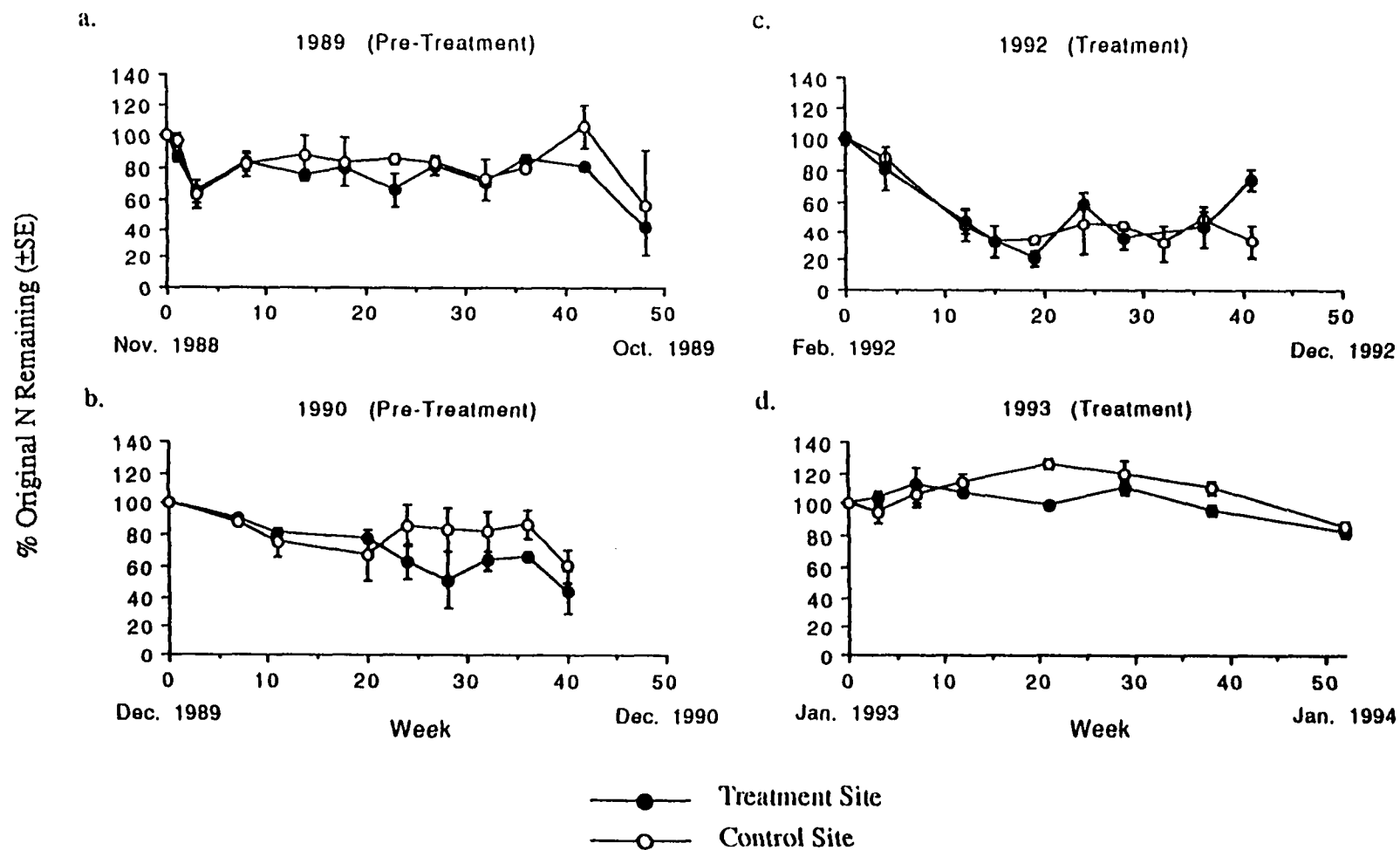


Figure 3.10. Percent of leaf original litter nitrogen remaining through time in the two Pointe au Chene swamp sites during two pre-effluent years, 1989 (a.) and 1990 (b.) and two post effluent years, 1992 (c.) and 1993 (d.).

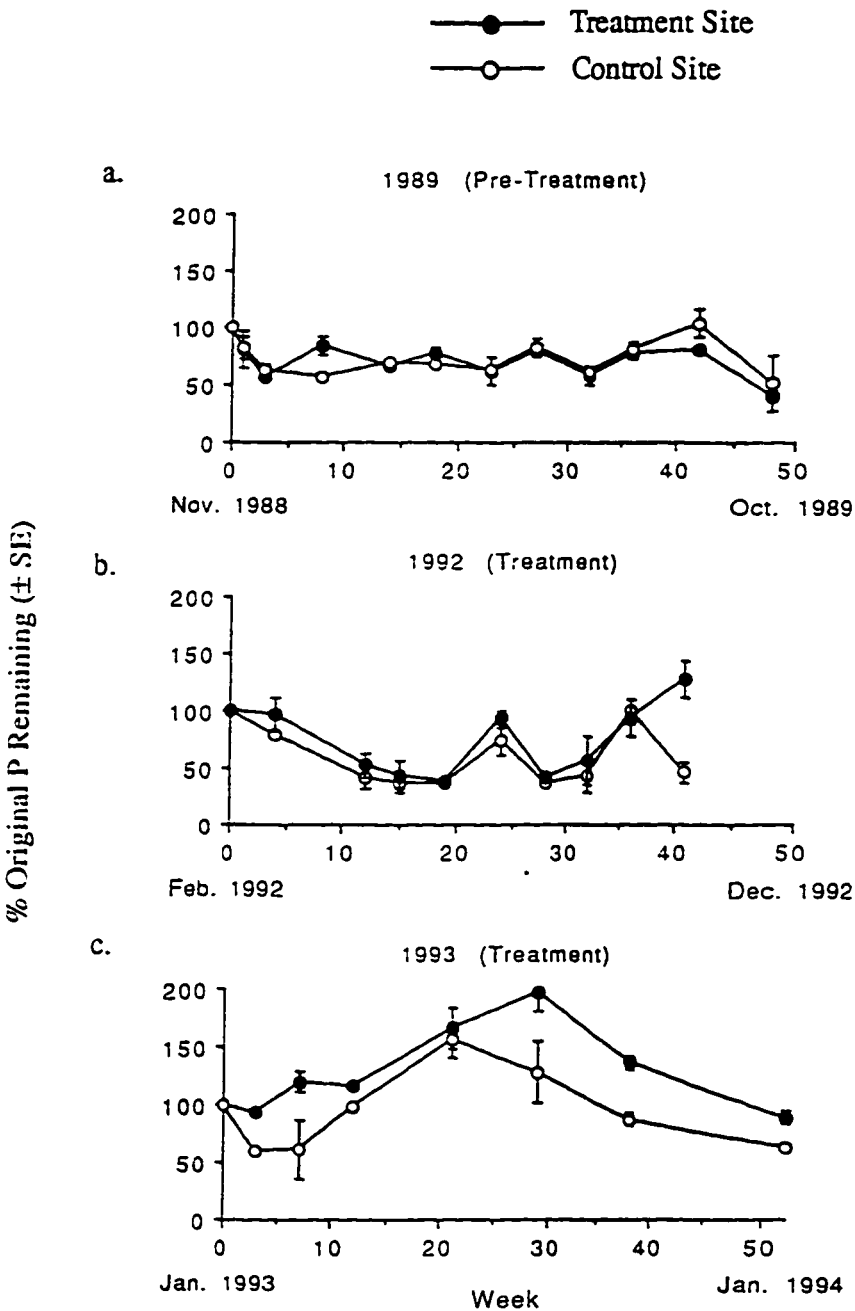


Figure 3.11. Percent of original leaf litter phosphorus remaining through time in the two Pointe au Chene swamp sites during one pre-effluent year, 1989 (a) and two post effluent years, 1992 (b) and 1993 (c).

Table 3.4. Maximum immobilization of nitrogen and phosphorus, expressed as a percent of the original absolute amount of N or P present in the litter, measured in decomposing litter during two pre-effluent (1989 and 1990) decomposition sets and two effluent application sets (1992 and 1992) in the Pointe au Chene swamp.

Date of Experimental Set.	Control <sup>1</sup>				Treatment <sup>2</sup>			
	% of original N (se)	week <sup>3</sup>	% of original P (se)	week	% of original N (se)	week	% of original P (se)	week
11/17/88 - 11/10/89	105.9 (13.4)	42	103.9 (12.1)	42	85.3 (3.9)	36	81.3 (2.2)	42
12/1/89 - 12/8/90	86.9 (9.1)	36	-	-	64.3 (6.4)	32	-	-
2/26/92 - 12/10/93	48.2 (5.7)	36	74.6 (13.9)	24	75.0 (6.8)	41	128.0 (16.2)	41
1/22/93 - 1/18/94	127.3 (3.6)	21	155.9 (15.5)	21	111.6 (4.6)	29	193.7 (7.6)	29

<sup>1</sup>Control site never received wastewater effluent

<sup>2</sup>Treatment site received wastewater effluent during the 1992 and 1993 decomposition sets only.

<sup>3</sup>week = number of weeks after the start of each respective decomposition set at which N or P immobilization peaked.

and finally, another period of net mineralization which lasted for the remainder of each experiment.

For the purposes of this study, maximum immobilization is defined as the maximum absolute amounts of N or P found in the litter after the initial net mineralization phase. In 1989 and 1993 in the control site, and 1992 and 1993 in the treatment site, absolute amounts of phosphorus, during the immobilization phase, increased to levels greater than the initial amounts (Table 3.4). However, due to funding constraints, only the initial and final litter collections were analyzed for P concentrations during 1990, so no determinations of peak immobilization could be made for that year. For nitrogen, absolute amounts increased to levels greater than original amounts in the control site during 1989 and 1993, but only in 1993 in the treatment site (Table 3.4).

A BACI analysis showed that the differences in absolute amounts of N during peak immobilization between the control and treatment sites before wastewater applications began were not significantly different than the difference between peak absolute amounts after effluent applications began in the treatment site, indicating that peak N immobilization was not affected by wastewater effluent (Figure 3.12). In contrast, the analysis did show a significant effluent effect on P immobilization ( $P < .05$ ) (Figure 3.12). Peak P immobilization was greater in the control site before effluent applications began in 1992, and greater in the treatment site after effluent additions began. However, this result must be viewed with caution since there was only one year of pre-effluent data for P, instead of two, it was not possible to test for additivity

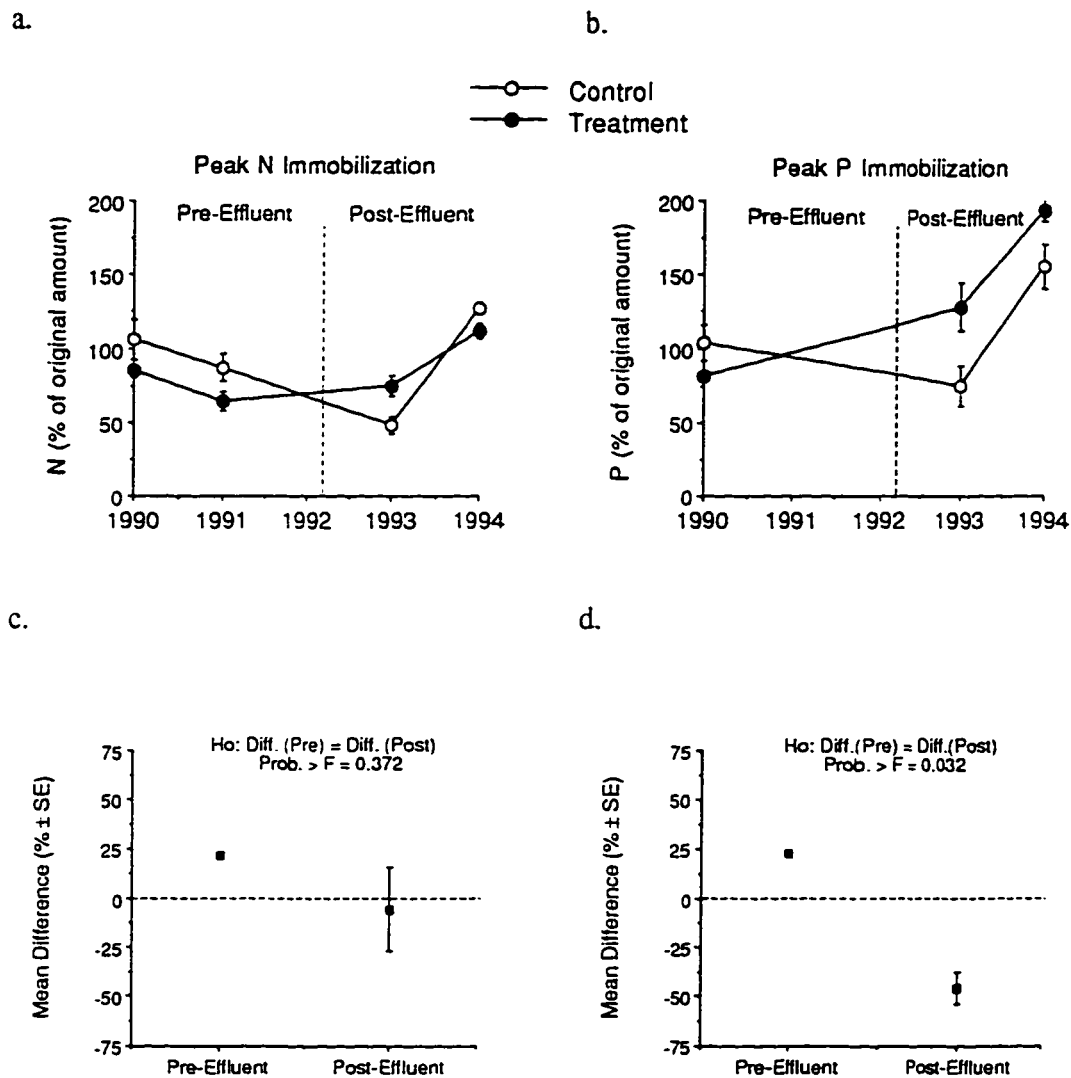


Figure 3.12. Peak immobilization (% of original amount) of litter nitrogen (a) and phosphorus (b) for the four yearly decomposition sets (two pre and two post effluent years) in the control and treatment sites, and the mean ( $\pm$  SE) pre and post-effluent differences, between the control and treatment sites (control - treatment), for peak immobilized nitrogen (c) and phosphorus (d). Figure 4.4.d shows that for P, the differences between sites in the pre-effluent period were significantly different than the differences between sites after effluent was added to the treatment site. No significant difference between differences was found for N (c.).

## DISCUSSION

### General Patterns

Overall, wastewater effluent had no effect on rates of decomposition, but did affect nutrient cycling within the decomposing litter matrix. Hydroperiods between the two sites were similar, despite the addition of wastewater to the treatment site, and there was no indication that decomposition rates within the treatment site were affected by changes in flooding frequency or duration due to wastewater inputs. 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 this source of variation as long as both the control and treatment site are exposed to the same annual fluctuations.

The general pattern of weight loss observed during this study is similar to the pattern recognized in numerous wetland systems (Hanson et al. 1984; Webster and Valiela et al. 1985; Benfield 1986; Moran et al. 1989) and has been described as three phase process (Valiela et al. 1985). The first phase is characterized by the rapid leaching of the extremely labile organic fractions and soluble compounds and usually lasts less than a month. The second phase is characterized by the microbial decomposition of labile organic matter, which can last for up to a year, and finally, the last phase is dominated by the slow decomposition of the remaining refractory compounds. Although, initially, all three processes actually occur at the same time,

it is first leaching, then the decomposition of labile carbon and, finally, the slow decomposition of refractory compounds that, in turn, dominate and control the overall rate of decomposition (Melillo et al. 1989).

The patterns of increasing litter N and P concentrations with time were also similar to those recognized in other wetland systems (Triska and Sedell 1976; Brinson 1977; Neely and Davis 1985; Moran et al. 1989; Conner and Day 1991; Hietz 1992). These patterns are the result of; (1) the loss of carbon as N and P is conserved, (2) the biologically mediated immobilization of exogenous N and P to meet the demand of decomposer organisms, or, (3) a combination of the two (Brinson 1977).

### **Effects of Effluent on Loss of Mass**

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). Therefore it is possible that the addition of supplementary N and P to a system would stimulate the rates of litter decomposition. However, while numerous studies have shown that this is the case, 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. First, the length of the study is of critical importance. In a review of 29 aquatic system decomposition/nutrient amendment experiments (Rybczyk et al. 1996), twelve of fourteen experiments that lasted less than 100 days showed that nutrient amendments positively affected decomposition rates. Conversely, only six of fifteen studies lasting 200 days or longer showed any nutrient amendment effect on decomposition rates. Furthermore,



nutrient amendments have been shown to 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 carbon as the leaf litter decomposes, from labile C to more resistant ligninous C forms, which is less affected by exogenous nutrient supplies (Melillo et al. 1984). Therefore, while short term experiments (two weeks to three months) reveal short term rate changes in response to nutrient amendments, these changes may ultimately have no effect on total loss of mass over one or more years. This suggests that long term experiments, especially ones which describe decomposition with simple one compartment exponential decay models, or ones which simply compare total loss of mass at the end of the experiment, are not sensitive to processes which affect only the early stages of decomposition. This probably explains the results found during this study because the litter used for each experimental set was allowed to decay for a long period (285 and 372 days), relative to the short term experiments reviewed by Rybczyk et al. (1996). Since this study was primarily concerned 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.

The second factor that affects nutrient enrichment and decomposition experiments is whether or not the initial litter nutrient concentrations were changed by the treatment. Nutrient amendment effect experiments fall into two general categories; 1) "external" application experiments in which litter is distributed between plots or tanks which then receive various nutrient amendment treatments, and, 2) "internal" substrate quality experiments in which decomposition rates are compared between plants that are first grown under various nutrient amendment regimes that alter the plant tissue nutrient quality (Rybczyk et al. 1996). With external

experiments, the treatment effect is, "nutrients applied to the wetland", whereas with internal experiments, the treatment effect is, "initial litter nutrient concentrations". In general, internal fertilization experiments show a clear fertilizer effect on rates of decomposition while results from external experiments are less conclusive (Rybczyk et al. 1996).

To determine whether effluent additions have any effect on rates of litter decomposition in specific wetlands of interest, this "internal effect" factor must be carefully considered. For example, in a two-year long experiment that closely paralleled both the objectives and methods used in this study, Deghi et al. (1980) found that sewage effluent amendments had no effect on litter decomposition rates in four cypress dome swamps (two wastewater amendment sites and two control sites) in central Florida. However, all of the litter used for the experiment was collected from an entirely separate location, therefore the possible effects of nutrient amendments on initial litter nutrient concentrations and therefore, the overall effect of effluent on decomposition rates could not be determined. In contrast, the Thibodaux study was designed to allow for an internal nutrient effect on decomposition rates because, except for this first baseline year, the litter used for each experimental set during this study was collected from the site where it decomposed. Even though wastewater effluent amendments were shown not to affect initial litter nutrient concentrations, the general conclusion that wastewater effluent did not affect decomposition rates is more robust than the conclusions drawn from the Florida study because the effect of effluent nutrients on initial litter concentrations were considered.

### **Effects of Effluent on Initial Litter N and P Concentrations**

In contrast to this study, several researchers have demonstrated that wastewater effluent additions to wetlands increased initial litter nutrient concentrations (Chamie 1976; Brown 1981; Bayley et al. 1985; Aschmann et al. 1990). The lack

of effect observed in this study may be due to 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, except for a brief period in October, 1990, 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.

### **Effects of Effluent on Final N and P Concentrations and Immobilization**

Final leaf litter C:N and C:P ratios of 16:1 and 200:1 respectively are indicative of complete microbial decay (Brinson 1977). Therefore, 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 utilize exogenous sources of N and P, if available, to satisfy their demand. Additionally, increases in exogenous supplies of N and P should be reflected as increased litter N and P, net immobilization and higher final N and P concentrations in the decomposing litter (Howarth and Fisher 1976; Neely and Davis 1985). Immobilization of P was greater in the control site before nutrient applications began, but greater in the treatment site after. Additionally, both final N and P litter concentrations were higher in the control site before effluent amendments, but higher in the treatment site after effluent applications began.

The implication of these results are twofold. First, increased immobilization and higher final litter nutrient concentrations in response to nutrient amendments indicates that decomposer communities are utilizing the exogenous supplies of N and P supplied by the effluent, since initial concentrations were not affected. However, there was no corresponding increase in annual rates of decomposition. This suggests

that effluent additions would not decrease relative elevation (due to increased rates of organic matter decomposition) even though these additional nutrients are being utilized. Second, in terms of water quality, leaf litter is an effective sink for N and P because more nutrients are sequestered in the litter, in response to effluent additions, without a simultaneous increase in decomposition and nutrient re-mineralization.

## CONCLUSIONS

This study was unique because it used BACI analyses to statistically detect the effects wastewater effluents on decomposition in a natural system. The strength of this design was its ability to isolate background variation from the treatment effect. However, the weakness of this design, as it was applied here, was a lack of power due to the low degrees of freedom, even though four years of decomposition data were collected.

It was originally suggested 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, it was 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, possibly because, 1) the effluent did not affect initial litter concentrations and 2) because long term experiments such as this one, are not sensitive to processes which may only affect the early stages of decomposition. These results suggest that wastewater effluent could potentially increase relative elevation in these systems if rates of organic matter production increased.

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## **CHAPTER 4**

### **THE EFFECTS OF WASTEWATER EFFLUENT ON ACCRETION IN A SUBSIDING FORESTED WETLAND IN COASTAL LOUISIANA**

#### **INTRODUCTION**

Several recent studies have examined the interaction between accretion, relative sea level rise (subsidence + eustatic sea level rise), sediment deprivation and wetland loss in coastal marshes of the Mississippi Delta (Delaune et al. 1978; Delaune et al. 1983; Hatton et al. 1983; Mendelssohn et al. 1983; Salinas et al. 1986; Kesel 1988; Cahoon and Turner 1989; Conner and Brody 1989; Day and Templet 1989; Delaune et al. 1989; Reed 1989; Delaune et al. 1991; Nyman et al. 1990; Nyman et al. 1993; Callaway 1994). Relatively few, however, have measured these interactions in coastal forests of this region. In forested wetlands, high rates of relative sea level rise (RSLR), coupled with sediment deficits, lead to increasingly long periods of inundation and nutrient limitations, which are associated with decreased productivity, reduced regeneration and tree mortality (Conner et al. 1981; Delaune et al. 1983; Delaune et al. 1987; Conner and Day 1988; Conner and Day 1989; Dicke and Toliver 1990; Conner and Day 1991b; Conner et al. 1993; Pezeshki et al. 1993). Contributing further to these problems, many forested wetlands in the deltaic region have been hydrologically isolated from surrounding wetlands and streams due to an exponential increase in the construction of canals and spoil banks during the past century (Turner and Cordes 1987). In addition to impeding drainage and, in many cases, physically impounding wetlands, these spoil banks also prevent the overland flow of sediments and nutrients into coastal wetlands, creating essentially ombrotrophic systems.

Wetlands have been shown to 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), and numerous recent efforts to restore wetlands in the subsiding delta region have focused on attempts to balance vertical accretion deficits (accretion - RSLR) by either physically adding mineral sediments or sediment-rich water to wetlands or by constructing sediment trapping mechanisms or landforms (Boesch et al. 1994). As an alternative to adding mineral sediments to subsiding wetlands, Day et al. (1992) hypothesized that adding nutrient rich secondarily treated wastewater to hydrologically isolated, nutrient limited and subsiding wetlands could promote vertical accretion through increased organic matter production and deposition. Although the interaction between insufficient mineral sedimentation and RSLR has been widely believed to be one of the most important causes of wetland loss in coastal Louisiana, organic matter accumulation has also been shown to be a critical component of vertical accretion in Louisiana coastal wetlands. Mineral mass has been associated with accretion because the accumulation of sediment mass is highly correlated with mineral matter (Bricker-Urso et al. 1989), however, recent studies have shown that it is sediment organic matter that is most correlated with accretion, and not sediment mineral matter (Hatton et al. 1983; Gosselink and Hatton 1984; Bricker-Urso et al. 1989; Nyman et al. 1990; Nyman and Delaune 1991; Callaway 1994). This is because, while mineral matter controls mass, it is sediment organic matter and pore space that accounts for most of the soil volume, and accretion is a function of volume, not mass (Bricker-Urso et al. 1989).

Whether or not nutrient enrichment will ultimately increase wetland elevation is dependent upon the interactions between organic matter production, accretion and subsequent decomposition and compaction. As hypothesized, nutrient amendments could directly increase relative wetland elevation by stimulating productivity and

subsequent organic matter accretion (Day et al. 1992). Indirectly, the water and nutrients associated with wastewater could also increase relative elevation by contributing to anaerobic conditions which could reduce decomposition rates and organic sediment loss (Godshalk and Wetzel 1978; Conner and Day 1991a).

Alternatively, nutrient amendments could decrease relative elevation by increasing the rate of decomposition of organic matter, either by improving litter nutrient quality (Coulson and Butterfield 1978; Valiela et al. 1985; Lukumbuzya et al. 1994), or by increasing, externally, 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 began applying secondarily treated municipal wastewater to the Pointe au Chene Swamp, a hydrologically isolated and subsiding forested wetland in the deltaic region, as part of its tertiary wastewater treatment program, thus providing an opportunity to test the hypothesis that effluent nutrients could stimulate organic matter accretion and offset accretion balance deficits. The specific objectives of this study were to; 1) characterize the soil profile (% organic and mineral matter by weight and volume, bulk density and pore space) in the swamp receiving wastewater effluent 2) measure mineral and organic matter accumulation rates and soil formation processes in the swamp receiving wastewater effluent, 3) measure and compare accretion rates before and after wastewater application in both a control and treatment site and, 4) estimate subsidence rates in the swamp, using selected data from previous studies in this region, and calculate accretion balance deficits both before and after the application of wastewater effluent.

## **SITE DESCRIPTION**

The Pointe au Chene swamp lies on the backslope of the abandoned Bayou Lafourche distributary approximately 10 km southwest of Thibodaux, Louisiana (Figure 4.1). This site has been cutoff from its historical source of sediments since the

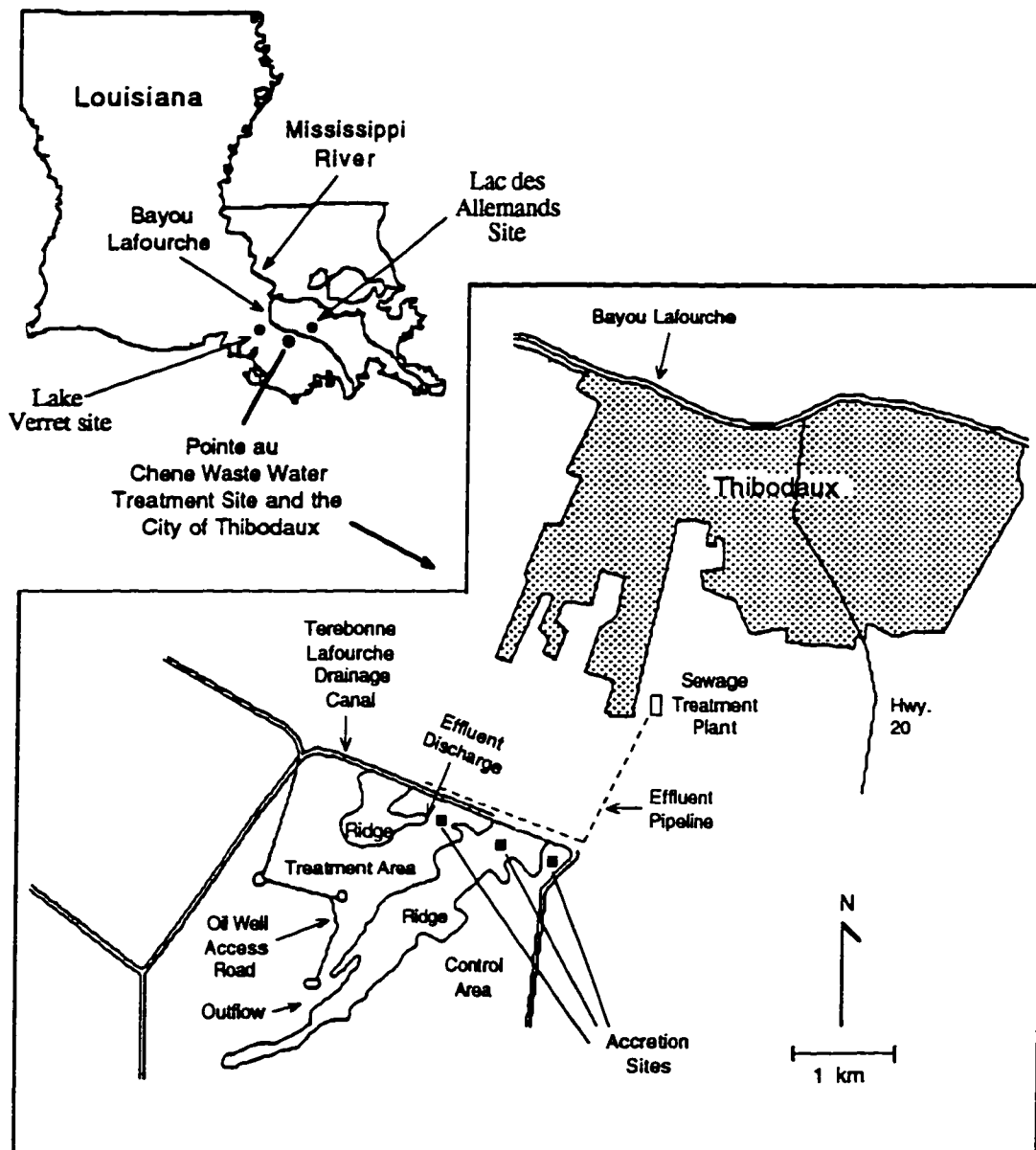


Figure 4.1. Map of the Pointe au Chene Swamp, located adjacent to the city of Thibodaux, Louisiana. An oil access road, a bottomland hardwood ridge, and the spoil banks associated with the Terrebonne-Lafourche drainage canal, hydrologically isolate the treatment site from the surrounding wetland.

1930's with the completion of flood control levees along the Mississippi. The study site consists of two almost continuously flooded forested wetlands, separated by a bottomland hardwood ridge, within a 1425 ha hydrologically restricted basin. The ridge site, oriented northeast to southwest (mean elevation = 1.16 meters above mean sea level (MSL)) is approximately 300 m wide and is vegetated primarily with oaks (*Quercus nigra* and *Q. obtusa*), sweetgum (*Liquidambar styraciflua*), American elm (*Ulmus americana*), palmetto (*Sabal minor*), and boxelder (*Acer negundo*). The two forested wetlands on either side of the ridge (mean elevation = 0.76 meters above MSL) are dominated by ash (*Fraxinus pennsylvanica*), black willow (*Salix nigra*), baldcypress (*Taxodium distichum*), water tupelo (*Nyssa aquatica*), red maple (*Acer rubrum*), and palmetto (*Sabal minor*).

Soils are classified as Fausse clay (very-fine, montmorillonitic, nonacid, thermic Typic Fluvaquents) and Sharkey clay (very-fine, montmorillonitic, nonacid, thermic Vertic Haplaquepts). They contain 60-80% clay in the B horizon, which effectively restricts groundwater flow (Conner et al., 1989b).

Mild climatic conditions in the region are 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/year. Rainfall can be quite variable and the total annual precipitation has ranged from 79 cm in 1962 to 222 cm in 1940.

Since March 1992, the 231 ha forested wetland site on the west side of the ridge has been receiving secondarily treated municipal wastewater at an average rate of  $7.5 \times 10^6$  L/day. Wastewater is discharged from 40 pipes located on the spoil bank that serves as the northern boundary of the site (Figure 4.1). The effluent then flows southward, between the ridge on the east and an oil access road on the west, and exits at a point where these two features nearly meet. In this report, the forested

wetland receiving wastewater effluent is referred to as the "treatment site", the swamp on the eastern side of the bottomland hardwood ridge as the "control site" and the ridge separating the two as the "ridge site".

Before wastewater effluent applications began, surface water nitrogen and phosphorus concentrations were similar in both the control and treatment sites (Day et al. 1989). Nitrogen concentrations were generally lower than, and phosphorus concentrations were similar to, values reported for surface waters in other southern bottomland hardwood and cypress swamps (Day et al. in press). The annual mean concentrations of  $\text{NH}_4\text{-N}$ ,  $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$  and TKN in the Pointe au Chene swamp during the 1988 - 1989 pre effluent period were 0.05 mg/L, 0.012 mg/L, and 1.34 mg/L respectively. During the same period, mean annual concentrations of  $\text{PO}_4\text{-P}$  and Total P were 0.24 mg/L and 0.43 mg/L respectively. Preliminary analyses of post-effluent surface water have shown that nitrogen and phosphorus concentrations have remained the same in the control site but have increased in the surface waters where all the treatment site experimental plots for this study were located. For example, in the treatment site, post effluent (1992 - 1994) mean annual concentrations of  $\text{NO}_3\text{-N} + \text{NO}_2\text{-N}$  increased to 5.0 mg/L and  $\text{PO}_4$  increased to 1.6 mg/L (Day et al. 1994).

#### **Relative Sea Level Rise at the Point au Chene Swamp**

Penland et al. (1988) analyzed the long term records from 33 USACE tidal gauges to estimate RSLR in the coastal region of Louisiana. Eleven of these gauges fell within a 40 km radius of the Pointe au Chene swamp and were considered for estimating RSLR in the Pointe au Chene swamp. Of these eleven, three were removed from the data set because gauge records did not span at least one lunar epoch as recommended by Penland et al. (1988). Additionally, the two stations located on Bayou Lafourche were not used because they measured RSLR within a confined channel regulated by numerous flood control structures such as dams, weirs, locks and levees and because

the weight of the Bayou Lafourche levees may have magnified the subsidence component of RSLR (Penland et al. 1988). Mean annual RSLR for the remaining seven gauges, all of which had complete records for the 1962 - 1982 lunar epoch, was  $1.23 \pm 0.34$  cm/yr.

Penland et al. (1988) also compared successive National Ocean Survey (NOS) first-order geodetic surveys to estimate the subsidence component of RSLR in the Terrebonne Parish region. In particular, NOS surveys L15631 (1955) and L2433-18 (1977) included two benchmarks situated along the southern border of the Pointe-au-Chene Swamp. By subtracting the 1977 survey elevation from the 1955 survey elevation, Penland calculated a subsidence rate of 0.53 cm/year for both stations. To estimate total RSLR, I added a eustatic sea level rise component of 0.12 cm/year (Gornitz 1982) to the geodetic subsidence to obtain a RSLR estimate of 0.65 cm/year for the Pointe au Chene Swamp.

## **METHODS**

### **Horizon Markers**

Two different horizon marker techniques were used to measure accretion. Feldspar markers were used to compare short term patterns of accretion between the control, treatment and ridge sites, before and after the application of wastewater to the treatment site, and  $^{137}\text{Cs}$ , a fallout by-product from above ground nuclear weapons testing was used as a marker to examine longer term accretional trends (30 years) in the Pointe-au Chene swamp.

### **Feldspar Markers**

Ten  $0.25 \text{ m}^2$  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 of 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 27 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 centimeters deep into the swamp sediment. Coring tubes were sealed in the field, to hold the cores in place, and stored in a vertical position until they were returned to the laboratory and frozen (unfrozen cores were too soft to extract from the cylinder and slice without disturbing the core and feldspar layer). Frozen cores were 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. October 1991 cores were collected during a natural drawdown period, while the August 1994 were collected when both sites were flooded. During flooded periods, accretion measurements were complicated by the presence of a flocculant fluid mud and organic muck layer (approximately 2 to 3 cm thick) in the standing water, and there was some question as to whether this layer represented actual accretion. Therefore, for the purposes of comparison, standing water in the flooded cores was allowed to evaporate in the laboratory, to mimic 1991 field conditions and to compact the fluid mud layer, before being frozen..

Measurements of accretion over the feldspar marker on the ridge site were taken in December 1991 (instead of October 1991 as in the control and treatment sites), and again in August 1994 (concomitantly with the control and treatment sites). For both sampling periods, measurements were made at five random locations within each plot by inserting a wide-bladed knife 15 to 20 centimeters 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.

A completely randomized design analysis of variance was used to test for differences in total accretion between all sites in October 1991 and in August 1994, and to test for differences in pre and post-effluent mineral and organic matter accumulation rates in the treatment site. An analysis of covariance (ANCOVA) was used to test for differences in accretion rates (separate slopes) between the control and treatment sites during the 34 month (December 1989 to October 1991) pre-effluent period and the 34 month (October 1991 to August 1994) post-effluent period. If significant differences were identified using these tests, individual sites were compared by linear contrasts using a Bonferoni correction for multiple comparisons (Neter et al. 1990).

### **$^{137}\text{Cs}$ Analyses**

Four cores were randomly collected from the treatment site in October 1993 for  $^{137}\text{Cs}$  analyses. Two of the cores (#s 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 centimeters 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. The standing water was allowed to evaporate in two of the coring tubes (#s 2 and 4) before freezing, in order to compact the fluid mud layer and two were frozen while still flooded. This was done to compare bulk density differences between the two sets of cores and calculate a sediment accretion correction factor for the fluid mud layer. Frozen cores were sectioned into 2 cm increments, dried at 60<sup>o</sup> C for at least 96 hours and ground to a fine powder with a Wiley Mill.  $^{137}\text{Cs}$  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<sup>3</sup> to over 1.0 g/cm<sup>3</sup> from the top to bottom of the cores,  $^{137}\text{Cs}$  activity was calculated on a volume basis, and

also normalized by weight. The  $^{137}\text{Cs}$  activity peak identified in each core profile corresponds with the soil surface during the peak of aboveground nuclear weapons testing and associated  $^{137}\text{Cs}$  fallout in 1963.

### **Sediment Accumulation and Soil Characteristics**

Since the measurement of  $^{137}\text{Cs}$  is non-destructive, each 2 centimeter increment was also subject to bulk density, % organic matter by weight, % mineral matter by weight and volume distribution (mineral, organic and pore space) analyses. Bulk density ( $\text{g cm}^{-3}$ ) 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 et al. (1974). Percent mineral matter (by weight) was then calculated as the remainder. Using an approximation of the particle density of mineral matter ( $2.62 \text{ g cm}^{-3}$ ) and organic matter ( $1.14 \text{ g cm}^{-3}$ ) (Delaune et al. 1983), volume distributions of organic and mineral matter were calculated as:

$$\% \text{ volume} = (\text{bulk density} \times \% \text{ weight min. or org.}) / \text{particle density}$$

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.

Using data obtained from the two cores that contained both a feldspar and a  $^{137}\text{Cs}$  marker, mean annual mineral and organic matter accumulation rates for the 25 year period bounded by the  $^{137}\text{Cs}$  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:

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

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<sup>2</sup>
- $A_y$  = Mineral or organic matter accumulation rate (g/m<sup>2</sup>/year) for period  $y$  (1963 to 1988 or 1988 to 1993)

## RESULTS

### Sediment Characteristics and Profiles in the Treatment Site

Water levels in the treatment site averaged  $25.1 \pm 1.5$  cm above the surface at the time that the cesium cores were extracted from the swamp and a thin layer (2-3 centimeters) of fluid mud and organic ooze was present at the water-sediment interface in all cores. Partially decomposed organic matter and live roots were abundant in the top 10 to 12 centimeters of the 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.

One of the cores (Core 1) extracted from the treatment site showed obvious evidence of bioturbation and was not used for analyses. Bulk density, organic and mineral matter, and volume distribution profiles were similar in the three remaining cores (Figure 4.2). Linear regression analyses revealed that there were significant ( $P < .05$ ) increases in bulk density ( $r^2 = .69$ ), % mineral matter by weight ( $r^2 = .69$ ) and % mineral matter by volume ( $r^2 = .68$ ) with depth. Bulk densities ranged from 0.11 g

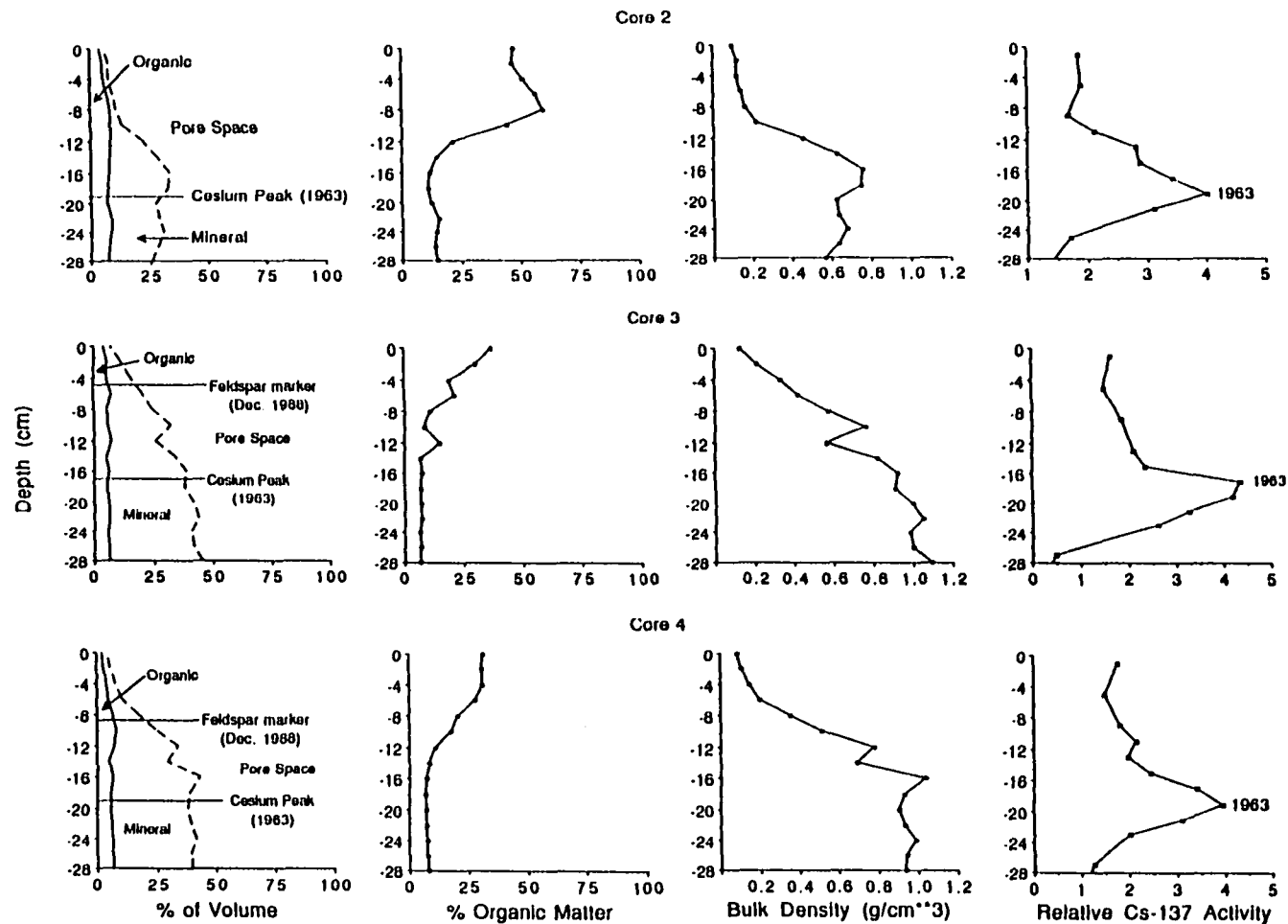


Figure 4.2. Volume distributions (percent organic, mineral and pore space), percent organic matter (by weight), bulk density and relative  $^{137}\text{Cs}$ , with depth, in cores taken from the treatment site in October 1993 at the Pointe au Chene Swamp near Thibodaux, Louisiana. Cores 3 and 4 also contained a feldspar layer that marked the surface in December 1988.

cm<sup>-2</sup> at the surface to over 1.0 g cm<sup>-2</sup> 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 2.0% at the surface to over 30.0% at 30 centimeters. In contrast, percent organic matter, by weight and % pore space, significantly decreased with depth ( $r^2 = 0.56$  and  $.69$  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 centimeters 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 ( $r^2 = .04$ ).

## Accretion

### Feldspar Markers

Total pre-effluent accretion (Figure 4.3), measured in October 1991, 34 months after the horizon markers were installed, averaged ( $\pm$  SE)  $22.3 \pm 3.2$  mm in the treatment site ( $n = 5$ ) and was not significantly different ( $P = 0.2633$ ) than the pre-effluent accretion measured in the control site ( $14.9 \pm 4.6$  mm,  $n = 7$ ) but was significantly greater ( $P = 0.0008$ ) than the accumulation measured on the ridge ( $9.5 \pm 1.4$  mm,  $n = 10$ ) (Table 4.1). Control and ridge site accretion was not significantly different (Table 4.1) ( $P = 0.1057$ ). Mean accretion rates for the pre-effluent period were  $7.8 \pm 1.1$ ,  $5.2 \pm 1.6$  and  $3.2 \pm 0.4$  mm/yr in the treatment, control and ridge sites respectively (Table 4.1). Total accretion measured in August 1994 (Figure 4.3), 68 months after the horizon markers were installed, and 29 months after effluent application began in the treatment site, averaged ( $\pm$  SE)  $54.6 \pm 1.5$  mm in the treatment site ( $n = 3$ ), and was significantly greater ( $P = 0.0002$ ) than total accretion after 68 months in the control ( $19.0 \pm 3.2$  mm,  $n = 5$ ) or the ridge site ( $14.9 \pm 2.8$  mm,  $n = 7$ ,

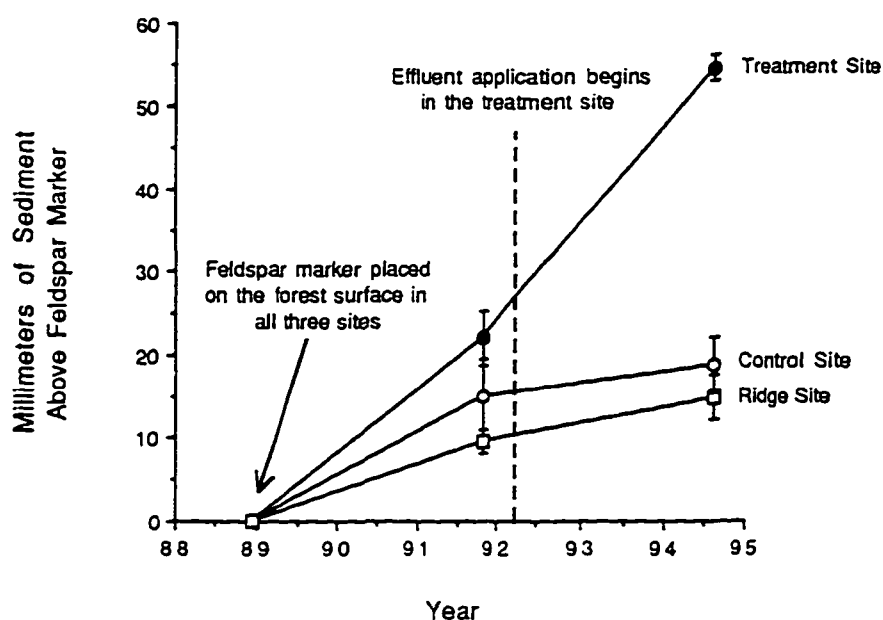


Figure 4.3. Vertical accretion of sediments (mean  $\pm$  se) in the treatment, control and ridge sites in the Pointe au Chene Swamp, between December 1988 and August 1994. Between the control and treatment sites, the rates of accretions (slopes of the sediment vs. time line) are not significantly different during the 1989 - 1991 time period, but are significantly different ( $P < .05$ ) after 1992.

Table 4.1. Total accretion and annual accretion rates (mean  $\pm$  se) measured in the Pointe au Chene swamp using feldspar and  $^{137}\text{Cs}$  markers. Underlines connect values that are not significantly different at the .05 level.

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

<sup>1</sup>Treatment site started receiving wastewater effluent in March 1992.

<sup>2</sup>Core samples on the ridge were collected two months after those in the ridge and control sites, therefore 10/91 date in the table is actually 12/91 for the ridge site.

<sup>3</sup>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.



$P = 0.0000$ ). Accretion in the control and ridge sites was not significantly different ( $P = 0.3612$ ). Accretion rates for the total 68 month period averaged  $9.7 \pm 0.3$  mm/year in the treatment site,  $3.4 \pm 0.6$  mm/year in the control site and  $2.7 \pm 0.5$  mm/year on the ridge. Sample size decreased with time because not all of the plots could be physically located after 34 months and even less could be located after 68 months.

The total accretion and accretion rates measured after 68 months integrate both pre and post-effluent conditions. Accretion during the October 91 to August 94, 29 month 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 4.1). Accretion rates during the post effluent time period were estimated at 11.4, 1.4 and 1.8 mm/yr in the treatment, control and ridge sites respectively. Accretion vs. time slopes (or the rates of accretion) were not significantly different (Figure 4.3) during the 34 month pre-effluent period ( $P = 0.1333$ ) but were significantly different during the 34 month post effluent period ( $P = 0.0043$ ).

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

### **$^{137}\text{Cs}$ Marker**

The three undisturbed cores extracted from the treatment site for  $^{137}\text{Cs}$  analyses all exhibited an obvious  $^{137}\text{Cs}$  peak (Figure 4.2). Cores 3 and 4 were extracted from the feldspar plots and contained two horizon markers, allowing for the comparison between background (1963 -1988) and background plus effluent (1963 - 1993)

accretion. Based on previous measurement of test cores and observed bulk density differences between wet and dry cores, 2.5 cm were subtracted from the surface of the cores that were frozen while flooded (Cores 2 and 4), before accretion rates were calculated, to correct for the fluid muck layer. Even though bulk density changed dramatically from the top to the bottom of the cores, the depth to peak  $^{137}\text{Cs}$  activity was the same whether activity was calculated on a core section (volume) basis, or normalized by section weight. The  $^{137}\text{Cs}$  distribution profiles shown in Figure 4.2 are normalized by weight. The mean thickness of the sediment layer between the 1963  $^{137}\text{Cs}$  marker and the 1993 surface ( $n = 3$ ) was  $166.7 \pm 8.3$  mm for the 30 year period, and accretion rates averaged  $5.5 \pm 0.3$  mm/yr. 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  $^{137}\text{Cs}$  marker. The average sediment thickness between these two layers was  $111.2 \pm 10.0$  mm for the 25 year period, resulting in a mean "background" accretion rate of  $4.4 \pm 0.4$  mm/year.

### **Accretion Balance Deficits**

To determine if sedimentation was keeping pace with RSLR, accretion balance deficits for the three sites were calculated by subtracting the RSLR, estimated from tidal gauge analyses, for the Point au Chene swamp (1.23 cm/yr) 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  $^{137}\text{Cs}$  cores were collected) was -7.9 mm/yr. All sites had negative accretion balances during the baseline, pre-effluent period (1988 - 1991), with sediment deficits ranging from -9.1 mm/yr on the ridge to -4.5 mm/yr in the treatment site (Figure 4.4). Accretion balances remained negative during the post-effluent period in all three sites, however,

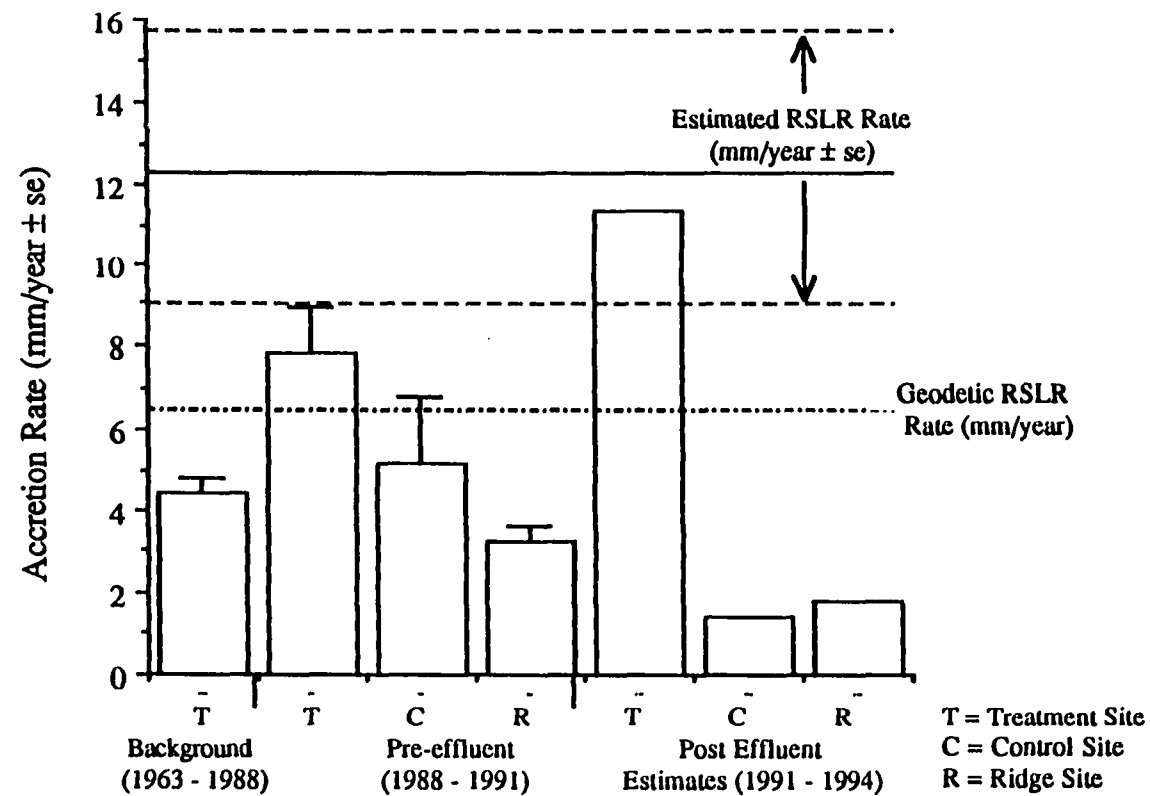


Figure 4.4. Background, pre-effluent and post-effluent accretion rates (mean ± se), relative sea level rise (RSLR) rates, and accretion balance deficits in the Pointe au Chene Swamp.

in the treatment site, the deficit was only -0.9 mm/year, and fell well within one standard error of estimated RSLR rate (Figure 4.4).

### **Mineral and Organic Matter Accumulation Rates**

In the treatment site, mean annual mineral accumulation rates between 1963 and 1988 were not significantly different ( $P = 0.0556$ ) than mineral accumulation rates during the 1988 through 1993 time period (Table 4.2). Organic matter accumulation rates, however, were significantly higher ( $P = 0.0157$ ) during 1988 - 1993 than in the earlier period (Table 4.2).

## **DISCUSSION**

### **Soil Profile Analyses**

Between depths of approximately 8 to 16 centimeters, cores 2 and 4, and to a lesser extent core 3 (Figure 4.2) exhibited sharp decreases in pore space and percent organic matter by weight with increasing depth, and, sharp increases in percent mineral matter by weight, percent mineral matter by volume and bulk density with increasing depth. This abrupt transitional zone might be explained by normal compaction and dewatering, sudden decreases in mineral inputs to the swamp, or sudden increases in organic matter inputs (or a sudden decrease in the loss of organic matter via reduced decomposition rates).

Percent mineral matter increases, both by weight and volume, and bulk density increases with depth are usually attributed to physical processes such as compaction and dewatering occurring within the sediment column (Gosselink and Hatton 1984), and these processes are probably partially responsible for the observed transition in the cores taken from the Pointe au Chene swamp, especially since there is a corresponding decrease in pore volume (Figure 4.2). However, the abruptness of this transitional layer suggests that other factors may also be responsible for the observed pattern. Sudden decreases in mineral inputs were probably not responsible for the transitions

Table 4.2. Average mineral and organic matter accumulation rates in the Pointe au Chene Swamp treatment site from 1963 - 1988 and from 1988 - 1993. Only organic matter accumulation rates were significantly different between time periods ( $P = 0.0157$ ).

<u>Time Period</u>	<u>Accumulation Rates (g dry weight /m<sup>2</sup>/year ± se)</u>	
	<u>Mineral Matter</u>	<u>Organic Matter</u>
1963 - 1988 <sup>1</sup>	2302.0 ± 29.4	275.9 ± 3.3
1988 - 1993 <sup>2</sup>	2004.6 ± 67.0	736.7 ± 58.3

<sup>1</sup>Soil horizon bounded by a <sup>137</sup>Cs marker on the bottom and a feldspar marker on the top.

<sup>2</sup>Soil horizon bounded by a feldspar marker on the bottom and the soil surface in 1993 at the top.

between 8 and 16 cm, because the accumulation data from this study indicates that mineral inputs have remained relatively constant since at least 1963 (Table 4.2).

Conner and Day (1988) examined the long term flooding history in two forested wetlands in coastal Louisiana, both located within 40 kilometers of the Pointe au Chene Swamp. They found that flooding duration increased suddenly in both swamps during the early 70's (from less than 5 days flooded per year to 300 days per year in one swamp and from approximately 100 days per year to 350 days per year in the other). After this period, flooding duration remained high through the end of their study in 1985. This sudden and eventually permanent increase was attributed to several successive years of major region-wide floods during the early 70's in combination with an accumulating accretion balance deficit throughout the period of record (1955 - 1985). If a similar inundation pattern occurred in the nearby Pointe au Chene swamp, the sudden onset of flooded and anaerobic conditions, and an associated decrease in decomposition rates after the early 1970's, might explain the increase in percent organic matter by weight, and corresponding decrease in bulk densities observed in cores taken from the treatment site. In support of this, the 16 centimeter depth at which the transitional zone begins, dates back to approximately 1972 (using the feldspar depth of 8.85 cm in the core 4 as a reference point, and an average accretion rate of 4.4 mm/year below the feldspar marker). Additionally, Bricker-Urso et al. (1989) also found abrupt transitional zones, similar to the ones described in this study, in some cores collected from salt marshes along the Atlantic coast. They similarly concluded that these changes corresponded with the period during which marsh development began in their sites, when organic materials began to accumulate.

### **Mineral and Organic Matter Accumulation Rates**

Since the primary source of mineral sediments to the Pointe au Chene swamp was eliminated when levees along the Mississippi were completed in the 1930's, I

initially suspected that I would find little or no mineral sediments above the 1963  $^{137}\text{Cs}$  marker horizon. However, post-1963 mineral accumulation rates measured during this study are higher than or within the range of published rates for coastal salt marshes in this region (Hatton et al. 1983; Callaway 1994) and comparable to accumulation rates for other bottomland hardwood forests (Johnston 1991). Because measured accretion rates on the ridge site were so low, and possibly overestimated, remnant bottomland hardwood ridges located within the confined wetland, such as the ridge site, may be the source of mineral sediments to the treatment swamp.

Unlike mineral matter accumulation, organic matter accumulation rates increased significantly during the 1988 - 1993 period. It is tempting to attribute this 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 (Culley and Epps 1973; 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 the 1963-1988 cohort and that the apparent increase in accumulation rates in the newer sediments may be an artifact of this.

### Accretion

Statistical comparisons (analyses of covariance: test for separate slopes) of total accretion over the feldspar markers between sites showed that there was no significant difference between the control and treatment site during the 34 month pre-effluent period, but revealed significant differences 34 months later, 27 months after effluent application began in the treatment site. More importantly, the *rates* of accretion

between the control and treatment site were similar during the 34 pre-effluent period, but were greater in the treatment site than the control during the post-effluent period. Although the estimated accretion rates in the treatment site were greater in the post-effluent period (11.4 mm/yr) than in the pre-effluent period ( $7.8 \pm 1.1$  mm/yr), no within site, between time, statistical comparison were made because post-effluent accretion was estimated by subtracting the December 1988 to October 91 accretion from the December 1988 to August 1994 accretion. This estimate would only be correct if it were assumed that there was no decomposition or compaction of the December 1988 to October 1991 sediment cohort, and this probably not the case. However, the net result is to underestimate post-effluent accretion, so the actual difference between pre and post-effluent accretion is probably greater.

These statistical models used in these analyses do not test whether wastewater effluent influenced accretion rates in the Pointe au Chene swamp. Studies such as this one, which attempt to detect the effect of some ecosystem-scale perturbation (wastewater effluent in this case) on one or more response variables are difficult to analyze statistically because of problems associated with inadequate or non-existent replication (Hurlbert 1984; Carpenter and Lodge 1986; Stewart-Oaten et al. 1986; Osenberg et al. 1994). Often, these problems arise because the perturbation was unplanned (e.g. oil spills, hurricanes) or, as in the case of this experiment, because of the expense, impracticality or impossibility of replicating treatments (e.g. another forested wetland used for wastewater effluent plus replicated controls). 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, simultaneous accretion measurements were taken in both a control and treatment site, both before and after effluent applications began in the treatment site, and we found that accretion rates in the treatment site changed in relationship to rates in the control after wastewater was



applied to the treatment site (an interaction effect). Second, estimated accretion in the treatment site was eight times greater than accretion in the control during the post-effluent period while pre-effluent accretion was not shown to be significantly different between the control and the treatment, even though the sample size was greater during the pre-effluent period. Third, 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. Fourth, soil profile analysis (Table 4.2) in the treatment site suggest that organic matter accretion has increased since wastewater effluent additions began. Finally, concomitant studies in this swamp showed that hydroperiods were similar in both sites over the 68 months of this experiment and that decomposition rates were not affected by wastewater effluent (Chapter 3). Therefore, I cannot propose any mechanism, other than effluent stimulated organic matter production, which would have caused the observed increase in accretion rates in the treatment site relative to the control.

### **Accretion Balance Deficits**

The ultimate success of this type of restoration effort will depend upon whether or not wastewater effluent can stimulate primary production to the point that organic 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 1994). As a result, accretion balance deficits are underestimated because accretion rates are overestimated. A conceptual diagram of this process is shown in Figure 4.5; although yearly inputs of sediments over the feldspar marker remain

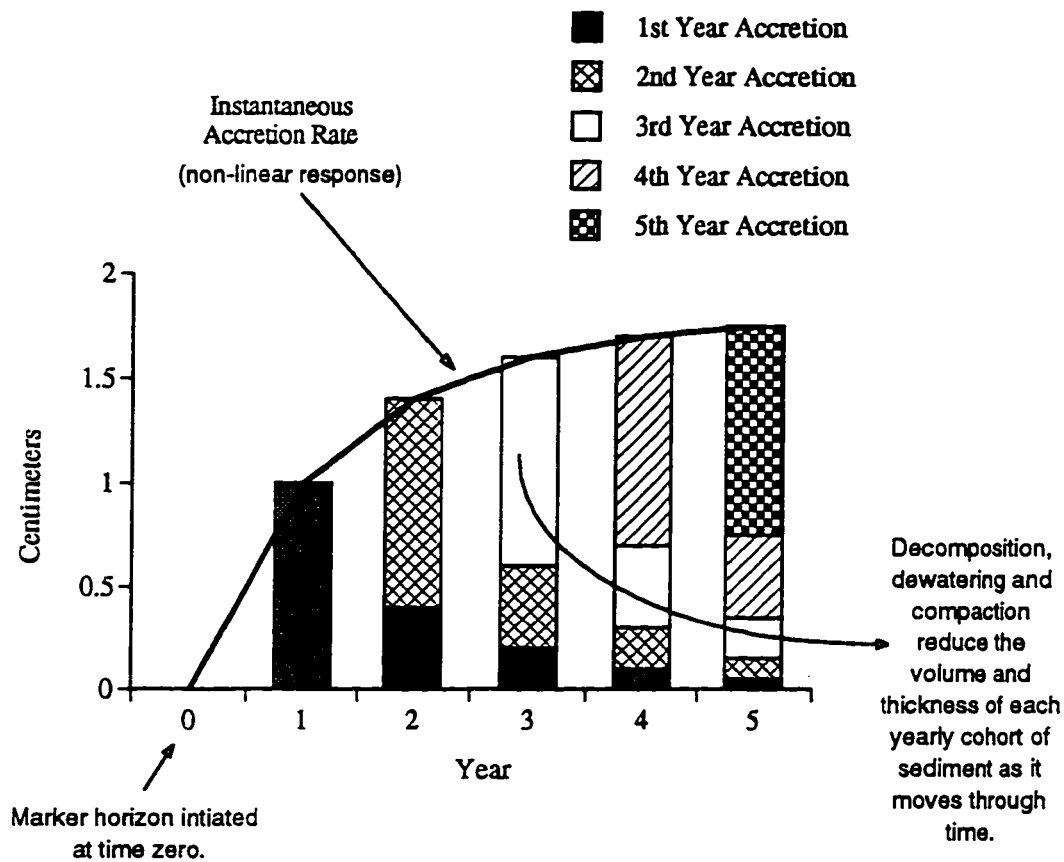


Figure 4.5. Conceptual graph of sediment accretion dynamics above a marker horizon over time. Given that the yearly cohort of sediment is constant, the instantaneous rate of accretion (solid line) decreases with time because of decomposition, dewatering and compaction.

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. Additionally, different methods of estimating RSLR yield different results. For example, the estimate of RSLR rates obtained from Penland's (1988) geodetic survey data was approximately half the mean rate obtained from tidal gauge analysis.

Therefore, in the short term, it is difficult to know for certain if post-effluent accretion is keeping pace with RSLR in the Pointe au Chene swamp. The post-effluent estimated rate of accretion in the treatment site (11.4 mm/yr) was less than the estimated rate of RSLR (12.3 mm/yr), but did fall within one standard error of the mean (Figure 4.4) and was well above the RSLR rate estimated from geodetic surveys (Penland 1988). As mentioned above, post-effluent accretion may be overestimated because decomposition and compaction have not been fully integrated into the rates equation calculated for recent sediments. However, this is partially balanced by the fact that post-effluent accretion was estimated by subtracting the December 1988 to October 91 accretion from the December 1988 to August 1994 accretion and, as discussed previously, this method underestimates post-effluent accretion.

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, because water level records vary substantially from year to year and from decade to decade (Turner 1991). Therefore, the accretion balance deficit of -7.9 mm/yr, calculated during this study using accretion rates obtained from  $^{137}\text{Cs}$

horizons (1963 - 1988) and RSLR rates obtained from Penland's (1988) tidal gauge analysis (1962 - 1982), 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 in the treatment is probably an underestimate of the long term deficit rate, it is an order of magnitude less than background deficits.

## CONCLUSIONS

Normal processes related to the deltaic cycle have led to high rates of submergence in Louisiana coastal wetlands. Additionally, the construction of extensive flood control levees along the Mississippi River have essentially eliminated annual subsidies of compensating sediments and nutrients associated with overbank flooding which once allowed these wetlands to keep pace with subsidence, resulting in extensive wetland loss in the coastal region. This and other studies suggest that the Point au Chene swamp has been more or less permanently flooded since the early 1970's. Results from this study suggest that wastewater effluent can potentially be used as a restoration tool in these types of subsiding systems by stimulating organic matter accretion and balancing accretion deficits. However, long term monitoring would be useful in order to fully integrate the effects of decomposition and compaction into the accretion balance equation. Furthermore, results from other long term wetland wastewater treatment studies suggest that it may be several years after the initiation of wastewater effluent before net primary production, and rates of organic matter accretion, reach a new and higher equilibrium (Knight 1992).

Currently, 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<sup>-1</sup> rate of RSLR (submergence + world wide

eustatic sea level rise) measured in the Louisiana delta region (Boesch et al. 1983; Penland and Ramsey 1990). While this study, and the issue of wetland loss due to high rates of RSLR, may presently only apply to the extensive wetlands associated with subsiding delta regions, the worldwide eustatic sea level rise component of RSLR is expected to steadily increase over the next century due to the impacts of global warming (Gornitz 1982). Therefore, this region can serve as a model for other coastal wetlands, which may also face problems associated with rising water levels in the near future.

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## **CHAPTER 5**

### **PRODUCTIVITY, STRUCTURE AND FOLIAR NUTRIENT CONCENTRATIONS IN A FORESTED WETLAND RECEIVING WASTEWATER EFFLUENT**

#### **INTRODUCTION**

In the hydrologically isolated, deltaic wetland forests of southeastern Louisiana, researchers have documented a trend towards decreasing productivity and increasing inundation, due to estimated vertical accretion deficits (relative sea level rise - accretion) ranging from 2.5 to 10.8 mm yr<sup>-1</sup> (Conner et al. 1981; Conner and Day 1988; Delaune et al. 1988; Conner and Brody 1989; Conner and Day 1989; Dicke and Toliver 1990; Conner and Day 1992; Conner et al. 1993). Productivity decreases observed in these types of wetlands (frequently flooded and hydrologically isolated) have been attributed to either the direct physio-chemical effects of flooding (i.e. anoxia or toxicity due to the reduced species of S and Fe), flood related nutrient limitations (i.e. denitrification or the inhibition of mineralization), nutrient limitations due to a reduction in allochthonous nutrient supplies or, most likely, some combination of these factors (Mitsch et al. 1979; Brinson et al. 1981; Brown 1981; Mitsch and Gosselink 1986; Conner and Day 1989). Community response to these factors include (i) increased tree mortality, (ii) shift in species composition towards less desirable and less productive water tolerant scrub species, and (iii) inhibition of germination and regeneration (Conner et al. 1981). Day et al. (1992) hypothesized that adding nutrient rich secondarily treated wastewater to these hydrologically isolated and subsiding wetlands could promote vertical accretion, and balance the accretion deficit, through increased organic matter production and deposition. This hypothesis operates under the

assumption that the observed decreases in productivity in these systems are due not only to extended periods of inundation, but also to decreased nutrient inputs.

In an extensive review, Lugo et al. (1988) suggested a forested wetland classification system, adapted from an earlier system proposed for mangrove wetlands (Lugo and Snedaker 1974), based on wetland hydrology, geomorphology and nutrient supply (Table 5.1). They concluded that nutritional factors interact with, and are sometimes more critical than, hydrologic factors in controlling productivity. Forested wetlands classified as "riverine" were found to be the most productive and were characterized by seasonal inundation's of nutrient rich floodwaters. Alternatively, forests classified as "basin" wetlands were the least productive and were described as nutrient poor with restricted hydrologic exchange. Hydrologically isolated, subsiding forested wetlands in the deltaic region of Louisiana, could essentially be considered as once productive systems that developed under a riverine regime but, due to recent anthropogenic alterations, are now growing under basin-like, productivity limiting, conditions. This is an important point and lends support to the hypothesis that wastewater effluent could increase organic matter production and accretion in these systems, because species that characteristically grow in nutrient rich habitats may be more responsive to nutrient subsidies than species adapted to growing in nutrient limited sites (Chapin et al. 1986).

The overall objective of this study was to describe the forest structure of the Pointe au Chene Swamp, a continually inundated and subsiding wetland located in the coastal zone of southeastern Louisiana and to measure the effect of wastewater effluent additions on above ground productivity, and, tree nutrient uptake and nutrient use efficiency (NUE). Specifically we hypothesized; 1) stem productivity and leaf litter fall would increase in response to effluent additions, 2) green leaf N, P, K, Mg, Ca and Fe concentrations would increase in response to effluent additions and, 3) litterfall

Table 5.1. Productivity and core environmental factors affecting two different types of forested freshwater wetlands (After Lugo et al. 1988).

<u>Wetland Type</u>	<u>Kinetic Energy of Water Flow</u>	<u>Hydroperiod</u>	<u>Nutritional Factors</u>	<u>Mean Aboveground Primary Production (T/ha/year)</u>
Riverine	High	Seasonal Flooding	Nutrient Rich	12.65
Basin	Low	Frequent Permanent Flooding	Variable, frequently low nutrient	5.96

N, P, K, Mg, Ca and Fe concentrations (an index of NUE (Vitousek 1982)) would increase in response to effluent additions. To address the question concerning primary productivity, we measured forest structure and aboveground productivity for three years before, and for four years after, applications of secondarily treated wastewater effluent began in a designated effluent treatment site within the Pointe au Chene wetland. Similar measurements were collected concurrently in a "control" section of the swamp that never received wastewater effluent.

To detect changes in patterns of nutrient uptake and use efficiency, we utilized indices of plant nutrient status and use efficiency originally proposed by Vitousek (1982). Vitousek (1982, 1984), analyzed litterfall nutrient data from numbers of published ecosystems studies in temperate and tropical forests and concluded that the dry mass biomass to nutrient mass ratio (or simply the inverse of the nutrient concentration) in litterfall was a good index of nutrient use efficiency (NUE). He found the highest ratios of litterfall dry mass to litterfall nutrient mass in nutrient limited sites, and the lowest ratios in non-limited sites. At that time, two possible mechanisms were proposed to account for this phenomenon; 1) increased NUE in active leaves (i.e. trees in lower nutrient sites would fix more carbon per unit nutrient than trees for high nutrient sites) and , 2) increased resorption of nutrients from leaves prior to leaf abscission (Vitousek 1982). Later, others found that, in the few studies that compared one species across a fertility gradient, there was no correlation between resorption of nutrients (mechanism 2) and plant nutrient status (Chapin 1991; delArco et al. 1991). Chapin (1991) did find, however, that within species, an increase in plant nutrient status was correlated with an increase in nutrients lost in via leaf litter. Given that the principal mechanism explaining Vitsousek's index of NUE is the amount of carbon fixed per unit nutrient in active leaves, then within species, green leaf nutrient concentrations would also be a good indicator of plant nutrient status.

In practice, both green leaf nutrient concentrations (Schlesinger 1978; Brinson 1984; Dierberg et al. 1986; Hopmans et al. 1990) and leaf litter (Schlesinger 1978; Gomez and Day 1982; Killingbeck 1986; Lugo et al. 1988; Hopkinson 1992) have been used as indicators of plant nutrient status in forested wetland systems. For this study we also measured green leaf and litter nutrient concentrations in the control and treatment site, both before and after effluent additions began in the treatment site.

## **SITE DESCRIPTION AND METHODS**

### **Site Description**

The Pointe au Chene study site (Figure 5.1) has been previously described in Chapters 3 and 4 of this document, and by Rybczyk et al. 1995. Aerial infrared imagery was used to identify the physical features within the study area and to delineate various vegetational zones during a pre-effluent baseline study. Preliminary surveys indicated that most of the forest was uniform; therefore, the pre and post-effluent studies were focused on the northern section of the study area. Baseline studies in both sites began in 1988. Effluent additions began in the treatment site in March 1992, and continue to present at an average rate of  $7.5 \times 10^6 \text{ L day}^{-1}$ . In the treatment site, all plots, described below, were located within 100 meters of the effluent impact zone. Control plots were located in an analogous region of the control site (Figure 5.1).

### **Productivity and Forest Structure**

Six 10 x 100 meter plots were established in the study area, three in the treatment site and three in the control site. Within each plot all trees  $\geq 10$  cm in diameter at breast height (dbh) were tagged. Initially, 130 and 167 trees were tagged in the treatment and control sites respectively in 1988, although the number of tagged trees per site changed from year to year due to recruitment into the size class, or tree mortality. Shrubs (individuals  $< 10$  cm dbh but  $\geq 2.5$  cm dbh) were also tagged in two 5 meter x 5 meter subplots established within each 10 meter x 100 meter plot. For both

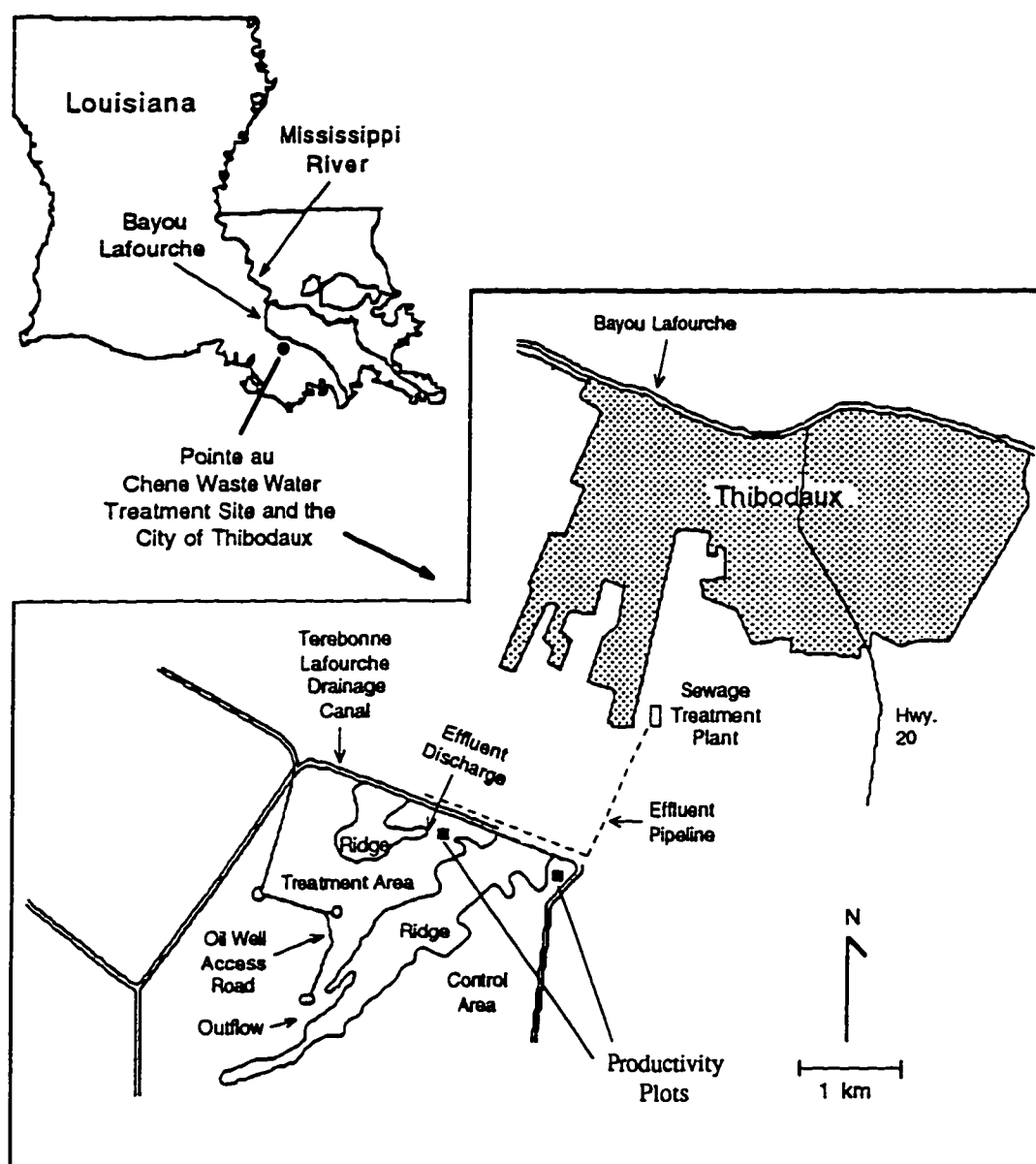


Figure 5.1. Map of the Pointe au Chene Swamp, located adjacent to the city of Thibodaux, Louisiana. An oil access road, a bottomland hardwood ridge, and the spoil banks associated with the Terrebonne-Lafourche drainage canal, hydrologically isolate the treatment site from the surrounding wetland.

trees and shrubs, species and dbh were recorded yearly from December 1988 through December 1995. Tree and shrub biomass, was estimated using dbh vs. tree biomass regression equations, calculated for each species in similar forests in southeastern United States (Conner and Day 1976). Change in biomass from year to year represented wood production for the year.

Table 5.2. Repeated measures design for detecting an impact on productivity due to effluent additions in the treatment site.

<u>Source of Variation</u>		<u>Degrees of Freedom</u>
Control vs. Treatment = CT		1
Plot (CT)	(random)	4
Before vs. After = BA		1
Year (BA)	(random)	4
CT x BA	(test for impact)	1
CT x Year (BA)	(random)	4
Residual		20

Within each plot, ten 0.25 m<sup>2</sup> traps, 15 cm deep, (n = 30 traps per site) were set one meter above the ground to collect leaf litter. Litter was collected monthly from November 1988 through December 1990 and resumed again from January 1992 until March 1996. Litter was separated into leaf and woody material, dried at 60° C and weighed. Monthly leaf litterfall was summed for each trap to obtain annual leaf fall for 1989, 1990, 1992, 1993, 1994 and 1995.

Yearly total aboveground productivity (wood production plus leaf litterfall) was calculated in both sites, for the two pre-effluent years 1989 and 1990, and four post-effluent years, 1992, 1993, 1994 and 1995. Because individual trees were repeatedly sampled over time, a repeated measures design, within a BACI format (Underwood



1991), was employed to test for a Control/Treatment vs. Before/After interaction, indicating a possible treatment effect on productivity (Table 5.2).

To examine forest structure over time, relative dominance and density was calculated for each year and site (1989 - 1995), for all recorded tree species. Relative dominance was calculated as :  $((\sum \text{ basal area for a given tree species} / \sum \text{ basal area of all tree species}) \times 100)$ . Relative density was calculated as  $((\sum \text{ individuals of a given tree species} / \sum \text{ of all individuals of all tree species}) \times 100)$ .

### **Green Leaf Nutrients**

Canopy leaves were randomly sampled in the control and treatment sites on August 1989 (before impact) and August 1993 (after impact) from swamp red maple (*Acer rubrum* var. *drummondii*), green ash (*Fraxinus pennsylvanica*), black willow (*Salix nigra*) and baldcypress (*Taxodium distichum*). Samples were collected from three different trees for each species, site and year. Samples were dried at 60° C for 72 hours and ground with a Wiley mill using a size 60 mesh screen. Tissue concentrations of P, Ca, Mg, K, and Fe for each sample were determined using an inductively coupled plasma spectrometer (ICP) after a nitric acid digest. Samples were analyzed for N by direct combustion with a C-H-N analyzer. Because leaf samples were collected randomly during each time period, a three way factorial ANOVA was performed for each nutrient, to detect any significant two way, site x year, interaction or a significant three way (species x time x location) interaction. Orthogonal contrasts were used to identify significant differences in tissue nutrient concentrations between the control and treatment for each year if a significant time x location interaction was identified, or to detect specific differences between the treatment and control for each year and species, if a significant three way interaction was indicated.

## **Litter Nutrients**

In 1989, and again in 1993, October leaf litter (peak litterfall month (Rybczyk et al. 1995)) was pooled by plot, ground with a Wiley mill through a 60 mesh screen, and analyzed for P, Ca, Mg, K and Fe, using the methods described for green leaf samples. Because Vitousek's NUE litter fall index is described as a collective property of the individuals of all species that occur in that community (Vitousek 1982), leaf litter was not separated by species. A two way factorial (site by year) was performed for each nutrient to detect any significant two way interaction. If any interaction was identified, orthogonal contrast were used to detect differences between sites for each year.

## **RESULTS AND DISCUSSION**

### **Forest Structure**

Within the eight year period of record, a total of seven different tree species were found in treatment site, while eight were located in the control site (Table 5.3). However, three flood tolerant species, red maple, black willow and green ash, classified as obligate, facultative-wetland and obligate species respectively, dominated the landscape, and together accounted for no less than 81 % of the total number of individuals in either the treatment or control site for any year. For all years, red maple was the most numerous species in the both the treatment and control sites (Table 5.3). However, as ranked by total basal area, green ash dominated the control site throughout the study, and was co-dominant with red maple in the treatment site (Table 5.4). In the treatment site, all species except for baldcypress decreased in absolute numbers from 1988 through 1995 with black willow exhibiting the most dramatic decline, from 37 trees in 1988 to only 6 in 1995. In the control site however, all species increased in absolute numbers during the same period except for three relatively

Table 5.3. Relative density values ((total number of stems for a given tree species/total number of stems for all tree species)x100), and the absolute densities, in parathesis, in the control and treatment sites in the Pointe au Chene Swamp, from 1988 through 1995. Effluent applications began in the treatment site in March 1992.

<u>Treatment Site</u>	<u>1988</u>	<u>1989</u>	<u>1990</u>	<u>1991</u>	<u>1992</u>	<u>1993</u>	<u>1994</u>	<u>1995</u>
<i>Acer rubrum</i>	48.5 (79)	50.9 (82)	52.4 (84)	51.5 (86)	54.0 (81)	54.7 (81)	51.7 (74)	62.6 (42)
<i>Salix nigra</i>	22.7 (37)	23.6 (38)	24.2 (39)	24.5 (51)	21.3 (32)	21.6 (32)	18.8 (27)	8.9 (6)
<i>Fraxinus pennsylvanica</i>	21.5 (35)	18.0 (29)	16.1 (26)	15.5 (26)	17.3 (26)	16.9 (25)	23.1 (33)	23.8 (16)
<i>Ulmus rubra</i>	3.7 (6)	3.7 (6)	3.7 (6)	3.7 (6)	2.7 (4)	2.7 (4)	2.8 (4)	1.5 (1)
<i>Liquidambar styraciflua</i>	1.8 (3)	1.8 (3)	1.8 (3)	1.8 (3)	2.0 (3)	1.3 (2)	0.7 (1)	0.0 (0)
<i>Quercus obtusa</i>	1.2 (2)	1.2 (2)	1.2 (2)	1.2 (2)	0.6 (1)	0.8 (1)	0.7 (1)	0.0 (0)
<i>Taxodium distichum</i>	0.6 (1)	0.6 (1)	0.6 (1)	1.8 (3)	2.0 (3)	2.0 (3)	2.1 (3)	2.9 (2)
<u>Control Site</u>								
<i>Acer rubrum</i>	38.8 (68)	39.3 (70)	39.9 (73)	40.4 (80)	41.3 (81)	41.5 (83)	43.6 (96)	40.8 (85)
<i>Fraxinus pennsylvanica</i>	37.7 (66)	37.6 (67)	37.1 (68)	36.4 (72)	36.2 (71)	35.5 (71)	33.6 (74)	35.6 (74)
<i>Salix nigra</i>	12.6 (22)	12.3 (22)	12.6 (23)	12.1 (24)	12.7 (25)	12.5 (25)	11.8 (26)	12.0 (25)
<i>Taxodium distichum</i>	3.4 (6)	3.4 (6)	3.3 (6)	4.5 (9)	4.6 (9)	5.5 (11)	6.8 (15)	6.7 (14)
<i>Ulmus rubra</i>	2.8 (5)	2.8 (5)	2.7 (5)	2.6 (5)	2.0 (4)	2.0 (4)	1.4 (3)	0.9 (2)
<i>Quercus nigra</i>	2.4 (4)	2.2 (4)	2.2 (4)	2.0 (4)	1.5 (3)	1.5 (3)	1.4 (3)	0.9 (2)
<i>Quercus obtusa</i>	1.8 (3)	1.7 (3)	1.6 (3)	1.5 (3)	1.5 (3)	1.5 (3)	1.4 (3)	1.4 (3)
<i>Liquidambar styraciflua</i>	0.5 (1)	0.6 (1)	0.5 (1)	0.5 (1)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)

minor species, slippery elm (*Ulmus rubra*), *Quercus obtusa* and sweetgum (*Liquidambar styraciflua*).

The observed decrease in absolute densities in the treatment site is, in part, a reflection of severe nutria (*Myocastor coypus*) herbivory observed and documented by Hesse et al. (1996) during 1994 and 1995, and in part due to the passage of Hurricane Andrew over the site in 1992 (Rybczyk et al. 1995). Through 1994, herbivory was restricted to the treatment site, and was essentially limited to black willow. By 1995, 91% of all black willow in the treatment site were impacted by nutria herbivory, commonly resulting in tree death (Hesse 1996). In 1995, some nutria herbivory was observed in the control site but it was not as extensive as in the treatment site. The intensity of the nutria impact in the treatment, compared to the control may be related to effluent applications and the improvement in food quality at this site. In a related study, Zhang (1995) found that after effluent applications began in 1992, concentrations of N, P, S and K in duckweed (*Lemna minor*), an important food source for nutria, were twice as high in the treatment site, compared to the control. Additionally, this study revealed that, in 1993, black willow green leaf N and P concentrations in the treatment site were significantly higher than green leaf N and P concentrations in red maple, green ash and baldcypress in the treatment site, possibly explaining why willow was selectively targeted. Others have also noted incidents of selective herbivory based upon food quality. In an extensive review, Mattson (1980) documented a herbivore preference for high N foods. Brown (1975) found that nutria densities in polluted farm ponds in Florida were four times higher than the densities found in similar non-polluted ponds.

Some portion of the observed decline in stem density in the treatment site can also be attributed to Hurricane Andrew. On 26 August 1992, the eye of the storm passed within approximately 80 km of the Pointe au Chene wetlands. Local observers

estimated sustained winds of 35 m/s (80 mph), with gusts to 44 m/s (100 mph) between the hours of 2:00 a.m. and 4:00 a.m. CST on the morning of the 26th. Rybczyk et al. (1995) documented moderate hurricane induced tree mortality in both the treatment and control sites. However, black willow, which at the time was more common in the treatment than in the control (Table 5.3 and Table 5.4) and had a higher relative density and dominance, was particularly vulnerable to storm damage. Doyle et al. (1995) also noted that black willow sustained a disproportionate amount of storm related damage during the passage of Hurricane Andrew over the Atchafalaya Basin.

### **Production**

Baseline (1989 and 1990) aboveground production values (stem growth plus litterfall) in the control and treatment sites (Table 5.5) were similar to the mean values reported for forested wetlands in general (Lugo et al. 1988), and for Louisiana wetlands specifically (Conner 1994). After effluent additions began in the treatment site in 1992, total aboveground production decreased in both sites (Figure 5.2), and statistical analyses revealed no significant time x site interaction ( $P = 0.4934$ ), indicating no treatment effect. In 1993, after the passage of Hurricane Andrew, leaf litterfall was reduced to approximately half the values recorded in both sites during 1990 (Table 5.5), and never recovered through 1995. Total stem production also decreased in both sites through the duration of the study (Table 5.5) and was probably the result of Hurricane Andrew (Rybczyk et al 1995), nutria herbivory (Hesse 1996) and prolonged inundation (This publication). Although some forested wetland tree species, such as cypress and tupelo, can thrive under flooded conditions (Conner 1989; Dicke and Toliver 1990), continual inundation can lead to decreased productivity and increased mortality even in these flood tolerant species (Eggler and Moore 1961; Harms et al. 1980). Other forested wetland species are less tolerant of prolonged inundation (Conner and Brody

Table 5.4. Relative dominance values ((total basal area for a given tree species/total basal area for all tree species)x100) in the control and treatment sites in the Pointe au Chene Swamp, from 1988 through 1995. Effluent applications began in the treatment site in March 1992.

<u>Treatment Site</u>	<u>1988</u>	<u>1989</u>	<u>1990</u>	<u>1991</u>	<u>1992</u>	<u>1993</u>	<u>1994</u>	<u>1995</u>
<i>Acer rubrum</i>	35.8	38.1	39.7	39.5	39.4	39.1	42.9	33.9
<i>Salix nigra</i>	12.1	13.6	14.7	15.2	13.6	13.3	11.6	4.2
<i>Fraxinus pennsylvanica</i>	42.7	38.7	35.6	34.9	37.5	36.8	39.0	60.2
<i>Ulmus rubra</i>	2.9	2.8	2.9	2.9	1.8	1.8	1.5	0.4
<i>Liquidambar styraciflua</i>	4.7	4.8	4.9	4.8	5.2	5.0	2.0	0.0
<i>Quercus obtusa</i>	0.6	0.6	0.8	0.8	0.3	0.3	0.4	0.0
<i>Taxodium distichum</i>	1.2	1.2	1.4	1.8	2.1	2.4	2.8	1.2
<u>Control Site</u>								
<i>Acer rubrum</i>	18.6	19.3	19.4	20.4	21.2	20.7	22.7	19.5
<i>Fraxinus pennsylvanica</i>	55.6	54.2	53.1	51.7	53.4	53.0	52.3	53.8
<i>Salix nigra</i>	7.5	8.2	9.0	9.6	10.9	11.2	10.5	11.5
<i>Taxodium distichum</i>	10.2	10.2	10.4	10.4	11.4	11.9	12.2	13.6
<i>Ulmus rubra</i>	2.8	2.7	2.7	2.5	1.5	1.5	0.7	0.5
<i>Quercus nigra</i>	4.2	4.3	4.2	4.3	0.9	0.9	0.9	0.5
<i>Quercus obtusa</i>	0.5	0.5	0.6	0.6	0.6	0.6	0.64	0.7
<i>Liquidambar styraciflua</i>	0.5	0.5	0.5	0.5	0.0	0.0	0.0	0.0

Table 5.5. Mean stem density ( $\pm$ s.e.), mean basal area ( $\pm$ s.e.), mean annual litterfall ( $\pm$ s.e.), biomass ( $\pm$ s.e.) mean annual stem productivity ( $\pm$  s.e.) and total aboveground productivity (stem productivity + litterfall) for the years 1988 - 1995 in the treatment and control sites in the Pointe au Chene wetlands.

	1988	1989	1990	1991	1992	1993	1994	1995
<b>A. Density (#/ha)</b>								
1. Shrubs								
a. Treatment	3200 (115)	3133 (176)	3200 (346)	3400 (529)	3200 (462)	3000 (611)	2866 (520)	-
b. Control	5000 (346)	5000 (346)	5133 (333)	5066 (480)	4600 (400)	4600 (416)	5066 (705)	4933 (867)
2. Trees								
a. Treatment	543 (20)	537 (22)	537 (23)	557 (27)	500 (38)	493 (38)	473 (57)	223 (57)
b. Control	583 (58)	593 (58)	610 (58)	660 (55)	653 (41)	667 (47)	723 (44)	690 (23)
<b>B. Basal Area (m<sup>2</sup>/ha)</b>								
1. Shrubs								
a. Treatment	4.6 (0.2)	5.0 (0.2)	5.2 (0.2)	5.5 (0.5)	5.6 (0.5)	5.3 (1.0)	4.8 (1.21)	-
b. Control	7.3 (0.6)	8.0 (0.6)	8.5 (0.6)	8.6 (0.3)	8.7 (0.2)	8.9 (0.4)	9.9 (1.1)	9.9 (1.4)
2. Trees								
a. Treatment	21.8 (1.7)	21.6 (1.7)	21.2 (2.5)	21.7 (2.6)	20.4 (2.3)	20.1 (2.8)	18.2 (3.0)	8.1 (0.2)
b. Control	24.6 (1.2)	25.6 (1.1)	26.9 (0.9)	28.2 (0.8)	26.6 (1.2)	27.1 (1.3)	28.3 (1.6)	27.2 (2.2)
<b>C. Biomass (kg/m<sup>2</sup>)</b>								
1. Shrubs								
a. Treatment	1.1 (0.0)	1.1 (0.0)	1.2 (0.1)	1.3 (0.1)	1.2 (0.1)	1.3 (0.2)	1.1 (0.2)	-
b. Control	1.7 (0.1)	1.8 (0.1)	1.9 (0.1)	2.0 (0.1)	2.0 (0.1)	2.1 (0.2)	2.2 (0.2)	2.2 (0.3)
2. Trees								
a. Treatment	12.5 (1.1)	12.3 (1.7)	12.0 (1.7)	12.3 (1.6)	11.8 (1.8)	11.4 (1.8)	10.3 (1.8)	4.7 (0.9)
b. Control	14.7 (1.3)	15.3 (1.3)	16.3 (1.2)	16.9 (1.1)	16.0 (0.9)	16.3 (1.0)	16.6 (1.1)	16.0 (1.5)
<b>D. Prod. (g/m<sup>2</sup>/yr)</b>								
1. Shrubs	-							
a. Treatment	-	52.9 (20.1)	91.1 (25.6)	125.0 (48.9)	48.8 (11.7)	42.5 (36.5)	17.6 (20.2)	-
b. Control	-	101.3 (20.3)	100.4 (2.1)	118.1 (14.8)	93.2 (12.6)	82.0 (34.9)	137.6 (55.2)	77.6 (13.0)
2. Trees								
a. Treatment	-	378.8 (26.9)	382.6 (17.8)	309.3 (111.1)	236.7 (71.8)	-111.0 (71.8)	0.7 (3.5)	39.6 (27.1)
b. Control	-	653.1 (153.5)	929.3 (166.1)	639.5 (147.4)	627.1 (140.0)	347.4 (119.8)	362.2 (115.1)	242.3 (136.9)
3. Leaf								
a. Treatment	-	386.1 (18.2)	358.4 (14.6)	-	272.5 (16.3)	166.1 (13.8)	121.6 (16.2)	149.8 (33.1)
b. Control	-	525.8 (20.2)	513.9 (16.5)	-	427.7 (24.1)	292.5 (19.0)	177.1 (15.8)	188.3 (20.9)
<b>E. Total Prod. (g/m<sup>2</sup>/yr)</b>								
1. Treatment	-	817.8	832.1	-	558.0	97.6	139.9	-
2. Control	-	1280.2	1543.6	-	1148.0	721.9	676.9	508.2

# Aboveground Productivity

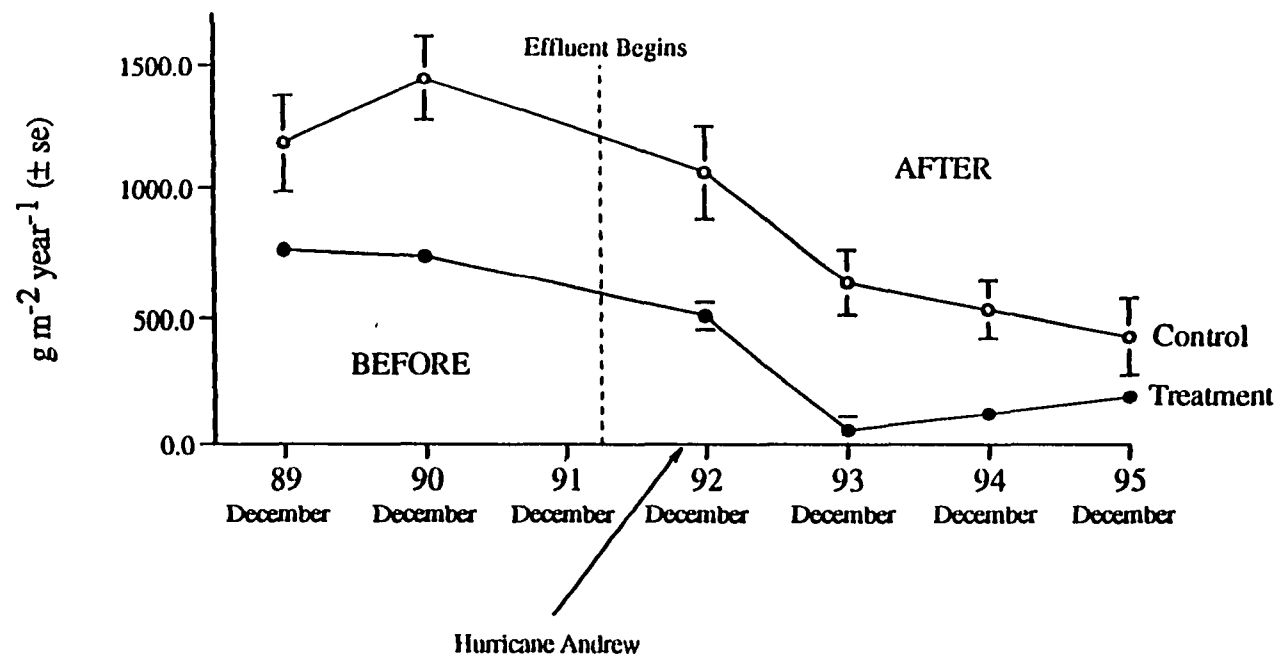


Figure 5.2. Aboveground productivity (wood increment plus litterfall) over a seven year period in two sites in the Pointe au Chene wetlands. The treatment site began receiving effluent in March 1992.



1989), and few species can survive for more than three years of continuous flooding (Bell and Johnson 1974).

### **Green Leaf and Litter Nutrient Concentrations**

Statistical analyses revealed a significant three way, time x site x species, interaction for N (Figure 5.3) and K (Figure 5.4) and a significant two way, treatment x time, interaction for P (Figure 5.5). There were no significant interactions for Mg (Figure 5.6), Ca (Figure 5.7) or Fe (Figure 5.8). The significant two way interaction indicates, that for all four species in general, the *difference* in green leaf P concentrations between the control and treatment sites, *before* effluent additions began in the treatment site (1989), was significantly different than the *difference* in concentrations between sites, in 1993, *after* effluent additions began in the treatment site (a treatment effect). Three way interactions indicate that the treatment effects were not similar across all species. For green leaf N, only two species, red maple and black willow, showed significant site x time interactions (Figure 5.3). Red maple N concentrations were significantly lower in the treatment site than in the control in 1989, but were not significantly different in 1993. Following a similar pattern, black willow leaf N concentrations were not significantly different between the control and treatment during 1989, but were significantly higher in 1993. In contrast, for baldcypress, green leaf N concentrations were significantly higher in the treatment site both before and after effluent applications began. For green ash, there was no significant difference between sites for either period.

For green leaf K, significant time x site interactions were also revealed for two species, red maple, and green ash (Figure 5.4). Red maple leaf K concentrations were not significantly different between sites during 1989, but were significantly lower in the treatment site in 1993. Alternatively, green ash leaf K concentrations were significantly

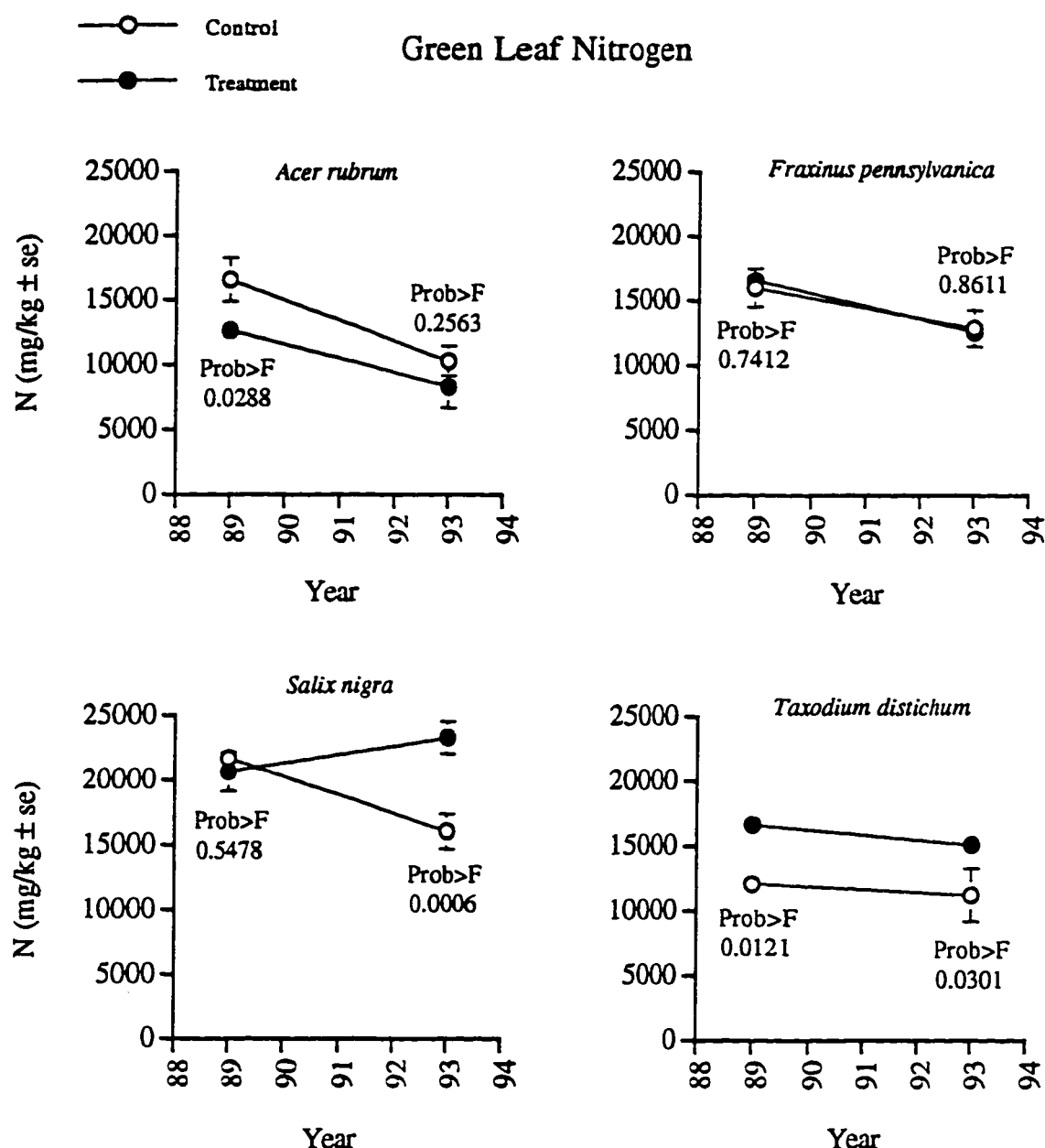


Figure 5.3. Mean green leaf N concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A three way factorial design indicated a significant three way (site  $\times$  species  $\times$  time) interaction (Prob > F = .0500) suggesting that there was a significant treatment effect for some, but not all species. Specifically, post ANOVA comparisons revealed that for *Acer rubrum*, control site green leaf N was significantly higher than in the treatment before effluent applications began in treatment site, but were not significantly different between sites in 1993 after applications began. Additionally, *Salix nigra* N concentrations were significantly higher in the treatment in 1993, but were not significantly different from the control during 1989. Results for each within year, control vs. treatment comparison, for each species, are shown on their respective graphs.

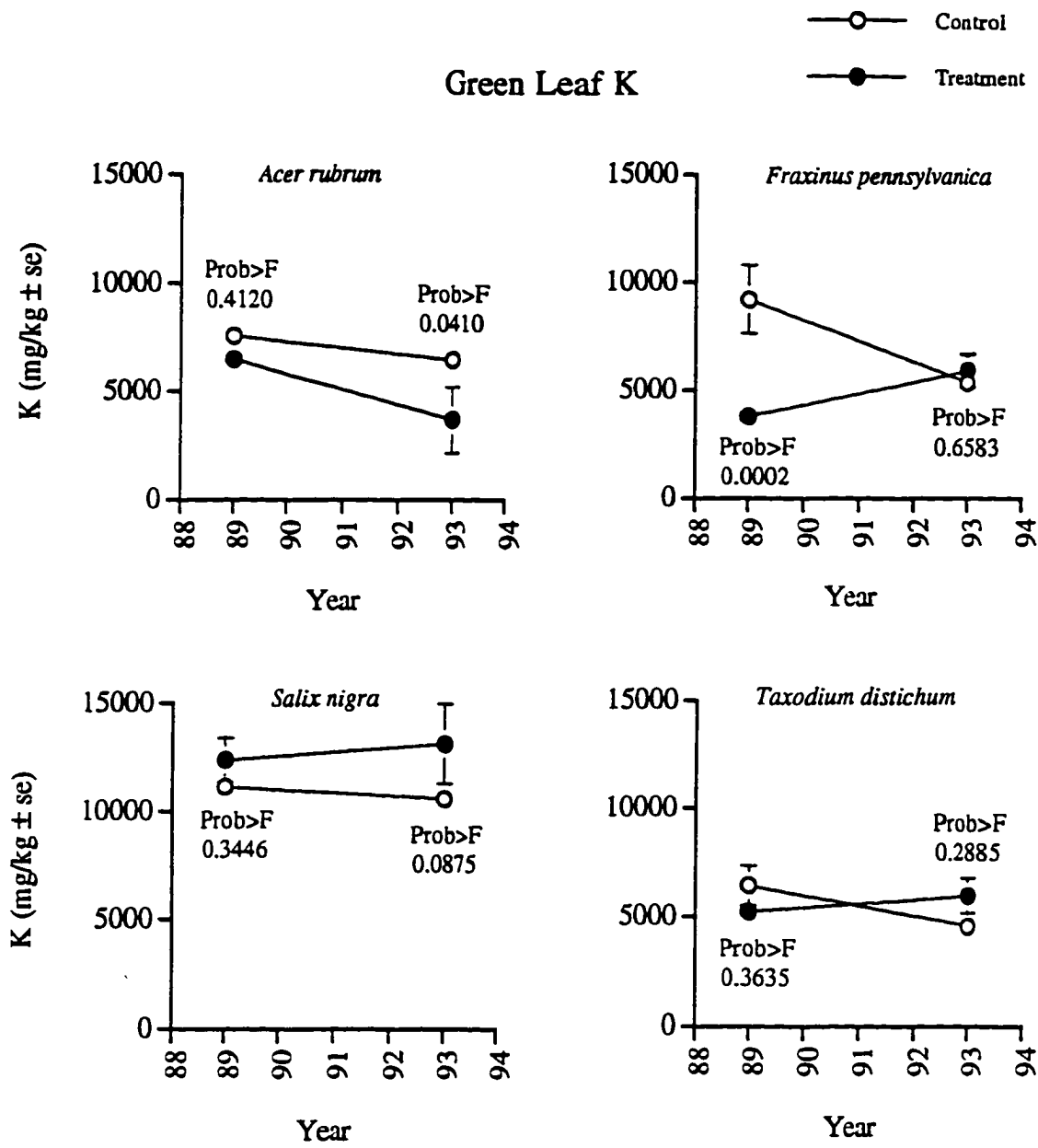


Figure 5.4. Mean green leaf K concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A 3 way factorial design indicated a significant three way (site  $\times$  species  $\times$  time) interaction (Prob > F = .0477) suggesting that there was a possible treatment effect for some, but not all species. Specifically, post ANOVA comparisons revealed that for *Fraxinus pennsylvanica*, control site green leaf K was significantly higher in the control than in the treatment before effluent applications began in treatment site, but were not significantly different between sites in 1993 after applications began. Additionally, *Acer rubrum* K concentrations were significantly higher in the control site in 1993, but were not significantly different from the treatment during 1989. Results for each within year, control vs. treatment comparison, for each species, are shown on their respective graphs.

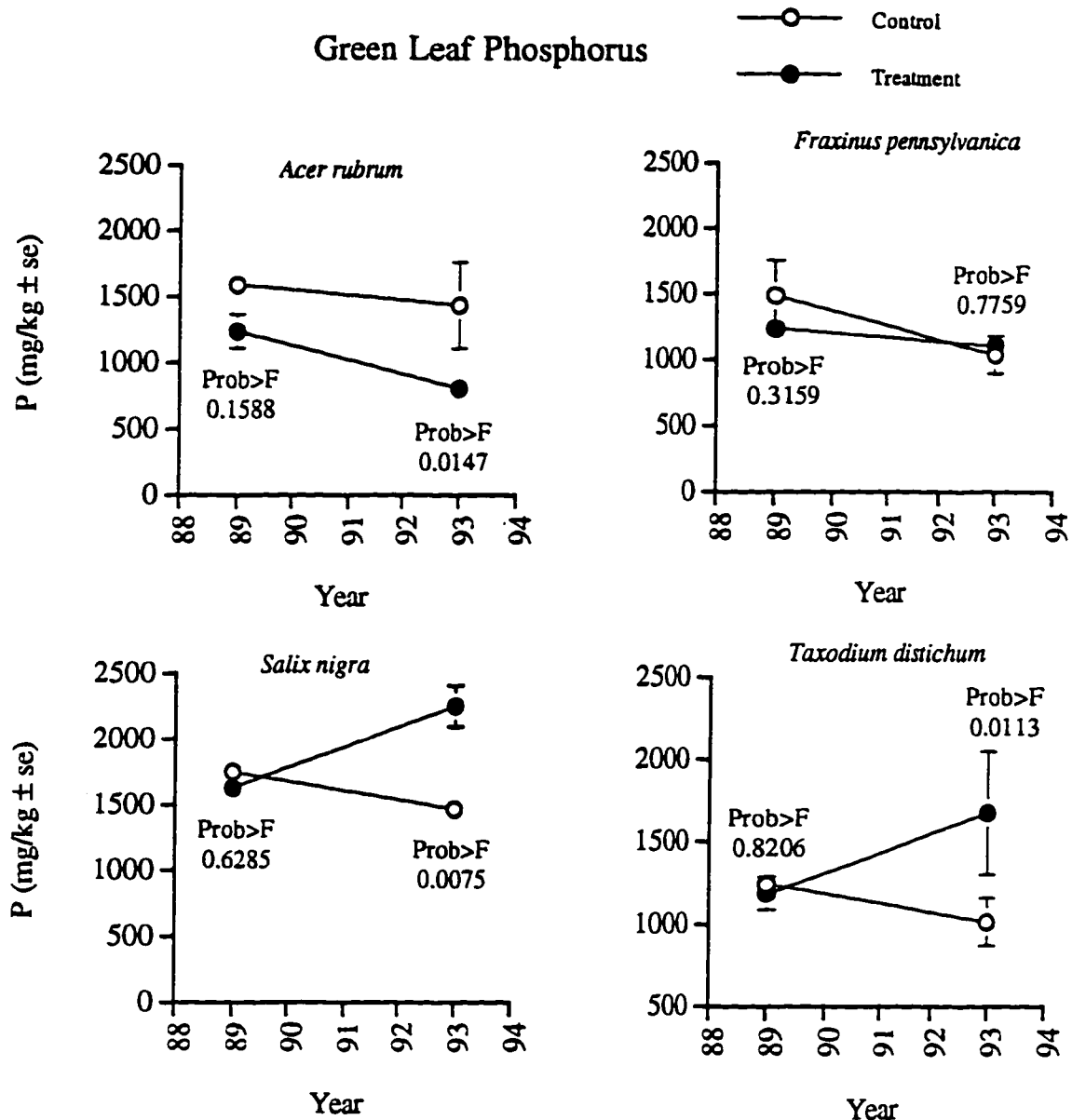


Figure 5.5. Mean green leaf P concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A 3 way factorial design indicated a significant two way (site x time) interaction ( $\text{Prob} > F = .0248$ ) and a weak three way interaction (site x time x species:  $\text{prob} > F 0.107$ ) suggesting that there was a significant treatment effect. Post ANOVA comparisons revealed that for *Salix nigra* and *Taxodium distichum* respectively, P concentrations were significantly higher in the treatment in 1993, but were not significantly different from the control during 1989. However, for *Acer rubrum*, N concentrations in the treatment site were significantly lower than the control during 1993 even though there was no significant difference between sites during 1989. Results for each within year, control vs. treatment comparison, for each species, are shown on their respective graphs.

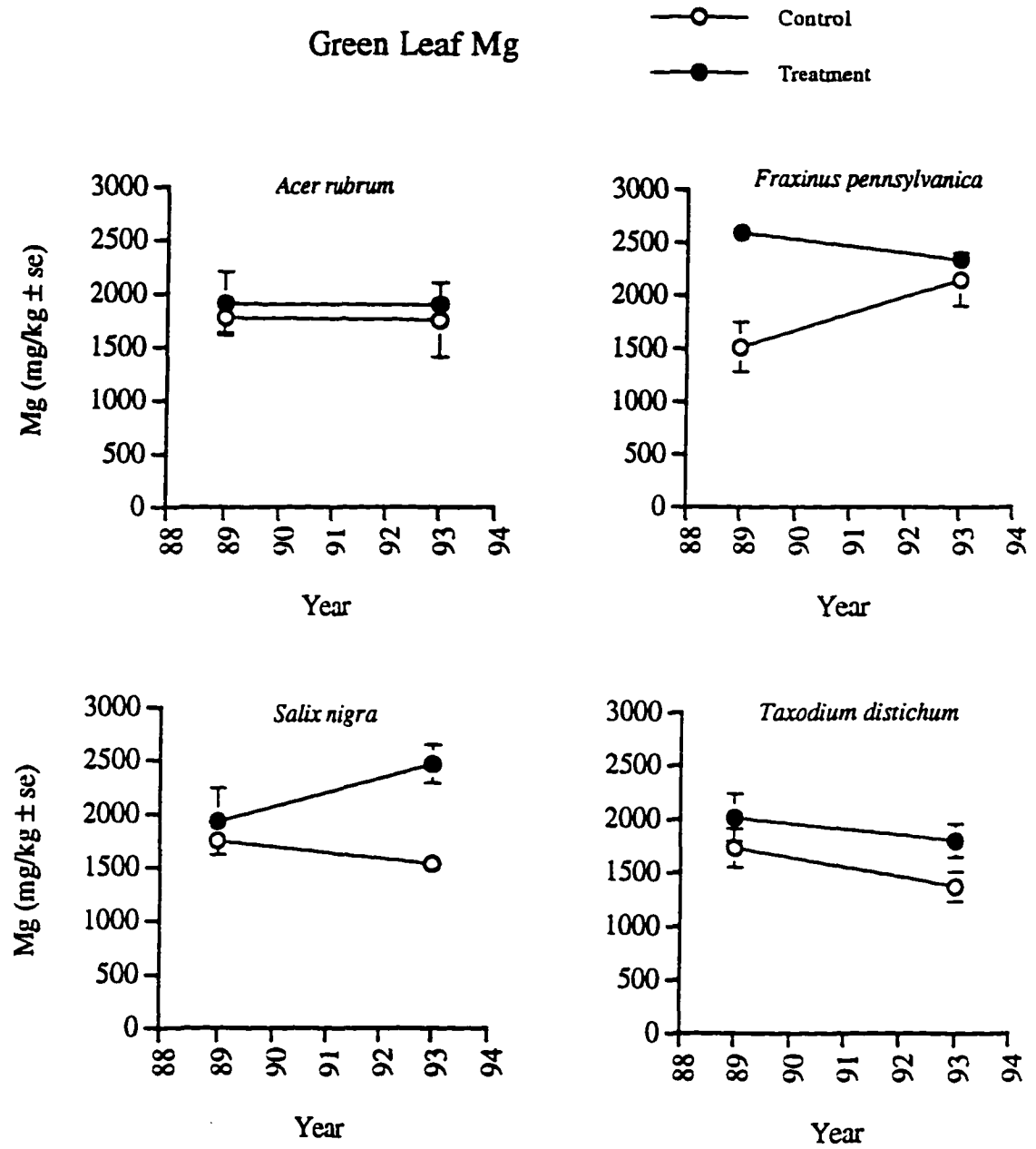


Figure 5.6. Mean green leaf Mg concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A three way factorial design indicated no significant two way interaction (time x site: Prob > F = 0.9792) or three way interaction (site x species x time: Prob > F = .0640) suggesting that there was no significant treatment effect.

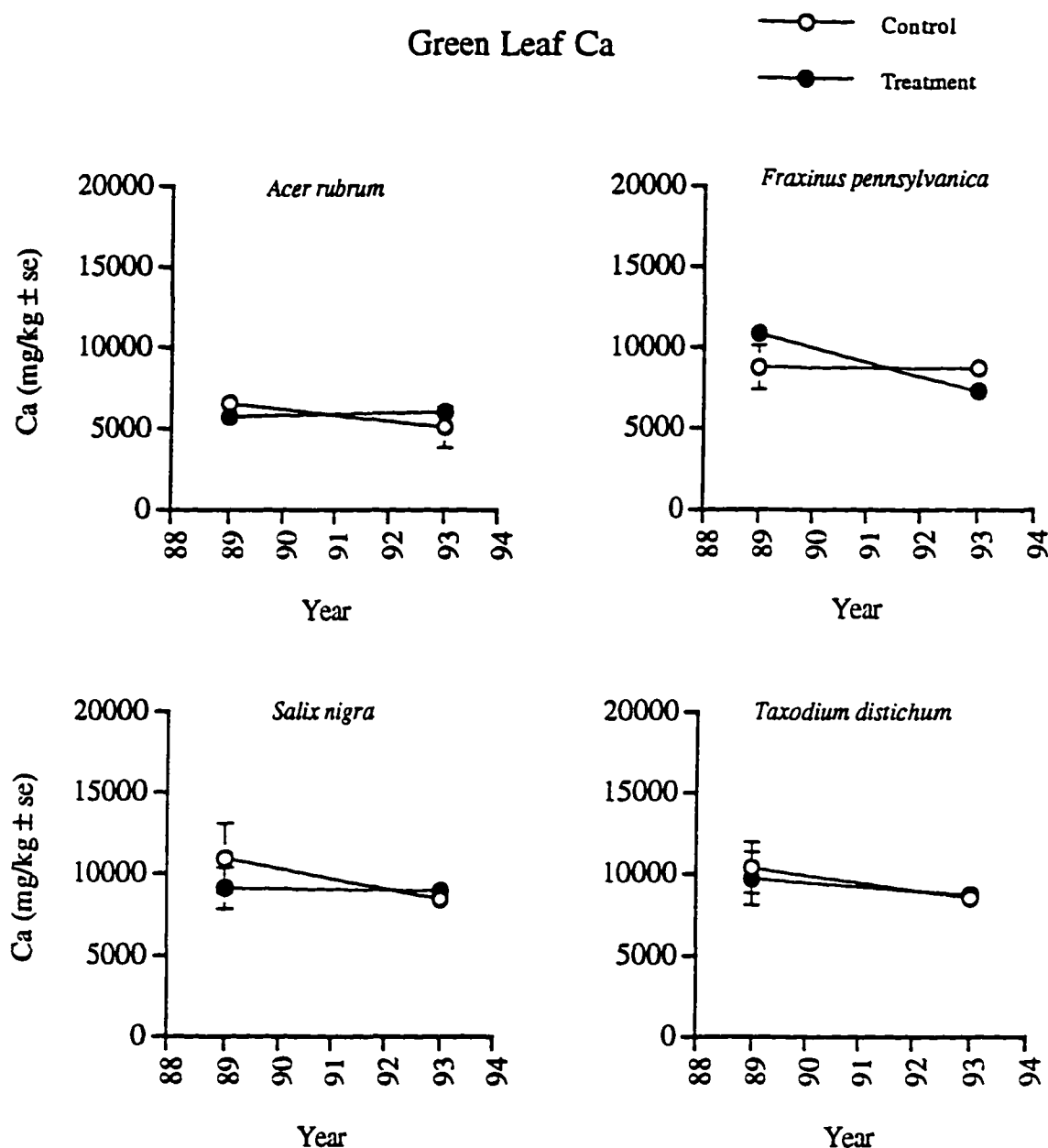


Figure 5.7. Mean green leaf Ca concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A three way factorial design indicated no significant two way interaction (time  $\times$  site: Prob  $> F = 0.7564$ ) or three way interaction (site  $\times$  species  $\times$  time: Prob  $> F = .2631$ ) suggesting that there was no significant treatment effect.

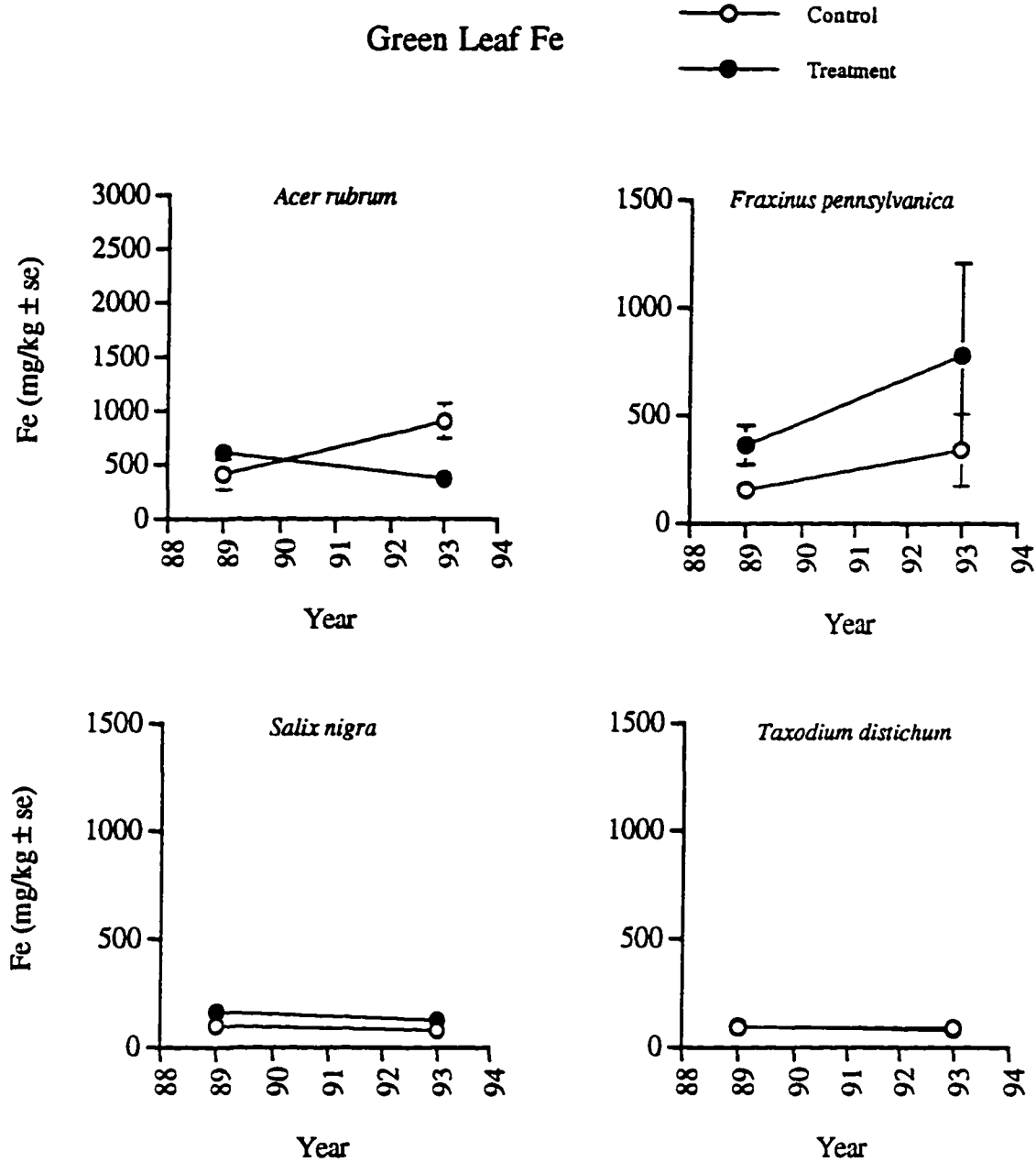


Figure 5.8. Mean green leaf Fe concentrations ( $\pm$  se) for four species, measured in the control (open circles) and treatment sites (closed circles), both before (1989) and after (1993) effluent additions began in the treatment site. A three way factorial design indicated no significant two way interaction (time  $\times$  site: Prob  $>$  F = 0.3338) or three way interaction (site  $\times$  species  $\times$  time: Prob  $>$  F = .0865) suggesting that there was no significant treatment effect.

lower in the treatment site during 1989, but were not significantly different between sites in 1993.

Although statistical analyses revealed a significant two way interaction for P, there was also an indication of a weak three way interaction ( $P = 0.1074$ ), therefore I examined each species individually (Figure 5.5). For both black willow and baldcypress respectively, leaf P concentrations were not significantly different between sites in 1989, but were significantly higher in the treatment site in 1993. For red maple however, P concentrations were not significantly different between sites in 1989, but were significantly lower in the treatment site during 1993. Green ash concentrations were not significantly different between sites for any date.

For leaf litter, two way, date x time, interactions were indicated for Ca, Fe and Mg nutrient concentrations (Figure 5.9), but not for N, P or K. For both Ca and Mg, nutrient concentrations were not significantly different between sites in 1989 but were significantly higher in the treatment site in 1992. Conversely, Fe concentrations were not significantly different between sites in 1989, but were significantly lower in the treatment site in 1993.

Among the six green leaf nutrients examined, N and P exhibited the most consistent and expected results (an increase in the treatment site, in respect to the control, after effluent additions began in the treatment site). This might be expected as Cole and Rapp (1980) pointed out that within a plant, N, P and Ca cycle primarily through leaf litterfall, while highly mobile K cycles largely via through fall and Mg is intermediate. Additionally, because Fe is mobile and available in these naturally anaerobic wetlands, it is probably not limiting at any time at either site. Among species, black willow was the most responsive to nutrient additions, exhibiting increases in green leaf N, K and P concentrations, with respect to the control, in 1993. This may be related to the plant's fundamental ecological strategy. For example,



### Litter Nutrients

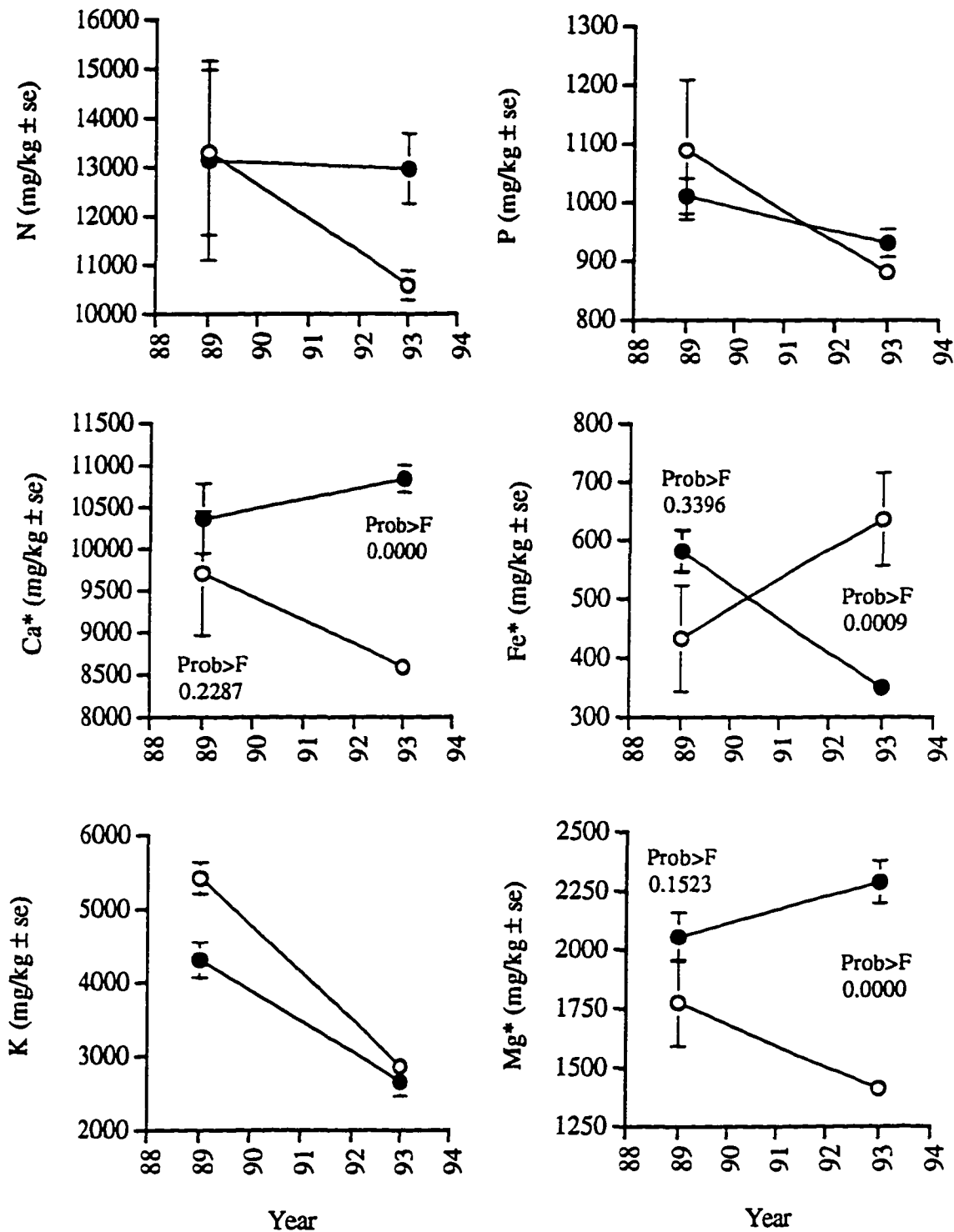


Figure 5.9. Mean litter nutrients ( $\pm$ se), measured in the control (open circles) and treatment site (closed circle) both before (1989) and after (1993) effluent additions began in the treatment site. A two way factorial ANOVA found a significant time x site interaction for Ca, Fe and Mg (indicated by a \*).

Mattson (1980) pointed out that, in general, fast growing, short lived species, such as black willow, contain more N than slower growing species. Litterfall nutrients did not reflect the patterns observed for green leaf nutrients. One probable explanation is that prolonged exposure in the littertraps, prior to collection, leached out labile pools of nutrients that may have reflected a change in nutrient status.

## CONCLUSIONS

Although plant tissue nutrient concentrations indicated some response to nutrient additions associated with wastewater effluent, dramatic decreases in aboveground productivity were observed in both the control and treatments sites from 1989 through 1995. This was attributed to the confounding effects of nutria herbivory, Hurricane Andrew and prolonged inundation due to region-wide accretion balance deficits. Coastal wetland forests of the southeast are subject to frequent disturbance from hurricanes and smaller tropical storms (Conner et al. 1989; Doyle et al. 1995). Episodic herbivory has also been shown to affect forest structure, patterns of regeneration and productivity (Conner and Day 1989; Hesse 1996). Unfortunately, continual inundation in these subsiding wetlands precludes the opportunity for regeneration after perturbances caused by storms or herbivores. Even the most flood tolerant tree species (e.g. baldcypress) cannot germinate in standing water (Demaree 1932).

Nutria can be particularly insidious in these systems. Although negative feedback loops are the most common in biological systems (Curtis and Barnes 1985), I believe that the interaction between nutria, wastewater effluent, and forest structure and productivity, provide an example of a positive feedback loop that accelerates forest destruction. As shown by Zhang (1995), nutrients associated with wastewater increase the internal nutrient quality of duckweed, an important food source for nutria. As a result, whether from immigration or fecundity, nutria populations increase in the

swamp. These nutria also strip the bark and cambium layers from canopy species, effectively girdling and killing the trees. I have observed these strips of bark being used for nesting materials in the swamp, but they are most probably also a food source, as nutria are known to eat entire seedlings and saplings (Brantley and Platt 1992). The death of these trees opens up the canopy, easing the light limitations operating on aquatic duckweed communities. Finally to complete the loop, duckweed standing crop and turnover increases, providing more food for more nutria.

As accretion balance deficits accumulate from year to year in these forested wetlands, flooding increases in duration and frequency and, eventually, the sites become continuously inundated (Conner and Day 1989; Rybczyk 1996). It was originally hypothesized that a treatment associated fertilizer effect at the Pointe au Chene wetlands could stimulate primary productivity to the point that accretion due to organic matter inputs would keep pace with the estimated rate of relative sea level rise. Other studies at this site have documented an increase in accretion rates in the treatment site, compared to the control, after effluent additions began (Rybczyk 1996). However, evidence from this study indicates that the aboveground production associated with trees and shrubs are not the cause of this observed increase. Unpublished percent cover data from the Pointe au Chene site, and observations from Zhang (1995), suggest that effluent stimulated increases in the production associated with floating aquatic plants (e.g. duckweed (*Lemna* sp.) and *Salvinia* sp.), may be the source for increased organic matter accumulation in the treatment site. Although numerous studies have shown that small-leaved floating aquatic plant production can be extremely high in wastewater treatment systems subject to frequent harvest ( $3200 \text{ g d.w. m}^{-2} \text{ yr}^{-1}$ : Reddy and Debusk 1984;  $5073 \text{ g d.w. m}^{-2} \text{ yr}^{-1}$ : Debusk and Ryther 1987), we know of no study that directly measured annual duckweed production and turnover, without harvest, in a natural wetland. Although we did not measure floating aquatic vegetation

production at this site either, in the next chapter, we use an integrated soil development/plant production simulation model to examine the possible contribution of floating aquatic vegetation to organic matter accumulation and accretion.

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## CHAPTER 6

### A WETLAND ELEVATION MODEL FOR THE POINTE AU CHENE SWAMP

#### INTRODUCTION

##### Background and Objectives

Coastal wetland elevation, relative to sea-level, is a function of numerous processes, such as eustatic sea-level rise, compaction, decomposition, subsidence and accretion, occurring at several time scales (Figure 6.1). Insufficient accretion, coupled with high rates of relative sea-level rise (subsidence plus eustatic sea-level rise), are two important processes contributing to wetland loss in the Mississippi Delta (Boesch et al. 1994). Additionally, the eustatic sea-level rise component of relative sea-level rise is expected accelerate over the next 100 years (Gornitz 1995). If deltaic wetlands are to survive rising water levels, they must be able to accrete at a rate such that surface elevation gain is sufficient to offset relative sea-level rise (RSLR). Several studies have attempted to predict the fate of selected coastal wetlands subject to an acceleration in eustatic sea-level rise (ESLR) rates 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; Templet and Meyer-Arendt 1988; Bricker-Urso 1989; Day et al. 1995). For example, in a coastal forested wetland in Louisiana, Delaune et al. (1987) measured RSLR rates of  $1.36 \text{ cm year}^{-1}$  and accretion rates of only  $0.63 \text{ cm year}^{-1}$ , to derive an accretion balance of  $-0.73 \text{ cm year}^{-1}$ . Conner and Day (1989) measured accretion deficits ranging from 0.25 to 1.08 cm/year, using similar methods, for coastal wetland forests in the Barataria and Lake Verret basins of southern Louisiana.



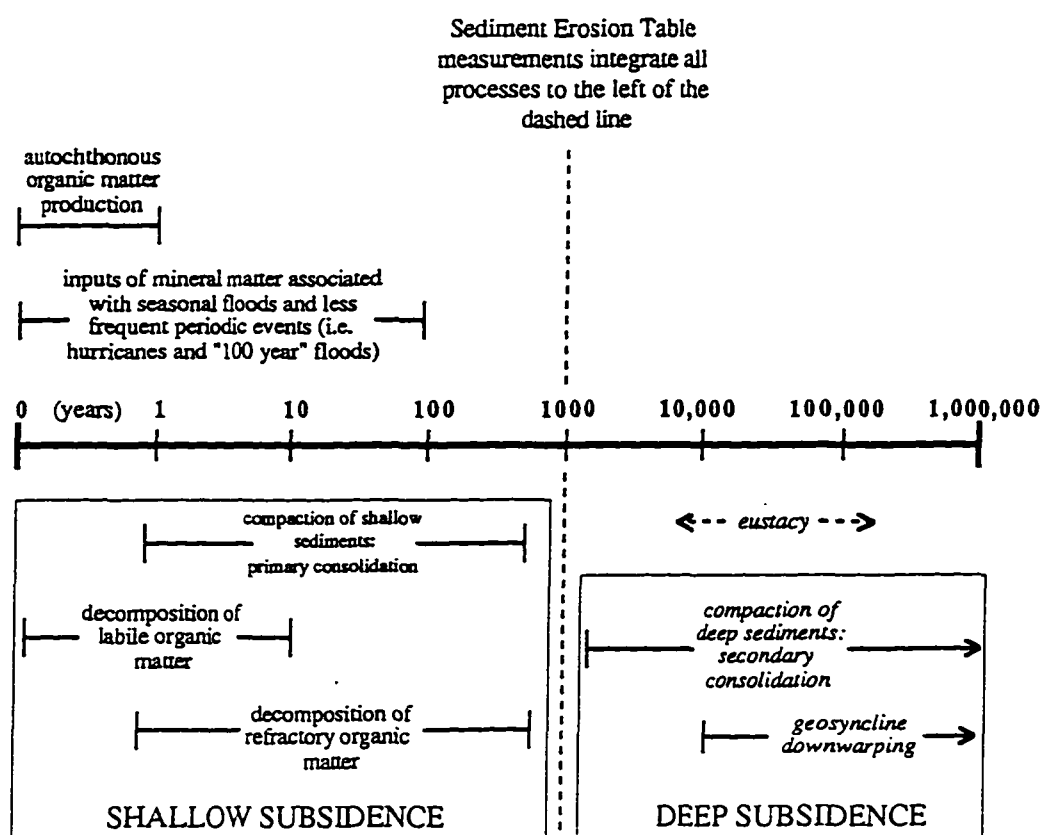


Figure 6.1. Processes that affect wetland elevation relative to sea level rise. Processes shown below the time line decrease elevation, while those shown above the line increase elevation. Processes shown in italics are commonly entered as forcing functions into existing wetland elevation models.

These types of calculations should be viewed with caution because short term field measurements of accretion and shallow subsidence (one or two years) do not fully integrate long term processes, such as compaction and decomposition, that affect wetland elevation. Additionally, these types of measurements do not take into account possible elevation feedback mechanisms on the processes themselves. Longer term sediment dating methods such as  $^{210}\text{Pb}$ , which typically integrate decomposition and compaction processes occurring within the first meter of sediment, can help to resolve this problem of scale. As an alternative to measuring accretion directly, sediment-erosion-tables, that integrate both accretion and shallow subsidence over several meters, can be used to measure changes in marsh elevation over time. These changes can then be compared to measured and predicted changes in relative sea-level (Boumans and Day 1993), although it may take several years using this method to discern a clear pattern of elevation change over time. However, the use of any of these methods to predict changes in wetland elevation, relative to sea-level, is still prone to error because they do not take into account possible elevation feedback mechanisms on the processes themselves. Specifically, changes in elevation can result in changes in allogenic sediment deposition, decomposition and autogenic primary production.

For this reason, site specific, field calibrated ecosystem models, that can simulate decomposition and compaction over decades or even centuries, and incorporate feedback mechanisms, can be useful for examining the response of wetland elevation to increasing rates of sea-level rise. I present here, an integrated wetland elevation/sediment accretion model for the Pointe au Chene swamp, a hydrologically isolated, subsiding forested wetland located within the coastal zone of Louisiana. The site consists of two permanently flooded forested wetlands (maple, ash, willow, cypress associations) separated by a slightly elevated bottomland hardwood ridge. Accretion balance deficits of  $0.79 \text{ cm year}^{-1}$  have been estimated for the forest

(Rybczyk 1996b). Since 1992, the 231 ha forested wetland on the western side of the ridge has received secondarily treated municipal wastewater at a rate of  $15,140 \text{ m}^{-3} \text{ day}^{-1}$ . A comprehensive site description is given by Rybczyk et al. (1995). Day et al. (1992) hypothesized that the addition of non-toxic secondarily treated wastewater effluent to this system could simulate organic matter accretion to the point that wetland elevation could keep pace with relative sea-level rise. The integrated model was used to; 1) examine the response of wetland elevation to possible effluent-related changes in the processes (e.g. production, decomposition) that contribute to wetland elevation, 2) determine the relative sensitivity of wetland elevation to the processes shown in Figure 6.1, and 3) predict the possible long term (50 year) effects of effluent additions on elevation.

### **Modeling Wetland Elevation**

Numerical simulation models designed specifically to simulate relative marsh elevation as a function of the processes outlined in Figure 6.1 are relatively few in number. We know of no such models for forested wetlands, although models have been developed to simulate forest production as a function of hydroperiod (Phipps 1979). Existing marsh elevation models have focused on simulating sub-sets of the processes shown in Figure 6.1 and have either ignored other processes, or included them as forcing functions. Models that are specifically designed to evaluate changes in relative wetland elevation in response to sea-level rise, and predict the consequences of various management scenarios, should consider all of these processes.

French (1993) and Allen (1990), for example, developed detailed mechanistic algorithms to simulate allochthonous sediment deposition in Great Britain salt marshes, as a feedback function of elevation. However, in both of these models, autochthonous organic matter was entered as a pre-compacted, pre-decomposed, forcing function. In contrast, Randerson's (1970) salt marsh development model focused primarily on

simulations of plant community structure and productivity as a function of elevation but relied on simple plant biomass vs. accretion regressions to simulate mineral sediment accretion. Morris and Bowden (1986) developed a yearly sediment cohort model to describe sediment formation in a freshwater tidal marsh that simulated some of the belowground processes that contribute to marsh subsidence, including labile and refractory organic matter decomposition. However, the primary focus of this model was to simulate N, C and P dynamics in a sediment column over time, and not to simulate changes in relative marsh elevation in response to sea-level rise. Chumura et al. (1992) developed a simple 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.

It has been recognized that compaction is an active process in shallow marsh sediments and that the compaction of recent sediments is one of major factors influencing subsidence in some wetlands (Penland et al. 1988). However, most marsh elevation/sediment dynamics models have either ignored this process (Randerson 1979; Morris and Bowden 1986), included it as a forcing function (Chumura 1992) or input sediments as precompacted units (Allen 1990; French 1993). Callaway (1994) developed a cohort sediment accretion model for coastal wetlands, similar in framework to the model developed by Morris and Bowden (1986), which simulated compaction as a function of the density of sediment above a given cohort. Because no universal compaction function was identified, Callaway (1994) calibrated the compaction algorithm separately for each wetland under consideration.

This model presented here is a logical extension of the soil models developed by Morris and Bowden (1986) and later, Callaway (1994). It further extends their body of work by modifying existing, and programming additional, sub-routines to address the characteristics of forested wetlands that are not associated with the annual systems that their models were intended for (i.e. perennial above and belowground plant biomass). It includes a primary production sub-model that simulates production as a function of elevation (organic matter was entered as a forcing function in both Morris and Bowden's (1986) and Callaway's (1994) model. Also, decomposition processes are simulated more mechanistically in this model, because this process may be affected by effluent additions (Rybczyk et al. 1996a). In the following sections I describe the model, show the results of calibration, sensitivity and validation exercises, and finally run the model under a variety of ESLR, subsidence, wastewater effluent and mineral input scenarios.

## **MODEL DESCRIPTION**

### **General Overview**

The model utilizes a cohort approach (tracking discrete packages of sediments through depth and time) to simulate sediment dynamics (organic and mineral matter accretion, decomposition, compaction, and belowground productivity). These dynamics produce model-generated changes in sediment characteristics including; bulk density, organic matter (o.m.) volume and mass, mineral matter volume and mass, pore volume and mass, and yield total sediment height as an output. Sediment height is then balanced with eustatic sea-level rise (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 floating aquatic vegetation) and mineral inputs, both of which are a feedback function of the model-generated marsh elevation (Figure 6.2).

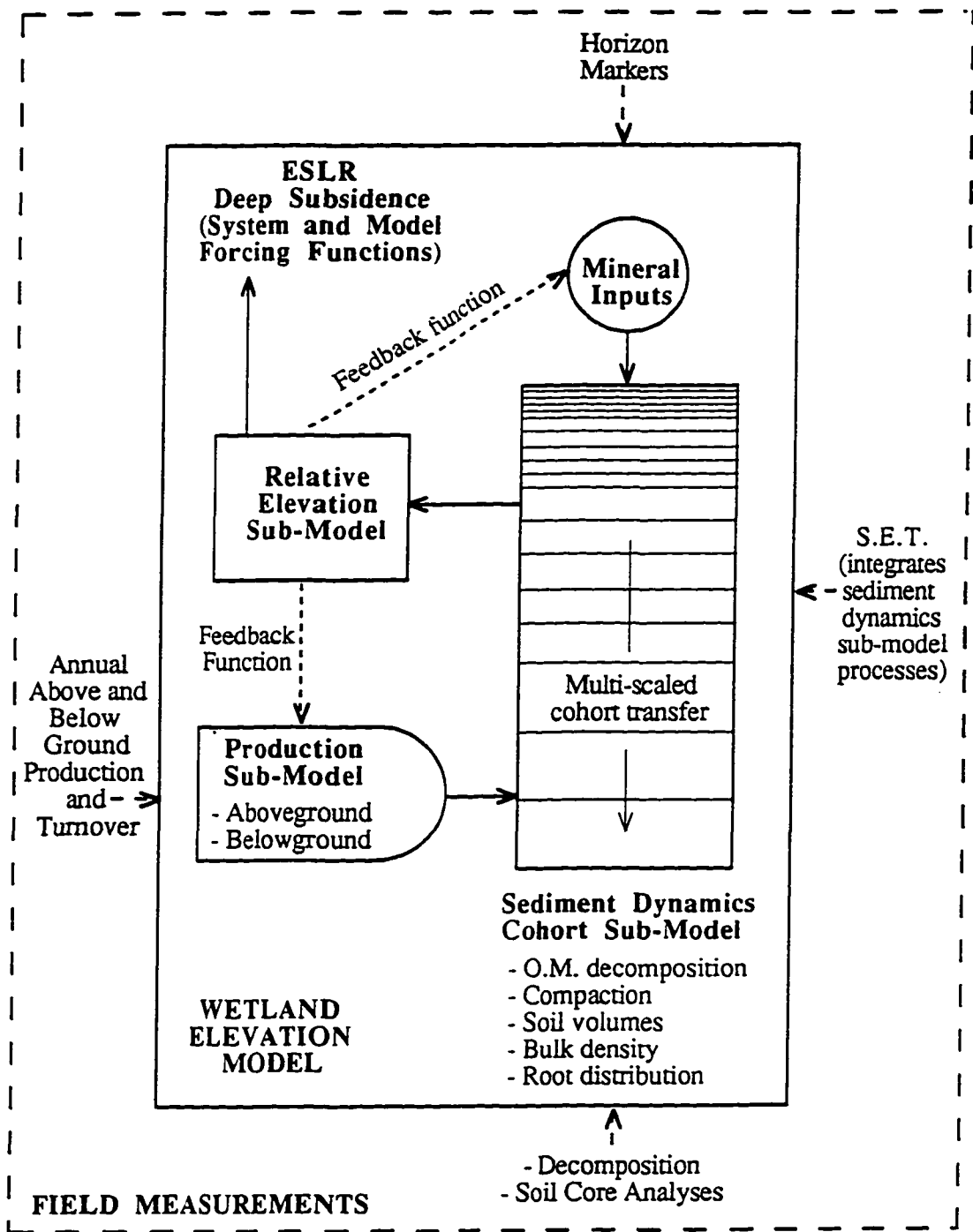


Figure 6.2. The integrated wetland elevation model and links to field data.

Field measurements collected as part of this dissertation, and as part of other research projects associated with the Pointe a Chene wetlands (Zhang 1995) were used for model initialization, calibration and validation (Figure 6.2). Specifically, measurements of annual aboveground production were used to calibrate and initialize the production sub-model, decomposition experiments and soil core analyses were used to calibrate the sediment dynamics sub-model, and horizon markers were used to estimate mineral inputs and to further calibrate the model

The integrated model consists of three linked sub-models or sectors (Figure 6.2): 1) Primary productivity; 2) Sediment dynamics and 3) Relative elevation. Each of these sectors will be described separately below. The model is programmed with STELLA simulation software and is currently being run on a Macintosh 520 upgraded with 20 Mg of RAM memory. An Euyler numerical method, with a time step of one week, is 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 6.1. Program code is attached as Appendix C.

### **Sediment Dynamics Sub-Model**

The sediment dynamics sub-model has four state variables, each replicated 18 times for each cohort layer. State variables include: 1) *lab\_below<sub>n</sub>*, representing labile o.m., 2) *ref\_below<sub>n</sub>*, representing refractory o.m., 3) *mineral<sub>n</sub>*, representing mineral matter, and 4) *root<sub>n</sub>*, representing live root biomass. The differential equations describing the changes in these state variables with time are shown in Table 6.2. 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 o.m., the inputs of mineral matter, the distribution of root biomass, sediment compaction, and the transfer of material from cohort to cohort. These process are outlined below.

Table 6.1. Model parameters, descriptions, values and sources.

Symbol	Description	Values and Units	Source
<b>A. State Variables</b>			
<i>Cleaf</i>	live leaf biomass	g d.w. m <sup>-2</sup>	field data
<i>Croot</i>	live root biomass	g d.w. m <sup>-2</sup>	Day & Mcgonigal 1993
<i>Cwood</i>	live tree biomass	g d.w. m <sup>-2</sup>	field data
<i>Float</i>	F.A.V. biomass	g d.w. m <sup>-2</sup>	Sklar 1983
<i>Lab_below<sub>n</sub></i>	labile organic matter in cohort <i>n</i>	g d.w. cm <sup>-2</sup>	field data/simulation
<i>Mineral<sub>n</sub></i>	mineral matter in cohort <i>n</i>	g d.w. cm <sup>-2</sup>	field data/simulation
<i>Ref_below<sub>n</sub></i>	refractory organic matter in cohort <i>n</i>	g d.w. cm <sup>-2</sup>	field data/simulation
<i>Root<sub>n</sub></i>	live root biomass in cohort <i>n</i>	g d.w. cm <sup>-2</sup>	field data/simulation
<b>B. Forcing Functions</b>			
<i>eslr_bau</i>	IPCC "business as usual" estimate eslr	66 cm in 100 years	Gornitz 1995
<i>eslr_bg</i>	IPCC "best guess" estimate eslr	48 cm in 100 years	Gornitz 1995
<i>eslr_c</i>	rate of eslr, initialized at current rate	15 cm in 100 years	Gornitz 1995
<i>max_min in</i>	maximum mineral input	0.00443 g cm <sup>-2</sup> week <sup>-1</sup>	estimated from field data
<i>surate</i>	local deep subsidence rate	0.0207 cm week <sup>-1</sup>	Penland 1988
<i>temp</i>	mean weekly temperature	°C	NOAA
<b>C. Rates and Constants</b>			
<i>comp k</i>	half sat. constant for soil compaction	2.5 cm <sup>3</sup> gram <sup>-1</sup>	calculated from field data
<i>float_lab_frac</i>	labile fraction of FAV litter	0.8 (unitless)	field data
<i>init_column</i>	initial height of sediment column	29.80 cm	model generated
<i>init_elev</i>	initial relative wetland elevation	0 cm	field data

(Table con'd.)



<u>Symbol</u>	<u>Description</u>	<u>Values and Units</u>	<u>Source</u>
<i>kdeep</i>	decomposition rate of deep refractory o.m.	0.0001 week <sup>-1</sup>	calibration
<i>klab</i>	decomposition rate of labile root o.m.	0.028 week <sup>-1</sup>	Day et al. 1989
<i>klabsurf</i>	decomposition rate of surface labile o.m.	0.028 week <sup>-1</sup>	field data
<i>krefr</i>	decomposition rate of refractory o.m.	0.0008 week <sup>-1</sup>	field data
<i>krefsur</i>	decomp. rate of surface refractory o.m.	0.0029 week <sup>-1</sup>	field data
<i>leaf_lab_frac</i>	labile fraction of aboveground biomass	0.3 unitless	estimated from field data
<i>llitter</i>	leaf litter production rate	0.015 week <sup>-1</sup> if week > 25 and < 45 else 1.0 if week > 45	field data
<i>maxnet</i>	max. net FAV production rate	2.66 week <sup>-1</sup>	Rejmankova 1975
<i>poremax</i>	maximum fraction of pore space in soil	0.9316 unitless (0-1)	field data
<i>poremin</i>	minimum fraction of pore space in soil	0.5813 unitless (0-1)	field data
<i>rlub%</i>	labile fraction of live roots	0.2 unitless	Day et al. 1989
<i>rlirate</i>	rate of root litter production	0.3 year <sup>-1</sup>	Day & Megonigal 1993
<i>root k</i>	root distribution constant	0.3 cm <sup>-1</sup>	calibration
<i>spacek</i>	FAV crowding constant	-0.0255 g d.w. <sup>-1</sup>	by calculation
<i>w</i>	tree flood tolerance constant	0.75 (unitless)	Phipps 1979
<i>wmort</i>	tree mortality rate	.0006 week <sup>-1</sup>	Rybczyk et al. 1995

**D. Functions**

<i>compact<sub>n</sub></i>	pore space compaction function	unitless (0-1)	model generated
<i>hydrofunc</i>	elevation function that modifies <i>maxnet</i>	unitless (0-1)	model generated
<i>maxlgrow</i>	maximum leaf vs. tree structure regression	g d.w. m <sup>2</sup> week	derived from field data
<i>spacefunc</i>	exponential FAV crowding function	unitless (0-1)	Rejmankova 1982
<i>ifunc</i>	temp. function that modifies <i>maxnet</i>	unitless (0-1)	model generated

**E. Output**

<i>bdensity<sub>n</sub></i>	bulk density of cohort n	g d.w. cm <sup>-2</sup>	model generated
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(table con'd.)

<u>Symbol</u>	<u>Description</u>	<u>Values and Units</u>	<u>Source</u>
<i>height<sub>n</sub></i>	total height of cohort n	cm	model generated
<i>theight<sub>n</sub></i>	height <sub>n</sub> + height of all cohorts above	cm	model generated
<i>land_el</i>	absolute land elev. not relative to elsr)	cm	model generated
<i>mass<sub>n</sub></i>	orgmass + minmass for cohort n	g d.w. cm <sup>-2</sup>	model generated
<i>min cm<sub>n</sub></i>	height of mineral matter in cohort n	cm	model generated
<i>minmass<sub>n</sub></i>	weight of mineral mass in cohort n	g d.w. cm <sup>-2</sup>	model generated
<i>minvol<sub>n</sub></i>	% mineral matter by volume	unitless	model generated
<i>omass%<sub>n</sub></i>	fraction of organic mass in cohort n	unitless (0-1)	model generated
<i>org cm<sub>n</sub></i>	height of organic matter in cohort n	cm	model generated
<i>orgmass<sub>n</sub></i>	weight of organic mass in cohort n	g d.w. cm <sup>-2</sup>	model generated
<i>pore cm<sub>n</sub></i>	height of pore space in cohort n	cm	model generated
<i>pore space<sub>n</sub></i>	fraction of pore space in cohort n	unitless (0-1)	model generated
<i>relative_el</i>	elevation of wetland above sea level	cm	model generated
<i>rlit<sub>n</sub></i>	root litter production for cohort n	g d.w. cm <sup>-2</sup> week <sup>-1</sup>	model generated
<i>rootprod</i>	total root production	g d.w. cm <sup>-2</sup> week <sup>-1</sup>	model generated
<i>rootin<sub>n</sub></i>	root distribution algorithm for cohort n	g d.w. cm <sup>-2</sup> week <sup>-1</sup>	model generated
<i>rpro<sub>n</sub></i>	weekly root production input to cohort n	g d.w. cm <sup>-2</sup> week <sup>-1</sup>	model generated
<i>Sub count</i>	ESLR rate + deep subsidence rate	cm week <sup>-1</sup>	model generated
<i>surfroot</i>	root production at surface (surface intercept)	g d.w. cm <sup>-2</sup> week <sup>-1</sup>	model generated
<i>lmass<sub>n</sub></i>	total mass of all cohorts above cohort <sub>n</sub>	g d.w. cm <sup>-2</sup>	model generated

Table 6.2. State variables and differential equations for the Sediment Dynamics Sub-Model

**Labile organic matter sediment cohorts,  $lab\_below_n$** 

$$d(lab\_below_n)/dt = (litter * leaf\_lab\_frac) + (rlit_n * rlab\%) + (tranl_{n-1} * lab\_below_{n-1}) - lab\_below_n * klab - (tranl_n * lab\_below_n)$$

where:

$lab\_below_n$	labile organic matter in cohort $n$ (g C cm <sup>-2</sup> )
$litter$	aboveground leaf litter inputs to surface cohort (g C cm <sup>-2</sup> week <sup>-1</sup> )
$leaf\_lab\_frac$	labile fraction of above ground biomass (unitless)
$rlit_n$	root litter inputs to cohort $n$ (g C cm <sup>-2</sup> week <sup>-1</sup> )
$rlab\%$	labile fraction of root litter (unitless)
$tranl_{n-1}$	transfer rate of labile o.m. from overlying cohort (g C cm <sup>-2</sup> year <sup>-1</sup> )
$lab\_below_{n-1}$	labile o.m. in overlying cohort (g C cm <sup>-2</sup> )
$klab$	decomposition rate of labile organic matter (week <sup>-1</sup> )
$tranl_n$	transfer rate of labile o.m. to underlying cohort (g C cm <sup>-2</sup> year <sup>-1</sup> )

**Refractory organic matter sediment cohorts,  $ref\_below_n$** 

$$d(ref\_below_n)/dt = (litter * (1 - leaf\_lab\_frac)) + (rlit_n * (1 - rlab\%)) + (tranr_{n-1} * ref\_below_{n-1}) - (ref\_below_n * kref) - (tranr_n * ref\_below_n)$$

where:

$ref\_below_n$	refractory o.m. in cohort $n$ (g C cm <sup>-2</sup> )
$tranr_{n-1}$	transfer rate of refractory o.m. from overlying cohort (g C cm <sup>-2</sup> year <sup>-1</sup> )
$ref\_below_{n-1}$	refractory o.m. in overlying cohort (g C cm <sup>-2</sup> )
$kref$	decomposition rate of refractory organic matter (week <sup>-1</sup> )
$tranr_n$	transfer rate of refractory o.m. to underlying cohort (g C cm <sup>-2</sup> year <sup>-1</sup> )

**Mineral matter in sediment cohorts,  $mineral_n$** 

$$d(mineral_n)/dt = (max\_min\_in * minelfunc) + (tranm_{n-1} * mineral_{n-1}) - (tranm_n * mineral_n)$$

where:

$mineral_n$	mineral matter in cohort $n$ (g cm <sup>-2</sup> )
$max\_min\_in$	maximum mineral input as a function of elevation (unitless)
$tranm_{n-1}$	transfer rate of mineral matter from overlying cohort (g cm <sup>-2</sup> year <sup>-1</sup> )
$mineral_{n-1}$	mineral matter in overlying cohort (g cm <sup>-2</sup> )
$tranm_n$	transfer rate of mineral matter to underlying cohort (g cm <sup>-2</sup> year <sup>-1</sup> )

**Live roots in sediment cohorts,  $root_n$** 

$$d(root_n)/dt = rootin_n - (rlirate * root_n)$$

where:

$root_n$	live root biomass in cohort $n$ (g C cm <sup>-2</sup> )
$rootin_n$	fraction of total root production (root ) distributed to cohort $n$ (g C cm <sup>-2</sup> )
$rlirate$	rate of root litter production (week <sup>-1</sup> )

Output includes the following sediment characteristics with depth: bulk density, sediment height, organic and mineral matter mass and volume, pore space and live root mass.

Changes within the cohort, due to decomposition and belowground production, which are both a function of model-generated depth, are calculated on a weekly basis. Sediment compaction, also calculated weekly, is a function of initial pore space and the mass of material above a particular cohort. The sector is particularly powerful, because the measurements obtained from a few soil cores (bulk density, % organic and mineral matter and % mineral matter) along with some measurement of accretion rates, ( $^{137}\text{Cs}$ , marker horizons or SET) provide a comprehensive set of data which can be used to calibrate the sub-model at several points.

### **Decomposition**

The model separates all o.m. (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 o.m. that is either liable or refractory, it is appropriate 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 o.m.). Finally, there is separate, depth dependent decomposition rate for deep refractory material. Decomposition for each o.m. state variable in each cohort is described by a simple negative exponential model. Required decomposition constants include: *kdeep*, *klab*, *krefr*, *ksurf*, *leaf\_lab\_frac*, *rlabfrac* and *klabsurf*, all described in Table 6.1.

### **Mineral Inputs**

Previous models have simulated mineral inputs as a function of marsh elevation and tidal range (French 1993; Callaway 1994). Because there are no measurable tides at the Pointe au Chene wetland, mineral inputs, *minin*, are simulated as a simple

linear function of wetland elevation. Mineral inputs are maximized when relative wetland elevation is below mean water levels and minimized as elevation increases above mean water levels. Maximum mineral inputs *max\_min\_in*, were estimated from accretion and soil core analysis data obtained in the field (see Chapter 4) and are entered into the model as a forcing function.

### Root Distribution

Although root production (*rootprod*) is simulated in the productivity sub-model, root biomass is distributed to the sediment cohorts in the sediment sub-model. I 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 (Figure 6.3). The fraction of the total root biomass allocated to each cohort is calculated as:

$$rootin_n = surfroot * [e^{(-rootk * height_b)} - e^{(-rootk * height_a)}] / -rootk \quad (1)$$

Where:

*rootin<sub>n</sub>* = root input to each sediment cohort *n* (g cm<sup>-2</sup>)

*surfroot* = weight of roots at sediment surface (g cm<sup>-2</sup>)

*rootk* = root depth distribution constant (cm<sup>-1</sup>)

*height<sub>b</sub>* = depth to the bottom of the cohort<sub>*n*</sub> (cm)

*height<sub>a</sub>* = depth to the top of the cohort<sub>*n*</sub> (cm)

The constant, *surfroot*, is the surface intercept of the exponential equation:

$$r = surfroot * e^{(-rootk * depth)} \quad (2)$$

Where:

*r* = root mass (g) at depth (cm)

To solve for *surfroot*, equation 3 can be re-arranged as:

$$surfroot = rootin_n / [e^{(-rootk * height_b)} - e^{(-rootk * height_a)}] / -rootk \quad (3)$$

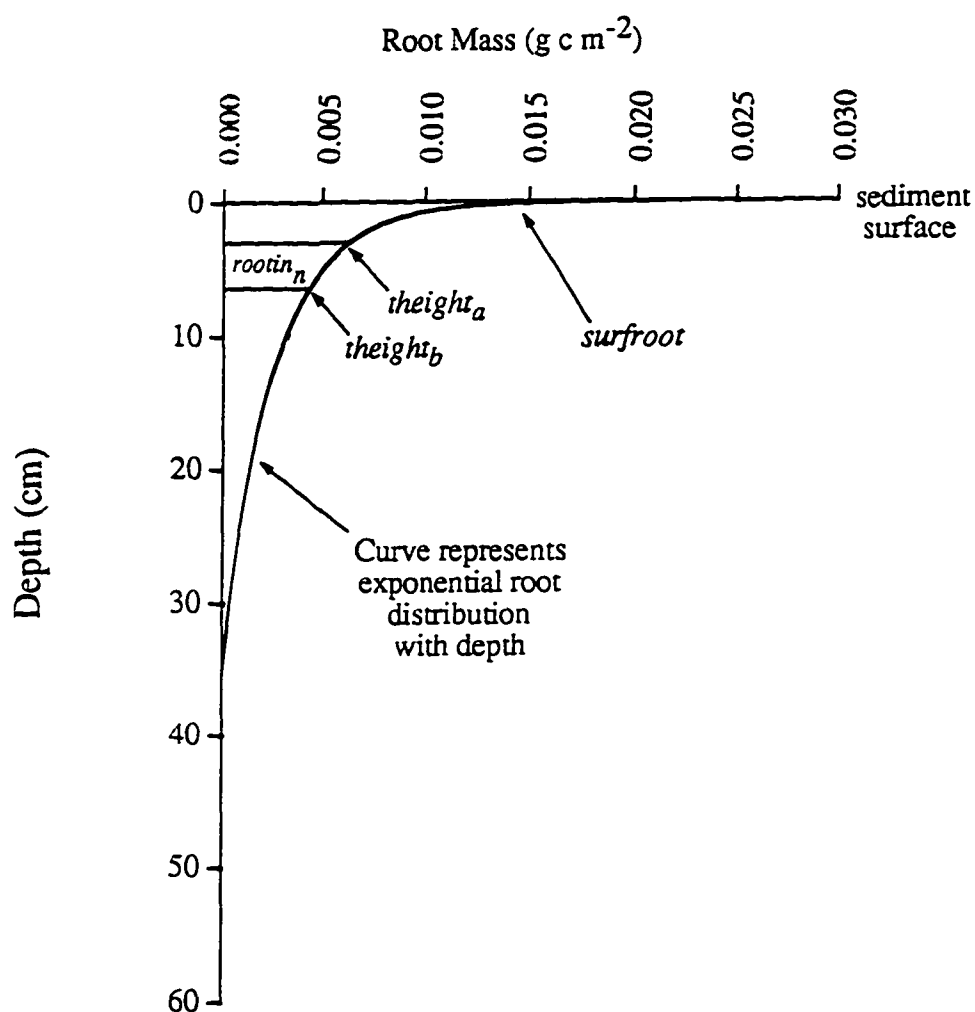


Figure 6.3. Parameters describing simulated root distribution with depth.

Then, given that total root production is known, we let  $rootin_n$  represent the entire sediment column, from the surface to a depth of infinity, so that  $rootin_n$  equals total root production. In this special case, the expression,  $e^{(-rootk*theight_b)}$ , approaches 0 and,  $e^{(-rootk*theight_a)}$ , approaches 1 for any value of  $rootk$ . Therefore, equation 6 (*surfroot*) simplifies to:

$$surfroot = rootprod/(-1/-rootk) \quad (4)$$

### **Sediment Compaction**

Within the temporal bounds of this model, soil compaction is a function of o.m. decomposition, simulated separately, and the reduction of sediment pore space (primary consolidation) (Penland and Ramsey 1990). Callaway (1994) simulated the compaction of pore space as an asymptotic decrease with depth, bounded by pre-set minimum and maximum pore space values. I use a modified version of Callaway's algorithm, where the decrease in pore space for a give cohort ( $pore\_space_n$ ) is a function of the mass of material above it:

$$porespace_n = poremin + ((poremax - poremin) * compact_n) \quad (5)$$

Where:

$pore\_space_n$  = pore space of cohort  $n$  (%)

$poremin$  = minimum pore space for the entire sediment column (%)

$poremax$  = maximum pore space for the entire sediment column (%)

$$compact_n = 1 - (tmass_i/(compk + tmass_i)) \text{ (unitless)} \quad (6)$$

The parameter,  $compact_n$ , describes a Michaelis-Menten type reduction in pore space where:

$tmass_i$  = mass of sediment overlying cohort  $n$  ( $g\ cm^{-2}$ )

$compk$  = half saturation compaction constant (unitless)

$Poremin$ ,  $poremax$  and  $compk$  values are derived from site specific soil cores collected to a depth of approximately 40 centimeters.

### **Cohort Transfer**

Sediment processes that can ultimately influence wetland elevation occur at several time scales (Figure 6.1). For example, annual fluxes of o.m. litter to the wetland surface, and the rapid decomposition of the labile component of that litter can cause short term oscillations in wetland elevation that may not reflect long term trends. Conversely, the primary compaction of shallow sediments and the decomposition of refractory o.m. influence wetland elevation over several decades. To address this problem of scale, cohort size in this model is a function of time and depth. The sediment state variables, labile o.m., refractory o.m., live root biomass and mineral matter are passed from cohort to cohort according to the sequence; 1, 1, 1, 1, 1, 2, 2, 2, 2, 2, 5, 5, 5, 5, 5, 10, 10 and 10+ years (a total of 18 cohort layers for each of the state variables). 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 processes, which for the most part occur at decades-long time scales, are simulated within the cohorts with the longest retention time.

### **Primary Productivity Sub-Model**

The purpose of the forested wetland primary production sub-model is to simulate the production of in-situ organic matter in a forested wetland, 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. Phipps (1979) constructed a detailed model of tree growth in southern forested wetlands that was based on an earlier model developed by Botkin et al. (1972) for eastern deciduous forests. These models "grow" individual trees and simulate processes such as competition (crowding and shading), mast production, reproduction and hydroperiod. Unfortunately, these well tested and documented models simulate only aboveground tree growth and ignore



the other processes, such as belowground production and the production associated with floating aquatic macrophyte communities, that also contribute to soil development and accretion. Since it has been estimated that as much as 60% of the annual increment to soil organic matter comes from roots (Megonigal and Day 1992), it is critical to simulate the effect of flooding on belowground production, as well as aboveground production. Additionally, because the critical output required from the productivity sub-model is merely overall net primary production ( $\text{g m}^{-2} \text{ week}^{-1}$ ), individual tree-growth type models, that simulate detailed community dynamics, are probably unnecessarily mechanistic for models primarily concerned with simulating wetland elevation. However, I was able to incorporate some useful sub-routines from these models, described below, into the primary production sub-model. Organic matter in the primary production sub-model was separated into three state variables associated with tree growth; *leaf* (tree leaf biomass), *wood* (woody stem biomass), and *root* (belowground biomass), and one state variable representing floating aquatic vegetation (FAV) biomass, *float*.

#### Aboveground Tree Production

Leaf biomass standing crop is calculated as:

$$d(\text{leaf})/dt = (\text{maxlgrow} * \text{hydrofunc}) - (\text{leaf} * \text{llitter}) \quad (7)$$

where:

*leaf* = leaf biomass ( $\text{g d.w. m}^{-2}$ )

*maxlgrow* = maximum net leaf productivity ( $\text{g d.w. m}^{-2} \text{ week}^{-1}$ )

*hydrofunc* = wetland flooding function that modifies *maxlgrow* (unitless)

*llitter* = leaf litter production rate ( $\text{week}^{-1}$ )

Simulated net tree leaf production is limited by the amount of live aboveground wood biomass (*wood*) available for support. To estimate the maximum leaf production (*maxlgrow*) for a given amount of wood biomass, I 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 aboveground wood biomass. These 87 data sets were grouped into 15 size classes according to the aboveground wood biomass ( $0 - 5 \text{ kg m}^{-2}$ ,  $> 5 - \leq 10 \text{ kg m}^{-2}$ ... $> 70 - \leq 75 \text{ kg m}^{-2}$ ). Then, using only the one data set within each size class with the greatest annual leaf productivity, aboveground wood biomass was regressed (second order polynomial) against leaf productivity to produce an estimate of maximum leaf productivity. Other techniques, such as the calculation of 90% expectile plots, that used the entire data set, were employed to obtain estimates of maximum leaf productivity given a certain biomass. All methods yielded similar results, however I used the simple polynomial regression of the restricted data set because it produced more realistic results as biomass approached zero.

*Maxlgrow* is limited by the water level function (*hydrofunc*) originally developed by Phipps (1979) for southeastern U.S. forested wetlands:

$$\text{hydrofunc} = 1 - 0.5511 (T - W)^2 \quad (8)$$

where:

*hydrofunc* = water-table growth factor (unitless multiplier between 0 and 1)

T = water table depth (meters)

W = optimum water table depth for a given species (meters)

Phipps' generalized equation was intended for water levels below the surface. If simulated water levels are above the surface, then leaf productivity is held constant at 78% of *maxlgro* (Conner and Day 1989). Leaf litter production is a function of time and the amount of plant biomass present, and is calibrated to reflect field measurements.

Baseline field measurements from the Thibodaux site showed that wood biomass production was roughly equivalent to leaf biomass production, therefore stem biomass standing crop was modeled as:

$$d(\text{wood})/dt = (\text{maxlgrow} * \text{hydrofunc}) - \text{wmort} \quad (9)$$

where:

*wood* = stem biomass (g d.w. m<sup>-2</sup>)

*wmort* = mortality rate derived from field data (week<sup>-1</sup>)

### **Belowground Production**

Simulating the effect of hydroperiod on root production is more complicated because recent studies suggest that belowground production is not simply a function or constant percentage of aboveground production. Originally, Mitsch and Ewel (1979) introduced the forested wetland stress-subsidy hypothesis which suggested that too much, or too little, water reduces total net primary productivity. However, most of the evidence for this hypothesis was derived from observations of aboveground production only. Recent studies have suggested that unflooded sites are actually the most productive when belowground productivity is also considered (Meronigal and Day 1992; Day and Meronigal 1993). The most comprehensive data sets related to belowground production and hydroperiod in forested wetlands of the southeastern U.S. come from the work of Day and Meronigal (1993) in the Dismal Swamp region of southeastern Virginia and northeastern North Carolina. They have shown that; 1) intermediately flooded sites were marginally more productive than drier and wetter sites when only aboveground productivity was considered and, 2) in the most flooded sites, belowground production ranged from 5.6% to 36% of above production, while in the drier mixed-hardwood sites, above and belowground production were equal. To reflect these recent findings, root production was simulated as:

$$d(\text{root})/dt = \text{rshunt} - (\text{root} * \text{rlirate}) \quad (10)$$

where:

$root$  = live root biomass (g d.w. m<sup>-2</sup>)

$rshunt = ((maxlgrow * hydrofunc) * 2)$  if water levels are  $\geq 0$  or

$((maxlgrow * hydrofunc) * .51)$  if water levels are  $\leq 0$

$rlirate$  = root litter rate (week<sup>-1</sup>)

### Floating Aquatic Vegetation

Many studies of annual net and gross primary production in floating aquatic macrophyte communities such as those found at the Thibodaux site (e.g. *Lemna* sp., *Salvinia* sp., *Wolffia* sp.) have been conducted in pools, ponds and wetlands associated with wastewater treatment. This would suggest that the literature would be rich with the data required for modeling population dynamics in natural systems, including; net and gross primary production, respiration, mortality, litter production, decomposition and turnover. However most of these studies did not consider limitations due to physical crowding and shading (either because the study included periodic harvesting, the data was collected only when space was not limiting, or, measurements were only collected for a few days or weeks and not for an entire year). Therefore, while rates of maximum gross or net productivity are common, estimates of turnover and annual production in systems not subject to period harvest are rare.

In natural systems, the relative growth rate of floating aquatic macrophytes has been found to decrease exponentially with increased crowding (Rejmankova 1982). Reflecting this, Sklar (1983) developed an aquatic materials flow model of a cypress tupelo forest in Louisiana that described primary production of *Lemna* as a function of its own biomass. We used a similar approach to simulate floating aquatic biomass, where:

$$d(float)dt = ((maxnet * spacefunc * ifunc1) * float) - (flitter * float) \quad (11)$$

where:

*float* = floating aquatic macrophyte standing crop (g d.w. m<sup>-2</sup>)

*maxnet* = maximum net primary production rate (week<sup>-1</sup>)

*spacefunc* = space limitation function (unitless from 0 to 1)

*tfunc1* = temperature imitation function:

*flitter* = litter production rate (week<sup>-1</sup>)

*Spacefunc* describes an exponential decrease with increasing macrophyte biomass,

where:

$$spacefunc = e(spacek * float) \quad (12)$$

and: spacek = exponential crowding coefficient (g d.w.<sup>-1</sup> m<sup>-2</sup>)

*Tfunc* describes a piecewise linear temperature function (Bowie 1985) of the form:

$$Tfunc = (1/(topt - tmin))T * (tmin/(topt - tmin)) \text{ if temp} > 13, \text{ else } 0 \quad (13)$$

where:

*topt* = optimum temperature for growth (°C)

*tmin* = minimum temperature for growth (°C)

Values for all rates and constants included in the primary production sub-model are shown in Table 6.1.

### Relative Elevation Sub-Model

Previous wetland accretion/subsidence models 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 in the treatment sites was measured at approximately 76 cm above sea-level. However, during the pre-treatment years 1989 and 1990, when precipitation was near normal, mean annual water depths in the Pointe au Chene treatment site were  $17.4 \pm 1.8$  and  $19.5 \pm 2.4$  cm respectively, and the site

was continually flooded during this period. Therefore, wetland elevation is simulated relative to mean annual water depth in this continually flooded wetland, 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 eustatic sea-level rise, deep subsidence, shallow subsidence and the accretion of organic and inorganic material. This is then added or subtracted from the initial wetland elevation at the start of the simulation. The accretion of mineral matter is modeled explicitly with the *minin* function described in the sediment dynamics sub-model. Inputs of o.m. is simulated in the primary productivity sub-model. Shallow subsidence is modeled explicitly with the decomposition and pore space compaction functions described in the sediment dynamics sub-model. The combination of inorganic and o.m. accretion, decomposition and compaction result in the development of a soil column over simulated time. The total height of this column, *theight18*, is calculated as the height of the deepest sediment cohort plus the total height of all overlaying cohorts.

The remaining parameters that affect simulated relative elevation, deep subsidence and ESLR are entered into the model as forcing functions. There are three ESLR scenario options programmed into the simulation, based on current IPCC reports; 1) "Current status" scenario: 15.0 cm in the next 100 years, 2) "Best estimate" scenario: 48 cm in the next 100 years and 3) "Business as usual" scenario: 66 cm in the next 100 years (Gornitz 1995).

## CALIBRATION AND INITIALIZATION

Data required for model initialization are shown in parts a, b and c of Table 6.1. Field data required for model calibration include annual aboveground production, some estimate of accretion and, most critically, sediment bulk density, % organic matter and

% pore space with depth (Figure 6.2). The model was first run for 100 simulated 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. Previous studies (Chapter 4) 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  $^{137}\text{Cs}$  analysis (Chapter 4). A comparison of simulated and actual water levels after 20 years provided an additional point for calibration.

The simulated soil column profile closely matched the profile obtained from cores collected at the Thibodaux site during 1993 (Figure 6.4). Additionally, simulated and actual production, accretion and elevation values were in close agreement. Simulated output from model-year 1989 was compared to pre-effluent baseline measurements taken from the Thibodaux site in 1989 (Table 6.3).

Table 6.3. Key calibration points for the Pointe au Chene simulation.

Parameter	Simulated Results	Field Measurement $\pm$ se
Cs-137 Accretion	0.40 cm yr <sup>-1</sup>	0.44 $\pm$ 0.04 cm yr <sup>-1</sup>
Annual Leaf Litter	413 g m <sup>2</sup> yr <sup>-1</sup>	386.1 $\pm$ 18.2 g m <sup>2</sup> yr <sup>-1</sup>
Wood Production	413 g m <sup>2</sup> yr <sup>-1</sup>	431.7 g m <sup>2</sup> yr <sup>-1</sup>
Tree Standing Crop	13.7 kg m <sup>2</sup>	13.6 kg m <sup>2</sup>
Relative Wetland Elev.	-16.8 cm	-17.4 $\pm$ 1.8 cm

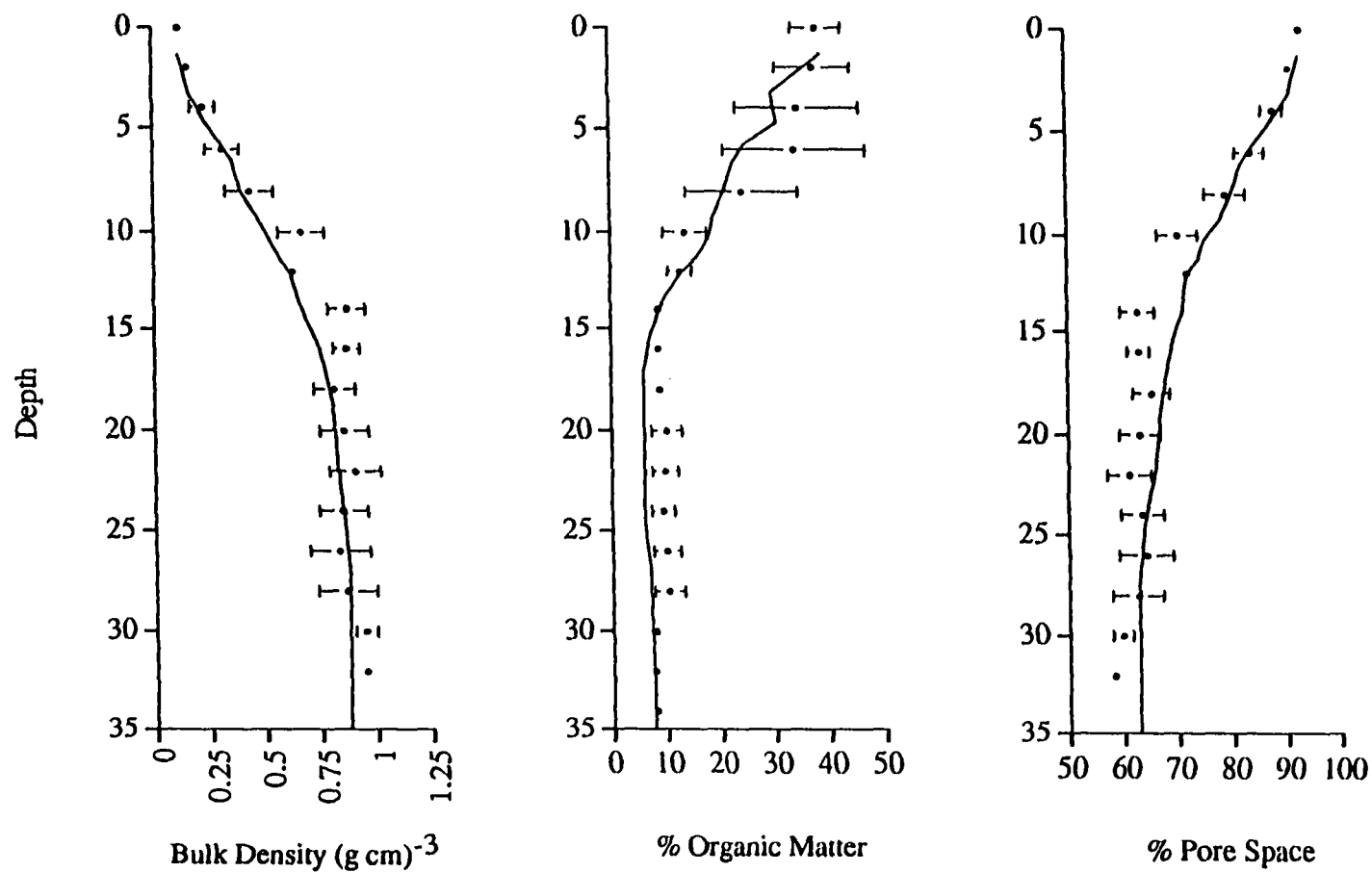


Figure 6.4. Pointe au Chene swamp sediment profiles. Field measurements are shown as dots with standard error bars. Solid lines represent simulated results.



Simulated litter and wood production were within six percent of field measurements. Model generated long term accretion rates were within one standard error of accretion rates measured in the field using  $^{137}\text{Cs}$  analyses. Simulated water depths (16.8 cm) were also within one standard error of actual values.

## **MODEL RESULTS AND DISCUSSION**

In this section I present the results of sensitivity and validation exercises. Then, using the results obtained from the sensitivity analyses as a guide, a series of simulation were conducted in order to examine the effects of, wastewater, subsidence rates and mineral inputs on wetland elevation.

### **Sensitivity Analyses**

Sensitivity analyses allows us to 1) detect parameters that markedly affect simulated output, 2) test the stability of the model under a wide range of values and 3) determine the influence of critical parameters that have a high degree of uncertainty (such as subsidence rates and predicted increases in sea-level. We examined the sensitivity of relative wetland elevation to changes in appropriate forcing functions and initialization parameters (Table 6.4). Because changes in wetland elevation over time are not always linear, analyses were run for five and fifty year time periods. Parameters were varied by plus and minus 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. The analyses revealed that simulated wetland elevation is most sensitive to initial minimum and maximum pore space, deep subsidence rates, the soil compaction constant, parameters associated with tree production, mineral inputs and eustatic sea-level rise (Table 6.4). It should be noted that a global analyses of this type may not reflect the probable or even possible range of variation for each parameter. However, once

Table 6.4. Relative sensitivity of wetland elevation to plus and minus 5% and 50% changes in model parameters. 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:  $(\% \text{ change in relative elevation} / \% \text{ change in parameter})$ .

Parameters	Description	<u>± 5%</u>		<u>± 50%</u>	
		<u>5 years</u>	<u>50 years</u>	<u>5 years</u>	<u>50 years</u>
<b>A. RSLR Parameters</b>					
eslr_c	current eustatic sea level rise	0.19	0.16	0.19	0.16
surate	local deep subsidence rate	1.49	1.21	1.46	1.21
<b>B. Production Parameters</b>					
hydrofunc	flooding stress multiplier	0.50	0.24	0.49	0.21
maxlgrow	tree net productivity	0.49	0.24	0.49	0.21
maxnet	FAV net prod.	0.02	0.007	0.03	0.07
rootk	root distribution constant	0.06	0.001	0.06	0.006
rshunt	production allocation to roots	0.27	0.08	0.27	0.08
wmort	tree mortality rate	0.03	0.16	0.03	0.10
<b>C. Soil Compaction Parameters</b>					
compk	soil compaction constant	1.85	2.32	2.02	.19
poremax <sup>1</sup>	maximum pore space	22.42	2.62	-	-
poremin <sup>1</sup>	minimum pore space	9.84	1.27	-	-
<b>D. Decomposition Parameters</b>					
floatlabfrac	% labile FAV	0.22	0.07	0.19	0.06
kdeep	decomp. rate of deep organics	0.18	0.02	0.02	0.02
klab	decomp. rate of labile roots	0.007	0.0004	0.008	0.0006
klabsurf	decomp. rate of surface labile org.	0.01	0.001	0.01	0.002
kref	decomp. rate of refractory roots	0.05	0.05	0.05	0.05
krefsurf	decomp. rate of surface refract.	0.03	0.007	0.02	0.007
leaflabfrac	fraction of labile leaf material	0.08	0.02	0.08	0.02
rlabfrac	fraction of labile root material	0.03	0.01	0.02	0.015
<b>E. Mineral Inputs</b>					
maxminin	maximum mineral inputs	0.31	0.23	0.20	0.22

<sup>1</sup>Values could not be varied  $\pm 50\%$  because doing so propagated logic errors in the compaction function algorithm.

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

Wetland elevation was most sensitive to changes in the *poremin* and *poremax* functions. Changes in these parameters trigger an immediate re-calculation of the compaction algorithm that affects the entire simulated soil column at one time. No other parameters have an instantaneous effect on the entire sediment column. However, sensitivity to these parameters decrease with time.

The uncertainty surrounding estimates of ESLR in the next century, and deep subsidence in the coastal zone have been well documented (Turner 1991; Gornitz 1995), and this analysis reveals that wetland elevation is relatively sensitive to both of these forcing function parameters, although both of these parameters are outside the influence of wetland wastewater effluent. Elevation is also shown to be sensitive to organic matter production, a parameter that may be affected by effluent associated nutrients. I examine, in greater detail, the effect of these parameters on wetland elevation in the following sections.

Sensitivity analyses indicated that wetland elevation was relatively insensitive to the parameters that control the rates of organic matter decomposition (Table 6.4). Day et al. (1992) originally hypothesized that nutrient amendments to the wetland would stimulate organic matter production and increase wetland elevation. However, some studies have shown that nutrient amendments could possibly stimulate organic matter decomposition (Rybczyk 1996a), and have a negative effect on wetland elevation. This analysis suggests that elevation would be relatively unaffected by effluent induced changes in decomposition rates.

## **Validation**

Model validation involves the comparison of model generated output with real world observations (Mayer and Butler 1993). Although validation has often been

confused with calibration, good model performance at the calibration stage does not necessarily mean that the model will be valid for predictive purposes, or for independent data sets, sites or subjects (Power 1993). Although a model can never be proven valid, only invalid, numerous validation techniques have been proposed as a guide to measure agreement between observed and simulated results. Ideally, validation exercises should be performed on a data set independent of the data set used for model calibration (Jorgensen 1986). This can be done by splitting the observed data into calibration and prediction data sets. Most commonly this technique is employed for time series data where the first half of the data is used to calibrate the model, and the second half is used for validation. Another technique is to calibrate the model using a data set from one site, and validate it using a data set from another site; this is the approach I employed for this model. To run a validation simulation, the only permitted modification to the original model is to enter the forcing functions and input parameters that apply to the new site (initialization). This modification is allowed only prior to the simulation. Although the temptation is huge, comparisons of observed and simulated results, for the purpose of validation, must be performed without the benefit of re-calibration.

To validate the model, I used a data set collected from a bottomland hardwood ridge adjacent to the treatment swamp. Field data included measurements of annual aboveground production (1988 - 1996), estimates of accretion using  $^{137}\text{Cs}$  analysis, and, sediment bulk density, % organic matter and % pore space with depth. The ridge site is approximately 40 cm higher than the treatment site, and is not inundated for most of the year (Rybczyk 1995). Vegetation is dominated by typical bottomland species such as oak (*Quercus nigra*), sweetgum (*Liquidambar styraciflua*) and elm (*Ulmus americana*).

After initialization, the model was run using the same procedure described for the calibration exercise. Model-generated and real world results were compared visually using predicted vs observed plots for bulk density, pore space and % organic matter (Figure 6.5). This technique represents goodness of fit as vertical deviations from the "perfect fit" line with a slope of one (Mayer and Butler 1993). I also used the dimensionless statistic; EF (modeling efficiency) (Loague and Green 1991), to relate these simulated values to observed values at the ridge site. EF is defined as :

$$EF = 1 - \sum(y_i - y_o)^2 / \sum(y_i - y_m)^2 \quad (14)$$

where:

$y_i$  = observed value

$y_o$  = predicted, or simulated value

$y_m$  = mean of observed values

EF parallels the commonly employed coefficient of determination ( $R^2$ ) except that the lower bounds for 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).

In general, observed vs. simulated results were in close agreement and all EF values were above zero (Figure 6.5). Deviations from the "perfect fit" line in the bulk density and % pore space plots can be attributed to a lens of low bulk density material occurring between 6 and 8 centimeters in the soil cores that the model did not simulate. Additionally, the model consistently under estimated the % organic matter in the soil column by two to five percent (Figure 6.5). This is probably due to inaccurate estimates of decomposition rates during initialization. The simulated long term accretion rate (30 years) was  $0.95 \text{ mm year}^{-1}$  compared to the mean observed  $^{137}\text{Cs}$  accretion rate of  $1.0 \pm 0.3 \text{ mm } (\pm \text{ s.e.}) \text{ year}^{-1}$ . Simulated aboveground production (leaf plus

wood) was 823.3 g d.w. m<sup>-2</sup> year, compared to an observed total aboveground production of 816.1 g d.w. m<sup>-2</sup> during 1990.

### **ESLR Scenarios**

To simulate the effect of various ESLR scenarios at the Pointe au Chene swamp under baseline conditions (not effluent), I used as forcing functions, an estimate for deep subsidence of 1.08 cm year<sup>-1</sup> (Penland et al. 1988), and three IPCC predicted increase in ESLR estimates; 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). For the first model run, simulations began in model-year 1970 under the "current trends" scenario. Then in model-year 1990, the "best estimate" forcing function was switched on for another 30 model-years. The same procedure was followed for the remaining IPCC scenarios. The simulations revealed that, under all three scenarios, given no intervention, relative wetland elevation would decrease and remain below zero for the entire 50 year simulation (Figure 6.6). After 50 years, the difference in wetland elevation between the highest elevation (current trends scenario) and the lowest (business as usual) was 11.67 cm, although it should be noted that the scenario switch did not come on until model-year 20, so the difference in elevation reflects only 30 model-years. It was unnecessary to run the model for additional years because mineral inputs reached a maximum early in the simulation (when relative elevation fell below 0 cm), autogenic organic matter production continued to decrease and decomposition rates did not change.

### **Wastewater Applications**

Effluent additions did not increase tree production at the Pointe au Chene site (Chapter 5), but did increase the percent cover of the floating aquatic vegetation (FAV) (Zhang 1995). In this section, the effect of increased F.A.V. productivity is examined. Before wastewater applications began, FAV was dominated by duckweed (*Lemna* sp.)

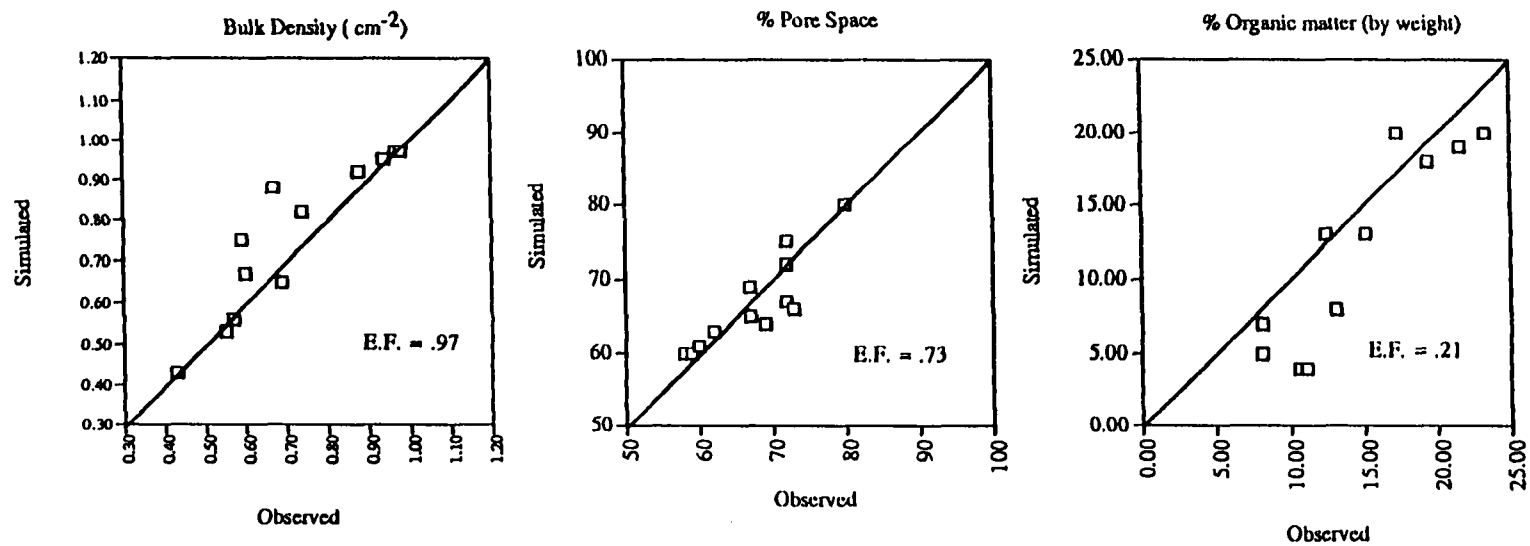


Figure 6.5. Observed vs. predicted plots for the Pointe au Chene Ridge site. E.F. = modeling efficiency (see text for explanation).

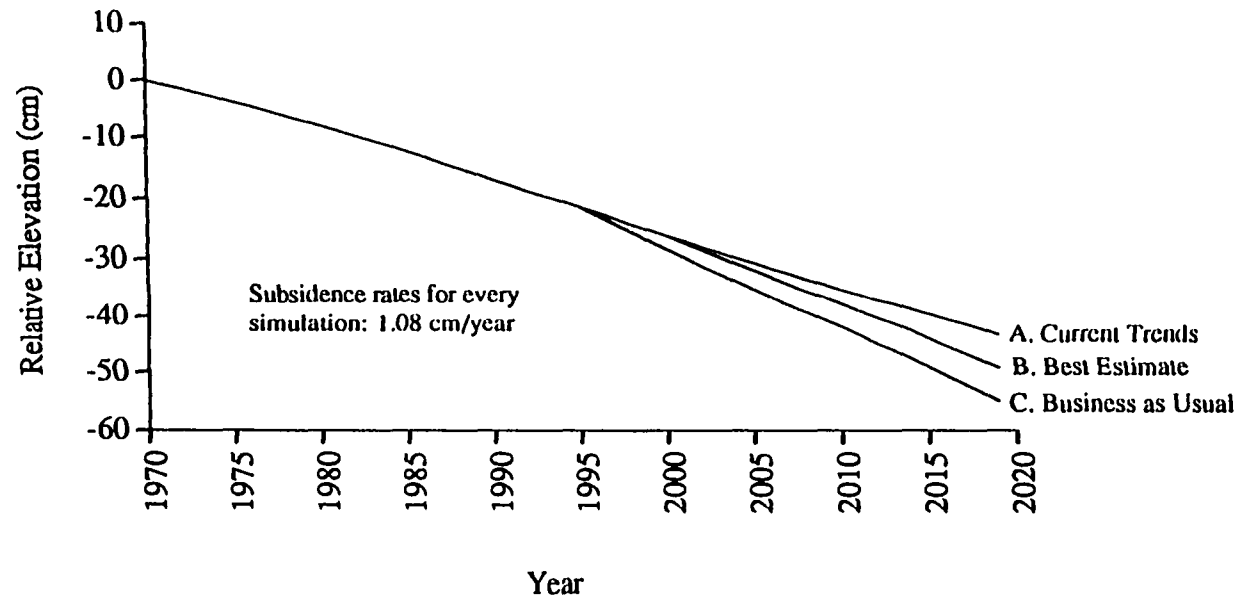


Figure 6.6. Changes in relative wetland elevation under three IPCC ESLR scenarios; A) "Current Trends" 15.6 cm in the next 100 years, B) "Best Estimate" 48 cm in the next 100 years, and C) "Business as usual" 66 cm in the next 100 years. After a 50 year simulation, final relative wetland elevations for A, B and C are - 43.3 cm, - 49.18 cm and - 54.97 cm respectively.



Since neither FAV production or standing crop was measured at the site, the model was initialized using data presented by Sklar (1983) for similar forests in southern Louisiana. Baseline (non-effluent) simulated peak standing crop and production for FAV were  $132.7 \text{ g d.w. m}^{-2}$  and  $467.0 \text{ g d.w. m}^{-2}$  respectively. This is similar to both Sklar's simulated production value of  $475.6 \text{ g d.w. m}^{-2}$  and Porath's (1979) estimate of duckweed production in natural systems ( $474.4 \text{ g d.w. m}^{-2}$ ). Simulated peak standing crop measurements are also in close agreement with observed values of  $167.0 \text{ g d.w. m}^{-2}$  (Sklar 1983). It is well known that wastewater effluent can stimulate FAV production in both natural wetland wastewater treatment systems (Ewel and Odum 1984) and in sewage oxidation ponds and lagoons (Reddy and Debusk 1985). Annual production rates of small leaved FAV have exceeded  $5000 \text{ g d.w. m}^{-2}$  in wastewater treatment systems with harvest (Debusk and Ryther 1987). In natural systems impacted by wastewater effluent, standing crops as high as  $400 \text{ g d.w. m}^{-2}$  have been observed. To simulate the effect of wastewater effluent in the Pointe au Chene swamp, the model was run under natural conditions settings from simulated year 1970 to 1992, and then FAV production values were switched so that peak standing crop reached  $400 \text{ g d.w. m}^{-2}$  for each of the remaining 28 years of the simulation (50 total years). Under these conditions simulated FAV production equaled  $1434 \text{ g d.w. m}^{-2}$ . This rate is approximately three times lower than the average rate reported by Debusk and Ryther (1987) for harvested wastewater systems in Florida. The model was run under IPCC "current conditions" scenario so that the contribution from ESLR would remain constant throughout the simulation. Increasing the simulated rate of F.A.V. production after 1992 had little effect, increasing the relative elevation only 1.12 cm over the baseline simulation (duckweed without effluent), in the remaining 28 years (Figure 6.7). This was primarily due to the rapid decomposition rate for duckweed, as 80% of the litter is

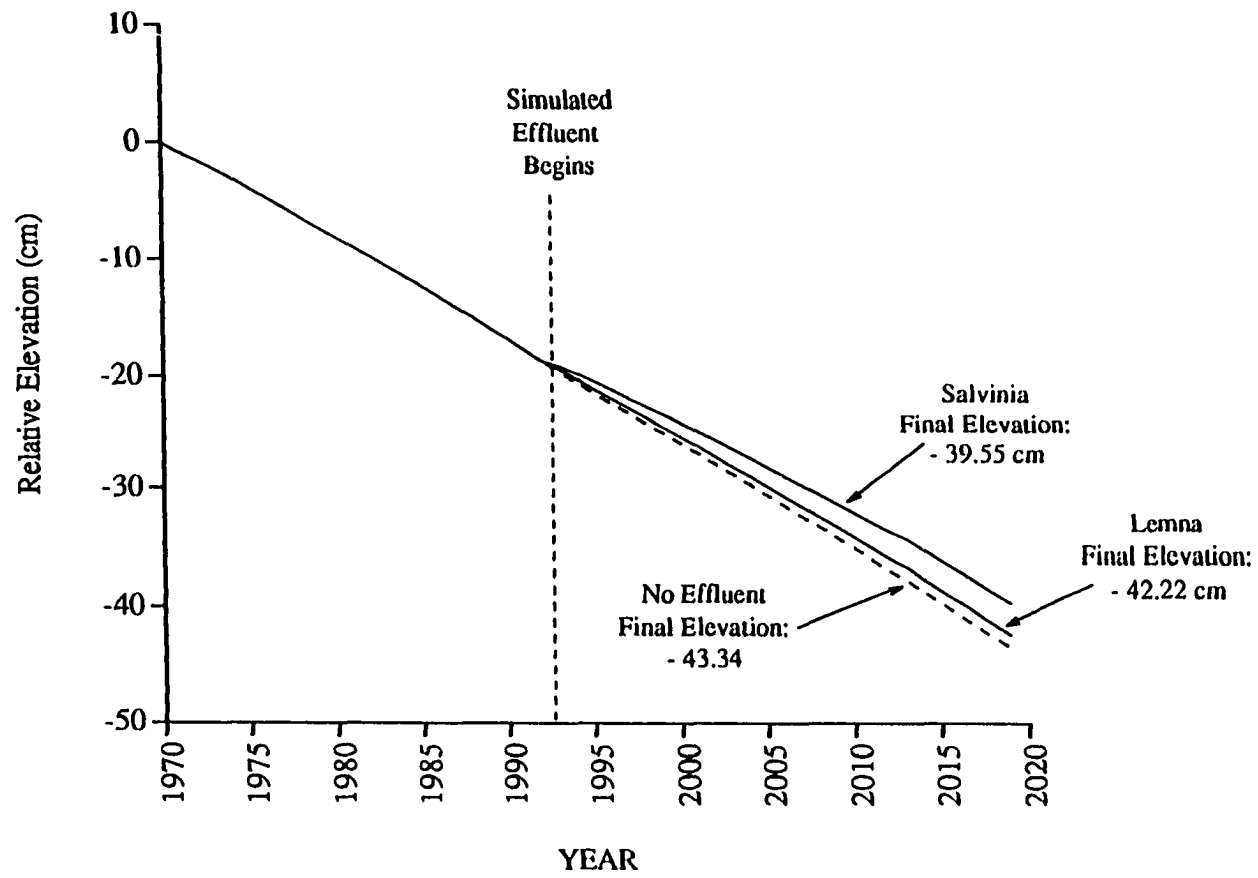


Figure 6.7. The effect of floating aquatic vegetation on relative wetland elevation. ESLR is fixed at  $0.15 \text{ cm year}^{-1}$ . Effluent additions begin in simulated year 1992.

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 duckweed as the dominant FAV in the treatment site. Preliminary decomposition experiments and subsequent non-linear regression analysis (Garson and Rybczyk; unpublished data) revealed that, unlike the primarily labile duckweed, 50% of *Salvinia* consisted of refractory material. In addition, the decomposition rate of the refractory component was markedly slower than published decomposition rates for duckweed. To simulate the effect of this species shift on relative wetland elevation, I followed the exact scenario described for the effluent stimulated duckweed simulation, except that after 1992, the *float\_lab\_frac* parameter was changed from 0.8 to 0.5 and the decomposition *krefsur* parameter was switched from  $0.028 \text{ week}^{-1}$  to  $0.0007 \text{ week}^{-1}$  (both of these parameters control FAV decomposition). Under this scenario, final simulated wetland elevation after 50 years was -39.55 cm, 2.67 centimeters higher than the duckweed with effluent simulation and 3.79 centimeters higher than the duckweed without effluent simulation (Figure 6.7). The effect of these simulated changes on long term accretion rates (equivalent to  $^{137}\text{CS}$  measurements in the field) were also examined. From 1970 to 1993, the baseline simulated long term accretion rate was  $0.35 \text{ cm year}^{-1}$ . Under the effluent stimulated duckweed scenario long term accretion rates increased to  $0.36 \text{ cm year}^{-1}$  over the remaining 28 years. For the equivalent time period, long accretion rates in the *Salvinia* simulation increased to  $0.46 \text{ cm year}^{-1}$ .

### **Subsidence Rates**

Numerous researchers have discussed the uncertainty surrounding the subsidence component of RSLR (Penland et al. 1988, Turner 1991; Cahoon et al. 1995). Furthermore, sensitivity analyses revealed that relative wetland elevation is

sensitive to changes in this forcing function. Estimates for deep subsidence in the vicinity of the Pointe au Chene swamp range from 0.53 to 1.08 cm year<sup>-1</sup> (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), varying only subsidence. Over 50 simulated years, starting in model year 1970, relative elevation decreased to -18.4 cm after 50 years when subsidence rates equaled the minimum 0.53 cm year<sup>-1</sup>, and decreased to -43.3 cm at the maximum subsidence rate of 1.08 cm year<sup>-1</sup> (Figure 6.8).

To simulate a "best case" scenario, subsidence rates were initialized at 0.53 cm year<sup>-1</sup>, ESLR rates were fixed at IPCC "current conditions" and simulated organic matter production included effluent stimulated *Salvinia* production. Even under these optimal conditions, relative elevation decreased to -14.6 cm by 2020 (Figure 6.8).

### **Mineral Inputs**

Like most other distributary wetlands of the Mississippi River delta, the historical source of mineral inputs to the Pointe au Chene wetlands were essentially eliminated after the completion of the levee along the Mississippi River in the early 1930's. Recent restoration efforts have focused on re-introducing inorganic sediments to wetlands to balance accretion deficits caused in large part by the construction of those levees (Boesch 1994). Since this model considers such processes as compaction, decomposition, above and belowground production and decomposition, that effect relative elevation, it can be used to estimate the amount of inorganic sediment required to maintain wetland elevation in the face of RSLR.

Using conservative forcing function estimates of ESLR (IPCC "current conditions scenario") and subsidence (0.53 cm year<sup>-1</sup>), and no wastewater effluent, I ran a series of 100 year simulations, starting in model year 1970, varying only the addition of mineral sediment. Given an initial elevation of 0 cm, the wetland required

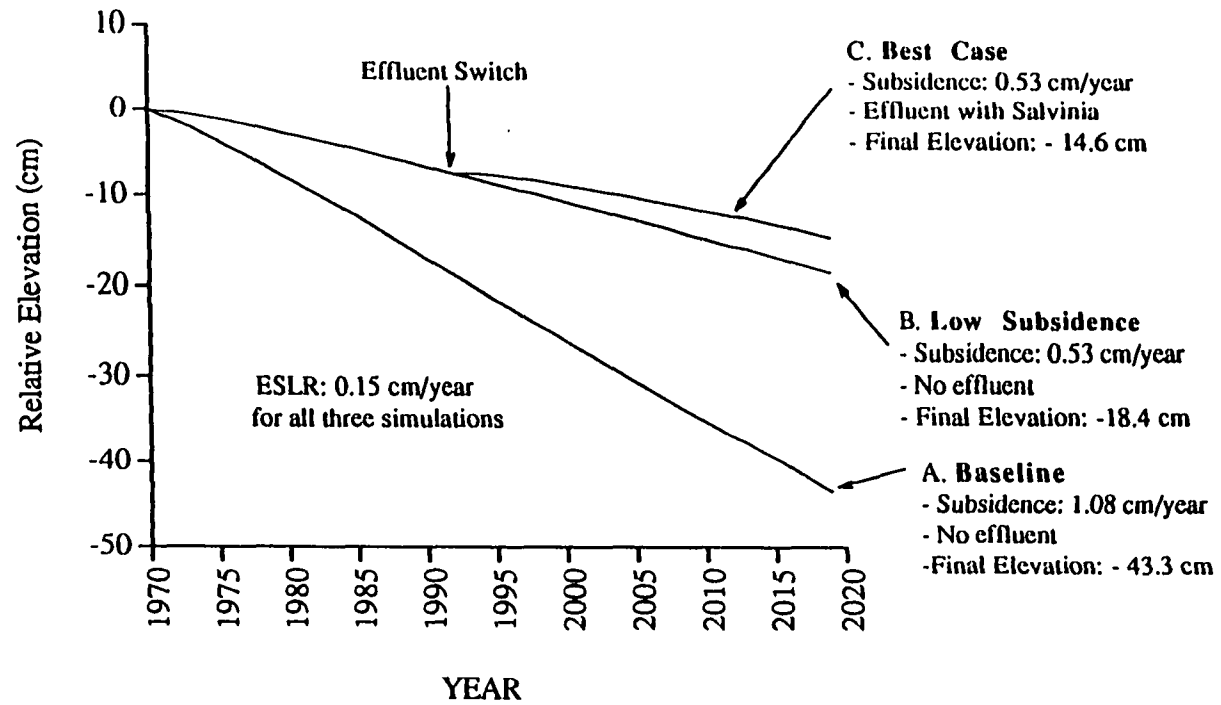


Figure 6.8. The effect of subsidence rates on relative elevation

an additional  $3000 \text{ g m}^{-2} \text{ year}^{-1}$  of mineral sediment to maintain a stable elevation over 100 years. However, given an initial elevation of -5 cm, the wetland required approximately  $4000 \text{ g m}^{-2} \text{ year}^{-1}$  to maintain that initial negative elevation, and  $4500 \text{ g m}^{-2} \text{ year}^{-1}$  to approach 0 cm elevation. In effect, when initial elevations are 0 cm or higher, overall accretion rates are higher than when initial elevations are below 0 cm, even though mineral inputs are equal (Table 6.5).

Net aboveground production is low in permanently flooded wetlands relative to those that are seasonally inundated (Conner et al. 1988). In addition, Day and Megonigal (1993) have shown that belowground production and root standing crop biomass is dramatically reduced in permanently flooded forested wetlands. For forested wetland systems in the temperate zone, there is little or no autogenic response to the addition of mineral sediments (not including the possible response involving nutrients associated with the mineral inputs (Nyman 1990)) to a permanently flooded system until a critical point, or elevation is reached, at which there is some relief from flooding stress during the growing season, or for re-generation. In other words, adding 5 cm of pre-compacted mineral sediment to a wetland that is permanently flooded with 30 cm of water, will have no effect but to raise the elevation to 25 cm below the water level until some critical elevation is obtained. Of course this point will vary by species (Phipps 1979), and by year, depending upon local hydrologic conditions, and for the purposes of this simulation, I have imposed the 0 cm elevation as this critical point. However, once a critical elevation is obtained, ecosystem response can include increased above and belowground production, seedling establishment and forest regeneration. Then, the addition of 5 cm sediment can result in a greater than 5 cm increase in elevation. The set of simulations shown in Table 6.5 illustrate this point. Under Scenario A, the addition of  $3000 \text{ g m}^{-2} \text{ year}^{-1}$  of sediments, in combination

with increased organic matter production, maintains the wetland above the critical elevation.

Table 6.5. Relationship between mineral inputs and accretion rates given two different initial elevations.

<b>Init. cond. for both scenarios:</b>		
Aboveground (leaf and wood)	838.0 g m <sup>-2</sup> yr <sup>-1</sup>	
Belowground root production	427.4 g m <sup>-2</sup> yr <sup>-1</sup>	
F.A.V. Production	467.0 g m <sup>-2</sup> yr <sup>-1</sup>	
Total Production	1732.4 g m <sup>-2</sup> yr <sup>-1</sup>	
Subsidence Rate	0.53 cm yr <sup>-1</sup>	
Mineral Inputs	3000 g m <sup>-2</sup> year <sup>-1</sup>	
	<b>Scenario A</b>	<b>Scenario B</b>
<b>Initial Relative Elevation:</b>	<b>0 cm</b>	<b>-10 cm</b>
<b>After 100 year Simulation</b>		
Aboveground (leaf and wood)	1588.4 g m <sup>-2</sup> yr <sup>-1</sup>	791.5 g m <sup>-2</sup> yr <sup>-1</sup>
Belowground root production	1588.4 g m <sup>-2</sup> yr <sup>-1</sup>	403.6 g m <sup>-2</sup> yr <sup>-1</sup>
F.A.V. Production	0 g m <sup>-2</sup> yr <sup>-1</sup>	467.0 g m <sup>-2</sup> yr <sup>-1</sup>
Total Production	3176.8 g m <sup>-2</sup> yr <sup>-1</sup>	1662.1 g m <sup>-2</sup> yr <sup>-1</sup>
Final Relative Elevation	1.2 cm	-13.3
Accretion Rate	0.72 cm year <sup>-1</sup>	0.59 cm year <sup>-1</sup>

Long term accretion rates equal 0.72 cm year<sup>-1</sup> for a return of 0.24 cm year<sup>-1</sup> of accretion for every kilogram m<sup>-2</sup> of sediment delivered. In the permanently flooded wetland, represented by scenario B, the same 3000 grams resulted in a long term accretion rate of only 0.59 cm year<sup>-1</sup> for a return of 0.19 cm year<sup>-1</sup> of accretion for every kilogram of sediment delivered.

## CONCLUSIONS

To simulate the response of wetland elevation to wastewater effluent, 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 using 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.

Calibration and validations exercises for this site revealed that simulated output and field measurements were in close agreement. The model revealed that relative wetland elevation in the Pointe au Chene swamp was more sensitive to the uncertainty surrounding predictions of ESLR and deep subsidence than in processes that could possibly be influenced by wetland wastewater treatment. Even under the best case scenario, effluent associated nutrients did not stimulate productivity enough to offset RSLR. A series of mineral input simulations were conducted that suggested that sediment supplement management strategies would be more effective when implemented before increasingly flood stressed wetlands became permanently inundated.

This model represent a logical progression from earlier wetland elevation/relative sea-level rise models and extends the boundaries of application to forested wetlands with perennial vegetation. Further refinements would include linking this model to a site specific hydrology sub-model and spatial articulation.

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## CHAPTER 7

### CONCLUSIONS

I utilized an integrated field and modeling approach to determine if wastewater effluent applications to a hydrologically isolated, subsiding coastal wetland forest in Louisiana could stimulate organic matter production enough to offset estimated rates of relative sea-level rise (subsidence plus eustatic sea-level rise). Field studies were designed to measure the processes that affect wetland elevation including; organic matter decomposition, primary production and sediment accretion. These processes were measured for several years in a control and treatment site, both before and after wastewater applications began in the treatment site. Therefore, I was able to use a Before-After-Control-Impact (BACI) statistical design to detect effluent effects. The strength of this design was its ability to isolate background variation from the treatment effect. However, its weakness was a lack of power to detect effects because of low degrees of freedom.

Although nutrient amendments could potentially increase rates of organic matter production and subsequent deposition, enrichment could also increase rates of organic matter decomposition, thus negating any affect of increased productivity. An extensive literature review suggested that nutrient amendments would not affect long term decomposition rates and field studies at the Thibodaux wastewater treatment site confirmed this. It was suggested that effluent additions did not affect rates of decomposition because, 1) the effluent did not affect initial litter concentrations and 2) long term experiments such as this one are not sensitive to processes which may only affect the early stages of decomposition.

This study also revealed that although N, P and K plant tissue concentrations increased in response to effluent additions, there was no corresponding increase in aboveground tree production. Instead, dramatic decreases in aboveground productivity were observed in both the control and treatment site over the course of this study (1988 - 1995). This decrease was attributed to the confounding effects of nutria herbivory, Hurricane Andrew and prolonged inundation due to region-wide accretion balance deficits. It is important to note that effluent additions were not the cause of permanent inundation, as both the control and the treatment sites were continuously flooded during the course of this study. These confounding effects also illustrate the strength of the BACI design, as it would have been impossible to factor out these perturbations without a "before" time period and a control site.

Despite an observed decrease in aboveground tree productivity, short term measurements revealed that rates of sediment accretion increased significantly in the treatment site, in respect to the control, after effluent additions began and approached the estimated rate of RSLR. This was attributed to increases in floating aquatic vegetation production in the treatment site after effluent applications began.

However, direct comparison between estimated rates of RSLR and observed sediment accretion rates, to predict the fate of coastal wetlands, must be made with caution because short term measurements of accretion do not fully integrate long term processes that affect elevation, such as compaction and decomposition. Additionally, these comparisons do not consider elevation feedback mechanisms. For example, changes in elevation can result in changes in primary production, decomposition and sediment deposition. To examine the response of wetlands to increasing rates of RSLR, and to predict the long term effects of effluent additions on wetland sustainability I developed a wetland elevation cohort model that incorporates elevation feedback mechanisms and simulates sediment dynamics over decades. The model utilizes a

cohort approach (tracking discrete packages of sediments through depth and time) to simulate sediment dynamics (organic and mineral matter accretion, decomposition, compaction, and belowground productivity). These dynamics produce model-generated changes in sediment characteristics including: bulk density, organic matter (o.m.) volume and mass, mineral matter volume and mass, pore volume and mass, and yield 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 floating aquatic vegetation) and mineral inputs, both of which are a feedback function of the model-generated marsh elevation.

Calibration and validation exercises for this site revealed that simulated output and field measurements were in close agreement. The model suggested that relative wetland elevation in the Pointe au Chene swamp was more sensitive to the uncertainty surrounding predictions of ESLR and deep subsidence than in processes that could possibly be influenced by wetland wastewater treatment. Even under the best case scenario, effluent associated nutrients did not stimulate productivity enough to offset RSLR.

The field studies presented as part of this dissertation, and the related model, suggest that flood stress limits autogenic response to nutrient amendments . Net aboveground production is low in permanently flooded wetlands relative to those that are seasonally inundated. In addition, it has been shown that belowground production and root standing crop biomass is dramatically reduced in permanently flooded forested wetlands. For forested wetland systems in the temperate zone, there is little or no production response to the addition of mineral sediments or nutrients (not including floating aquatic vegetation) to a permanently flooded system until a critical point, or elevation is reached, at which there is some relief from flooding stress during

the growing season. In addition continuous flooding precludes the opportunity for forest re-generation. However, once a critical elevation is obtained, ecosystem response can include increased above and belowground production, seedling establishment and forest regeneration. This suggests that sediment or nutrient supplement management strategies would be more effective when implemented before increasingly flood stressed wetlands became permanently inundated.

Finally, while this study, and the issue of wetland loss due to high rates of RSLR, may presently only apply to the extensive wetlands associated with subsiding delta regions, the worldwide ESLR component of RSLR is expected to steadily increase over the next century due to the impacts of global warming. Therefore, this region can serve as a model for other coastal wetlands that may also face problems associated with rising water levels in the near future. The generic model presented here can readily be programmed for other coastal wetland systems such as fresh and salt water emergent marshes. Easily obtained site specific field measurements, such as accretion rates and soil core analyses, could be used for model calibration and validation. These models could then be used to identify vulnerable wetlands and to predict the outcome of various management options employed to counter rising rates of ESLR.

**APPENDIX A**  
**LETTER OF PERMISSION**



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24 November 1996

Dr. John M. Rybczyk  
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**APPENDIX B**  
**BACI ANOVA DESIGN FOR DECOMPOSITION DATA SET**

BACI Source Table

<u>Source</u>	<u>DF</u>	<u>Comments</u>
BA	1	Before or After
CT	1	Control or Treatment
BA x CT	1	This is the test for a treatment effect
Year[BA]	2	Random Effect, must be marked as such
<u>CT x Year[BA]</u>	<u>2</u>	<u>Error term not included in SAS model</u>

**APPENDIX C**  
**STELLA PROGRAM CODE**

## PRIMARY PRODUCTIVITY SUB-MODEL

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[ ] cleal(t) = cleal(t - dt) + (lshunt - llitter) * dt
INIT cleal = 0
INFLOWS:
  * lshunt = net_leaf
OUTFLOWS:
  * llitter = if cleal < 25 then 0 else if jday < 43 then (.015*cleal) else
    (1*cleal)
[ ] crool(t) = crool(t - dt) + (rshunt - rlitter) * dt
INIT crool = 1000
INFLOWS:
  * rshunt = if crool ≥ 2500 then 0 else if Relative_El ≥ 30 then (net_leaf+wshunt) else ((net_leaf+wshunt) * .51)
OUTFLOWS:
  * rlitter = liratemod*crool
[ ] cwood(t) = cwood(t - dt) + (wshunt - wmort) * dt
INIT cwood = 14000
INFLOWS:
  * wshunt = net_leaf
OUTFLOWS:
  * wmort = .0006*cwood
[ ] Floal(t) = Floal(t - dt) + (lnetpro - flitter) * dt
INIT Floal = 1
INFLOWS:
  * lnetpro = (maxnet*spacelunc*tlunc1)*Floal
OUTFLOWS:
  * flitter = if temp < 13 then (Floal*.30) else Floal*.08
[ ] littersum(t) = littersum(t - dt) + (flitter - purge) * dt
INIT littersum = 0
INFLOWS:
  * flitter = if temp < 13 then (Floal*.30) else Floal*.08
OUTFLOWS:
  * purge = if jday = 51 then littersum else 0
[ ] net_leaf(t) = net_leaf(t - dt) + (tgrow - lshunt) * dt
INIT net_leaf = 0
INFLOWS:
  * tgrow = if jday < 10 or jday > 20 then 0 else (Maxlgrow*hydrotunc)
OUTFLOWS:
  * lshunt = net_leaf
[ ] xannleaf(t) = xannleaf(t - dt) + (inl - outl) * dt
INIT xannleaf = 0
INFLOWS:
  * inl = lshunt
OUTFLOWS:
  * outl = if jday = 51 then xannleaf else 0
[ ] xannroot(t) = xannroot(t - dt) + (inr - outr) * dt
INIT xannroot = 0
INFLOWS:
  * inr = rshunt
OUTFLOWS:
  * outr = if jday = 51 then xannroot else 0
[ ] xannwood(t) = xannwood(t - dt) + (inw - outw) * dt
INIT xannwood = 0
INFLOWS:
  * inw = wshunt
OUTFLOWS:
  * outw = if jday = 51 then xannwood else 0
[ ] xann_float(t) = xann_float(t - dt) + (inc - outf) * dt
INIT xann_float = 0
INFLOWS:
  * inc = lnetpro
OUTFLOWS:
  * outf = if jday = 51 then xann_float else 0
[ ] year(t) = year(t - dt) + (Year_counter) * dt
INIT year = 0
INFLOWS:
  * Year_counter = if (count = 51) then 1/dt else 0
○ annual_max_grow = Maxlgrow*10

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○ con = Relative_EI/100
○ count = COUNTER(1,53)
○ hydrolunc = if con ≥ 3 then (1 - (0.05511*((con-w)^2))) else .78
○ jday = counter(1,52)
○ illratemod = if jday = 43 then (Rlirate) else 0
○ Maxlgrow = (11.0 - (37.8*(cwood/1000)) - (.267*(cwood/1000)^2))/10
○ maxnet = if Relative_EI > 0 then 0 else 2.66
○ Rlirate = .3
○ spacefunc = if Float = 0 then 1 else EXP(spacek*Float)
○ spacek = if year < 22 then -.0255 else -.0255
○ ifunc1 = if temp < 13 then 0 else ((.05 * temp) - .5)
○ Total_weeks = COUNTER(1,10000)
○ w = 75
○ temp = GRAPH(jday)
(0.00, 12.3), (4.73, 12.6), (9.45, 16.6), (14.2, 20.5), (18.9, 23.9), (23.6, 26.6), (28.4, 28.7), (33.1, 27.3),
(47.3, 16.7), (52.0, 12.8)

Not in a sector
☐ ESLR_total(t) = ESLR_total(t - dt) + (a) * dt
INIT ESLR_total = 0
INFLOWS:
  a = ESLR_C
☐ lab_below(t) = lab_below(t - dt) + (lb_in + Litter_in - trani - decomp1) * dt
INIT lab_below = .008836
INFLOWS:
  lb_in = rlab%*rit1
  Litter_in = (Litter*Leaf_Lab_Frac)+(Floatlitter*Float_lab_frac)
OUTFLOWS:
  trani = if (count=30) then (lab_below)/dt else 0
  decomp1 = klabsurt*lab_below
☐ lab_below_10(t) = lab_below_10(t - dt) + (lb_in_10 + trani_9 - decomp1_10 - trani_10) * dt
INIT lab_below_10 = .000016257
INFLOWS:
  lb_in_10 = rlab%*rootin_10
  trani_9 = if (count=30) then (lab_below_9*.5)/dt else 0
OUTFLOWS:
  decomp1_10 = klab*lab_below_10
  trani_10 = if (count=30) then (lab_below_10*.5)/dt else 0
☐ lab_below_11(t) = lab_below_11(t - dt) + (lb_in_11 + trani_10 - decomp1_11 - trani_11) * dt
INIT lab_below_11 = .00003744509
INFLOWS:
  lb_in_11 = rlab%*rootin_11
  trani_10 = if (count=30) then (lab_below_10*.5)/dt else 0
OUTFLOWS:
  decomp1_11 = klab*lab_below_11
  trani_11 = if (count=30) then (lab_below_11*.2)/dt else 0
☐ lab_below_12(t) = lab_below_12(t - dt) + (lb_in_12 + trani_11 - decomp1_12 - trani_12) * dt
INIT lab_below_12 = .000024164
INFLOWS:
  lb_in_12 = rlab%*rootin_12
  trani_11 = if (count=30) then (lab_below_11*.2)/dt else 0
OUTFLOWS:
  decomp1_12 = klab*lab_below_12
  trani_12 = if (count=30) then (lab_below_12*.2)/dt else 0
☐ lab_below_13(t) = lab_below_13(t - dt) + (lb_in_13 + trani_12 - decomp1_13 - trani_13) * dt
INIT lab_below_13 = .00001505
INFLOWS:
  lb_in_13 = rlab%*rootin_13
  trani_12 = if (count=30) then (lab_below_12*.2)/dt else 0
OUTFLOWS:
  decomp1_13 = klab*lab_below_13
  trani_13 = if (count=30) then (lab_below_13*.2)/dt else 0
☐ lab_below_14(t) = lab_below_14(t - dt) + (lb_in_14 + trani_13 - decomp1_14 - trani_14) * dt
INIT lab_below_14 = .00000966
INFLOWS:
  lb_in_14 = rlab%*rootin_14
  trani_13 = if (count=30) then (lab_below_13*.2)/dt else 0

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OUTFLOWS:
  * decomp1_14 = klab*lab_below_14
  * trani_14 = if (count=30) then (lab_below_14*.2)/dt else 0
[ ] lab_below_15(t) = lab_below_15(t - dt) + (lb_in_15 + trani_14 - decomp1_15 - trani_15) * dt
INIT lab_below_15 = .00001032
INFLOWS:
  * lb_in_15 = rlab%*rootin_15
  * trani_14 = if (count=30) then (lab_below_14*.2)/dt else 0
OUTFLOWS:
  * decomp1_15 = klab*lab_below_15
  * trani_15 = if (count=30) then (lab_below_15*.1)/dt else 0
[ ] lab_below_16(t) = lab_below_16(t - dt) + (lb_in_16 + trani_15 - decomp1_16 - trani_16) * dt
INIT lab_below_16 = .000004800
INFLOWS:
  * lb_in_16 = rlab%*rootin_16
  * trani_15 = if (count=30) then (lab_below_15*.1)/dt else 0
OUTFLOWS:
  * decomp1_16 = klab*lab_below_16
  * trani_16 = if (count=30) then (lab_below_16*.1)/dt else 0
[ ] lab_below_17(t) = lab_below_17(t - dt) + (lb_in_17 + trani_16 - decomp1_17 - trani_17) * dt
INIT lab_below_17 = .0000021788
INFLOWS:
  * lb_in_17 = rlab%*rootin_17
  * trani_16 = if (count=30) then (lab_below_16*.1)/dt else 0
OUTFLOWS:
  * decomp1_17 = klab*lab_below_17
  * trani_17 = if (count=30) then (lab_below_17*.1)/dt else 0
[ ] lab_below_18(t) = lab_below_18(t - dt) + (lb_in_18 + trani_17 - decomp1_18) * dt
INIT lab_below_18 = .0000020366
INFLOWS:
  * lb_in_18 = rlab%*rootin_18
  * trani_17 = if (count=30) then (lab_below_17*.1)/dt else 0
OUTFLOWS:
  * decomp1_18 = klab*lab_below_18
[ ] lab_below_2(t) = lab_below_2(t - dt) + (lb_in_2 + trani - trani_2 - decomp1_2) * dt
INIT lab_below_2 = .00431
INFLOWS:
  * lb_in_2 = rlab%*rli2
  * trani = if (count=30) then (lab_below)/dt else 0
OUTFLOWS:
  * trani_2 = if (count=30) then (lab_below_2)/dt else 0
  * decomp1_2 = klab*lab_below_2
[ ] lab_below_3(t) = lab_below_3(t - dt) + (lb_in_3 + trani_2 - trani_3 - decomp1_3) * dt
INIT lab_below_3 = .001163
INFLOWS:
  * lb_in_3 = rlab%*rli3
  * trani_2 = if (count=30) then (lab_below_2)/dt else 0
OUTFLOWS:
  * trani_3 = if (count=30) then (lab_below_3)/dt else 0
  * decomp1_3 = klab*lab_below_3
[ ] lab_below_4(t) = lab_below_4(t - dt) + (lb_in_4 + trani_3 - decomp1_4 - trani_4) * dt
INIT lab_below_4 = .00038317
INFLOWS:
  * lb_in_4 = rlab%*rli4
  * trani_3 = if (count=30) then (lab_below_3)/dt else 0
OUTFLOWS:
  * decomp1_4 = klab*lab_below_4
  * trani_4 = if (count=30) then (lab_below_4)/dt else 0
[ ] lab_below_5(t) = lab_below_5(t - dt) + (lb_in_5 + trani_4 - decomp1_5 - trani_5) * dt
INIT lab_below_5 = .00017435
INFLOWS:
  * lb_in_5 = rlab%*rli5
  * trani_4 = if (count=30) then (lab_below_4)/dt else 0
OUTFLOWS:
  * decomp1_5 = klab*lab_below_5
  * trani_5 = if (count=30) then (lab_below_5)/dt else 0

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☐ lab_below_6(t) = lab_below_6(t - dt) + (lb_in_6 + trani_5 - decomp1_6 - trani_6) * dt
INIT lab_below_6 = .00018464
INFLOWS:
  ⚙ lb_in_6 = rlab%*rit6
  ⚙ trani_5 = if (count=30) then (lab_below_5)/dt else 0
OUTFLOWS:
  ⚙ decomp1_6 = klab*lab_below_6
  ⚙ trani_6 = if (count=30) then (lab_below_6*.5)/dt else 0
☐ lab_below_7(t) = lab_below_7(t - dt) + (lb_in_7 + trani_6 - decomp1_7 - trani_7) * dt
INIT lab_below_7 = .00011566
INFLOWS:
  ⚙ lb_in_7 = rlab%*rit7
  ⚙ trani_6 = if (count=30) then (lab_below_6*.5)/dt else 0
OUTFLOWS:
  ⚙ decomp1_7 = klab*lab_below_7
  ⚙ trani_7 = if (count=30) then (lab_below_7*.5)/dt else 0
☐ lab_below_8(t) = lab_below_8(t - dt) + (lb_in_8 + trani_7 - decomp1_8 - trani_8) * dt
INIT lab_below_8 = .000080021
INFLOWS:
  ⚙ lb_in_8 = rlab%*rit8
  ⚙ trani_7 = if (count=30) then (lab_below_7*.5)/dt else 0
OUTFLOWS:
  ⚙ decomp1_8 = klab*lab_below_8
  ⚙ trani_8 = if (count=30) then (lab_below_8*.5)/dt else 0
☐ lab_below_9(t) = lab_below_9(t - dt) + (lb_in_9 + trani_8 - decomp1_9 - trani_9) * dt
INIT lab_below_9 = .00005876
INFLOWS:
  ⚙ lb_in_9 = rlab%*rit9
  ⚙ trani_8 = if (count=30) then (lab_below_8*.5)/dt else 0
OUTFLOWS:
  ⚙ decomp1_9 = klab*lab_below_9
  ⚙ trani_9 = if (count=30) then (lab_below_9*.5)/dt else 0
☐ Mineral(t) = Mineral(t - dt) + (minin - tranm) * dt
INIT Mineral = .09746
INFLOWS:
  ⚙ minin = MAX(minin1,.000825)
OUTFLOWS:
  ⚙ tranm = if (count = 30) then (Mineral)/dt else 0
☐ Mineral_10(t) = Mineral_10(t - dt) + (tranm_9 - tranm_10) * dt
INIT Mineral_10 = .4607
INFLOWS:
  ⚙ tranm_9 = if (count=30) then (Mineral_9*.5)/dt else 0
OUTFLOWS:
  ⚙ tranm_10 = if (count=30) then (Mineral_10*.5)/dt else 0
☐ Mineral_11(t) = Mineral_11(t - dt) + (tranm_10 - tranm_11) * dt
INIT Mineral_11 = 1.1517
INFLOWS:
  ⚙ tranm_10 = if (count=30) then (Mineral_10*.5)/dt else 0
OUTFLOWS:
  ⚙ tranm_11 = if (count=30) then (Mineral_11*.2)/dt else 0
☐ Mineral_12(t) = Mineral_12(t - dt) + (tranm_11 - tranm_12) * dt
INIT Mineral_12 = 1.1517
INFLOWS:
  ⚙ tranm_11 = if (count=30) then (Mineral_11*.2)/dt else 0
OUTFLOWS:
  ⚙ tranm_12 = if (count=30) then (Mineral_12*.2)/dt else 0
☐ Mineral_13(t) = Mineral_13(t - dt) + (tranm_12 - tranm_13) * dt
INIT Mineral_13 = 1.1516
INFLOWS:
  ⚙ tranm_12 = if (count=30) then (Mineral_12*.2)/dt else 0
OUTFLOWS:
  ⚙ tranm_13 = if (count=30) then (Mineral_13*.2)/dt else 0
☐ Mineral_14(t) = Mineral_14(t - dt) + (tranm_13 - tranm_14) * dt
INIT Mineral_14 = 1.1508
INFLOWS:
  ⚙ tranm_13 = if (count=30) then (Mineral_13*.2)/dt else 0
OUTFLOWS:

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    ⚙️ tranm_14 = if (count=30) then (Mineral_14*.2)/dt else 0
    ☐ Mineral_15(t) = Mineral_15(t - dt) + (tranm_14 - tranm_15) * dt
    INIT Mineral_15 = 2.258
    INFLOWS:
    ⚙️ tranm_14 = if (count=30) then (Mineral_14*.2)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_15 = if (count=30) then (Mineral_15*.1)/dt else 0
    ☐ Mineral_16(t) = Mineral_16(t - dt) + (tranm_15 - tranm_16) * dt
    INIT Mineral_16 = 2.122
    INFLOWS:
    ⚙️ tranm_15 = if (count=30) then (Mineral_15*.1)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_16 = if (count=30) then (Mineral_16*.1)/dt else 0
    ☐ Mineral_17(t) = Mineral_17(t - dt) + (tranm_16 - tranm_17) * dt
    INIT Mineral_17 = 1.867
    INFLOWS:
    ⚙️ tranm_16 = if (count=30) then (Mineral_16*.1)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_17 = if (count=30) then (Mineral_17*.1)/dt else 0
    ☐ Mineral_18(t) = Mineral_18(t - dt) + (tranm_17) * dt
    INIT Mineral_18 = 4.414
    INFLOWS:
    ⚙️ tranm_17 = if (count=30) then (Mineral_17*.1)/dt else 0
    ☐ Mineral_2(t) = Mineral_2(t - dt) + (tranm - tranm_2) * dt
    INIT Mineral_2 = .2303
    INFLOWS:
    ⚙️ tranm = if (count = 30) then (Mineral)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_2 = if (count=30) then (Mineral_2)/dt else 0
    ☐ Mineral_3(t) = Mineral_3(t - dt) + (tranm_2 - tranm_3) * dt
    INIT Mineral_3 = .2303
    INFLOWS:
    ⚙️ tranm_2 = if (count=30) then (Mineral_2)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_3 = if (count=30) then (Mineral_3)/dt else 0
    ☐ Mineral_4(t) = Mineral_4(t - dt) + (tranm_3 - tranm_4) * dt
    INIT Mineral_4 = .2303
    INFLOWS:
    ⚙️ tranm_3 = if (count=30) then (Mineral_3)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_4 = if (count=30) then (Mineral_4)/dt else 0
    ☐ Mineral_5(t) = Mineral_5(t - dt) + (tranm_4 - tranm_5) * dt
    INIT Mineral_5 = .2303
    INFLOWS:
    ⚙️ tranm_4 = if (count=30) then (Mineral_4)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_5 = if (count=30) then (Mineral_5)/dt else 0
    ☐ Mineral_6(t) = Mineral_6(t - dt) + (tranm_5 - tranm_6) * dt
    INIT Mineral_6 = .4607
    INFLOWS:
    ⚙️ tranm_5 = if (count=30) then (Mineral_5)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_6 = if (count=30) then (Mineral_6*.5)/dt else 0
    ☐ Mineral_7(t) = Mineral_7(t - dt) + (tranm_6 - tranm_7) * dt
    INIT Mineral_7 = .4607
    INFLOWS:
    ⚙️ tranm_6 = if (count=30) then (Mineral_6*.5)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_7 = if (count=30) then (Mineral_7*.5)/dt else 0
    ☐ Mineral_8(t) = Mineral_8(t - dt) + (tranm_7 - tranm_8) * dt
    INIT Mineral_8 = .4607
    INFLOWS:
    ⚙️ tranm_7 = if (count=30) then (Mineral_7*.5)/dt else 0
    OUTFLOWS:
    ⚙️ tranm_8 = if (count=30) then (Mineral_8*.5)/dt else 0
    ☐ Mineral_9(t) = Mineral_9(t - dt) + (tranm_8 - tranm_9) * dt
    INIT Mineral_9 = .4607
    INFLOWS:

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    ⚙️ tranm_8 = if (count=30) then (Mineral_8*.5)/dt else 0
OUTFLOWS:
    ⚙️ tranm_9 = if (count=30) then (Mineral_9*.5)/dt else 0
□ ref_below(t) = ref_below(t - dt) + (rb_in + RLitter_in - tranr - decomp_r) * dt
INIT ref_below = .013345
INFLOWS:
    ⚙️ rb_in = (1-riab%)*rlit1
    ⚙️ RLitter_in = ((1-Leaf_Lab_Frac)*Litter)+((1-Float_Lab_Frac)*Floatlitter)
OUTFLOWS:
    ⚙️ tranr = if (count = 30) then (ref_below)/dt else 0
    ⚙️ decomp_r = krestr*ref_below
□ ref_below_10(t) = ref_below_10(t - dt) + (rb_in_10 + tranr_9 - decomp_r_10 - tranr_10) * dt
INIT ref_below_10 = .027417
INFLOWS:
    ⚙️ rb_in_10 = (1-riab%)*rootin_10
    ⚙️ tranr_9 = if (count=30) then (ref_below_9*.5)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_10 = kretr*ref_below_10
    ⚙️ tranr_10 = if (count = 30) then (ref_below_10*.5)/dt else 0
□ ref_below_11(t) = ref_below_11(t - dt) + (rb_in_11 + tranr_10 - decomp_r_11 - tranr_11) * dt
INIT ref_below_11 = .071022
INFLOWS:
    ⚙️ rb_in_11 = (1-riab%)*rootin_11
    ⚙️ tranr_10 = if (count = 30) then (ref_below_10*.5)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_11 = if (theight_11 > 10) then (kdeep*ref_below_11) else (kretr*ref_below_11)
    ⚙️ tranr_11 = if (count=30) then (ref_below_11*.2)/dt else 0
□ ref_below_12(t) = ref_below_12(t - dt) + (rb_in_12 + tranr_11 - decomp_r_12 - tranr_12) * dt
INIT ref_below_12 = .075401
INFLOWS:
    ⚙️ rb_in_12 = (1-riab%)*rootin_12
    ⚙️ tranr_11 = if (count=30) then (ref_below_11*.2)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_12 = if (theight_12>10) then (kdeep*ref_below_12) else (kretr*ref_below_12)
    ⚙️ tranr_12 = if (count=30) then (ref_below_12*.2)/dt else 0
□ ref_below_13(t) = ref_below_13(t - dt) + (rb_in_13 + tranr_12 - decomp_r_13 - tranr_13) * dt
INIT ref_below_13 = .079458
INFLOWS:
    ⚙️ rb_in_13 = (1-riab%)*rootin_13
    ⚙️ tranr_12 = if (count=30) then (ref_below_12*.2)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_13 = if (theight_13>10) then (kdeep*ref_below_13) else (kretr*ref_below_13)
    ⚙️ tranr_13 = if (count=30) then (ref_below_13*.2)/dt else 0
□ ref_below_14(t) = ref_below_14(t - dt) + (rb_in_14 + tranr_13 - decomp_r_14 - tranr_14) * dt
INIT ref_below_14 = .08293
INFLOWS:
    ⚙️ rb_in_14 = (1-riab%)*rootin_14
    ⚙️ tranr_13 = if (count=30) then (ref_below_13*.2)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_14 = if (theight_14>10) then (kdeep*ref_below_14) else (kretr*ref_below_14)
    ⚙️ tranr_14 = if (count=30) then (ref_below_14*.2)/dt else 0
□ ref_below_15(t) = ref_below_15(t - dt) + (rb_in_15 + tranr_14 - decomp_r_15 - tranr_15) * dt
INIT ref_below_15 = .172067
INFLOWS:
    ⚙️ rb_in_15 = (1-riab%)*rootin_15
    ⚙️ tranr_14 = if (count=30) then (ref_below_14*.2)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_15 = if (theight_15>10) then (kdeep*ref_below_15) else (kretr*ref_below_15)
    ⚙️ tranr_15 = if (count=30) then (ref_below_15*.1)/dt else 0
□ ref_below_16(t) = ref_below_16(t - dt) + (rb_in_16 + tranr_15 - decomp_r_16 - tranr_16) * dt
INIT ref_below_16 = .169164
INFLOWS:
    ⚙️ rb_in_16 = (1-riab%)*rootin_16
    ⚙️ tranr_15 = if (count=30) then (ref_below_15*.1)/dt else 0
OUTFLOWS:
    ⚙️ decomp_r_16 = if (theight_16>10) then (kdeep*ref_below_16) else (kretr*ref_below_16)

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    * trnr_16 = if (count=30) then (ref_below_16*.1)/dt else 0
    [ ] ref_below_17(t) = ref_below_17(t - dt) + (rb_in_17 + trnr_16 - decomp_r_17 - trnr_17) * dt
    INIT ref_below_17 = .157053
    INFLOWS:
    * rb_in_17 = (1-rlab%)*rootin_17
    * trnr_16 = if (count=30) then (ref_below_16*.1)/dt else 0
    OUTFLOWS:
    * decomp_r_17 = if theight_17>10 then (kdeep*ref_below_17) else (kreir*ref_below_17)
    * trnr_17 = if (count=30) then (ref_below_17*.1)/dt else 0
    [ ] ref_below_18(t) = ref_below_18(t - dt) + (rb_in_18 + trnr_17 - decomp_r_18) * dt
    INIT ref_below_18 = .481719
    INFLOWS:
    * rb_in_18 = (1-rlab%)*rootin_18
    * trnr_17 = if (count=30) then (ref_below_17*.1)/dt else 0
    OUTFLOWS:
    * decomp_r_18 = if theight_18>10 then (kdeep*ref_below_18) else (kreir*ref_below_18)
    [ ] ref_below_2(t) = ref_below_2(t - dt) + (rb_in_2 + trnr - trnr_2 - decomp_r_2) * dt
    INIT ref_below_2 = .021769
    INFLOWS:
    * rb_in_2 = (1-rlab%)*rlit2
    * trnr = if (count = 30) then (ref_below)/dt else 0
    OUTFLOWS:
    * trnr_2 = if (count=30) then (ref_below_2)/dt else 0
    * decomp_r_2 = kreir*ref_below_2
    [ ] ref_below_3(t) = ref_below_3(t - dt) + (rb_in_3 + trnr_2 - trnr_3 - decomp_r_3) * dt
    INIT ref_below_3 = .02047
    INFLOWS:
    * rb_in_3 = (1-rlab%)*rlit3
    * trnr_2 = if (count=30) then (ref_below_2)/dt else 0
    OUTFLOWS:
    * trnr_3 = if (count=30) then (ref_below_3)/dt else 0
    * decomp_r_3 = kreir*ref_below_3
    [ ] ref_below_4(t) = ref_below_4(t - dt) + (rb_in_4 + trnr_3 - decomp_r_4 - trnr_4) * dt
    INIT ref_below_4 = .02111
    INFLOWS:
    * rb_in_4 = (1-rlab%)*rlit4
    * trnr_3 = if (count=30) then (ref_below_3)/dt else 0
    OUTFLOWS:
    * decomp_r_4 = kreir*ref_below_4
    * trnr_4 = if (count=30) then (ref_below_4)/dt else 0
    [ ] ref_below_5(t) = ref_below_5(t - dt) + (rb_in_5 + trnr_4 - decomp_r_5 - trnr_5) * dt
    INIT ref_below_5 = .019489
    INFLOWS:
    * rb_in_5 = (1-rlab%)*rlit5
    * trnr_4 = if (count=30) then (ref_below_4)/dt else 0
    OUTFLOWS:
    * decomp_r_5 = kreir*ref_below_5
    * trnr_5 = if (count=30) then (ref_below_5)/dt else 0
    [ ] ref_below_6(t) = ref_below_6(t - dt) + (rb_in_6 + trnr_5 - decomp_r_6 - trnr_6) * dt
    INIT ref_below_6 = .037759
    INFLOWS:
    * rb_in_6 = (1-rlab%)*rlit6
    * trnr_5 = if (count=30) then (ref_below_5)/dt else 0
    OUTFLOWS:
    * decomp_r_6 = kreir*ref_below_6
    * trnr_6 = if (count=30) then (ref_below_6*.5)/dt else 0
    [ ] ref_below_7(t) = ref_below_7(t - dt) + (rb_in_7 + trnr_6 - decomp_r_7 - trnr_7) * dt
    INIT ref_below_7 = .034549
    INFLOWS:
    * rb_in_7 = (1-rlab%)*rlit7
    * trnr_6 = if (count=30) then (ref_below_6*.5)/dt else 0
    OUTFLOWS:
    * decomp_r_7 = kreir*ref_below_7
    * trnr_7 = if (count=30) then (ref_below_7*.5)/dt else 0
    [ ] ref_below_8(t) = ref_below_8(t - dt) + (rb_in_8 + trnr_7 - decomp_r_8 - trnr_8) * dt
    INIT ref_below_8 = .032108

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INFLOWS:
  * rb_in_8 = (1-riab%)*rlit8
  * tranr_7 = if (count=30) then (ref_below_7*.5)/dt else 0
OUTFLOWS:
  * decomp_r_8 = kreir*ref_below_8
  * tranr_8 = if (count=30) then (ref_below_8*.5)/dt else 0
[ ] ref_below_9(t) = ref_below_9(t - dt) + (rb_in_9 + tranr_8 - decomp_r_9 - tranr_9) * dt
INIT ref_below_9 = .029719
INFLOWS:
  * rb_in_9 = (1-riab%)*rlit9
  * tranr_8 = if (count=30) then (ref_below_8*.5)/dt else 0
OUTFLOWS:
  * decomp_r_9 = kreir*ref_below_9
  * tranr_9 = if (count=30) then (ref_below_9*.5)/dt else 0
[ ] Root_1(t) = Root_1(t - dt) + (rpro1 - rlit1) * dt
INIT Root_1 = .00672323
INFLOWS:
  * rpro1 = rootin
OUTFLOWS:
  * rlit1 = Root_1*litratemod
[ ] Root_10(t) = Root_10(t - dt) + (rpro10 - rlit10) * dt
INIT Root_10 = .0004370757
INFLOWS:
  * rpro10 = rootin_10
OUTFLOWS:
  * rlit10 = Root_10*litratemod
[ ] Root_11(t) = Root_11(t - dt) + (rpro11 - rlit11) * dt
INIT Root_11 = .0007670872
INFLOWS:
  * rpro11 = rootin_11
OUTFLOWS:
  * rlit11 = Root_11*litratemod
[ ] Root_12(t) = Root_12(t - dt) + (rpro12 - rlit12) * dt
INIT Root_12 = .0004583935
INFLOWS:
  * rpro12 = rootin_12
OUTFLOWS:
  * rlit12 = Root_12*litratemod
[ ] Root_13(t) = Root_13(t - dt) + (rpro13 - rlit13) * dt
INIT Root_13 = .00028858
INFLOWS:
  * rpro13 = rootin_13
OUTFLOWS:
  * rlit13 = Root_13*litratemod
[ ] Root_14(t) = Root_14(t - dt) + (rpro14 - rlit14) * dt
INIT Root_14 = .00018725
INFLOWS:
  * rpro14 = rootin_14
OUTFLOWS:
  * rlit14 = Root_14*litratemod
[ ] Root_15(t) = Root_15(t - dt) + (rpro15 - rlit15) * dt
INIT Root_15 = .00020564
INFLOWS:
  * rpro15 = rootin_15
OUTFLOWS:
  * rlit15 = Root_15*litratemod
[ ] Root_16(t) = Root_16(t - dt) + (rpro16 - rlit16) * dt
INIT Root_16 = .000090504
INFLOWS:
  * rpro16 = rootin_16
OUTFLOWS:
  * rlit16 = Root_16*litratemod
[ ] Root_17(t) = Root_17(t - dt) + (rpro17 - rlit17) * dt
INIT Root_17 = .0000411795
INFLOWS:
  * rpro17 = rootin_17
OUTFLOWS:

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    * rlit17 = Root_17*litratemod
    Root_18(t) = Root_18(t - dt) + (rpro18 - rlit18) * dt
    INIT Root_18 = .00039055
    INFLOWS:
    * rpro18 = rootin_18
    OUTFLOWS:
    * rlit18 = Root_18*litratemod
    Root_2(t) = Root_2(t - dt) + (rpro2 - rlit2) * dt
    INIT Root_2 = .0039748
    INFLOWS:
    * rpro2 = rootin_2
    OUTFLOWS:
    * rlit2 = Root_2*litratemod
    Root_3(t) = Root_3(t - dt) + (rpro3 - rlit3) * dt
    INIT Root_3 = .002291311
    INFLOWS:
    * rpro3 = rootin_3
    OUTFLOWS:
    * rlit3 = Root_3*litratemod
    Root_4(t) = Root_4(t - dt) + (rpro4 - rlit4) * dt
    INIT Root_4 = .001549589
    INFLOWS:
    * rpro4 = rootin_4
    OUTFLOWS:
    * rlit4 = Root_4*litratemod
    Root_5(t) = Root_5(t - dt) + (rpro5 - rlit5) * dt
    INIT Root_5 = .001141214
    INFLOWS:
    * rpro5 = rootin_5
    OUTFLOWS:
    * rlit5 = Root_5*litratemod
    Root_6(t) = Root_6(t - dt) + (rpro6 - rlit6) * dt
    INIT Root_6 = .001636725
    INFLOWS:
    * rpro6 = rootin_6
    OUTFLOWS:
    * rlit6 = Root_6*litratemod
    Root_7(t) = Root_7(t - dt) + (rpro7 - rlit7) * dt
    INIT Root_7 = .001075103
    INFLOWS:
    * rpro7 = rootin_7
    OUTFLOWS:
    * rlit7 = Root_7*litratemod
    Root_8(t) = Root_8(t - dt) + (rpro8 - rlit8) * dt
    INIT Root_8 = .0007630136
    INFLOWS:
    * rpro8 = rootin_8
    OUTFLOWS:
    * rlit8 = Root_8*litratemod
    Root_9(t) = Root_9(t - dt) + (rpro9 - rlit9) * dt
    INIT Root_9 = .00056827
    INFLOWS:
    * rpro9 = rootin_9
    OUTFLOWS:
    * rlit9 = Root_9*litratemod
    subcount(t) = subcount(t - dt) + (transf) * dt
    INIT subcount = 0
    INFLOWS:
    * transf = Surate
    subcount(t) = subcount(t - dt) + (ASLR) * dt
    INIT subcount = 0
    INFLOWS:
    * ASLR = ESLR_C+Surate
    bdensity = if (height=0) then 0 else (minmass+orgmass)/height
    bdensity_10 = if (height_10=0) then 0 else (minmass_10+orgmass_10)/height_10
    bdensity_11 = if (height_11=0) then 0 else (minmass_11+orgmass_11)/height_11
    bdensity_12 = if (height_12=0) then 0 else (minmass_12+orgmass_12)/height_12

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○ bdensity_13 = d (height_13=0) then 0 else (minmass_13-orgmass_13)/height_13
○ bdensity_14 = d (height_14=0) then 0 else (minmass_14-orgmass_14)/height_14
○ bdensity_15 = d (height_15=0) then 0 else (minmass_15-orgmass_15)/height_15
○ bdensity_16 = d (height_16=0) then 0 else (minmass_16-orgmass_16)/height_16
○ bdensity_17 = d (height_17=0) then 0 else (minmass_17-orgmass_17)/height_17
○ bdensity_18 = d (height_18=0) then 0 else (minmass_18-orgmass_18)/height_18
○ bdensity_2 = d (height_2 = 0) then 0 else (minmass_2-orgmass_2)/height_2
○ bdensity_3 = d (height_3=0) then 0 else (minmass_3-orgmass_3)/height_3
○ bdensity_4 = d (height_4=0) then 0 else (minmass_4-orgmass_4)/height_4
○ bdensity_5 = d (height_5=0) then 0 else (minmass_5-orgmass_5)/height_5
○ bdensity_6 = d (height_6=0) then 0 else (minmass_6-orgmass_6)/height_6
○ bdensity_7 = d (height_7=0) then 0 else (minmass_7-orgmass_7)/height_7
○ bdensity_8 = d (height_8=0) then 0 else (minmass_8-orgmass_8)/height_8
○ bdensity_9 = d (height_9=0) then 0 else (minmass_9-orgmass_9)/height_9
○ compact_1 = 1-(lmass/(comp_k+lmass))
○ compact_10 = 1-(lmass_10/(comp_k+lmass_10))
○ compact_11 = 1-(lmass_11/(comp_k+lmass_11))
○ compact_12 = 1-(lmass_12/(comp_k+lmass_12))
○ compact_13 = 1-(lmass_13/(comp_k+lmass_13))
○ compact_14 = 1-(lmass_14/(comp_k+lmass_14))
○ compact_15 = 1-(lmass_15/(comp_k+lmass_15))
○ compact_16 = 1-(lmass_16/(comp_k+lmass_16))
○ compact_17 = 1-(lmass_17/(comp_k+lmass_17))
○ compact_18 = 1-(lmass_18/(comp_k+lmass_18))
○ compact_2 = 1-(lmass_2/(comp_k+lmass_2))
○ compact_3 = 1-(lmass_3/(comp_k+lmass_3))
○ compact_4 = 1-(lmass_4/(comp_k+lmass_4))
○ compact_5 = 1-(lmass_5/(comp_k+lmass_5))
○ compact_6 = 1-(lmass_6/(comp_k+lmass_6))
○ compact_7 = 1-(lmass_7/(comp_k+lmass_7))
○ compact_8 = 1-(lmass_8/(comp_k+lmass_8))
○ compact_9 = 1-(lmass_9/(comp_k+lmass_9))
○ comp_k = 2.0
○ ESLR_BAU = if year < 20 then ESLR_C else if year < 25 then .0028846 else if year ≥ 25 and year < 40 then .011538
○ ESLR_BG = if year < 20 then ESLR_C else if year < 30 then .0028846 else if year ≥ 30 and year < 40 then .0076923 else .01
○ ESLR_C = .00288
○ Floatlitter = flitter/100/100
○ Float_lab_lrac = if year < 22 then .8 else .8
○ height = Min_cm+org_cm+pore_cm
○ height_10 = Min_cm_10+org_cm_10+pore_cm_10
○ height_11 = Min_cm_11+org_cm_11+pore_cm_11
○ height_12 = Min_cm_12+org_cm_12+pore_cm_12
○ height_13 = Min_cm_13+org_cm_13+pore_cm_13
○ height_14 = Min_cm_14+org_cm_14+pore_cm_14
○ height_15 = Min_cm_15+org_cm_15+pore_cm_15
○ height_16 = Min_cm_16+org_cm_16+pore_cm_16
○ height_17 = Min_cm_17+org_cm_17+pore_cm_17
○ height_18 = Min_cm_18+org_cm_18+pore_cm_18
○ height_2 = Min_cm_2+org_cm_2+pore_cm_2
○ height_3 = Min_cm_3+org_cm_3+pore_cm_3
○ height_4 = Min_cm_4+org_cm_4+pore_cm_4
○ height_5 = Min_cm_5+org_cm_5+pore_cm_5
○ height_6 = Min_cm_6+org_cm_6+pore_cm_6
○ height_7 = Min_cm_7+org_cm_7+pore_cm_7
○ height_8 = Min_cm_8+org_cm_8+pore_cm_8
○ height_9 = Min_cm_9+org_cm_9+pore_cm_9
○ init_column = 22.99
○ init_elev = 35
○ kdeep = .0001
○ klab = .028
○ klabsurf = .30
○ kretr = if Relative_EI > 0 then .002 else .002
○ krelsuf = if year < 22 then .002 else .002

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○ landel = (height_18-sub1count) * (init_elev-init_column)
○ Leaf_Lab_Frac = .2
○ Litter = Litter/100/100
○ mass = minmass+orgmass
○ mass8 = minmass_8+orgmass_8
○ mass_10 = minmass_10+orgmass_10
○ mass_11 = minmass_11+orgmass_11
○ mass_12 = minmass_12+orgmass_12
○ mass_13 = minmass_13+orgmass_13
○ mass_14 = minmass_14+orgmass_14
○ mass_15 = minmass_15+orgmass_15
○ mass_16 = minmass_16+orgmass_16
○ mass_17 = minmass_17+orgmass_17
○ mass_18 = minmass_18+orgmass_18
○ mass_2 = minmass_2+orgmass_2
○ mass_3 = minmass_3+orgmass_3
○ mass_4 = minmass_4+orgmass_4
○ mass_5 = minmass_5+orgmass_5
○ mass_6 = minmass_6+orgmass_6
○ mass_7 = minmass_7+orgmass_7
○ mass_9 = minmass_9+orgmass_9
○ Max_min_in = .00443
○ minin1 = (if Relative_EI < 0 then Max_min_in else (Max_min_in-(Relative_EI*.00018846)))
○ minmass = Min_cm^2.51
○ minmass_10 = Min_cm_10^2.51
○ minmass_11 = Min_cm_11^2.51
○ minmass_12 = Min_cm_12^2.51
○ minmass_13 = Min_cm_13^2.51
○ minmass_14 = Min_cm_14^2.51
○ minmass_15 = Min_cm_15^2.51
○ minmass_16 = Min_cm_16^2.51
○ minmass_17 = Min_cm_17^2.51
○ minmass_18 = Min_cm_18^2.51
○ minmass_2 = Min_cm_2^2.51
○ minmass_3 = Min_cm_3^2.51
○ minmass_4 = Min_cm_4^2.51
○ minmass_5 = Min_cm_5^2.51
○ minmass_6 = Min_cm_6^2.51
○ minmass_7 = Min_cm_7^2.51
○ minmass_8 = Min_cm_8^2.51
○ minmass_9 = Min_cm_9^2.51
○ minvol = Min_cm/height
○ minvol10 = Min_cm_10/height_10
○ minvol11 = Min_cm_11/height_11
○ minvol12 = Min_cm_12/height_12
○ minvol13 = Min_cm_13/height_13
○ minvol14 = Min_cm_14/height_14
○ minvol15 = Min_cm_15/height_15
○ minvol16 = Min_cm_16/height_16
○ minvol17 = Min_cm_17/height_17
○ minvol18 = Min_cm_18/height_18
○ minvol2 = Min_cm_2/height_2
○ minvol3 = Min_cm_3/height_3
○ minvol4 = Min_cm_4/height_4
○ minvol5 = Min_cm_5/height_5
○ minvol6 = Min_cm_6/height_6
○ minvol7 = Min_cm_7/height_7
○ minvol8 = Min_cm_8/height_8
○ minvol9 = Min_cm_9/height_9
○ Min_cm = Mineral/2.51
○ Min_cm_10 = Mineral_10/2.51
○ Min_cm_11 = Mineral_11/2.51
○ Min_cm_12 = Mineral_12/2.51

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☐  $\text{Min\_cm\_13} = \text{Mineral\_13}/2.61$   
☐  $\text{Min\_cm\_14} = \text{Mineral\_14}/2.61$   
☐  $\text{Min\_cm\_15} = \text{Mineral\_15}/2.61$   
☐  $\text{Min\_cm\_16} = \text{Mineral\_16}/2.61$   
☐  $\text{Min\_cm\_17} = \text{Mineral\_17}/2.61$   
☐  $\text{Min\_cm\_18} = \text{Mineral\_18}/2.61$   
☐  $\text{Min\_cm\_2} = \text{Mineral\_2}/2.61$   
☐  $\text{Min\_cm\_3} = \text{Mineral\_3}/2.61$   
☐  $\text{Min\_cm\_4} = \text{Mineral\_4}/2.61$   
☐  $\text{Min\_cm\_5} = \text{Mineral\_5}/2.61$   
☐  $\text{Min\_cm\_6} = \text{Mineral\_6}/2.61$   
☐  $\text{Min\_cm\_7} = \text{Mineral\_7}/2.61$   
☐  $\text{Min\_cm\_8} = \text{Mineral\_8}/2.61$   
☐  $\text{Min\_cm\_9} = \text{Mineral\_9}/2.61$   
☐  $\text{omass\%} = \text{orgmass}/\text{mass}$   
☐  $\text{omass\%10} = \text{orgmass\_10}/\text{mass\_10}$   
☐  $\text{omass\%11} = \text{orgmass\_11}/\text{mass\_11}$   
☐  $\text{omass\%12} = \text{orgmass\_12}/\text{mass\_12}$   
☐  $\text{omass\%13} = \text{orgmass\_13}/\text{mass\_13}$   
☐  $\text{omass\%14} = \text{orgmass\_14}/\text{mass\_14}$   
☐  $\text{omass\%15} = \text{orgmass\_15}/\text{mass\_15}$   
☐  $\text{omass\%16} = \text{orgmass\_16}/\text{mass\_16}$   
☐  $\text{omass\%17} = \text{orgmass\_17}/\text{mass\_17}$   
☐  $\text{omass\%18} = \text{orgmass\_18}/\text{mass\_18}$   
☐  $\text{omass\%2} = \text{orgmass\_2}/\text{mass\_2}$   
☐  $\text{omass\%3} = \text{orgmass\_3}/\text{mass\_3}$   
☐  $\text{omass\%4} = \text{orgmass\_4}/\text{mass\_4}$   
☐  $\text{omass\%5} = \text{orgmass\_5}/\text{mass\_5}$   
☐  $\text{omass\%6} = \text{orgmass\_6}/\text{mass\_6}$   
☐  $\text{omass\%7} = \text{orgmass\_7}/\text{mass\_7}$   
☐  $\text{omass\%8} = \text{orgmass\_8}/\text{mass\_8}$   
☐  $\text{omass\%9} = \text{orgmass\_9}/\text{mass\_9}$   
☐  $\text{orgmass} = \text{org\_cm}^{1.14}$   
☐  $\text{orgmass\_10} = \text{org\_cm\_10}^{1.14}$   
☐  $\text{orgmass\_11} = \text{org\_cm\_11}^{1.14}$   
☐  $\text{orgmass\_12} = \text{org\_cm\_12}^{1.14}$   
☐  $\text{orgmass\_13} = \text{org\_cm\_13}^{1.14}$   
☐  $\text{orgmass\_14} = \text{org\_cm\_14}^{1.14}$   
☐  $\text{orgmass\_15} = \text{org\_cm\_15}^{1.14}$   
☐  $\text{orgmass\_16} = \text{org\_cm\_16}^{1.14}$   
☐  $\text{orgmass\_17} = \text{org\_cm\_17}^{1.14}$   
☐  $\text{orgmass\_18} = \text{org\_cm\_18}^{1.14}$   
☐  $\text{orgmass\_2} = \text{org\_cm\_2}^{1.14}$   
☐  $\text{orgmass\_3} = \text{org\_cm\_3}^{1.14}$   
☐  $\text{orgmass\_4} = \text{org\_cm\_4}^{1.14}$   
☐  $\text{orgmass\_5} = \text{org\_cm\_5}^{1.14}$   
☐  $\text{orgmass\_6} = \text{org\_cm\_6}^{1.14}$   
☐  $\text{orgmass\_7} = \text{org\_cm\_7}^{1.14}$   
☐  $\text{orgmass\_8} = \text{org\_cm\_8}^{1.14}$   
☐  $\text{orgmass\_9} = \text{org\_cm\_9}^{1.14}$   
☐  $\text{org\_cm} = ((\text{lab\_below} + \text{ref\_below} + \text{Root\_1}))/1.14$   
☐  $\text{org\_cm\_10} = ((\text{lab\_below\_10} + \text{ref\_below\_10} + \text{Root\_10}))/1.14$   
☐  $\text{org\_cm\_11} = ((\text{lab\_below\_11} + \text{ref\_below\_11} + \text{Root\_11}))/1.14$   
☐  $\text{org\_cm\_12} = ((\text{lab\_below\_12} + \text{ref\_below\_12} + \text{Root\_12}))/1.14$   
☐  $\text{org\_cm\_13} = ((\text{lab\_below\_13} + \text{ref\_below\_13} + \text{Root\_13}))/1.14$   
☐  $\text{org\_cm\_14} = ((\text{lab\_below\_14} + \text{ref\_below\_14} + \text{Root\_14}))/1.14$   
☐  $\text{org\_cm\_15} = ((\text{lab\_below\_15} + \text{ref\_below\_15} + \text{Root\_15}))/1.14$   
☐  $\text{org\_cm\_16} = ((\text{lab\_below\_16} + \text{ref\_below\_16} + \text{Root\_16}))/1.14$   
☐  $\text{org\_cm\_17} = ((\text{lab\_below\_17} + \text{ref\_below\_17} + \text{Root\_17}))/1.14$   
☐  $\text{org\_cm\_18} = ((\text{lab\_below\_18} + \text{ref\_below\_18} + \text{Root\_18}))/1.14$   
☐  $\text{org\_cm\_2} = ((\text{lab\_below\_2} + \text{ref\_below\_2} + \text{Root\_2}))/1.14$   
☐  $\text{org\_cm\_3} = ((\text{lab\_below\_3} + \text{ref\_below\_3} + \text{Root\_3}))/1.14$   
☐  $\text{org\_cm\_4} = ((\text{lab\_below\_4} + \text{ref\_below\_4} + \text{Root\_4}))/1.14$



```

○ org_cm_5 = ((lab_below_5-rel_below_5-Root_5)/1.14
○ org_cm_6 = ((lab_below_6-rel_below_6-Root_6)/1.14
○ org_cm_7 = ((lab_below_7-rel_below_7-Root_7)/1.14
○ org_cm_8 = ((lab_below_8-rel_below_8-Root_8)/1.14
○ org_cm_9 = ((lab_below_9-rel_below_9-Root_9)/1.14
○ poremax = .8025
○ poremin = .5590
○ pore_cm = ((pore_space)/(1-pore_space))*(Min_cm+org_cm)
○ pore_cm_10 = ((pore_space_10)/(1-pore_space_10))*(Min_cm_10+org_cm_10)
○ pore_cm_11 = ((pore_space_11)/(1-pore_space_11))*(Min_cm_11+org_cm_11)
○ pore_cm_12 = ((pore_space_12)/(1-pore_space_12))*(Min_cm_12+org_cm_12)
○ pore_cm_13 = ((pore_space_13)/(1-pore_space_13))*(Min_cm_13+org_cm_13)
○ pore_cm_14 = ((pore_space_14)/(1-pore_space_14))*(Min_cm_14+org_cm_14)
○ pore_cm_15 = ((pore_space_15)/(1-pore_space_15))*(Min_cm_15+org_cm_15)
○ pore_cm_16 = ((pore_space_16)/(1-pore_space_16))*(Min_cm_16+org_cm_16)
○ pore_cm_17 = ((pore_space_17)/(1-pore_space_17))*(Min_cm_17+org_cm_17)
○ pore_cm_18 = ((pore_space_18)/(1-pore_space_18))*(Min_cm_18+org_cm_18)
○ pore_cm_2 = ((pore_space_2)/(1-pore_space_2))*(Min_cm_2+org_cm_2)
○ pore_cm_3 = ((pore_space_3)/(1-pore_space_3))*(Min_cm_3+org_cm_3)
○ pore_cm_4 = ((pore_space_4)/(1-pore_space_4))*(Min_cm_4+org_cm_4)
○ pore_cm_5 = ((pore_space_5)/(1-pore_space_5))*(Min_cm_5+org_cm_5)
○ pore_cm_6 = ((pore_space_6)/(1-pore_space_6))*(Min_cm_6+org_cm_6)
○ pore_cm_7 = ((pore_space_7)/(1-pore_space_7))*(Min_cm_7+org_cm_7)
○ pore_cm_8 = ((pore_space_8)/(1-pore_space_8))*(Min_cm_8+org_cm_8)
○ pore_cm_9 = ((pore_space_9)/(1-pore_space_9))*(Min_cm_9+org_cm_9)
○ pore_space = poremin+((poremax-poremin)*compact_1)
○ pore_space_10 = poremin+((poremax-poremin)*compact_10)
○ pore_space_11 = poremin+((poremax-poremin)*compact_11)
○ pore_space_12 = poremin+((poremax-poremin)*compact_12)
○ pore_space_13 = poremin+((poremax-poremin)*compact_13)
○ pore_space_14 = poremin+((poremax-poremin)*compact_14)
○ pore_space_15 = poremin+((poremax-poremin)*compact_15)
○ pore_space_16 = poremin+((poremax-poremin)*compact_16)
○ pore_space_17 = poremin+((poremax-poremin)*compact_17)
○ pore_space_18 = poremin+((poremax-poremin)*compact_18)
○ pore_space_2 = poremin+((poremax-poremin)*compact_2)
○ pore_space_3 = poremin+((poremax-poremin)*compact_3)
○ pore_space_4 = poremin+((poremax-poremin)*compact_4)
○ pore_space_5 = poremin+((poremax-poremin)*compact_5)
○ pore_space_6 = poremin+((poremax-poremin)*compact_6)
○ pore_space_7 = poremin+((poremax-poremin)*compact_7)
○ pore_space_8 = poremin+((poremax-poremin)*compact_8)
○ pore_space_9 = poremin+((poremax-poremin)*compact_9)
○ r1 = (Root_1*100*100)
○ r10 = (Root_10*100*100)
○ r11 = (Root_11*100*100)
○ r12 = (Root_12*100*100)
○ r13 = (Root_13*100*100)
○ r14 = (Root_14*100*100)
○ r15 = (Root_15*100*100)
○ r16 = (Root_16*100*100)
○ r17 = (Root_17*100*100)
○ r18 = (Root_18*100*100)
○ r2 = (Root_2*100*100)
○ r3 = (Root_3*100*100)
○ r4 = (Root_4*100*100)
○ r5 = (Root_5*100*100)
○ r6 = (Root_6*100*100)
○ r7 = (Root_7*100*100)
○ r8 = (Root_8*100*100)
○ r9 = (Root_9*100*100)
○ Relative_El = (theight_18-subcount) + (init_elev-init_column)
○ riab% = 2

```

```

rootin = surtfoot*{(EXP(-rootk*height1))-(EXP(-rootk*0))}/-rootk
rootin_10 = (surtfoot*{(EXP(-rootk*height_10))-(EXP(-rootk*height_9))}/-rootk)
rootin_11 = (surtfoot*{(EXP(-rootk*height_11))-(EXP(-rootk*height_10))}/-rootk)
rootin_12 = (surtfoot*{(EXP(-rootk*height_12))-(EXP(-rootk*height_11))}/-rootk)
rootin_13 = (surtfoot*{(EXP(-rootk*height_13))-(EXP(-rootk*height_12))}/-rootk)
rootin_14 = (surtfoot*{(EXP(-rootk*height_14))-(EXP(-rootk*height_13))}/-rootk)
rootin_15 = (surtfoot*{(EXP(-rootk*height_15))-(EXP(-rootk*height_14))}/-rootk)
rootin_16 = (surtfoot*{(EXP(-rootk*height_16))-(EXP(-rootk*height_15))}/-rootk)
rootin_17 = (surtfoot*{(EXP(-rootk*height_17))-(EXP(-rootk*height_16))}/-rootk)
rootin_18 = (surtfoot*{(EXP(-rootk*height_18))-(EXP(-rootk*height_17))}/-rootk)
rootin_2 = surtfoot*{(EXP(-rootk*height_2))-(EXP(-rootk*height1))}/-rootk
rootin_3 = surtfoot*{(EXP(-rootk*height_3))-(EXP(-rootk*height_2))}/-rootk
rootin_4 = surtfoot*{(EXP(-rootk*height_4))-(EXP(-rootk*height_3))}/-rootk
rootin_5 = surtfoot*{(EXP(-rootk*height_5))-(EXP(-rootk*height_4))}/-rootk
rootin_6 = surtfoot*{(EXP(-rootk*height_6))-(EXP(-rootk*height_5))}/-rootk
rootin_7 = surtfoot*{(EXP(-rootk*height_7))-(EXP(-rootk*height_6))}/-rootk
rootin_8 = surtfoot*{(EXP(-rootk*height_8))-(EXP(-rootk*height_7))}/-rootk
rootin_9 = surtfoot*{(EXP(-rootk*height_9))-(EXP(-rootk*height_8))}/-rootk
rootk = .3
rt_transform = rshunt/100/100
Srate = .0207
surtfoot = rt_transform/(1/rootk)
height1 = height
height_10 = height_10+height_9
height_11 = height_11+height_10
height_12 = height_12+height_11
height_13 = height_13+height_12
height_14 = height_14+height_13
height_15 = height_15+height_14
height_16 = height_16+height_15
height_17 = height_17+height_16
height_18 = height_18+height_17
height_2 = height_2+height1
height_3 = height_2+height_3
height_4 = height_4+height_3
height_5 = height_4+height_5
height_6 = height_5+height_6
height_7 = height_6+height_7
height_8 = height_7+height_8
height_9 = height_8+height_9
tmass = 0
tmass_10 = tmass_9+mass_9
tmass_11 = mass_10+tmass_10
tmass_12 = mass_11+tmass_11
tmass_13 = mass_12+tmass_12
tmass_14 = mass_13+tmass_13
tmass_15 = mass_14+tmass_14
tmass_16 = mass_15+tmass_15
tmass_17 = mass_16+tmass_16
tmass_18 = mass_17+tmass_17
tmass_2 = mass+tmass
tmass_3 = tmass_2+mass_2
tmass_4 = tmass_3+mass_3
tmass_5 = tmass_4+mass_4
tmass_6 = tmass_5+mass_5
tmass_7 = tmass_6+mass_6
tmass_8 = tmass_7+mass_7
tmass_9 = tmass_8+mass_8

```

## VITA

John M. Rybczyk was born on November 30, 1959 in Salina, Kansas. He started his undergraduate degree at the University of Hawaii in 1977, in the Department of Tropical Agriculture, and received his bachelor of science degree from Michigan State University in 1981, majoring in wildlife biology. After working as a District Executive for the great Munhacker District of the Boy Scouts of America for several years in Ann Arbor, Michigan, he returned to school in 1986 and received his master of science degree in ecosystems biology from Eastern Michigan University in 1990. Under the direction of Dr. Patrick Kangas (a former student of H.T. Odum's), he evaluated patterns of nutrient use efficiency across a wetland gradient in southeastern Michigan for his master's thesis. While attending E.M.U., he worked as a Physical Scientist for the Great Lakes Environmental Research Laboratory in Ann Arbor, Michigan where he learned the science, art and frustration of ecosystem modelling from the director of his research group, Dr. Tom Fontaine (a former student of H.T. Odum's). He began his doctoral program, under the direction of Dr. John Day (a former student of H.T. Odum's), in the Department of Oceanography at Louisiana State University in the fall of 1990, and accepted a position as a Research Associate at the Coastal Ecology Institute at L.S.U. in 1994. He currently works on numerous wetlands related projects in Louisiana and Italy where his research interests include; systems ecology and ecological modeling, the statistics of detecting environmental impacts, wetland ecology and wetland restoration, wetland sustainability and eustatic sea-level rise, wetland soil development, nutrient cycling and nutrient use efficiency, the effects of humans on ecosystem function and applied ecology.


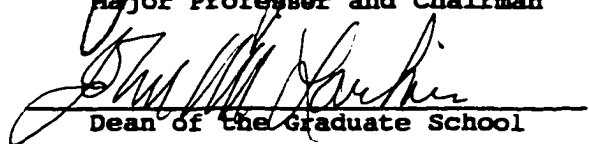
DOCTORAL EXAMINATION AND DISSERTATION REPORT

**Candidate:** John M. Rybczyk

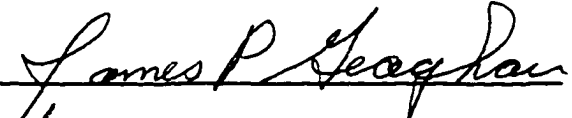


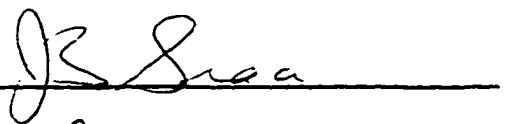

**Major Field:** Oceanography and Coastal Sciences

**Title of Dissertation:** The Use of Secondarily Treated Wastewater Effluent for Forested Wetland Restoration in a Subsiding Coastal Zone

**Approved:**

  
Major Professor and Chairman  
  
Dean of the Graduate School

**EXAMINING COMMITTEE:**

**Date of Examination:**

1/13/97