

**Ecosystem Health of the
Maurepas Swamp:
Feasibility and Projected Benefits of a
Freshwater Diversion**

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Executive Summary

Many questions remain about the quantitative benefits that diversions would have on the deteriorating swamps of southeastern Louisiana. Addressing these questions, the present report summarizes the findings of the feasibility study of re-introducing Mississippi River water into the Lake Maurepas Swamp, a highly degraded baldcypress-tupelogum (*Taxodium distichum-Nyssa aquatica*) swamp system located in the northern Lake Pontchartrain Basin, Louisiana. The purpose of this feasibility study is to evaluate the current condition of these swamp forests and to assess the potential benefits the whole ecosystem would derive from a freshwater diversion into this area. The wetlands of concern are part of the Blind and Amite River mapping units within Region 1 of the Louisiana coastal zone as defined in the Coast 2050 (1998) planning effort and restoration report, an area identified as stressed and dying, and in need of restoration. The proposed freshwater diversion is being sponsored by the Environmental Protection Agency (EPA) as the recommended strategy for restoring these wetlands under funding from the Coastal Wetland Planning, Protection, and Restoration Act (CWPPRA, 1990).

This study consists of an investigation of the potential effects of a freshwater diversion on the rate of local wetland subsidence. We then address the specific abiotic conditions found at the study sites to provide insight into which factors most affect the observed vegetative conditions of the swamp and how these factors may be affected by a diversion. Finally, we evaluate the health and rates of primary production of the woody and herbaceous components of the vegetation at these sites.

Most of the Maurepas swamp appears to be converting to marsh and open water primarily due to the lack of riverine input (Barris et al. 1994). Salt stress is killing trees that are proximal to Lake Maurepas, whereas stagnant standing water and nutrient deprivation appear to be the largest stressors at interior swamp sites. Furthermore, as increasing periods of flooding have been found to decrease the allocation of carbon to the root system (Powell and Day 1991), sites with stagnant standing water such as interior swamp sites are expected to show a greater

rate of subsidence than sites that are only seasonally flooded. On average, flood durations in the Maurepas swamps have doubled over the past half century (Thomson et al. 2002).

Severe increases in salinity, like those experienced during the drought in 1999-2000, however, may be prevented or greatly ameliorated by the increased fresh water throughput that the proposed diversion would bring. It is likely that the influences of freshening would be felt in areas as remotely located as Jones Island and the Manchac landbridge, as the proposed diversion could replace all of the water in Lake Maurepas roughly twice each year, and Pass Manchac and North Pass (adjacent to Jones Island and Manchac) are the only two direct conduits that will allow the additional fresh water to eventually reach Lake Pontchartrain.

Besides decreasing the detrimental effects of salinity throughout the Maurepas swamp, the proposed diversion would also increase the sediment load and nutrient supply to these wetlands. Hydrologic modeling showed that due to the low water-holding capacity of Hope Canal (the proposed sitting of the diversion), most of the diversion water is likely to flow as sheet-flow through the interior Maurepas swamps (Kemp et al. 2001). The resulting, evenly distributed influx of sediments is expected to strengthen the highly organic soils of the Maurepas swamp and to increase elevation in certain areas sufficiently to make the natural regeneration of several wetland forest species possible. The potentially negative impacts of lake eutrophication due to the increase in nutrient loading to the swamp are unlikely to occur, as nutrient models indicate high nutrient retention in the swamp with nutrient removal efficiencies of 94-99% from the time diversion water enters the swamp until it reaches Lake Maurepas (Day et al. 2001). Our experimental nutrient augmentation enhanced biomass production of the herbaceous vegetation by up to 300%. Furthermore, several studies conducted over the last decade have demonstrated that nutrient augmentation to baldcypress (*Taxodium distichum*) seedlings doubles growth rates in the Manchac/Maurepas area (Greene 1994, Forder 1995, Myers et al. 1995, Boshart 1997). Swamps as nutrient poor, stagnant, and impounded as the interior Maurepas swamps would be expected to at least double their rates of production if they received an infusion of freshwater and nutrients from the Mississippi River. This enhanced

productivity is essential for subsiding coastal wetlands to offset RSLR, as roots may contribute as much as 60% of the annual increment to soil organic matter (Meronigal and Day 1988). According to Keddy (2000), the exact duration and depth to cause the transition from swamp to marsh remains an interesting unresolved mystery. Without a diversion from the Mississippi River, however, the Maurepas swamp may soon resolve this issue all too clearly.

INTRODUCTION

The modern Mississippi River Delta and its associated deltas lobes encompasses an area that accounts for roughly 40% of the coastal wetlands found in the 48 contiguous states of the United States (Pezeshki and DeLaune, 1995; Coleman et al., 1998). It has a history that has its beginnings in the Late Cretaceous Period and has since undergone many major geologic changes, such as ice ages and continental drift. One of the defining characteristics of the present shape of the Mississippi River Delta system is the vast network of current and abandoned river deltas that have formed during the natural progression of the Mississippi delta cycle (Coleman et al., 1998). The time span that governs the deltaic cycle from the construction of a river delta to its eventual abandonment and deterioration is about 1,000 to 2,000 years, a process that has sped up considerably through human intervention (Coleman et al., 1998; Gosselink et al., 1999). The Mississippi River delivers annually about 240 billion kg of sediments to the Gulf coast (Goolsby, 2000). Historically, these sediments and accompanying freshwater were distributed throughout the various distributaries in the active Mississippi River Delta, of which the present familiar "bird-foot" Balize Delta (extending to the continental shelf) is the most recent. While the velocity with which these sediments are carried off the continental shelf currently prevents the production of new land, these sediments historically maintained old deltas and slowed their natural degradation and subsidence (Coleman et al., 1998; Day et al., 2000). The process of deltaic deterioration has sped up manifold (Pezeshki and DeLaune, 1995) since the construction of levees and channels. Constriction of the channel limits or prevents this natural distribution of freshwater and sediments, alters natural hydrology, and causes impoundments (Turner and Rao, 1990; Lane et al., 1999; Day et al., 2000). Due to these multiple stressors, coastal Louisiana currently experiences approximately 80% of the nation's coastal wetland losses, totaling roughly 66 km² of wetland loss per year (Dunbar et al., 1990, as referenced in Pezeshki and DeLaune, 1995; Barras et al., 1994). This situation has been recognized by federal and state agencies as a problem of immediate concern and has led to the passing of the Coastal Wetlands Planning,

Protection and Restoration Act (CWPPRA) that provides funding to support remediation and restoration projects intended to slow or reverse coastal wetland loss.

One of the most promising restoration techniques investigated appears to be the construction of controlled and/or uncontrolled river diversions (Lane, et al., 1999; Shaffer et al., 1992), a technique designed to restore the natural flow of sediments, nutrients, and freshwater into degrading wetlands to slow or halt the process of deterioration (Coleman et al., 1998; Day et al. 2000). Current restoration planning in Louisiana (Coast 2050, 1998) incorporates outfall management as an implicit part of any diversion. The purpose of outfall management is to ensure that the freshwaters and any sediments they carry are distributed through the receiving area to most effectively maintain existing wetlands and maximize new land building.

Many questions remain about the quantitative benefits that diversions would have on the deteriorating swamps of southeastern Louisiana. Addressing these questions, the present report summarizes the findings of the feasibility study of re-introducing Mississippi River water into the Lake Maurepas Swamp, a highly degraded baldcypress-tupelogum (*Taxodium distichum-Nyssa aquatica*) swamp system located in the northern Lake Pontchartrain Basin, Louisiana. The purpose of this feasibility study is to evaluate the current condition of these swamp forests and to assess the potential benefits the whole ecosystem would derive from a freshwater diversion into this area. The wetlands of concern are part of the Blind and Amite River mapping units within Region 1 of the Louisiana coastal zone as defined in the Coast 2050 (1998) planning effort and restoration report, an area identified as stressed and dying, and in need of restoration. The proposed freshwater diversion is being sponsored by the Environmental Protection Agency (EPA) as the recommended strategy for restoring these wetlands under funding from the Coastal Wetland Planning, Protection, and Restoration Act (CWPPRA, 1993).

Feasibility Study Objectives

Before a diversion can be constructed, a feasibility study is required that would address the information objectives outlined in the project proposal and restated below:

1. Provide an understanding of the nutrient assimilation capacity of the south Maurepas swamp, to estimate the size of a prospective diversion with regard to nutrient loading limits, so as to protect adjacent lake water quality and prevent algal blooms.
2. Develop a basis for differentiating influences and responses of the proposed diversion over a gradient of least to greatest apparent stress and to associate swamp conditions with causal factors (especially riverine influences).
3. Determine estimates of baseline environmental conditions in the swamp to provide input to biological/ecological models and to support predictions of conditions in the future without a restoration project, therefore determining benefits to be gained from a diversion.

To address these information objectives, the first section of this study consists of an investigation of the potential effects of a freshwater diversion on the rate of local wetland subsidence. We then address the specific abiotic conditions found at the study sites to provide insight into which factors most affect the observed vegetative conditions of the swamp and how these factors may be affected by a diversion. Finally, we evaluate the health and rates of primary production of the woody and herbaceous components of the vegetation at these sites.

Wetland Subsidence Rates

The effects of a freshwater diversion on the rate of wetland subsidence is under investigation through a comparison of sediment subsidence in areas located near the Amite River Diversion Canal and other sites located further away from any sources of sediment input. We hypothesize that subsidence rates near the Amite River Diversion Canal will be significantly lower than elsewhere in the swamp, especially lower than in the interior swamp regions that are cut off from any direct sources of sediment input. Furthermore, we hypothesize that wetland subsidence will be greatest at the most degraded study sites located near the lakeshore, since at these sites belowground productivity might be negatively impacted by the multiple stresses from increased salinity and low sediment and nutrient input. Because increasing periods of flooding

have been found to decrease the allocation of carbon to the root system (Powell and Day, 1991), sites characterized by prolonged stagnant standing water (i.e., much of the Maurepas swamps) are expected to show a greater rate of subsidence than sites only seasonally flooded.

Abiotic Site Characterization

To quantify the impact of the various hypothesized stressors that may be affecting the different sites, special emphasis was placed on investigating differences in bulk density and salinity. We hypothesized that sites located close to Lake Maurepas would exhibit the highest levels of both interstitial and surface water salinity due to salt-water intrusion. Conversely, we hypothesized that salinity levels would decrease with increasing distance from the lake or when located near waterways carrying a steady supply of fresh water, which would create a barrier to salt-water intrusion. Thus, swamp interior sites and sites located near the Amite River Diversion Canal were expected to have the lowest salinity levels measured in this study.

Furthermore, we hypothesized that study sites located near the Amite River Diversion Canal would exhibit the highest bulk densities within the system studied due to the increased mineral sediment load the diversion canal supplies to these sites. Similarly, the lowest bulk densities were expected to occur at interior swamp sites, since most sediments are hypothesized to settle out of the water column before reaching these sites. We also speculated that nutrient levels would be highest at the Amite River Diversion sites and that these sites would respond with relatively high rates of primary production.

Woody Vegetation

We tested the hypotheses that there are significant differences in tree health and primary productivity between the degraded baldcypress-tupelogum swamp forests located close to the southern lakeshore of Lake Maurepas, interior swamp forests at slightly higher elevations, areas located near the existing, relatively small Amite River Diversion Canal, as well as areas along Hope Canal. Furthermore, we hypothesized that the lower rates of primary productivity would

be correlated with higher salinity levels (Pezeshki et al., 1987), lower soil bulk densities (and thus lower concentrations of minerals and nutrients), and prolonged flooding at the degraded study sites (i.e. Lake sites), when compared to the relatively healthy sites in the study area (i.e. Throughput sites). Hydrologic and nutrient inflows have been found to be coupled in most swamp systems studied to date (Mitsch and Gosselink, 2000; Megonigal et al., 1996; Messina and Conner, 1998) and, thus, both low nutrient inflow and stagnant, standing water have been shown to decrease productivity in cypress swamps (Brown, 1981). Moreover, we hypothesized that the major portion of tree primary productivity would be attributed to tupelogum and baldcypress trees, since these are the dominant tree species in the Maurepas swamp. Of these two tree species, baldcypress was expected to be more productive, since tupelogum is more susceptible to the combined stresses of prolonged flooding and salinity (Conner et al., 1997). Also, physical stress was apparent on the vast majority of tupelo trees in the Maurepas swamp, as exemplified by broken canopies.

Herbaceous Vegetation

We hypothesized that the cover of herbaceous vegetation would be inversely related to overstory vegetation cover and that the Lake sites would, therefore, have the highest herbaceous productivity. The stressed Lake sites appear to be following the same trend as occurred on the Manchac land bridge and Jones Island, both just east of the Maurepas system. Since the mid-1950s these two areas have undergone transition from forest swamp to open marsh, and are transitioning to open water at present. We expect these trends in the Maurepas swamp to lag behind Manchac and Jones Island because it is slightly more buffered from the influence of the Gulf of Mexico.

Overall, primary productivity was hypothesized to be low in the entire Maurepas swamp due to the multiple stressors of prolonged flooding, little hydrologic throughput, sediment and nutrient deprivation, and salt-water intrusions, when compared to other, healthier swamps. We also expect to find a change in species composition that shows a shift towards salt-tolerant

species at sites likely to be subjected to salt-water intrusions. Furthermore, we hypothesized that fertilized plots within each site would have significantly higher rates of herbaceous biomass production than unfertilized plots.

MATERIALS AND METHODS

Study Site

The study area is located in the Pontchartrain Basin, a marginal deltaic basin that was enclosed by the progradation of the St. Bernard delta complex some 2,500 years ago. Regional estimates place RSLR between 3.6 and 4.5 mm yr⁻¹ in this basin (Penland and Ramsey 1990). To accurately characterize the Maurepas swamp, twenty study sites with two replicates each were selected in the southern wetlands of Lake Maurepas (Figure 1). These sites were chosen to capture a variety of different hydrological regimes within the swamp. Of these sites, sites 1 (Red Top), 2 (Cher Bayou), 3 (Blind River/Amite Flood-relief Diversion Canal confluence), 6 (Potato Run), 7 (Peter's Run), 9 (Interior Mississippi Bayou), and 13 (Black Lake) were the most swamp interior sites (hereafter referred to as "Interior"), located away from any direct water exchange with Lake Maurepas and only accessible by airboat. Sites 10 (Tent Bayou/Hope Canal), 11 (middle of Hope Canal) and 12 (top of Hope Canal) are located along Hope Canal, a manmade canal that crosses beneath I-10. These sites receive non-point source runoff from the town of Garyville and are located far enough from Lake Maurepas to make any water exchange with the lake minimal. In combination with site 5 (Alligator Island), which receives freshwater and sediments

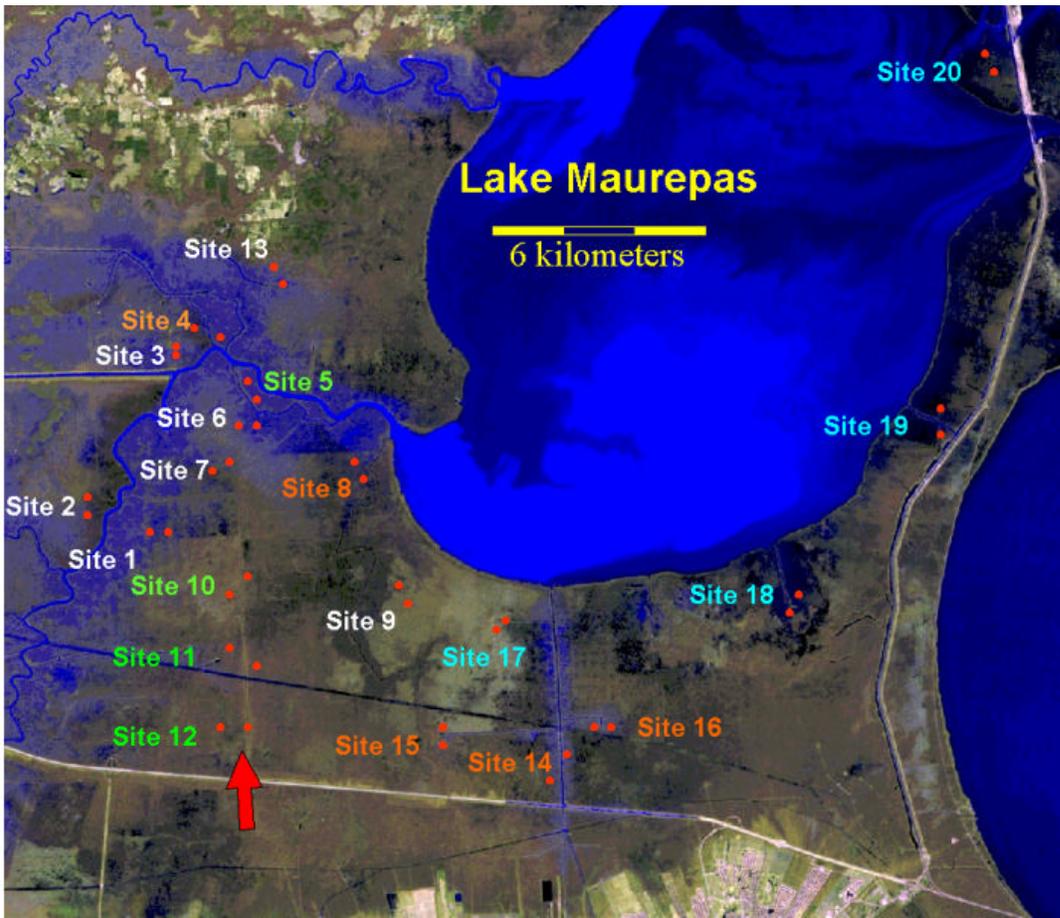


Figure 1. Site locations on Lake Maurepas. To accurately characterize the Maurepas swamp, twenty study sites with two replicates each were selected in the southern wetlands of Lake Maurepas. Different colors characterize different site types, chosen to capture a variety of different hydrological regimes within the swamp. Sites 1,2,3,6,7,9, and were the most swamp interior (Interior) sites, located away from any direct water exchange with Lake Maurepas and only accessible by airboat. Sites 10, 11, and 12 are located along Hope Canal, a manmade canal that crosses beneath I-12. These sites receive non-point source runoff from the town of Garyville and are located far enough from Lake Maurepas to make any water exchange with the lake minimal. In combination with site 5, which receives freshwater and sediments from the Amite River Flood-relief Diversion Canal, the sites along Hope Canal are the healthiest swamp areas of this study (Throughput) sites. Sites 4, 8, 14, 15, and 16 (Intermediate) sites, are located closer to the lake and in the vicinity of larger bayous or canals that make direct water exchange with the lake probable. Lastly, sites 17-20 (Lake sites) grade towards the eastern side of the Maurepas swamp towards Pass Manchac, the main waterway between Lake Maurepas and Lake Pontchartrain. These lake sites are close enough to Lake Maurepas to make water exchange with the lake likely.

from the Amite River Flood-relief Diversion Canal, the sites along Hope Canal are the healthiest swamp areas of this study and will, hereafter, be referred to as the "Throughput" sites. Sites 4 (Lil' Chene Blanc), 8 (Dutch Bayou), 14 (Reserve Relief Canal, near I-55), 15 (Reserve Relief Canal west), and 16 (Reserve Relief Canal east), hereafter referred to as "Intermediate" sites, are located closer towards the lake and in the vicinity of larger bayous or canals that make direct water exchange with the lake probable. Lastly, sites 17 (Reserve Relief Canal near Lake Maurepas), 18 (Tobe Canal), 19 (Ruddock), and 20 (Jones Island), hereafter referred to as "Lake" sites, grade towards the eastern side of the Maurepas swamp towards Pass Manchac, the main waterway between Lake Maurepas and Lake Pontchartrain. These lake sites are close enough to Lake Maurepas to make water exchange with the lake likely. The "Interior" and "Intermediate" groups represent the majority of the swamp area located on the southern shore of Lake Maurepas (Figure 1). Taken together, all study sites characterize an area roughly 180 km² in size.

A discriminant function analysis was used to evaluate *a priori* site groupings. The discriminant model was interactively modified only to include variables significant at $P < 0.05$. To avoid tolerance problems (shared variation among variables, Hair et al. 1998), variables were screened for multicollinearity by principal component analysis prior to being used in the discriminant model. No tolerance problems were found. The data used were also screened for multivariate outliers (Tabachnik and Fidell 2001), but none were found. Missing data was treated by case-wise exclusion. Model validity was tested by comparing the actual classification to a calculated Proportional Chance Criterion (PCC, Hair et al. 1998) and by a chi-square significance test (using Press' Q, Hair et al. 1998). Variable loadings in the discriminant function model were evaluated to determine the relative importance of particular predictors.

Wetland Subsidence

The net subsidence in the study area at twelve representative study sites will be evaluated using a combination of sediment elevation tables (SETs) and feldspar marker horizons (Baumann and Day, 1993; Cahoon et al., 1999; Cahoon and Turner, 1989; Day et al. 1999) to determine the total sediment subsidence and sediment accretion.

Sediment elevation tables:

Two SETs were installed at twelve representative study sites throughout the Maurepas swamp (and they are where?). Each permanent SET setup consists of a 5 m long aluminum pipe (8 cm in diameter) that was driven into the ground to “refusal.” Since subsidence and soil compaction is expected to occur primarily within the top two meters (Cahoon et al., 1999), this setup captures shallow subsurface subsidence to ± 2 mm (Cahoon et al., 1999). This pipe was then fenced in with a 4-m x 4-m wooden frame designed to allow access to the pipe via a sliding bench, while excluding external disturbance of the SET plot. The top of this insert pipe fits a specially constructed table that has a bubble level to allow the table to be adjusted plumb and level to the ground. From this table, which can be set in four compass directions, nine pin readings of the soil elevation were taken (Baumann and Day, 1993). The SET readings were averaged across the nine pin readings; the four compass directions have been shown to be true replicates (Cahoon et al., 1999), while the two SET plots at each study site will be analyzed as split plot main effects. The first reliable reading of these SET tables occurred during the months of October and November 2001. The difference in elevation between this reading and the most recent set of readings taken during October and November 2002, provides an accurate estimate of the net subsidence rates currently occurring within the Maurepas Swamp.

Feldspar markers

Feldspar markers were used as a measure of the rate of accretion that is concurrently occurring in the Maurepas Swamp. The marker horizons were laid down at the northwest and

southeast sides within each SET plot frames to minimize disturbance. At the time that the 2002 SET readings were taken, frozen soil cores were extracted from these areas to measure the total vertical accretion of sediments since the feldspar was applied (Knaus and Cahoon, 1990). In conjunction, the SET measurements and the feldspar markers should provide a measure of the absolute rate of subsidence at the study sites, as well as a measure of how much of this subsidence is currently offset by accretion.

Abiotic Characterization

To test the effects of salinity, sediment deprivation, and prolonged flooding, surface-water salinity and soil variables such as soil water salinity, pH, bulk density, redox potential (Eh), sulfide concentrations, and concentrations of nitrate, ammonia, and selected elements were monitored at the twenty study sites. A discriminant function analysis was performed to investigate the predictive capacity of the measured edaphic factors in separating the drought year (2000) from the non-drought year (2001). Again, discriminant models were interactively modified to include only significant variables ($\alpha = 0.05$). The data used were screened for multivariate outliers, and one case with too large a Mahalanobis D^2 for its own group (evaluated as chi-square (χ^2) values with degrees of freedom equal to the number of predictors at $\alpha = 0.001$, Tabachnik and Fidell 2001) was excluded from the analysis. Missing data, model validity, and variable loadings were treated and evaluated as described above.

Well salinity

Two 1-m 6-cm diameter PVC wells were inserted 0.75 m into the ground at each of the forty stations. Wells were capped at both ends. Horizontal slits were cut into the wells every 2 cm from a depth of 5 cm to a depth of 70 cm below the soil surface to enable ground water to enter. Wells were outfitted with plastic skirts at the soil surface to prevent rainwater seepage (Thomson 2000). Each well was completely evacuated and allowed to refill before salinity measurements were taken, using a YSI salinity / conductivity / temperature meter. Well-water

salinity was measured during most site visits and averaged to yield a measure of yearly mean salinity at each study plot. Well salinity was analyzed in a 2x3 factorial treatment arrangement with years and groups as the independent variables. In a subsequent analysis, well salinity was also analyzed for site differences so that spatial trends in salinity could be analyzed with orthogonal polynomial contrasts to address specific hypotheses stated above (Zar 1999).

Bulk density

Soil cores for bulk density analysis were collected during the fall sampling period in 2001 and 2002, using an aluminum soil corer with a 1.6 cm inner diameter. Samples were collected by coring to a depth of 10 cm, carefully removing the soil corer from the surrounding substrate, and extruding the cores into plastic sample bags. To minimize the influence of micro-scale heterogeneity of soil strength, five replicate cores were taken at two locations within each study plot. The five replicate cores were combined into a single sample in the field, while the two samples from different locations within the same study plot were processed independently before being averaged to yield a plot average for the final analysis. Each sample was dried to constant mass at 65°C in a ventilated oven before soil core weights were measured. Bulk density was analyzed in a one-way ANOVA using groups as the only independent variable.

Soil reduction

The degree of soil reduction was measured at the surface (1-2 cm depth) and at a depth of 15 cm. Measurements were performed using brightened, calibrated electrodes and a calomel reference electrode as described in Faulkner (1989). Redox probes were allowed to equilibrate for a minimum of 15 minutes. Millivolt readings were used to calculate soil redox potential (Eh) by adding the potential of the calomel electrode against a standard hydrogen electrode (+244 mV, Faulkner 1989). Surface and deep measurements were replicated three times each at two different locations within each study plot and then averaged.

Soil chemical properties

Each year soil water samples for chemical analysis were collected as soil cores using an aluminum soil corer with a 5.25 cm inner diameter. Samples were collected by coring to a depth of 15 cm, carefully removing the soil corer, retaining both sediment and interstitial water, and then extruding the cores into 500-ml, acid-washed centrifuge bottles. The samples were immediately gassed with nitrogen for a minimum of 3 minutes to maintain an anaerobic environment inside the centrifuge bottle. The centrifuge bottles were then placed on ice until they could be refrigerated upon their return from the field. Due to the unusual drought conditions throughout the southeastern parts of the United States during 1999-2000, soil samples from the sampling round during summer 2000 did not have sufficient water available for interstitial water extraction and analysis. Therefore a dry extraction was employed to achieve an estimate of interstitial soil characteristics. Samples were dried, ground up, and subsequently rehydrated with distilled water to gain a sample of interstitial water for chemical analysis (see Shaffer et al. 2001 for a detailed description of this procedure). Soil samples from the were generally processed within 24 hours after collection, or frozen until processing was possible. Samples were removed from the refrigerator or freezer, dried of any external condensation, and then weighed. Samples were then centrifuged at 4° C for 15 minutes at 5000 RPM. Interstitial soil water collected after centrifugation was partitioned into aliquots for sulfide analysis, auto-analyzer analysis, ICP analysis, and for salinity and pH determination. Samples for sulfide analysis were immediately filtered and placed into an antioxidant buffer. Sulfide concentrations were then determined using a Jenco pH/mV meter, a LAZAR sulfide selective electrode, and a reference electrode. Samples for auto-analyzer and ICP analyses were placed into acid-washed vials and refrigerated until suspended sediments settled. The remaining supernatant was then filtered through a 2- μ m syringe filter into approximately 10-ml samples for ICP analysis and two 5-ml samples for auto-analyzer analysis. Samples for ICP analysis were placed into acid-washed scintillation vials, preserved with 2 drops of concentrated HNO₃, and refrigerated until analysis. Auto-analyzer samples were placed into auto-analyzer vials and

frozen. These samples were later transported to Louisiana State University (LSU) for analysis. Samples for pH and salinity determination were allowed to warm to room temperature. Salinity was measured using a YSI salinity / conductivity / temperature meter and pH was determined using a calibrated Jenco pH/mV meter. The dry extraction method used for the summer 2000 samples was also applied to a set of replicate cores taken during fall 2000 to determine if a predictable relationship existed between dry extraction and wet interstitial extraction methods. These replicate samples were treated similarly to the summer samples described above, except that they were extracted into 1-gallon sampling bags and were not purged with nitrogen gas.

During 2001 and 2002, soil water samples were extracted using different methods. In the summer of 2001, soil water samples were collected by placing approximately one gallon-portion of wet soil into a fine-meshed cloth filter-bag and catching the filtrate in two 20-ml acid-washed, glass sample bottles. In the fall of 2001 and both sampling rounds of 2002, soil water samples were extracted from the soil using a soil sipper. The soil sipper consisted of a 20 cm, perforated copper pipe connected via clear vinyl tubing to a 60-ml syringe. The copper pipe was inserted completely into the soil until all perforations were below the soil surface before soil water was extracted into the syringe and subsequently expunged into two 20-ml acid-washed, glass sample bottles. Following extraction, soil water samples were then placed on ice until they could be frozen in the laboratory. One out of each two sample bottles was later thawed and allowed to reach room temperature before salinity and pH measurements were taken. The remaining set of soil water samples from each season of each year was later thawed, filtered through a 2- μ m syringe filter into 15-ml samples for auto-analyzer analysis and frozen. Again, auto-analyzer samples were transported to LSU for analysis.

Sulfide concentrations were below detection levels for all samples during 2000. ICP metal data from 2000 correlated strongly with phosphate levels and thus provided no new information. Therefore, as the results from both sulfide and ICP analyses did not provide any essential new information during 2000, these analyses were not repeated during 2001 or 2002. Soil salinity was analyzed in a 2x4 factorial design with split plot, which used three years and

four groups as the main plot independent variables and season in the sub-plot. Soil water nutrients and pH were analyzed using discriminant function analysis for year and group differences.

Herbaceous Vegetation

Within each of the forty 25 m x 25 m (625 m²) permanent stations, four 4m x 4m (16 m²) permanent herbaceous plots were established at a diagonal distance of 5 m in from each of the corners of each station. A 4 m² plot was established in the center of each 16-m² plot for cover value estimates.

Cover values

Cover values were obtained by two independent estimates for all 160 plots during mid-June to early July and again in early September of 2000 and for all 200 (additional 'E' and 'F') plots during spring and fall of 2001 and 2002. Percentage cover of vegetation by species was determined by ocular estimation in 5% increments in the field and classified according to a modification (Shaffer et al. 1992) of the Braun Blanquet cover method (Braun-Blanquet 1932).

Nutrient augmentation

During the first year of the study, half of the 16-m² plots were fertilized with 59 g/m² of Osmocote 18-6-12 timed-release fertilizer. This dosage emulates a loading rate of 11.25 g N m²/year, similar to what a 1,500 cfs diversion would carry. During years two and three the 'A' and 'C' plots, of 'A' stations were double fertilized (i.e., 118 g/m² of Osmocote) during the spring to emulate a 3,000 cfs diversion. In addition, 'E' and 'F' plots were added to 'A' stations and these were fertilized with 59 g/m² of Osmocote during the spring and again during late summer. This treatment was added to emulate a diversion open year-round.

Herbivore-exclusion experiment

During the second year of the study we observed mammal herbivores foraging on fertilized plots with increased intensity. Therefore, we installed herbivore-exclusion cages around 35 herbaceous plots representing all of the nutrient augmentation regimes including the control, meaning each caged plot was paired with the same treatment of an uncaged plot (e.g., 'A' = double fertilized and caged paired with 'C' = double fertilized but uncaged).

Annual production

During 2000, herbaceous (understory) primary productivity was estimated within one fertilized and one unfertilized herbaceous (4 m x 4 m) plot by clipping two randomly chosen (non-repeating) replicate subplots (of 0.25 m² area) twice during the growing season as outlined in Whigham et al. (1978) and Wohlgemuth (1988). Plots were clipped during late June – early July (summer) and again during late September - early October (fall). All clipped subplots were randomly selected without replacement at the beginning of the study. This ensured that a subplot would never be clipped more than once during the study period. Plant material was clipped at the soil surface, placed in a labeled bag, and transported to the lab, where it remained in cold storage until it could be sorted into live and dead tissue and then oven-dried to constant weight and weighed. As stated above, these clip plots were harvested from one fertilized and one unfertilized herbaceous plot within the each of the forty 625 m² stations. The other half of the herbaceous plots were not disturbed by clipping and were used to measure the soil edaphic variables outlined above.

We expected biomass to peak during the fall “late season” clip plots (Mitsch and Gosselink, 2000). However, salt water intrusions during mid-summer caused sufficient stress to send most herbaceous species into senescence during 2000 and 2001. During 2002, extremely high flood levels primarily associated with hurricanes Lily and Isidore, combined with salinity intrusions, persisted from August through the end of the growing season and these caused almost complete senescence to occur in all rooted species. Fortunately, we had micro-mapped

each species' location in all 160 plots during early season cover value estimates of 2000. We were thus able to assess, for each species, the percent of second-harvest cover that could be attributed to new growth. We estimated the range of new growth for each species and used the upper limit to augment the early-season biomass measurements with that proportion of late-season biomass measurements. In other words, we added the proportion of late-season biomass that could be new biomass (biomass produced after first clip plots) to the early-season biomass measurements to estimate annual production. If any bias was introduced, we believe that annual production could have been slightly overestimated. During 2002 a large bloom of the floating aquatic *Salvinia molesta* occurred in much of the Maurepas swamp. We analyze the data with and without this component.

Forest Structure and Tree Mortality

Forest structure and tree mortality were measured through the collection of tree diameter data. The measures of forest structure used in this study were basal areas per hectare and stem density per hectare. Percent tree mortality was partitioned between the three years of study and as the yearly, average rate of tree mortality per group.

The majority of all trees within each of the two 625m² plots at each of twenty study sites in the Maurepas swamp were tagged using 8 penny galvanized nails and pre-numbered 5-cm metal ID tags in February and March of 2000. All *Taxodium distichum* and *Nyssa aquatica* trees in each plot were tagged, as well as trees and large shrubs of other species, such as *Fraxinus pennsylvanica*, *Acer rubrum* var. *drummondii*, *Nyssa sylvatica* var. *biflora*, *Quercus obtusa*, *Salix nigra*, *Myrica cerifera*, *Cephalanthus occidentalis*, and *Sapium sebiferum* (Table 1). Trees were tagged at breast height, unless the fluting bases of *T. distichum* and *N. aquatica* or the complex branching structure of other trees required the tags to be somewhat higher. Using fiberglass metric diameter tapes, initial tree diameters of the approximately 1600 tagged trees were measured during February and March of 2000 at the bottom of the freely hanging metal tags. To efficiently locate trees during subsequent field sampling events, the location of

each tagged tree was mapped. During October 2000, a second diameter measure was taken of every tree previously tagged - as far as these trees could be found - and further trees were added to the study to include all trees with diameters larger than 5 cm. A third round of tree diameter measurements and additional tagging of new trees, that had previously been overlooked, was

Table 1. Common plants in the Maurepas swamp.

Scientific Name	Common Name
<u>Trees and Shrubs</u>	
<i>Acer rubrum</i> var. <i>drummondii</i>	swamp red maple
<i>Cephalanthus occidentalis</i>	buttonbush
<i>Fraxinus pennsylvanica</i>	green ash
<i>Myrica cerifera</i>	southern waxmyrtle
<i>Nyssa aquatica</i>	water tupelo
<i>Nyssa sylvatica</i> var. <i>biflora</i>	blackgum
<i>Quercus obtusa</i>	diamond oak
<i>Salix nigra</i>	black willow
<i>Sapium sebiferum</i>	Chinese tallow
<i>Taxodium distichum</i>	baldcypress
Herbaceous Plants	
<i>Alternanthera philoxeroides</i>	alligatorweed
<i>Amaranthus australis</i>	pigweed
<i>Apium lepiphyllum</i>	marsh parsly
<i>Aster</i> spp.	white and purple asters

<i>Baccharis helimifolia</i>	eastern baccharis
<i>Bacopa monnieri</i>	coastal water hyssop
<i>Echinochloa walterii</i>	Walter's millet
<i>Eleocharis</i> spp.	spikerush
<i>Galium tictorium</i>	marsh bedstraw
<i>Hydrocotyle</i> spp.	dollarweed
<i>Iris virginica</i>	blue flag
<i>Ludwigia leptocarpa</i>	false loostrife
<i>Panicum dicotomiflorum</i>	fall panicgrass
<i>Panicum hemotomon</i>	maidencane
<i>Peltandra virginica</i>	arrow arum
<i>Polygonum punctatum</i>	smartweed
<i>Pontedaria chordata</i>	pickerelweed
<i>Sable minor</i>	palmetto
<i>Sagittaria lancifolia</i>	bulltongue
<i>Vigna luteola</i>	deer pea

conducted during October 2001 and 2002. At the end of the 2002 sampling round, a total of 2012 trees had been tagged and measured at all forty study plots. All tree diameter data was entered into SYSTAT 10.0 (2001) for data management and statistical analysis.

Stem density

The number of stems per plot was counted and multiplied by 16 (as there are sixteen 625m² plots to a hectare) to yield an estimate of the number of trees per hectare. A further part of the analysis subdivided the trees into four size classes (5-10 cm, 10-20 cm, 20-30 cm, and > 40 cm) to be able to plot average stand structure per group as a qualitative indicator of forest health. Stem density was analyzed using a 3x4 factorial treatment arrangement with three species categories and four groups.

Basal area

The measured tree diameters in each plot were converted into basal wood areas in m², so that

$$\text{basal wood area per tree} = \pi (d/200)^2$$

where d is the diameter of the tree in cm. These basal wood areas per tree were then summed per plot and multiplied by sixteen to yield the total area of basal wood per hectare for each of the forty study plots. Basal area was analyzed in a one-way ANOVA using groups as the only independent variable.

Percent mortality

Percent mortality during the study period could be calculated from the number of trees that were initially tagged and the number of trees that were left at the end of the study period as:

$$\text{percent mortality} = (1 - n_s/n_o) * 100$$

where n_s is the number of survivors and n_o is the original number of trees. The data obtained in this manner was not used in any statistical analysis but rather as a qualitative measure of forest health.

Average mortality rate :

This measure of forest health was calculated by adjusting percent mortality for the time period that had elapsed, so that

$$\text{annual percent mortality} = [(1 - n_s/n_o) * 100] / y$$

where y is the number of years elapsed. Similar to percent survival, this measure was used as a qualitative indicator of forest health.

Primary Productivity of Trees

Tree primary productivity was measured through the collection of annual litter and the measurement of annual tree diameter growth (Mitsch and Ewel 1979, Brown 1981, Conner and Day 1992) at the forty study plots located throughout the forested wetlands of southern Lake Maurepas.

Litter production

Five litter traps were installed at approximately even spacing at each of two plots at twenty study sites throughout the Maurepas swamp (Figure 2), to yield a total of 200 litter traps deployed. Each of these litter traps was 0.25 m² in area and was constructed to catch biomass in a fine (1 mm) mesh approximately 1 meter above the ground to prevent loss from flooding events. The litter was collected frequently during site visits, which occurred as often as once every two weeks or as infrequent as once every two months during periods of the growing

season when few leaves were falling (i.e., spring). During or after collection, the litter from each of the five litter traps at each plot was combined to yield one total sample of litter per plot, because the five traps are considered pseudo-replicates. For this study, we use the term litter for both woody and non-woody (leaves, flowers, fruits, and seeds) tissue. Collected litter was then dried to constant mass at 65 °C. After drying, the litter was sorted into *T. distichum*, *N. aquatica* and other litter. This enabled us to monitor productivity effects at the species level for at least the two most dominant tree species in the swamp. All litter data was then entered into SYSTAT 10.0 (2001).

Wood production

The diameter data, obtained from the yearly measurements of tree diameters according to the procedures described above, was used to calculate the amount of wood production per tree for the dominant tree species. The dominant species of interest were *T. distichum* and *N. aquatica*, with all remaining species being grouped together into the category 'Other.' Wood production was calculated as the difference in wood biomass per year, which, in turn, was calculated from the measured tree diameters using wood biomass regression formulas found in the literature (Clark et al. 1985, Scott et al. 1985, Muzika et al. 1987 - see Appendix A). Wood production per tree was then summed by species per plot and then divided by 625 to yield the total wood production per square meter per year.

Statistical Analysis

All statistical analyses were performed using SYSTAT 10.0 (Wilkinson, 2000, 2001). Discriminant function analysis was used to evaluate *a priori* site groupings. To avoid tolerance problems, variables were screened for multicollinearity by principle components analysis prior to being used in the discriminant model. Principle components analysis utilized a varimax rotation and minimum eigenvalue of 1.0 (Hair et al. 1998). Furthermore, discriminant analysis utilized an automatic backward-stepping model with probability to enter and remove set equal to

0.15. All data, with the exception of the wet-extracted nutrient and ICP results, were subjected to repeated measures ANOVA. When a significant effect of time occurred, univariate ANOVAs were then employed within each time to determine trends. Wet-extracted nutrient and ICP results were analyzed using univariate ANOVAs. Linear contrasts were used to address specific *a priori* hypotheses.

Tree and herbaceous production, and herbaceous cover data were analyzed with site groupings (location) and species as the independent variables and bulk density, salinity, and woody basal area as potential covariables. Bonferroni-adjusted LSDs were used to determine significant differences in herbaceous, wood, and litter production among the site groupings and between species. Linear contrasts were used to address specific *a priori* hypotheses. In addition, wood and litter production were also analyzed as total (all species) wood and litter production per m² per year, again with bulk density, interstitial salinity, and basal area as potential covariables. Finally, herbaceous, wood, and litter production were combined for total primary production and analyzed for site grouping differences, also using the three potential covariables. Unless otherwise noted, all statistical findings were significant at a protected $\alpha = 0.05$ level (Zar, 1996).

RESULTS

Wetland Subsidence

The original SET measurements were taken during October and December of 2000. These readings were later deemed useless because the severe drought caused the highly organic soils of the Maurepas swamp to shrink. SET readings were also taken during the fall and winter of 2001 and 2002, under subaerially -exposed conditions in most cases. At present, changes in elevation range from net subsidence of 5.8 mm (s.e. + 5.2 mm) at Intermediate sites to net accretion of 12.0 mm (s.e. + 6 mm) at marsh sites, with most site grouping appearing roughly stable (Figure 2). The prolonged high water during late summer and fall of 2002 was accompanied by two tropical storms (Hurricane Isadore and tropical storm Lilly) that resuspended sediments from Lakes Ponchartrain and Maurepas and these carried deep into the surrounding marshes and swamps. Deposition of greater than 2 cm occurred in the marshes contiguous with North Pass (Shaffer, unpublished data). Feldspar marker horizon data would enable quantification of these accretionary events, but thus far we have been unable to locate any of the 44 horizons that we placed down in 2001. It appears that the soils of the Maurepas swamp are so weak (i.e., bulk densities are very low) that the feldspar marker horizons are not supported. That is, the feldspar appears to be of sufficient density to move through the soils. We have spent tens of tanks of liquid nitrogen in attempt to find the horizons and have yet to detect a single one.

Validity of *a priori* site grouping

The *a priori* grouping of the sites was tested with a backward stepping discriminant function analysis using all soil and productivity variables measured in 2001. Three soil variables (nitrate concentration, phosphate concentration, and soil salinity), one forest structure variable (basal area), and one productivity variable (total litter) loaded in the discriminant function analysis as predictors of group membership. Of these, basal area and soil salinity were

Sediment Elevation Table: Net Elevation Change by Site Type 2001-2002

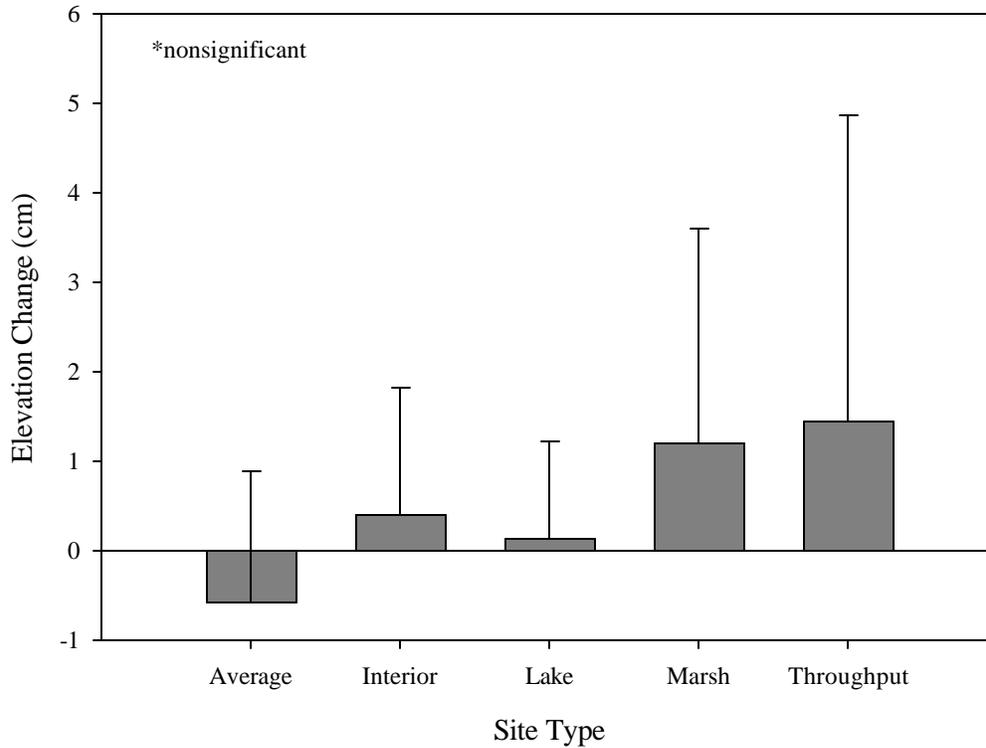


Figure 2. Elevation change (cm) across site types between 2001-2002. Elevation was measured with 22 Sediment Elevation Table sites located at North Pass (marsh), Manchac (marsh), and the southern swamps of Lake Maurepas (other sites) see comment on location.

the most potent variables (Tabachnik and Fidell 2001), followed by phosphate concentration and total litter production as less potent variables, and nitrate concentration as the least potent variable. The first of the three discriminant functions (Figure 3) explained roughly 72% of the total variation in site characteristics and correlated negatively with total litter production ($r = -0.619$) and basal area ($r = -0.518$), as well as positively with soil salinity ($r = 0.490$) and nitrate concentration ($r = 0.434$). This factor appears to be an indicator of salt stress. The second function explained another 21% of the total variation in sites and correlated positively with total litter production ($r = 0.635$) and soil salinity ($r = 0.512$), as well as with nitrate concentration to a lesser degree ($r = 0.390$). This second function appears to indicate a measure of throughput. The last function accounted for less than 7% of the total variation among sites and correlated positively with phosphate concentration ($r = 0.658$) and basal area ($r = 0.609$), as well as negatively with soil salinity ($r = -0.570$). Due to its complexity, this function was not named. The overall model hit ratio for the correct classification of sites into groups was 80% in the main model and 75% in the jackknifed model solution, indicating an overall good model stability. Model validity was tested (Hair et al. 1998) by using the Proportional Chance Criterion plus 25% (PCC = 33.13%), and Press' Q (Press' Q = 64.53, $p < 0.001$) and was found to be significant. Intermediate sites, with correct classifications as low as 70% in the main and jackknifed model, and throughput sites, with correct classifications as low as 75% in the main model and only 63% in the jackknifed model, were the most difficult sites to classify.

Abiotic Characterization of the Study Area

Well salinity

The analysis of the well water salinity data revealed that salinity differed significantly between site groupings ($F_{3,72} = 28.21$, $p < 0.000001$; Figure 4) and between years ($F_{1,72} = 146.85$, $p < 0.000001$; Figure 4). Well water salinity levels at the Lake sites were higher than anywhere else with a mean of 4.15 ± 0.29 ppt (mean \pm standard error) during the drought year

Canonical Scores Plot

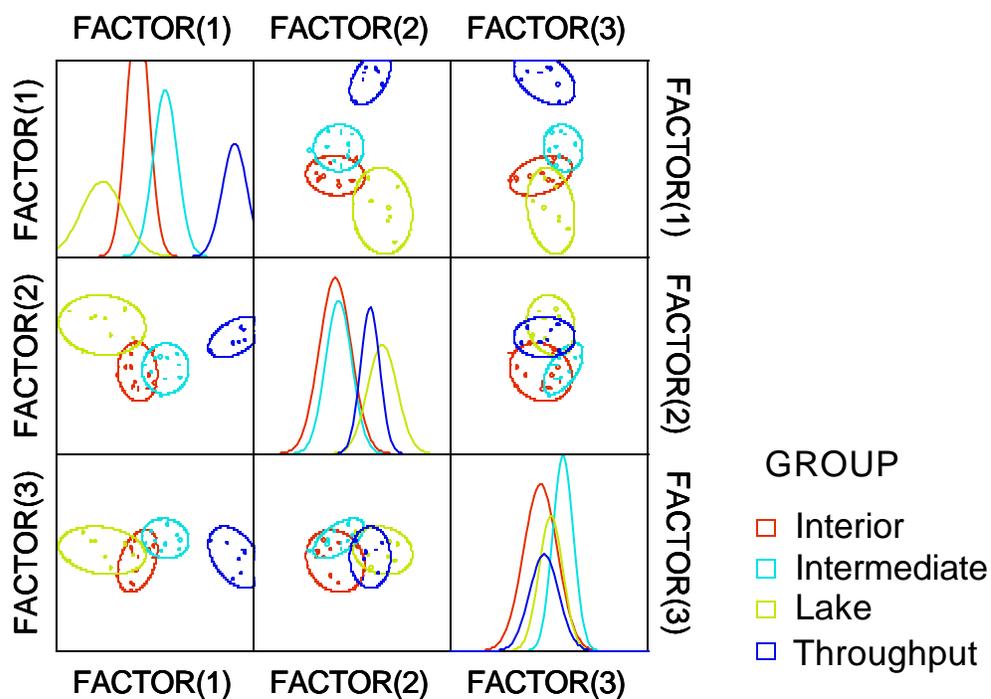


Figure 3. Results of the Discriminant Function Analysis to classify sites into groups using all available, significant year 2001 data. Factor one accounts for 79.5% of the total variation and correlates positively with total litter ($r = 0.848$), basal area ($r = 0.733$) and bulk density ($r = 0.608$), as well as negatively with light reaching the ground ($r = -0.811$). Factor two captures 17.0% of the total variation and correlates positively with nitrate concentration ($r = 0.633$). Factor three explains less than 5% of the total variation and correlates negatively with phosphate concentration ($r = -0.674$).

Maurepas Well Salinity 2000-2002 by Site Type and Year

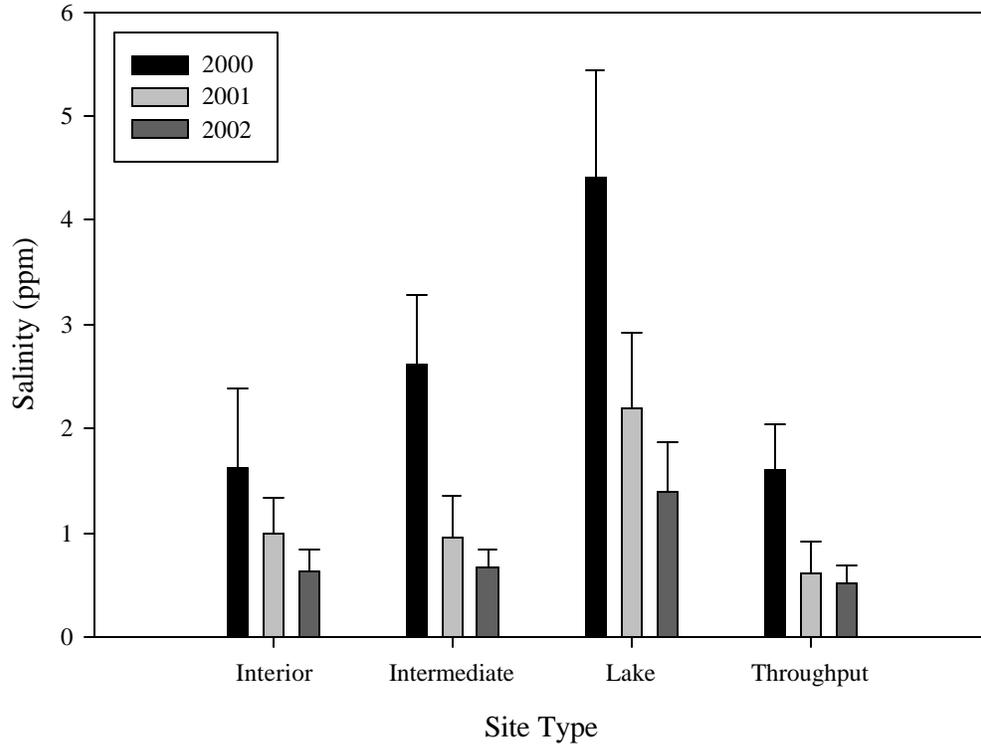


Figure 4. Well Salinities exhibited a downward trend over the three year study ($p < 0.001$), and were highest at Lake Sites.

and a mean of 2.19 ± 0.26 ppt during the following year and $1.5 + .40$ during 2002.

Intermediate plots had higher salinities than Interior and Throughput plots during 2000, but very similar salinities during 2001 and 2002 (Figure 4). Analyzing the salinity data in a site-specific model, polynomial contrasts were calculated to investigate specific spatial trends of salinity changes between four sites at a time. Well-salinity was found to decrease significantly with increasing distance from Lake Maurepas into the interior swamp (site 8 > site 10 > site 11 > site 12; $F_{1,40} = 41.11$, $p < 0.000001$; Figure 5). Increasing proximity to Pass Manchac along the southern shore of Lake Maurepas further increased well-salinity levels (site 17 < site 18 < site 19 < site 20; $F_{1,40} = 54.39$, $p < 0.000001$; Figure 6). Distance from the lake, however, did not affect well-salinity levels along Blind River ($F_{1,40} = 0.04$, $p = 0.836$) or within the areas along Reserve Relief Canal ($F_{1,40} < 0.01$, $p = 0.997$). Overall, well salinity at all study plots was approximately twice as high during the drought year as during years with normal precipitation, as represented by the year 2001 and 2002 data.

Bulk density

Bulk densities differed among site groupings ($F_{3,36} = 7.60$, $p < 0.000463$; Figure 7). The highest bulk densities were found at the Throughput sites (mean = 0.145 ± 0.035 g/cm³). Interior and Intermediate sites had lower bulk densities than the Throughput sites, ranging from a mean of 0.076 ± 0.006 g/cm³ at the Interior sites to a mean of 0.103 ± 0.012 g/cm³ at the Intermediate sites, and did not differ significantly from one another. Lake sites had the lowest bulk densities (mean = 0.054 ± 0.003 g/cm³). Site 5 (Alligator Island), which is impacted by the Amite Flood-relief Diversion Canal, has significantly higher bulk densities than all other sites along Blind River (orthogonal polynomial contrast $F_{1,20} = 10.05$, $p < 0.005$; Figure 7).

Yearly Average Well Salinity by Distance from Lake Maurepas

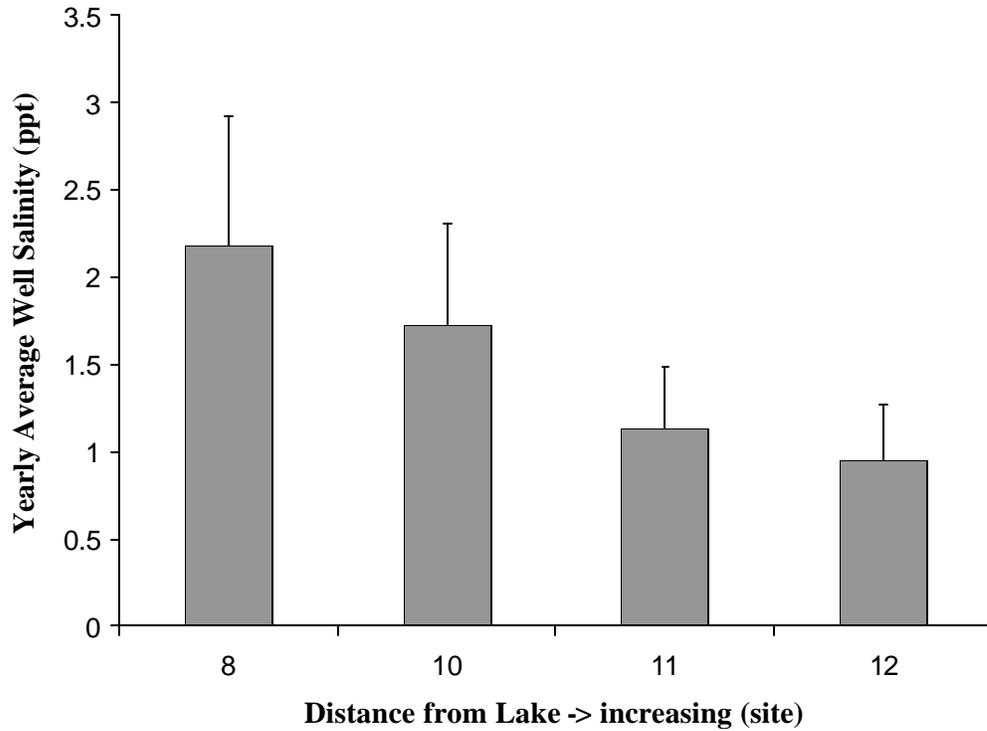


Figure 5. Yearly average well salinity (mean \pm standard error) at four sites increasingly removed from Lake Maurepas. Orthogonal polynomial contrast $F_{1,40} = 41.11$, $p < 0.000001$. Salinity decreases as distance from the lake increases.

Yearly Average Well Salinity by Distance from Pass Manchac

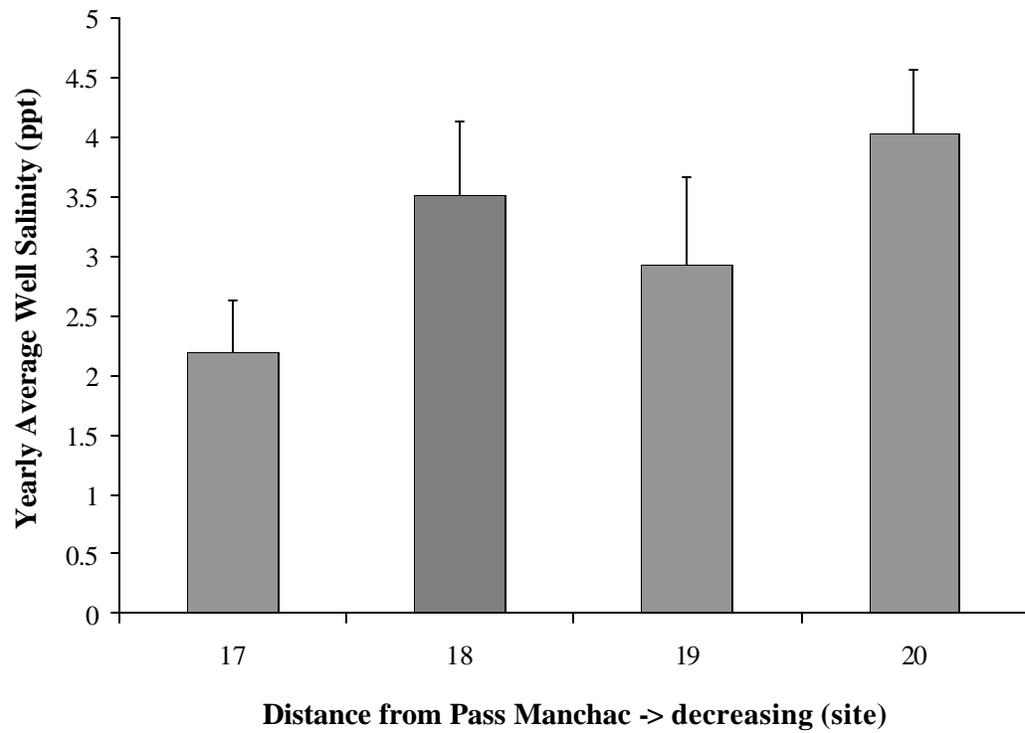


Figure 6. Yearly average well salinity (mean \pm standard error) at four sites at decreasing distances from Pass Manchac. Orthogonal polynomial contrast $F_{1,40} = 54.39, p < 0.000001$. Salinity increases with proximity to Pass Manchac.

Bulk Density by Site Type

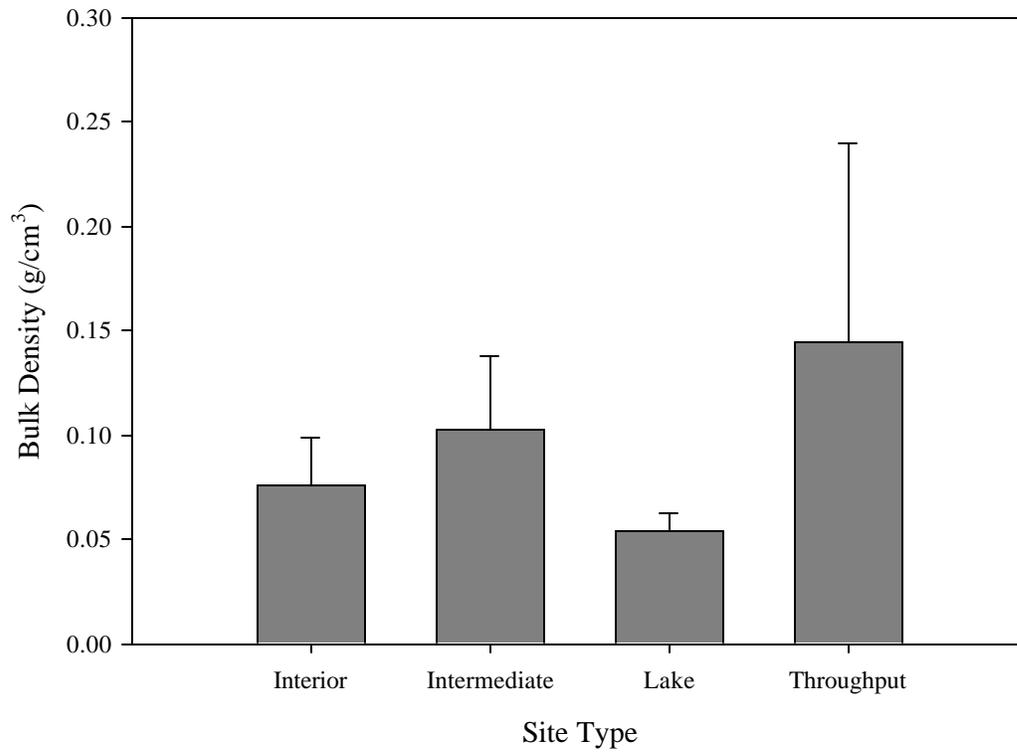


Figure 7. Bulk densities at the different habitat types located in southern Lake Maurepas

Drought effects

As dry extraction soil data was collected with a greatly different extraction method, no direct statistical comparisons were made between dry extraction and wet extraction soils data. Overall averages of all sites within the Maurepas study area show, however, that soil pH decreased to an average of 4.93 ± 0.06 during the June sampling period in 2000 (compared to an average pH of 6.00 ± 0.06 during June 2001). Furthermore, redox potentials were more oxidized at all sites that were sampled during June 2000 (mean = 294.2 ± 14.1 mV, as compared to mean = 183.1 ± 9.9 mV during August 2001). The maximum well salinity recorded in the study area was 8.0 ppt during the drought year and 4.3 ppt during the subsequent year, both at sites on Jones Island (site 20, Figure 1).

Of all the soil variables measured (during wet conditions), only well salinity, pH, nitrate concentrations, and redox potential entered as significant variables into the discriminant function analysis to classify drought year (2000) from normal precipitation year (2001) data. Of these, well salinity was by far the most significant and potent variable and pH was the least potent. Twenty percent of the data were used as a holdout sample to validate the model. Only one function was needed to separate years. This function correlated negatively with well salinity ($r = -0.820$) and positively with phosphate concentration ($r = 0.529$). The overall model hit ratio for the correct classification of soil samples was 95% in the main model, 100% in the holdout sample, and 95% in the jackknifed model solution, indicating an overall good model stability. Model validity was tested by using Proportional Chance Criterion plus 25% (Hair et al. 1998) ($PCC_{\text{main model}} = 65.0\%$ and $PCC_{\text{holdout sample}} = 66.41\%$), and Press' Q ($\text{Press' Q}_{\text{main model}} = 43.65$, $p < 0.001$; $\text{Press' Q}_{\text{holdout sample}} = 16.00$, $p < 0.05$). In short, drought year soils after re-flooding could be characterized by higher salinity and lower redox potentials at 15 cm depth, nitrate concentrations, and pH.

Percentage organic matter

Percent organic matter was highest for the Interior and Lake sites and lowest for Intermediate sites (Figure 8). Throughput sites did not differ from any other sites with respect to percent organic matter. No significant difference between fertilized and unfertilized herbaceous plots was detected in percentage organic matter levels.

Prediction of interstitial soil characteristics from dry collection

All simple linear regressions to predict interstitial soil variable concentrations from dry extractions resulted in R^2 values of less than 0.001. This indicates that there is little predictive ability in utilizing the dry extraction soil data to predict wet extraction (interstitial) soil variable concentrations. Overall, values generated from dry soil extractions were much more variable than those generated from wet (interstitial) extractions. Despite the fact that dried soil was ground and thoroughly mixed, only a sub-sample of the complete core is extracted in this procedure. Conversely, when interstitial water is collected (following centrifugation) it represents a true composite of the soil and, hence, is less variable.

Nutrients

During 2000, nitrate levels were elevated in the fertilized plots over the control plots, except for Throughput sites where they did not differ (Figure 9a). In contrast, no differences occurred between fertilized and control plots for phosphate and ammonium (Figure 9b, c). With respect to site groupings, Lake sites had elevated phosphate and ammonium.

During 2001, nitrate levels did not differ across site types or fertilizer levels (Figure 10a, d). Phosphate levels at Interior sites were higher than those at Lake sites

Percent Organic Matter of Soil in 2001 by Site Type

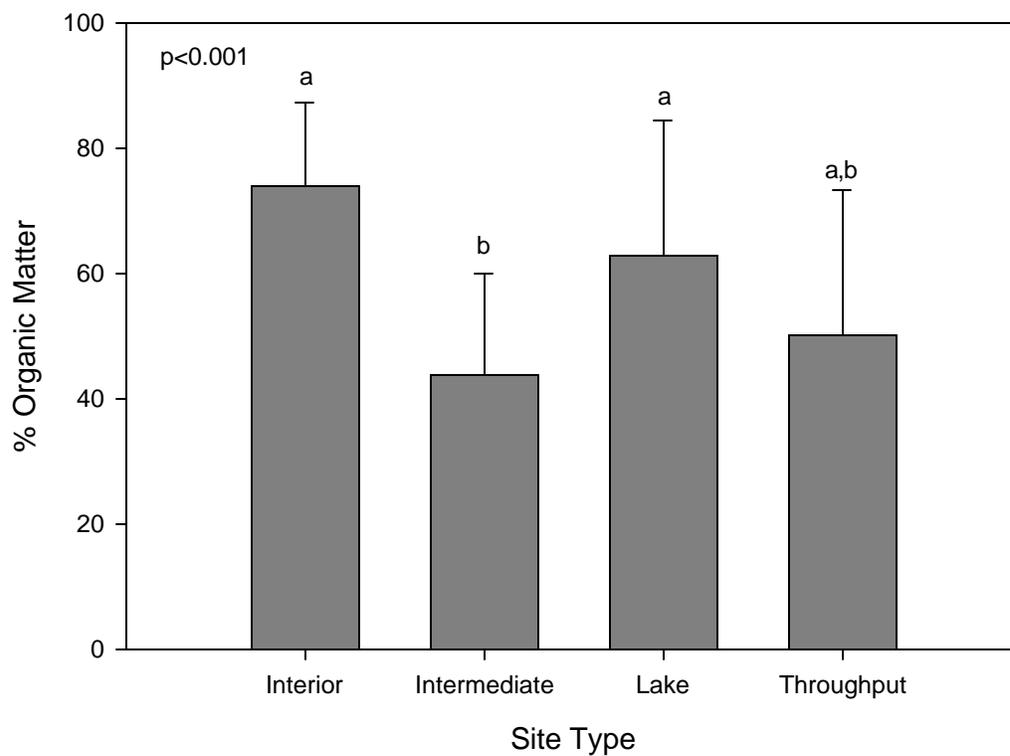


Figure 8. Percent organic matter is significantly different by site type ($p < 0.001$). Bars with shared letters are not significantly different according to a Bonferonni-adjusted Least Significant Difference.

Ammonium, Phosphate, and Nitrogen Levels in 2000 by Site Type and Nutrient Augmentation

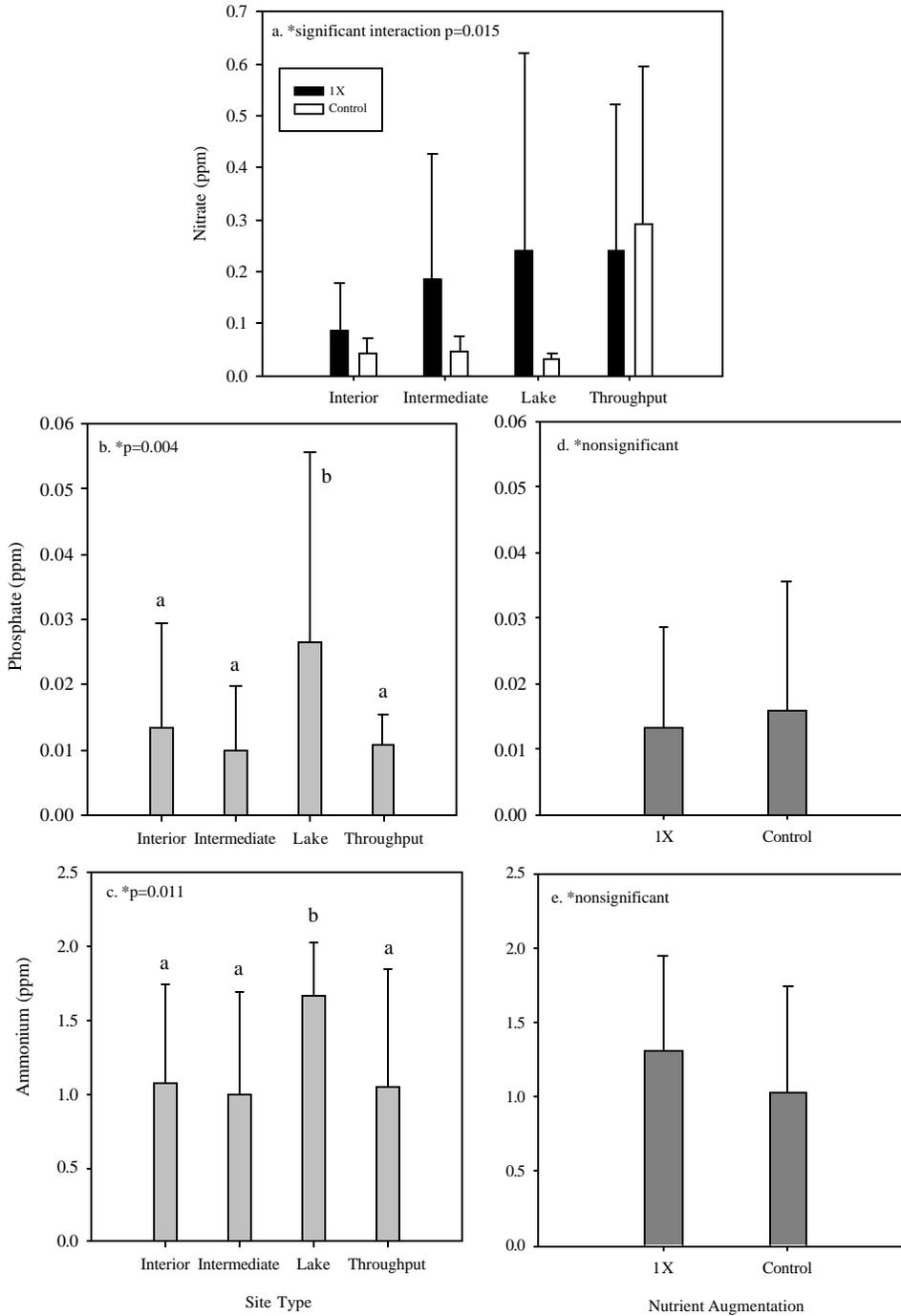


Figure 9. Nutrient levels in 2000 for a) Interaction of nitrate levels between site type and nutrient augmentation (1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer), b) Phosphate levels under nutrient augmentation, c) Ammonium levels under nutrient augmentation, d) Phosphate levels for the different site types and e) Ammonium levels for the different site types. Bares with shared letters are not significantly different according to a Bonferroni-adjusted LSD.

Ammonium, Phosphorous, and Nitrate Levels in 2001 by Site Type and Nutrient Augmentation

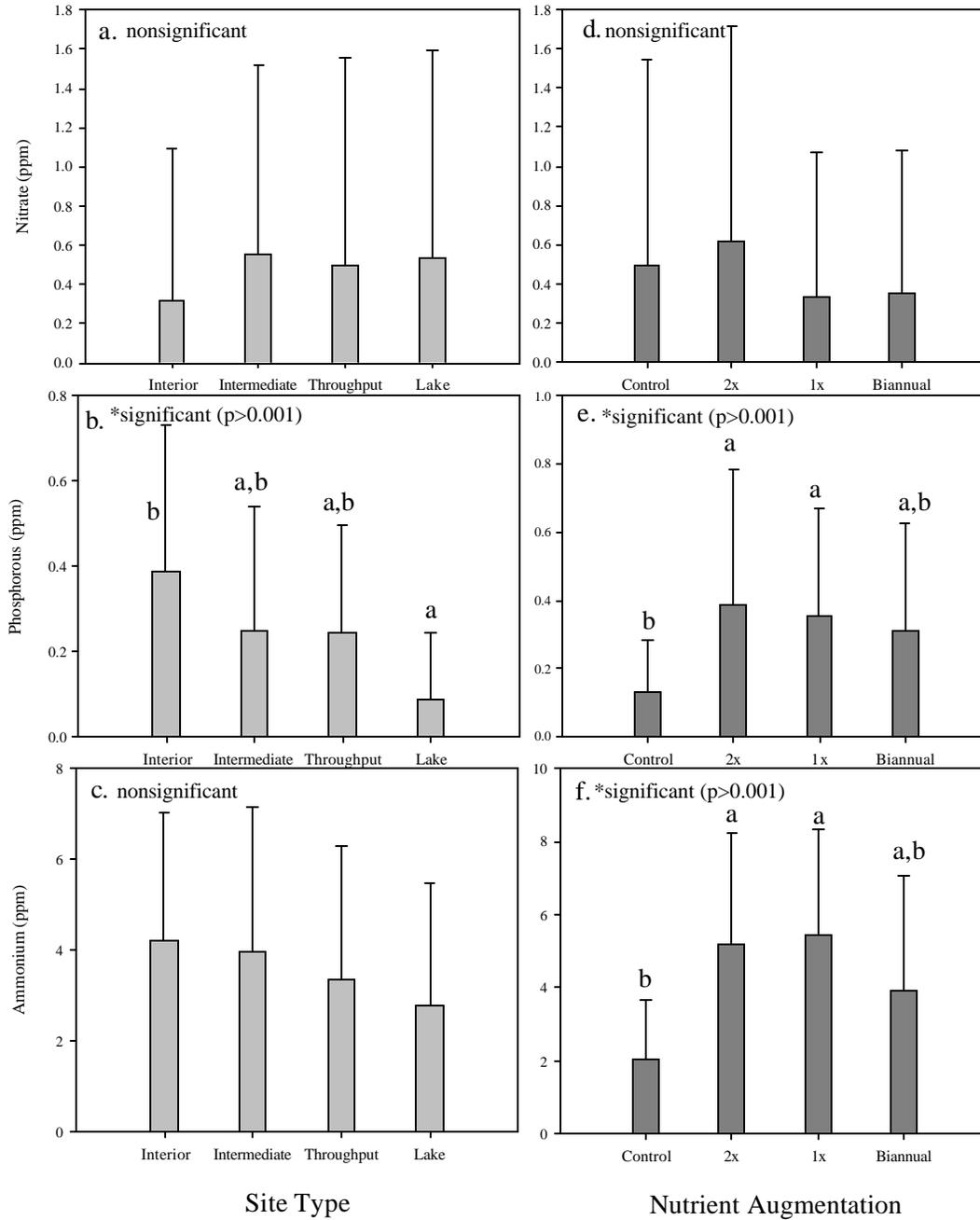


Figure 10. Nitrate, phosphorous, and ammonium levels by site type (a-c) and nutrient augmentation levels (d-f; 1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer. Bars that share letters are not significantly different according to a Bonferonni-adjusted LSD.

(Figure 10b) and single and double fertilized plots had higher levels of phosphates than the controls (Figure 10e). Ammonium levels did not differ across site types (Figure 10c), but the single and double fertilized plots did display elevated levels over control plots (Figure 10f).

During 2002, nitrate levels did not differ between site types or fertilizer levels (Figure 11a, d). Phosphate levels were highest at Lake sites and the double fertilized and biannually fertilized plots had elevated phosphate levels over the controls. Ammonium levels were lower at Throughput sites than Lake sites (Figure 11c) and did not differ for fertilized and non-fertilized plots (Figure 11f).

Overall, nutrient levels are very low in the soil waters of the Maurepas swamp (Figure 12). In general, the drought year of 2000 contained significantly lower concentrations of ammonium, nitrate, and phosphate than the non-drought years of 2001 and 2002 (Figure 12).

Soil redox

The Throughput sites became more reduced at the surface (1 cm depth) than all other sites in the fall compared with the spring, resulting in an interaction of location x season ($F_{4,40} = 24.290$, $P < 0.001$; Figure 13). There was also a consistent and significant trend of surface being more reduced in the fall than the summer ($F_{1,40} = 88.878$, $P < 0.001$; Figure 13). Although not significant, there was also a consistent trend towards deeper soils being more reduced during the fall season ($F_{1,40} = 3.646$, $P = 0.064$; Figure 13). The Lake and Throughput sites did not show as strong of a change in soil reduction as the other sites during fall, resulting in a time by location interaction ($F_{4,40} = 14.651$, $P < 0.001$). It should be noted that all soil redox measurements were greater than -75 mV and that site averages ranged between 26-411 mV. This explains

Ammonium, Phosphate, and Nitrate Levels in 2002 by Site Type and Nutrient Augmentation

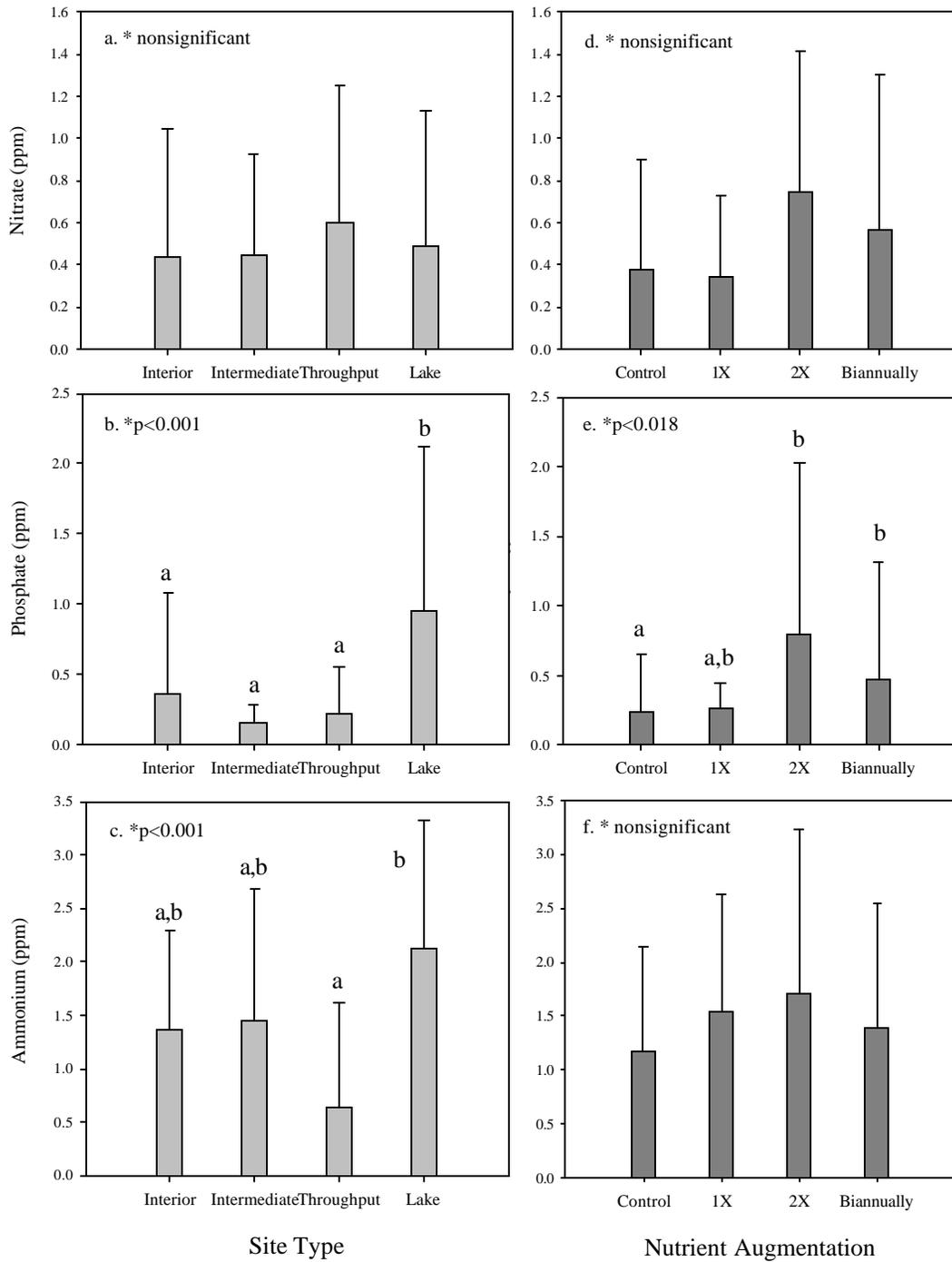


Figure 11. Nitrate, phosphorous, and ammonium levels by site type (a-c) and nutrient augmentation levels (d-f; 1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer). Bars with shared levels are not significantly different. Bars that share letters are not significantly different according to a Bonferonni-adjusted LSD.

Nutrient Levels in Control Plots by Year

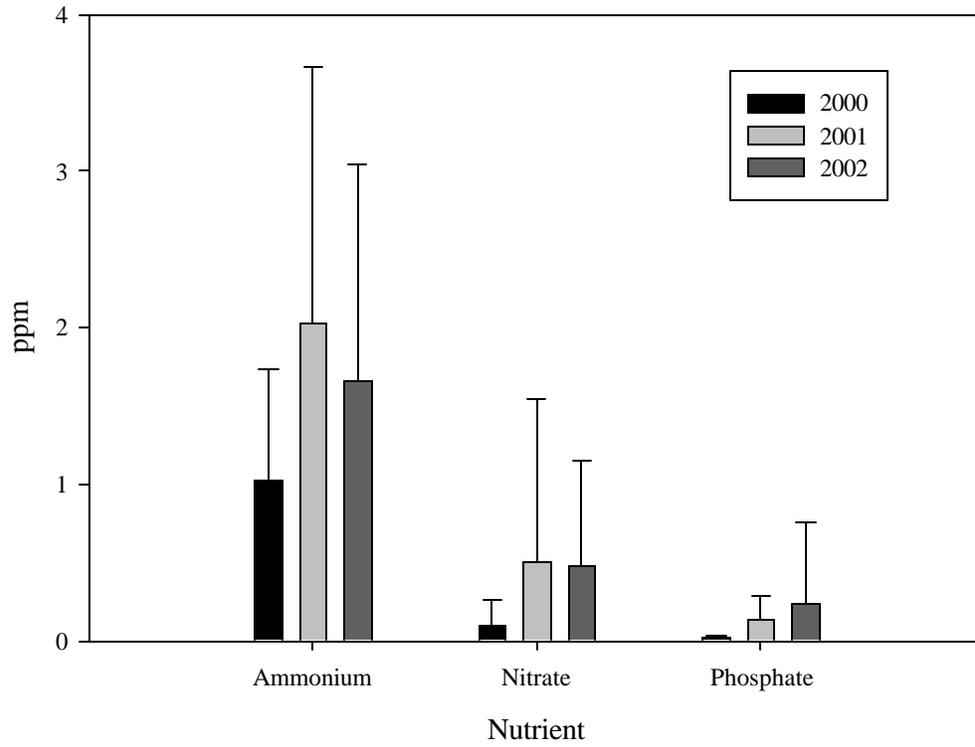


Figure 12. Nutrient levels in control plots across years ($p < 0.001$). Bars are new standard errors and are not adjusted for the experimental design variables of covariables.

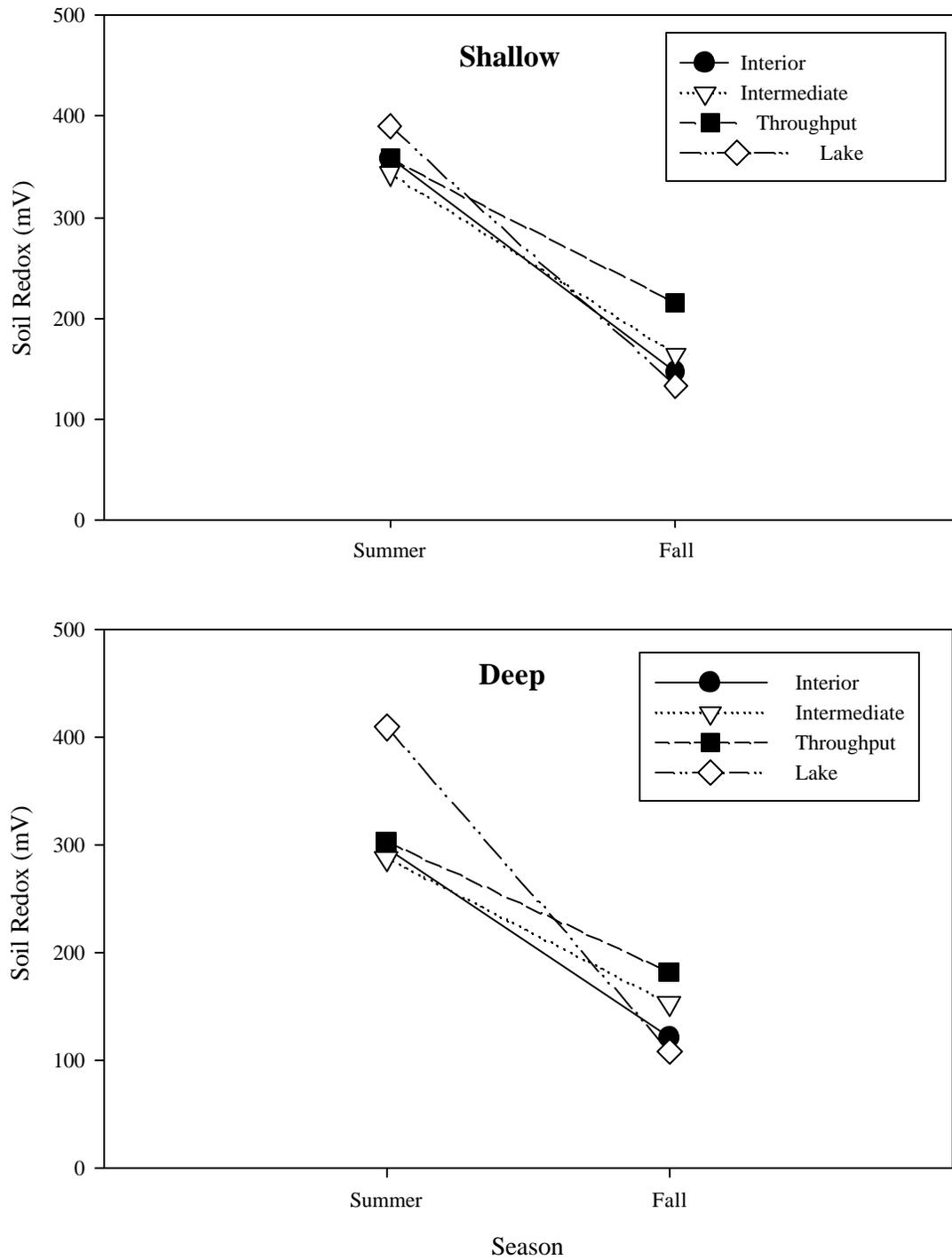


Figure 13. There is a decrease in redox potential from the summer to the fall season at all depths and site types.

the lack of detectable sulfides as sulfide production does not occur until soil reduction reaches -75 mV (Mitsch and Gosselink 2000).

pH

During 2000, Interstitial pH levels were not significantly different in fertilized plots than unfertilized plots (Figure 14a). pH levels during 2001 were significantly lower than during 2000 (Figure 14b), but did not differ between control plots and the various fertilizer regimes (Figure 14c).

Herbaceous Vegetation

Species Richness

Species richness of herbaceous vegetation was highest at the Lake sites, lowest at the Throughput sites and intermediate at Intermediate and Interior sites (Figure 15a), across the three years and six sampling seasons. Nutrient augmentation, regardless of the amount or the timing, had no effect on richness (Figure 15b). The relatively low species richness of the Throughput sites is almost certainly due to dense overstory and mid-story cover of woody species.

Cover values

Early season cover value estimates of the 15 dominant species, that represent 97% of the total cover for 2001 (Figure 16) and 2002 (Figure 17), revealed that Alligatorweed (*Alternanthera philoxeroides*), smartweed (*Polygonum punctatum*), and arrow arum (*Peltandra virginica*) appeared to be the most ubiquitous species in the swamps of southern Maurepas, as they were present in all habitat types. Pickerelweed (*Pontederia chordata*) decrease in abundance as habitats become more

pH by Site Type, Year, and Nutrient Augmentation Level

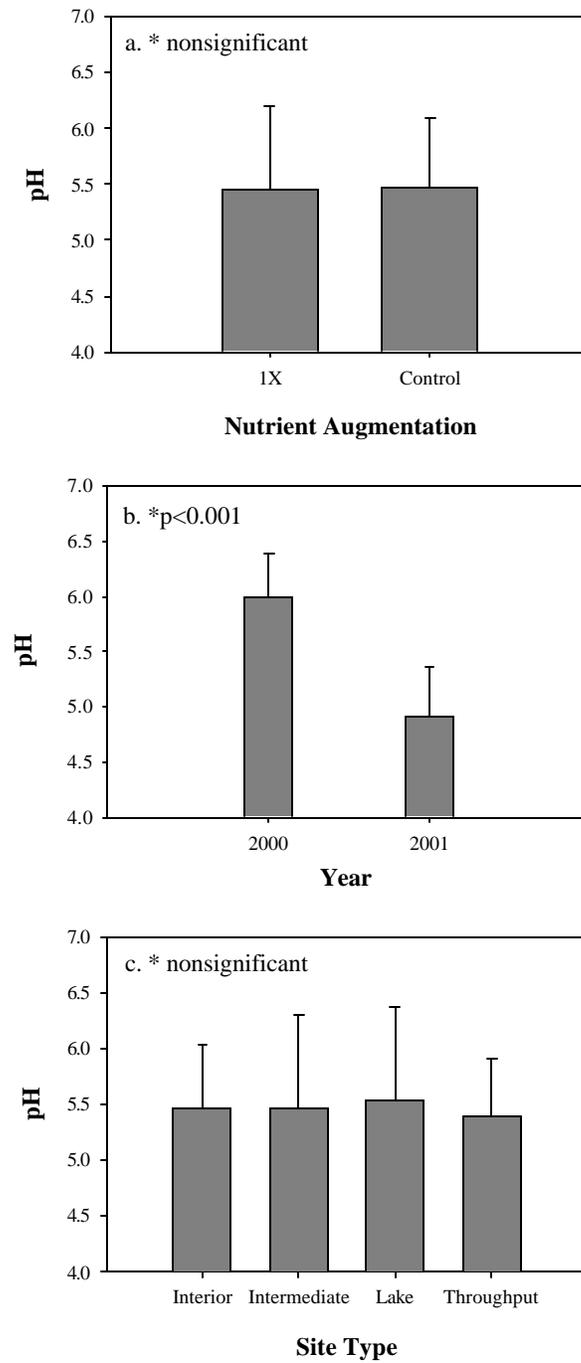


Figure 14. pH across (a) 2000 fertilizer levels (1X= 59g Osmocote/m²), (b) years and (c) site types.

Herbaceous Species Richness (2001-2002) by Site Type and Nutrient Augmentation Level

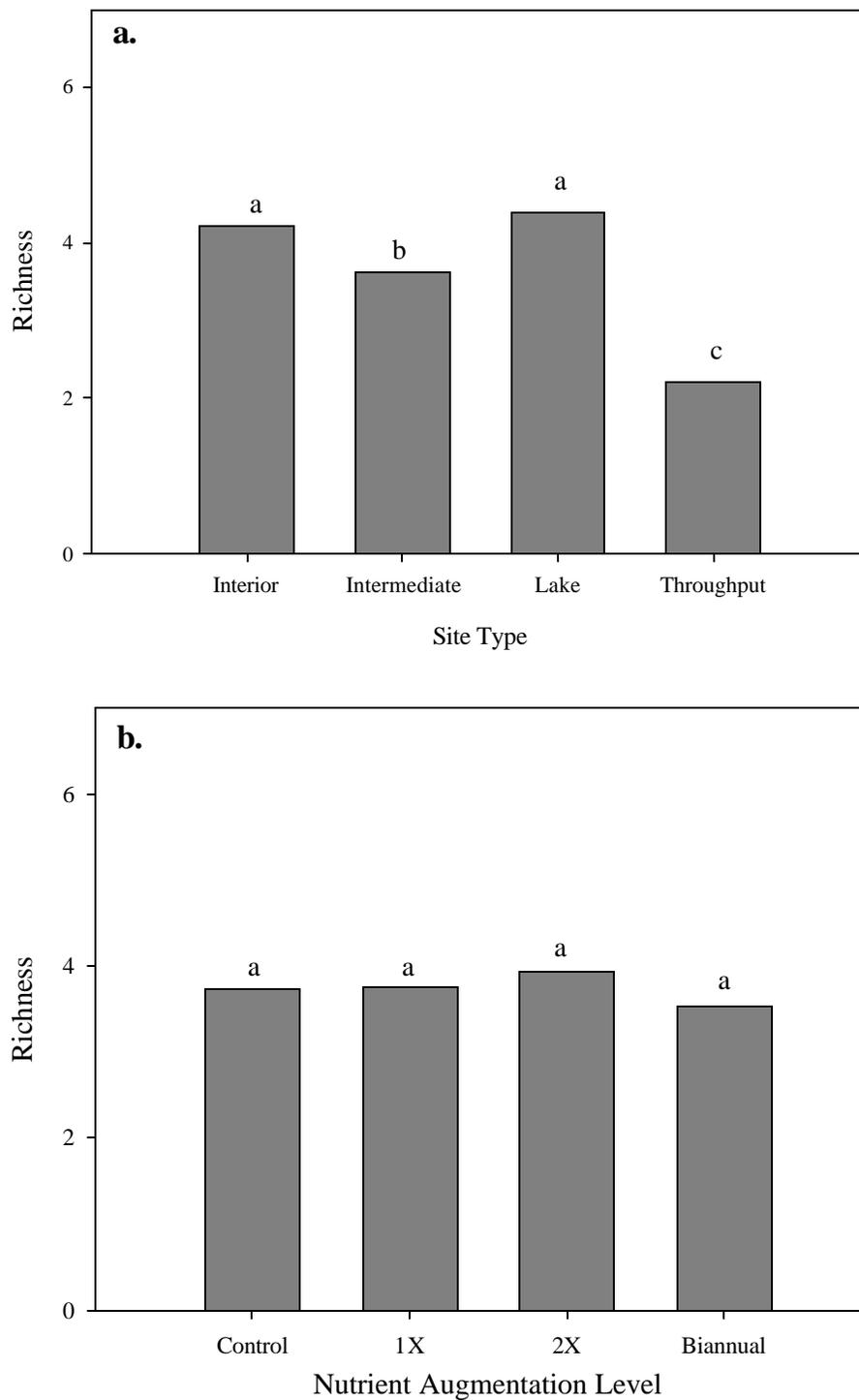


Figure 15. Vegetative species richness across (a) site types and (b) fertilizer levels (1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer). Bars that share letters are not significantly different according to a Bonferonni-adjusted LSD.

Percent Cover Signatures of Site Types in 2001

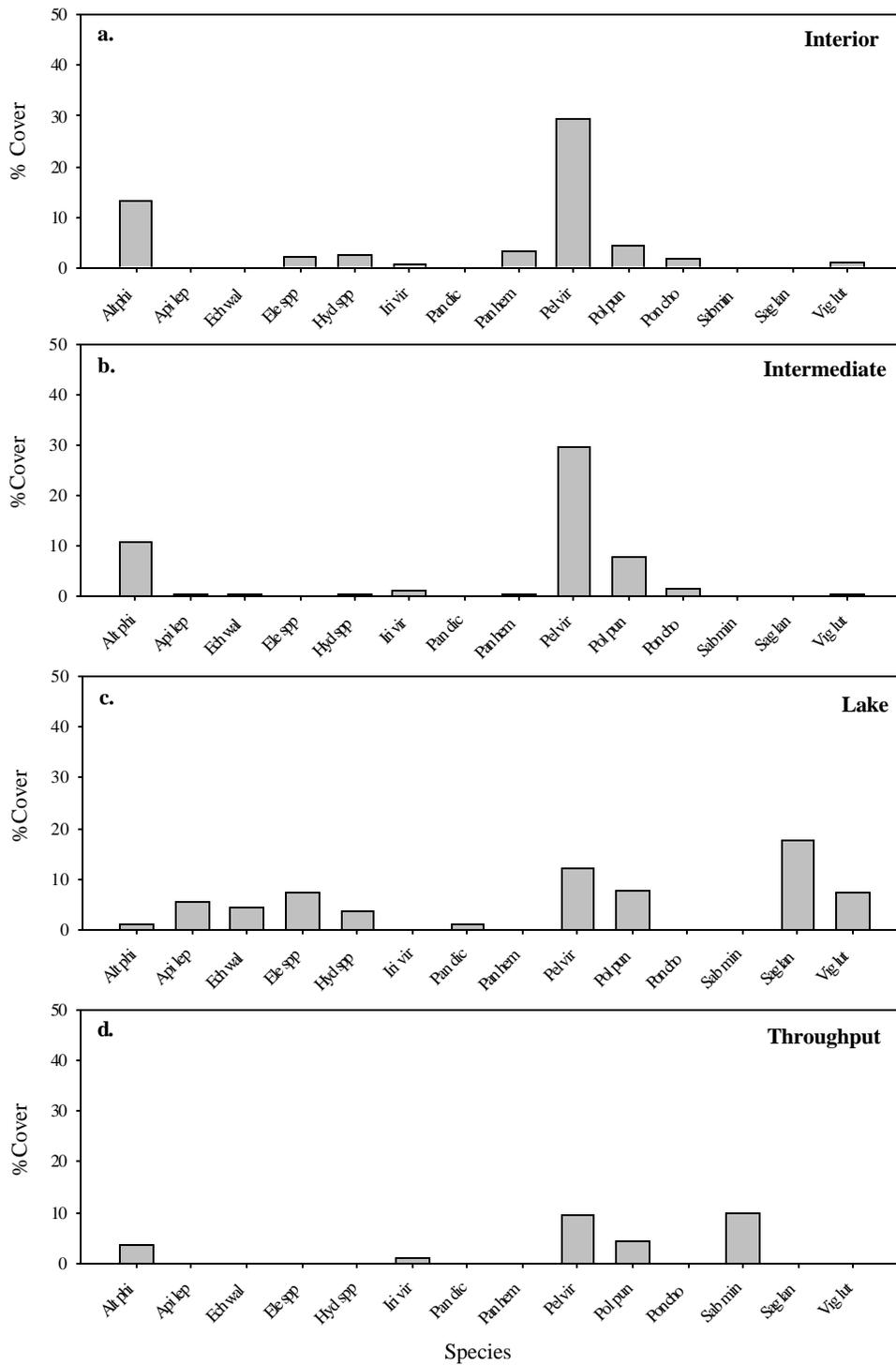


Figure 16. Vegetative species cover in 2001 for the 15 dominant species in (a) Interior, (b) Intermediate, (c) Lake and (d) Throughput sites.

Percent Cover Signatures of Site Types in 2002

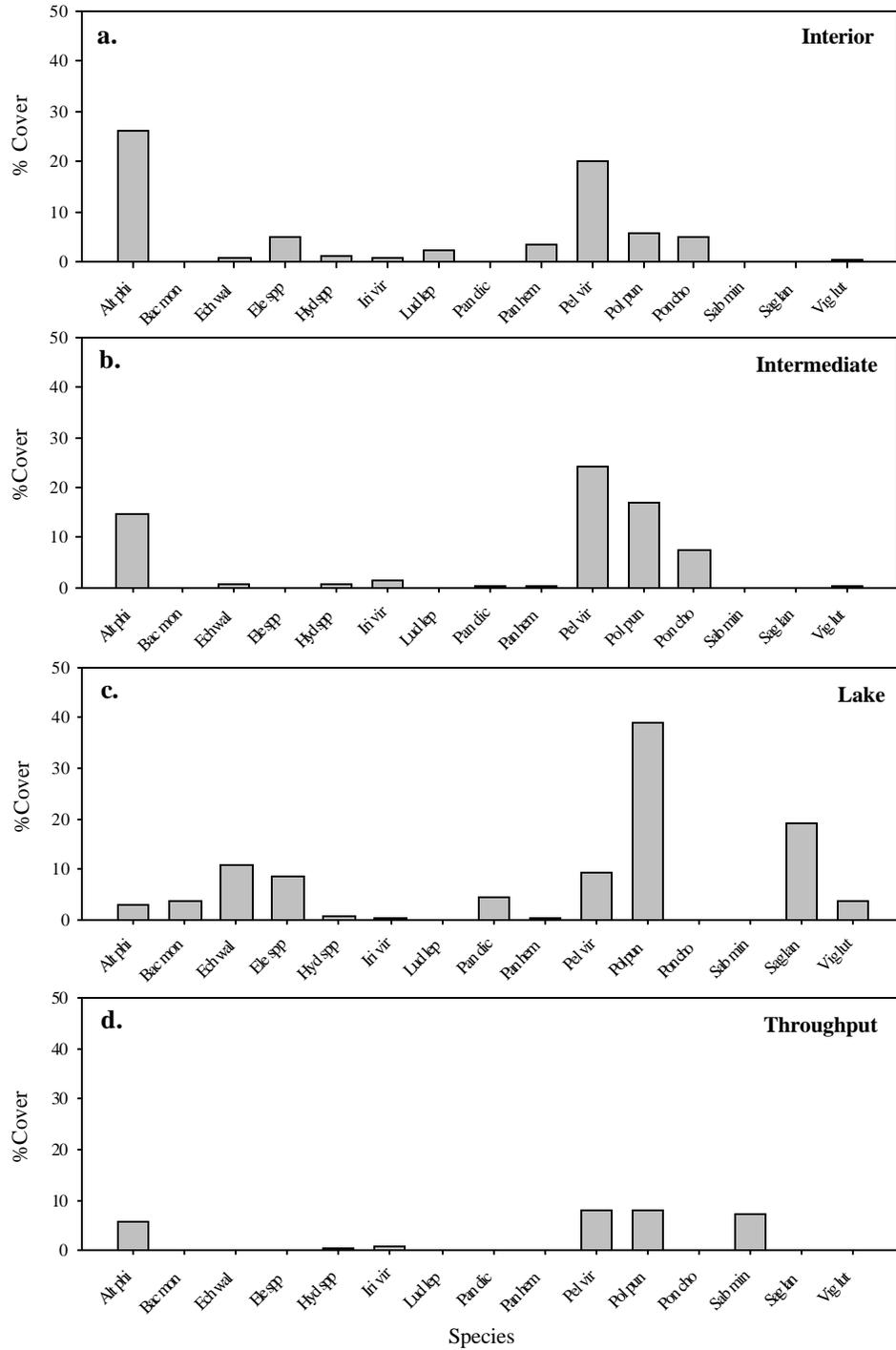


Figure 17. Vegetative species cover in 2002 for the 15 dominant species in (a) Interior, (b) Intermediate, (c) Lake and (d) Throughput sites.

degraded, whereas bulltongue (*Sagittaria lancifolia*) and fall panicum (*Panicum dicotomiflorum*) become more abundant. Maidencane (*Panicum hemitomon*) and spike rush (*Eleocharis* spp.) are generally only represented in the Interior sites and in ponding areas of Lake sites. These two species may indicate that a transition to open water is imminent as these are common floating marsh species and the Interior and Lake sites have the lowest soil strength (i.e., bulk densities). Together, these species may serve as indicators of ecosystem health, as may palmetto (*Sabal minor*), which occurs only at the Alligator Island Diversion sites (Figure 1); these two stations have the highest soil strength of the 40 stations interspersed throughout the swamps of southern Maurepas.

Nutrient augmentation

The effects of timed-release fertilizer on vegetative standing crop were most pronounced during the third year of the study (2002). During the summer sampling period, the three nutrient addition treatments produced similar standing crops, which were significantly greater than those of the controls (Figure 18a). By the end of the growing season, standing crop of the plots double-fertilized in the spring, and those fertilized during the spring and again in summer, produced significantly greater standing crops than the once-fertilized and control plots (Figure 18b).

Far more biomass was produced during the first half of the growing seasons in both 2001 (Figure 19a) and 2002 (Figure 19b). As demonstrated by summer of 2002 (Figure 19b), standing crop was highest at the Lake sites and lowest at Throughput sites. This pattern was consistent across seasons and years and is negatively correlated with the basal areas of woody stems at each site.

Herbaceous Biomass by Nutrient Augmentation Level 2002

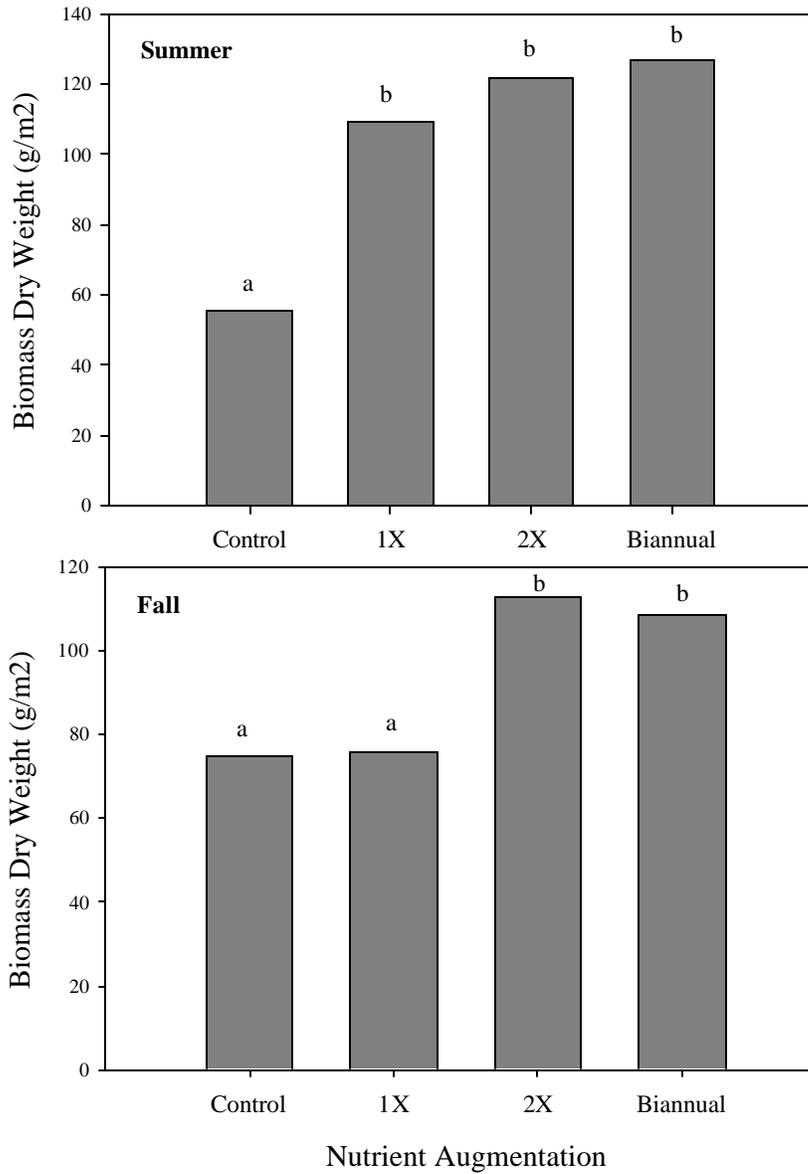


Figure 18. Biomass (dry weight g/m²) across fertilizer levels (1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer) during (a) summer and (b) fall growing seasons of 2002. Bars that share letters are not significantly different according to a Bonferonni-adjusted LSD.

Herbaceous Biomass by Season, Year, and Site Type

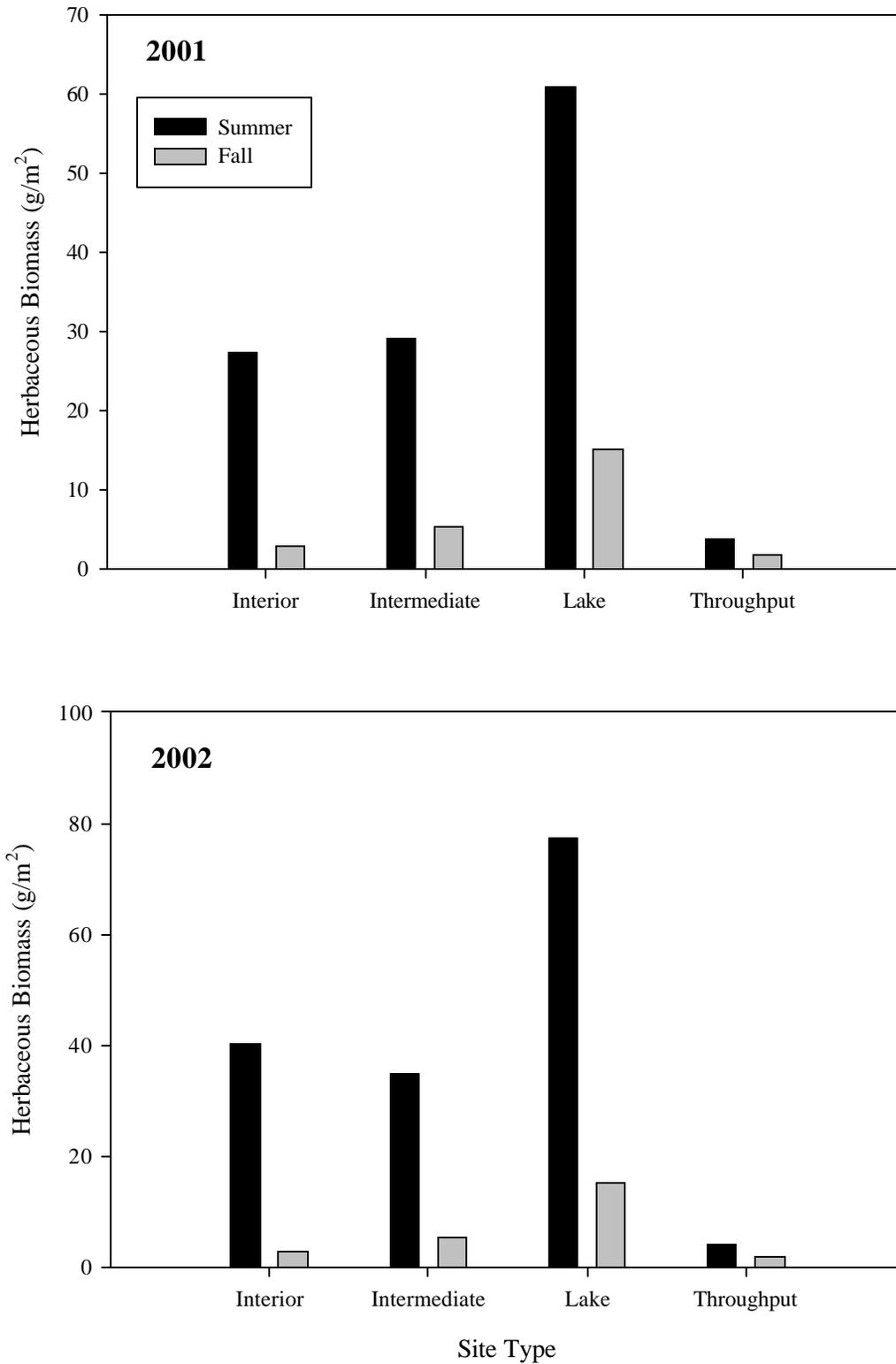


Figure 19. Herbaceous standing crop (dry weight g/m²) for (a) summer and (b) fall growing seasons in the various site types for 2001-2002.

Mammal Exclusion Experiment

Mammals appear to have a dramatic effect on herbaceous standing crop, regardless of nutrient regime (Figure 20). Our mammal-exclusion experiment reveals that the actual biomass production of herbaceous material is double that which is clipped twice annually (compare caged and uncaged control plots in Figure 20a and b). The ratio of (caged to uncaged) biomass in those plots has a mean of 2.04. We use this ratio to adjust our annual production estimates for herbivory.

Annual production

Herbaceous production was highest at the Lake sites, followed by Intermediate and Interior sites (which did not differ), and lowest for Throughput sites (Figure 21). Adjusting for mammalian herbivores doubles production. Furthermore, during 2002 a bloom of *Salvinia molesta* further increased production by about one third (Figure 22). It appears that the low herbaceous production for the southern Maurepas swamps as a whole is primarily due to nutrient limitation. The fall caged plots that were fertilized during the spring and again during the summer contained nearly three-fold higher standing crop than the uncaged control plots (Figure 20b).

**Herbaceous Biomass in 2002 by Season,
Cage Exclusion and Nutrient Augmentation**

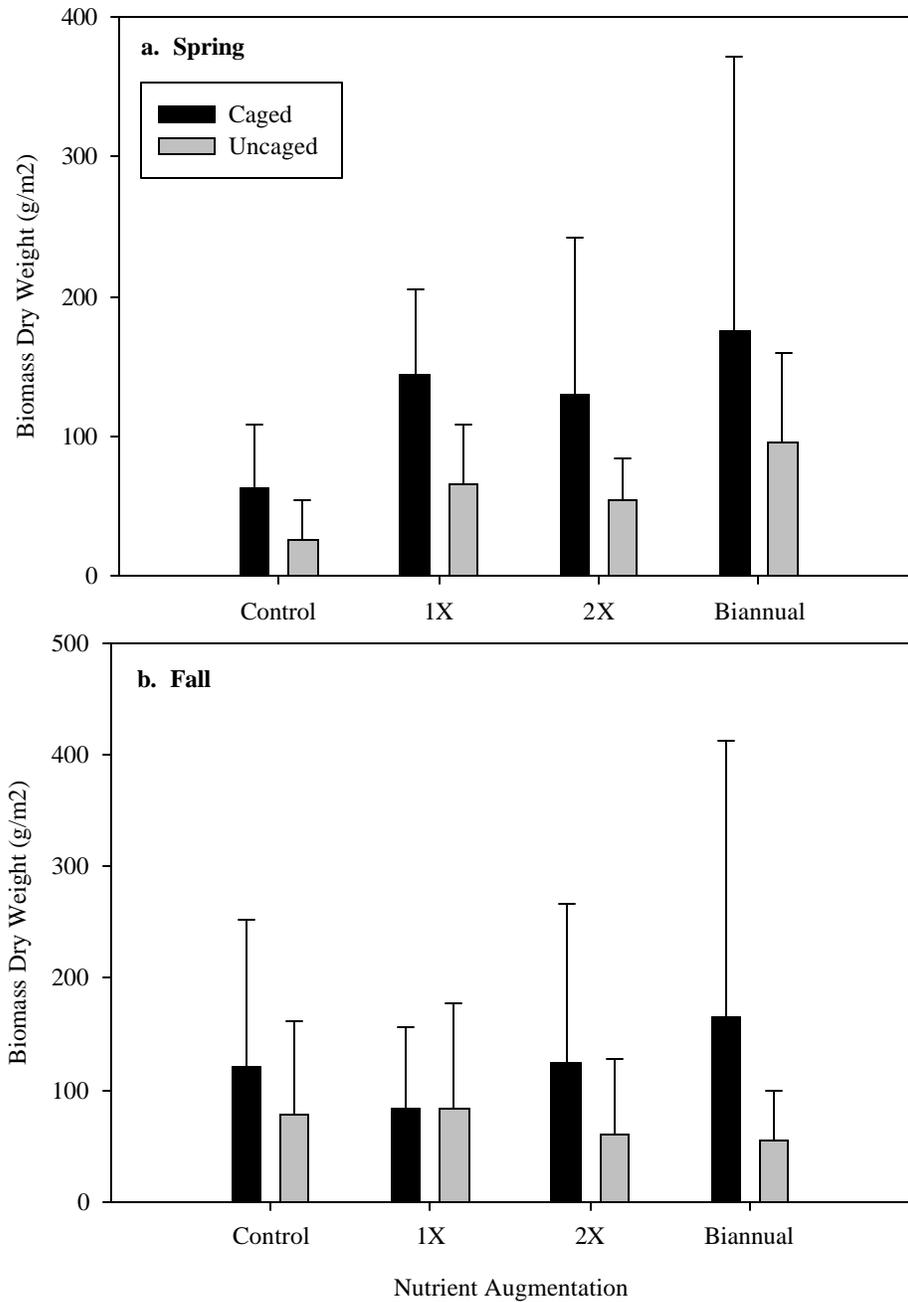


Figure 20. Herbivore exclusion plots across four nutrient augmentation regimes (1X=59g Osmocote/ m², 2X= 118g Osmocote/ m² applied during spring, Biannual=59g Osmocote/ m² applied during spring and summer) during (a) Spring 2002 and (b) Fall

Corrected Herbaceous Production (g/m^2) by Year

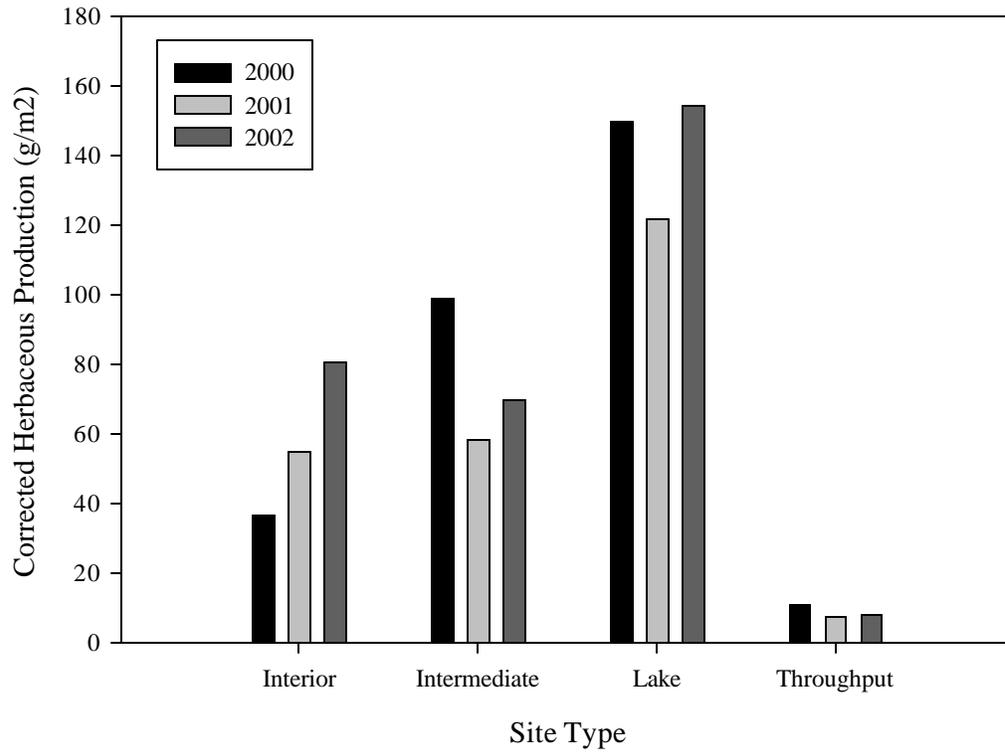


Figure 21. Net primary production of herbaceous vegetation across site types for each of the three years (2000 - 2002).

Herbaceous Production with Floaters by Site Type (2000-2002)

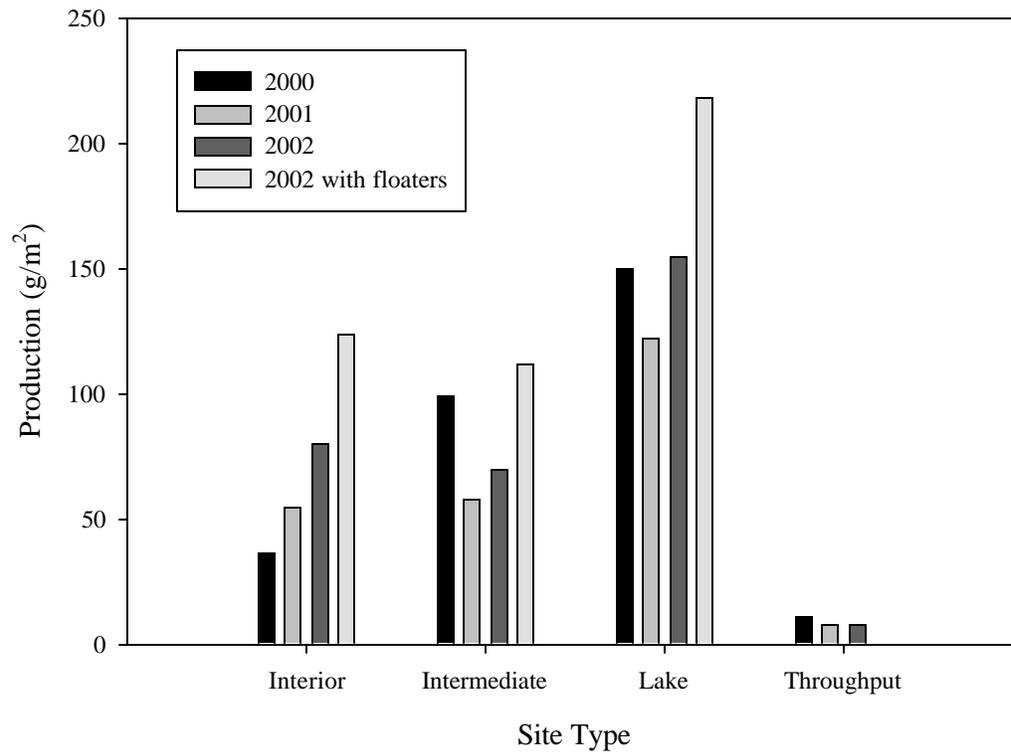


Figure 22. Net primary production of herbaceous vegetation as clipped and adjusted for herbivory across site types and years. Also included is floating vegetation productivity dominated by *Salvinia molesta*.

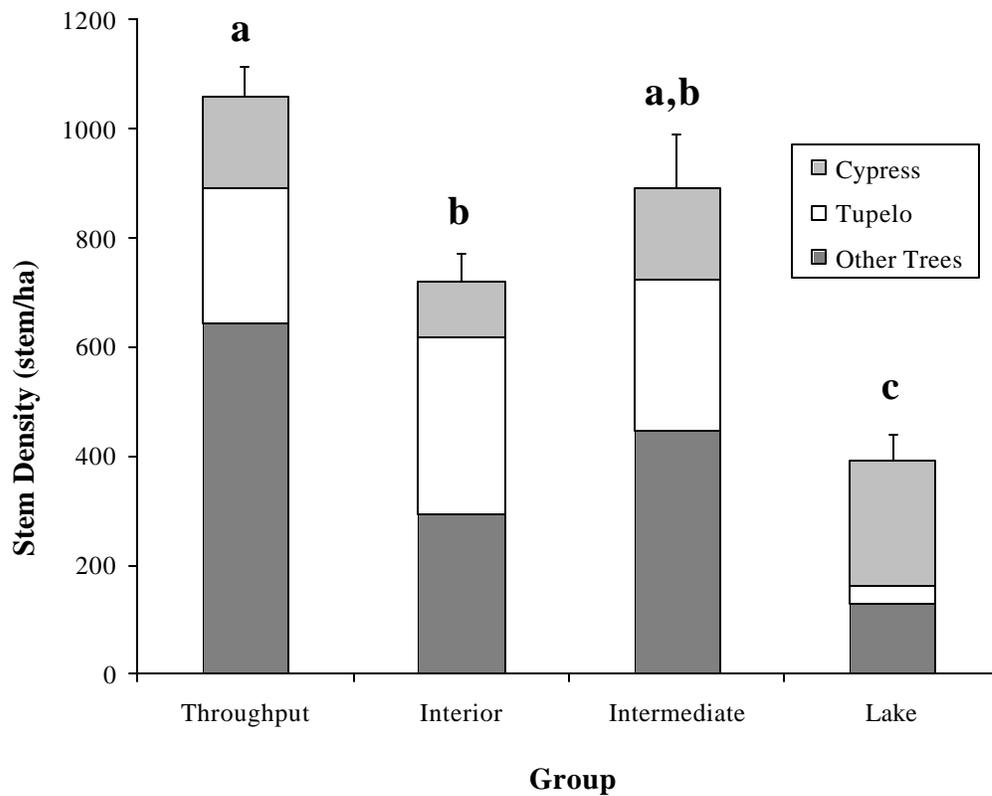


Figure 23. Stem density (mean \pm standard error) across groups. $F_{3,108} = 8.83$, $p = 0.000028$. Other trees are significantly more abundant than *Taxodium distichum* or *Nyssa aquatica* trees, which are equally abundant. $F_{2,108} = 18.38$, $p < 0.000001$. Bars that share letters are not significantly different according to Bonferroni-adjusted multiple comparison tests.

Forest Structure and Tree Mortality

Stem density

Site groupings differed significantly in stem density of trees ($F_{3,108} = 8.83, p < 0.00003$; [Figure 23](#)). Throughput sites had the greatest number of trees (mean = 1058.0 ± 53.8 stems ha^{-1}). Interior and Intermediate sites had significantly lower stem densities than the Throughput sites (mean = 718.9 ± 52.4 stems ha^{-1} and mean = 891.2 ± 96.5 stems/ha, respectively) and did not differ significantly from one another. Stem densities were lowest at the Lake sites, averaging only 392.0 ± 46.0 stems ha^{-1} . Both Throughput and Intermediate sites are numerically dominated by 'Other' trees up to roughly 60 and 50%, respectively. At both of these types of sites, *N. aquatica* is the next most abundant tree (25% and 30%, respectively), followed by *T. distichum* as the least abundant tree (15% and 20%, respectively; [Figure 23](#)). *N. aquatica* and 'Other' trees were approximately co-dominant at the Interior site, accounting for 80-90% of the trees present. *T. distichum* dominated the Lake site with densities around 60%, followed by 'Other' trees with densities around 30-35%. Overall, species categories differed significantly in stem density ($F_{2,108} = 18.38, p < 0.000001$). 'Other' trees, generally dominated by *Acer rubrum* and *Fraxinus pennsylvanica*, were, overall, the most abundant category throughout the study area. *T. distichum* and *N. aquatica* trees were significantly less abundant, but did not differ significantly in abundance from one another. A significant interaction between group and species effects ($F_{6,108} = 7.16, p < 0.000002$; [Figure 24](#)) indicated that there was a higher abundance of *T. distichum* at the Lake sites and a higher abundance of *N. aquatica* at Interior sites than was expected.

Basal area

Basal areas per hectare differed significantly among site groupings ($F_{3,35} = 24.89, p < 0.000001$; [Figure 25](#)). The greatest basal areas were found at the Throughput sites (mean = 48.62 ± 5.80 $\text{m}^2 \text{ha}^{-1}$). Interior and Intermediate sites had significantly lower basal areas than

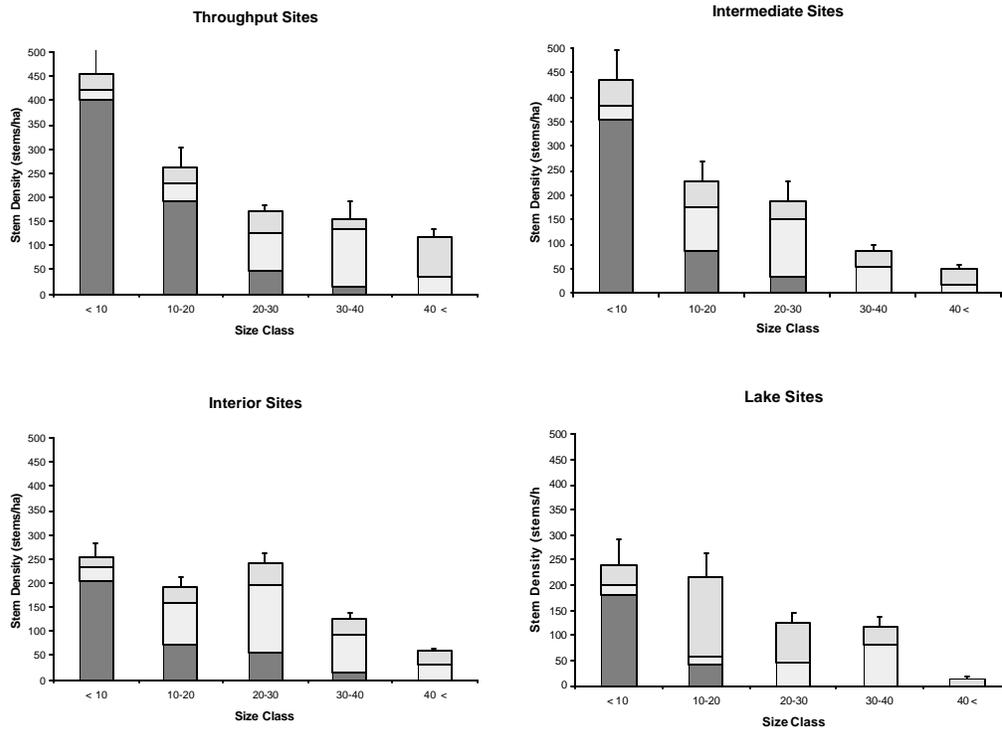


Figure 24. Group differences in stand structure by size class (mean \pm standard error). Horizontal hatching denotes *Taxodium distichum*, dots mark *Nyssa aquatica*, and diagonal hatching shows other trees. Stem density scales are scaled to 500 stems per hectare. The size class of trees less than 10 cm in diameter is dominated by other trees throughout the study area. Large size classes of 30 cm diameter trees and larger are dominated or exclusively represented by *N. aquatica* or *T. distichum*. Interior and Lake sites are particularly impoverished in small trees.

Basal Area/ Ha 2002 by Species

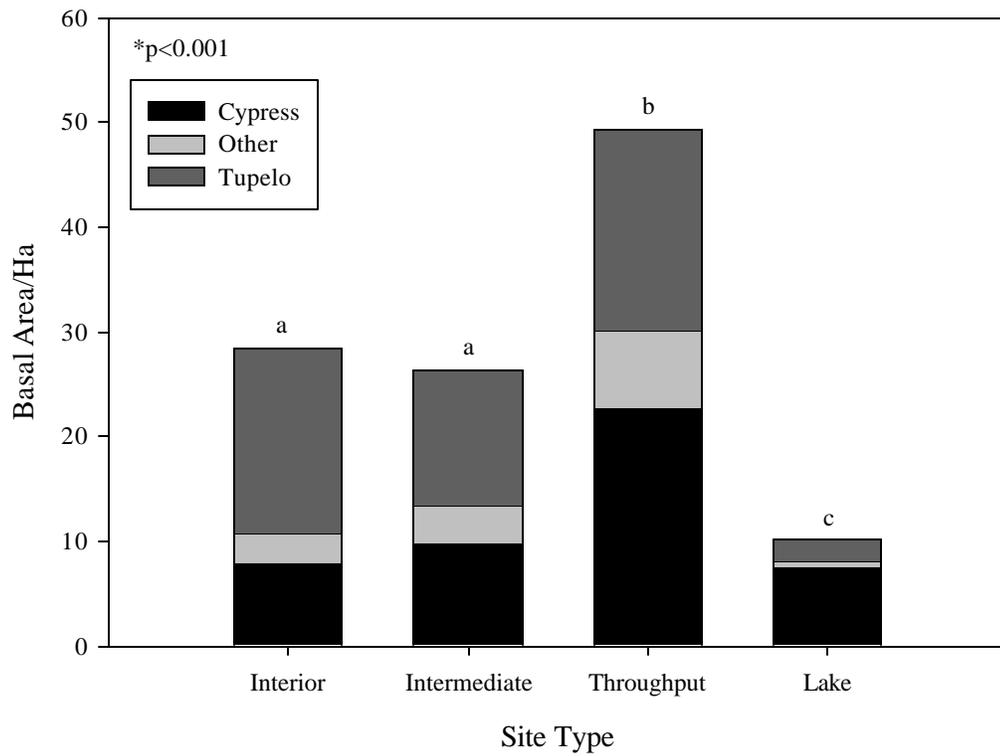


Figure 25. Basal Area/Ha is significantly different by site type and species. Bars with the same letters are not significantly different in a Bonferonni-adjusted LSD.

the Throughput sites (mean = $28.80 \pm 2.01 \text{ m}^2 \text{ ha}^{-1}$ and mean = $26.19 \pm 3.46 \text{ m}^2 \text{ ha}^{-1}$, respectively) and did not differ from one another. Basal areas were lowest at the Lake sites, averaging only $10.27 \pm 2.92 \text{ m}^2 \text{ ha}^{-1}$. *T. distichum* and *N. aquatica* were co-dominant at the Throughput and Intermediate sites, accounting for as much as 80-90% of the basal areas at these sites. *N. aquatica* dominated the basal areas of Interior sites, contributing roughly 60-65% of the basal areas at these sites, and interspersed with 25% of *T. distichum* and generally less than 10% 'Other' trees. Conversely, Lake site basal areas were dominated by roughly 75% *T. distichum*, followed by 15-20% other trees and roughly 5% *N. aquatica* (Figure 25).

Percent mortality

For all but the Lake sites, the highest percent mortality for tupelo and 'other' species occurred during 2002, two years following the severe drought that was accompanied by salt-water intrusions (Figure 26). Cypress only suffered substantial mortality at the lake sites (Figure 26). In general, most of the mortality occurred at the sites most closely adjacent to Pass Manchac and Lake Maurepas (Figure 27). Cumulatively (Figure 28) nearly 10 percent of all individuals alive at the beginning of the study are now dead.

Average mortality rate

Adjusting the overall percent mortality for the time span elapsed, yearly average mortality rates were at or below roughly 2% at Throughput, Interior, and Intermediate sites. Lake site trees were dying at a rate of roughly 10% per year, with mortality rates as high as 25% per year on Jones Island (Site 20; Figure 1) and accordingly lower at sites farther removed from Pass Manchac.

Percent Mortality by Species and Year (2000-2002)

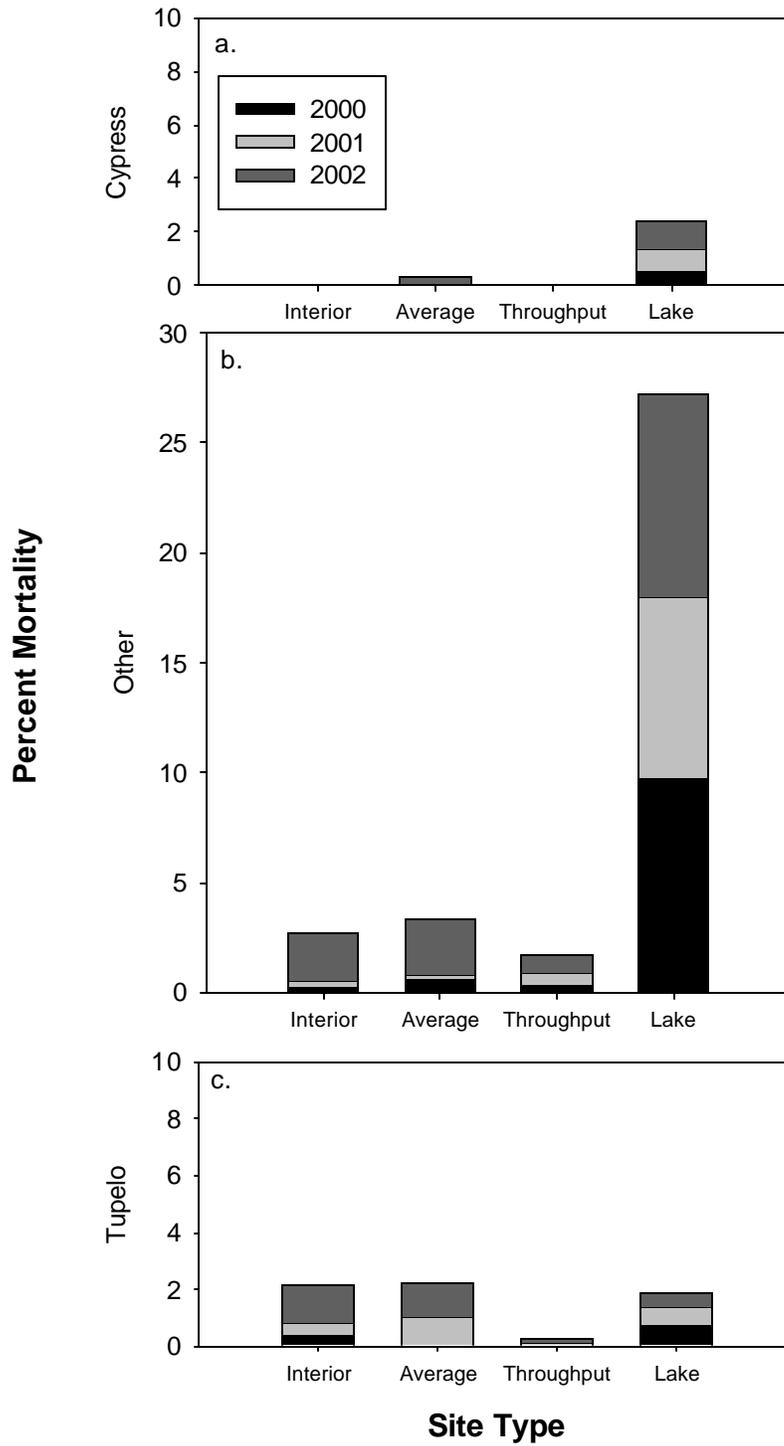


Figure 26. Percent mortality for (a) cypress (b) other and (c) tupelo by site type for 2000-2002. Mortality continued after the drought of 2000 and sustained into 2001-2002.

Total % Tree Mortality Since 2000 by Site Type and Species

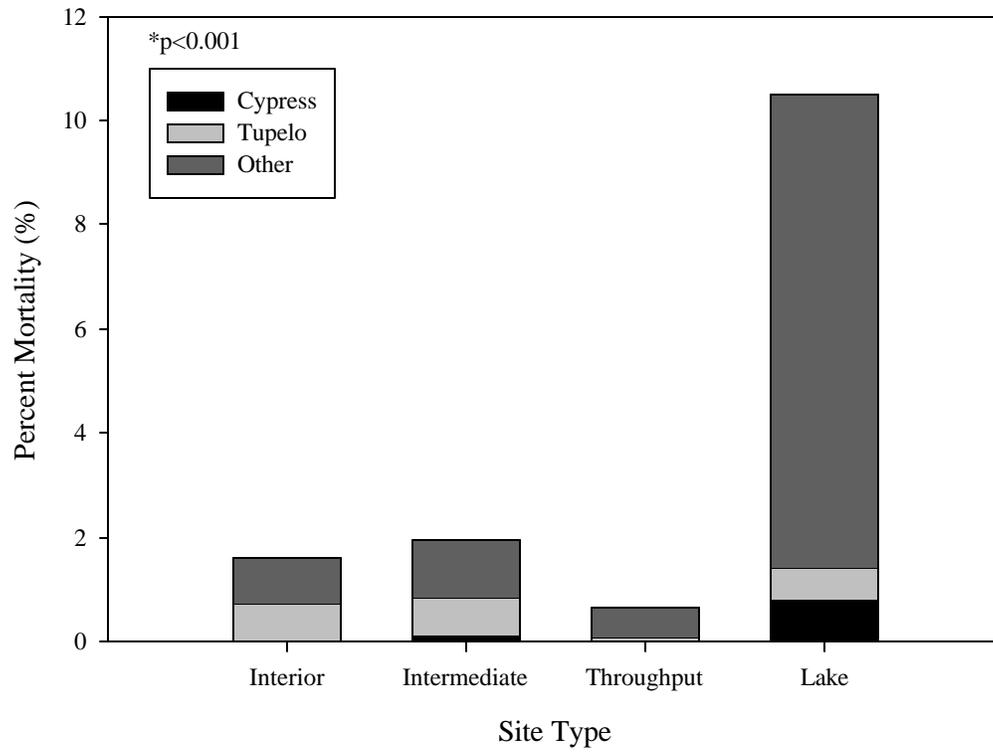


Figure 27. Percent mortality is significantly different between site types ($p<0.001$).

Cumulative Percent Tree Mortality 2000-2002 by Species

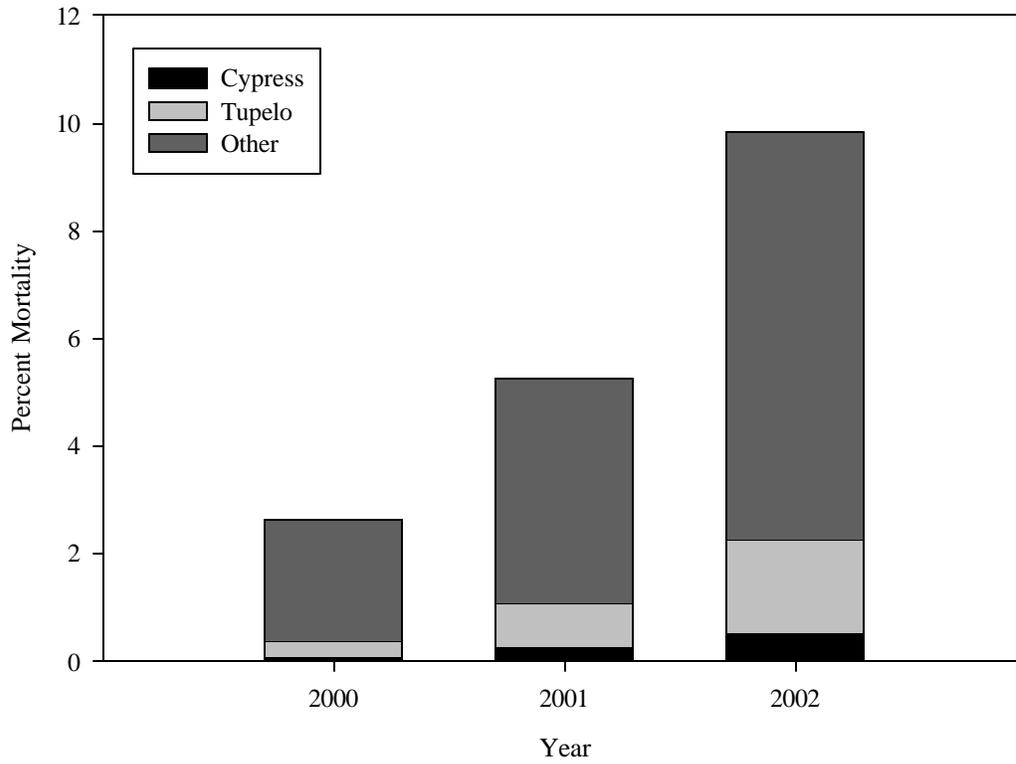


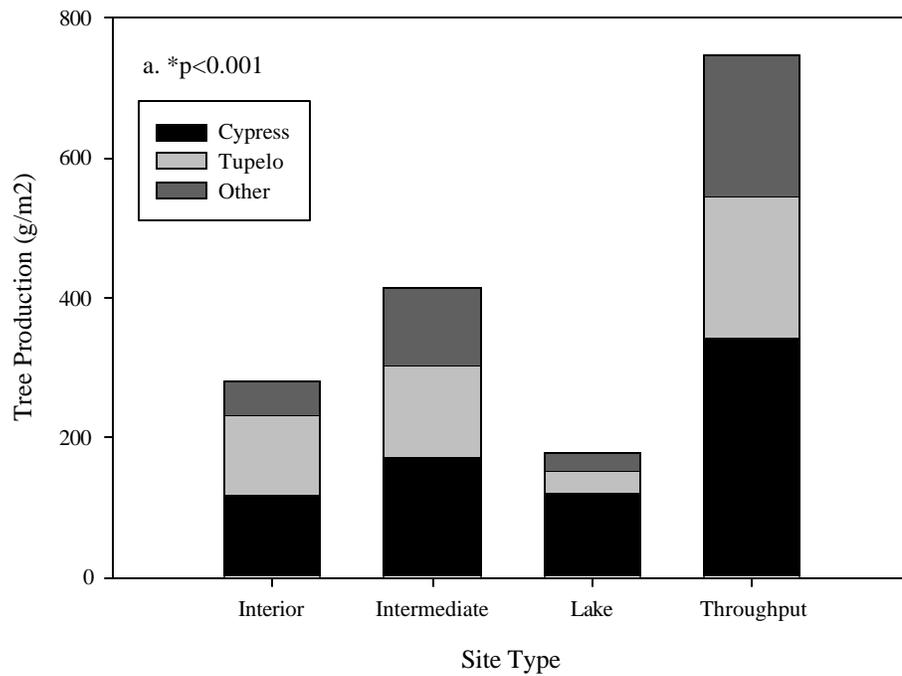
Figure 28. Cumulative Percent Mortality by Species and Year.

Primary Productivity of Trees

Tree Productivity

Total tree primary productivity differed significantly between site groupings ($F_{3,447} = 16.75, p < 0.000001$; [Figure 29](#)), years ($F_{1,447} = 20.62, p < 0.000007$; [Figure 29](#)), species ($F_{2,447} = 79.94, p < 0.000001$ [Figure 29](#)), and the biomass partitions wood and litter ($F_{1,447} = 78.66, p < 0.000001$; [Figure 30](#)). Bulk density and basal area were significant covariables in the model ($F_{1,447} = 4.58, p < 0.03$ and $F_{1,447} = 50.23, p < 0.000001$, respectively), indicating that tree primary productivity was higher at sites with higher bulk densities and greater basal areas. The highest rates of total tree primary productivity were found at the Throughput sites (mean = $752 \pm 78.5 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$; [Figure 29](#)) and during 2001 ([Figure 29](#)). Intermediate sites were the next most productive sites (mean = $410.0 \pm 27.2 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$) followed by Interior sites (mean = $286.7 \pm 28.2 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$) and Lake sites were significantly less productive than Intermediate sites (and mean = $184.7 \pm 62.3 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$; [Figure 29](#)). Tree primary productivity displayed the same pattern of production through time across site types ([Figure 31](#)) and was lowest during 2002, followed by the drought year of 2000 ([Figure 29b](#)). An interaction between group and year effects ($F_{3,447} = 2.94, p < 0.03$) showed that total tree primary productivity was relatively unaffected by the drought at Interior sites ([Figure 31](#)). Almost certainly, leaf litter was lost from litterfall traps during the tropical storms of 2002, as the ratio of leaf litter to wood production was higher considerably for 2000 and 2001 ([Figure 30](#)). Wood production did decrease from 2000 to 2002, but not nearly as much as the sharp decrease in litter during 2002, again probably an artifact of loss during the two tropical storms. Using the average ratio of litterfall to wood production for 2000 and 2001 (ratio = 1.55), we may have lost roughly $72 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ of leaf litter from the litterfall traps during 2002. This estimates total wood production during 2002 to be roughly $330 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, still considerably below that of the other two years. We believe the reduction was attributable to ten weeks of continuous high water during late summer and fall of 2002.

Total Tree Production by Site Type (averaged 2000-2002)



Total Tree Production (g/m²) by Year

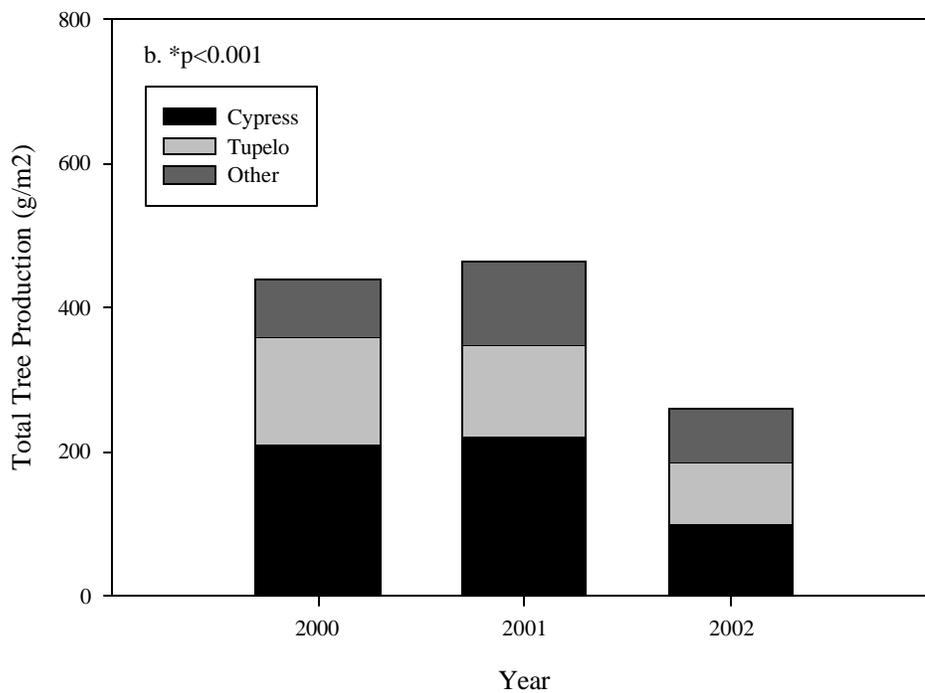


Figure 29. Total tree production (g/m^2) for (a) site types and (b) years for cypress, tupelo and other.

Litter vs. Wood Production for 2000-2002

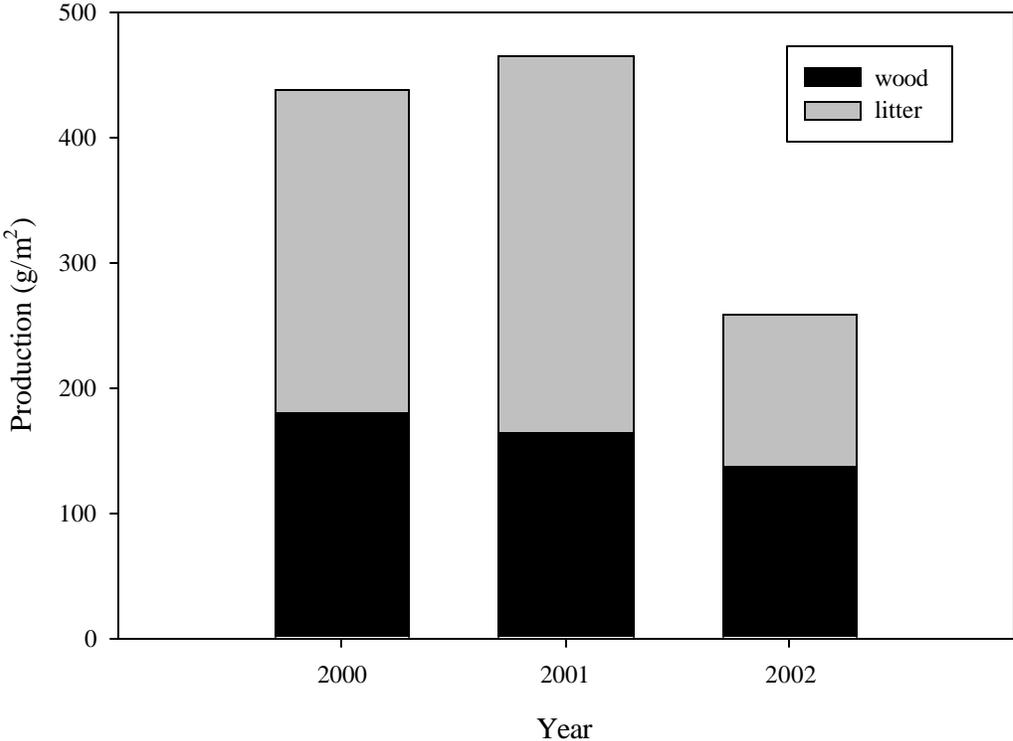


Figure 30. Litter-to-wood production ratio is reduced in the year 2002, indicating loss of litter during the hurricanes and floods of Fall 2002.

Total Tree Production (g/m²) by Site Type and Year

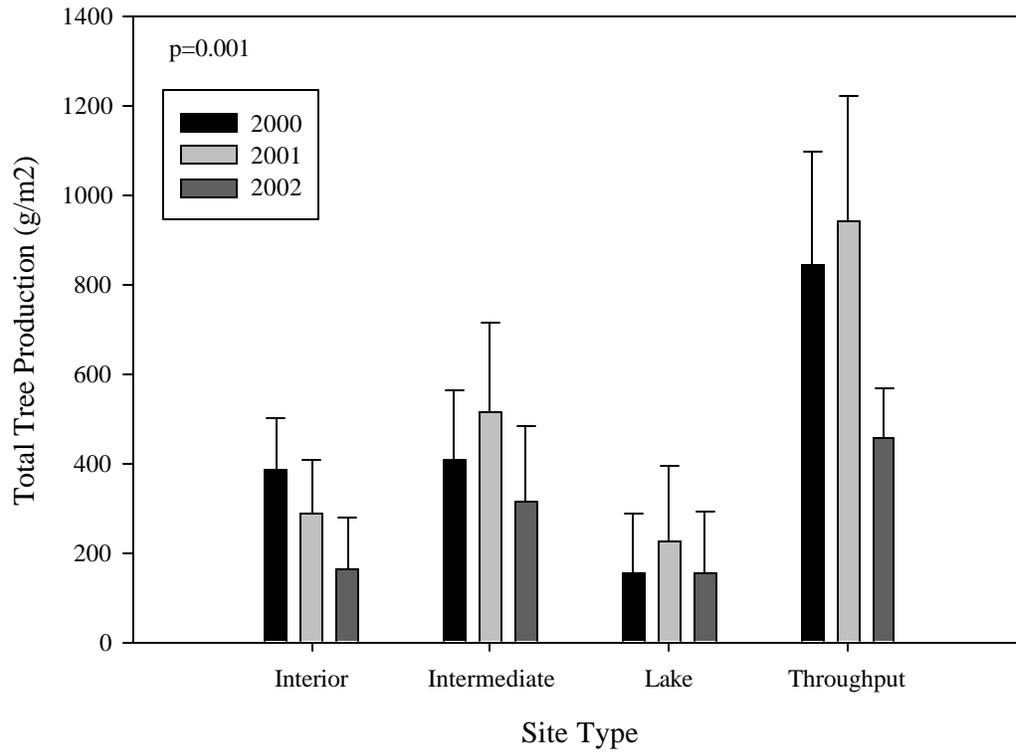


Figure 31. Tree primary productivity displayed the same pattern of production through time across site types.

Of the tree species categories examined, *T. distichum* was by far the most productive species in the study area (Figure 29). *N. aquatica* and 'Other' trees were overall similarly productive, though *N. aquatica* were slightly but significantly more productive than 'Other' trees. *T. distichum* alone accounted for nearly 50% of all tree primary production throughout the study. *N. aquatica* at Interior sites produced significantly more biomass than expected, as indicated by the interaction between site grouping and species ($F_{6,447} = 12.19$, $p < 0.000001$; Figure 31a).

After re-analyzing the total tree productivity data (summed over species and biomass partitions) in a site-specific model, polynomial contrasts were calculated to investigate specific spatial trends of productivity changes between four sites at a time. Tree primary productivity was found to increase significantly with increasing distance from Lake Maurepas into the interior swamp (site 8 < site 10 < site 11 < site 12; $F_{1,59} = 145.34$, $p < 0.000001$; Figure 32). Conversely, increasing proximity to Pass Manchac along the southern shore of Lake Maurepas further decreased tree primary productivity (site 17 > site 18 > site 19 > site 20; $F_{1,59} = 12.19$, $p = 0.00092$; Figure 33). Along Blind River and within the impact zone of the Amite Flood-relief Diversion Canal, only site 5 showed increased tree primary productivity ($F_{1,59} = 45.20$, $p < 0.000001$; Figure 34). Similarly, though opposite, among the sites located along Reserve Relief Canal, only site 17, located closest to the lake, showed a significant decrease in tree primary productivity ($F_{1,59} = 15.95$, $p = 0.00018$; Figure 35).

Effect of Distance from Lake Maurepas on Tree Primary Production

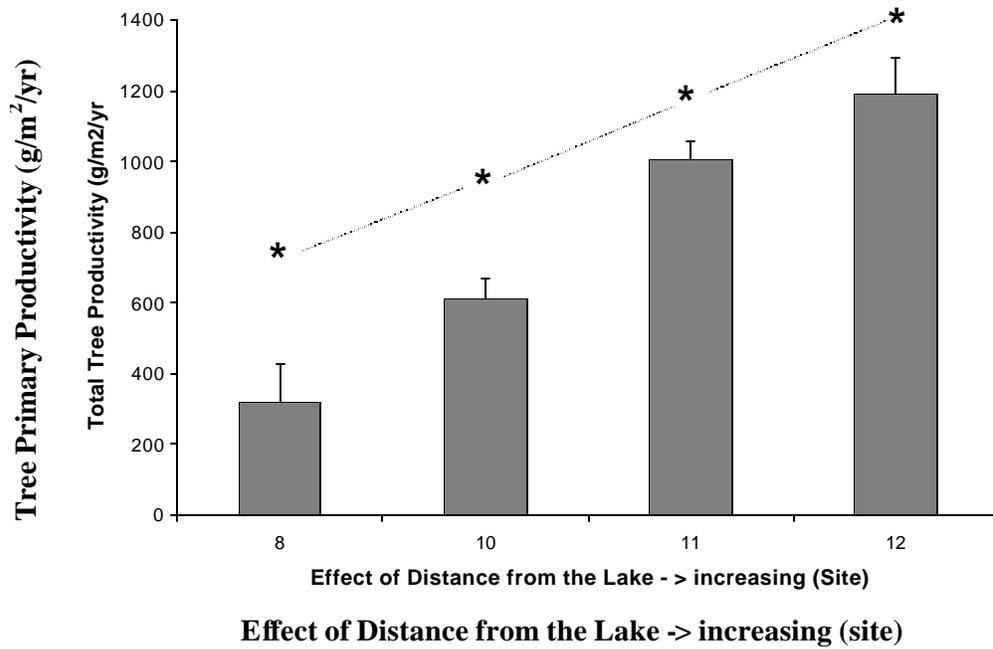


Figure 32. The effect of distance from Lake Maurepas on total tree production (mean \pm standard error). Orthogonal polynomial contrast $F_{1,59} = 145.34, p < 0.000001$. Site 12 is the Throughput site farthest removed from Lake Maurepas, Site 8 is an Intermediate site relatively close to the Lake (see Figure 2).

Effect of Distance from Pass Manchac on Tree Primary Production

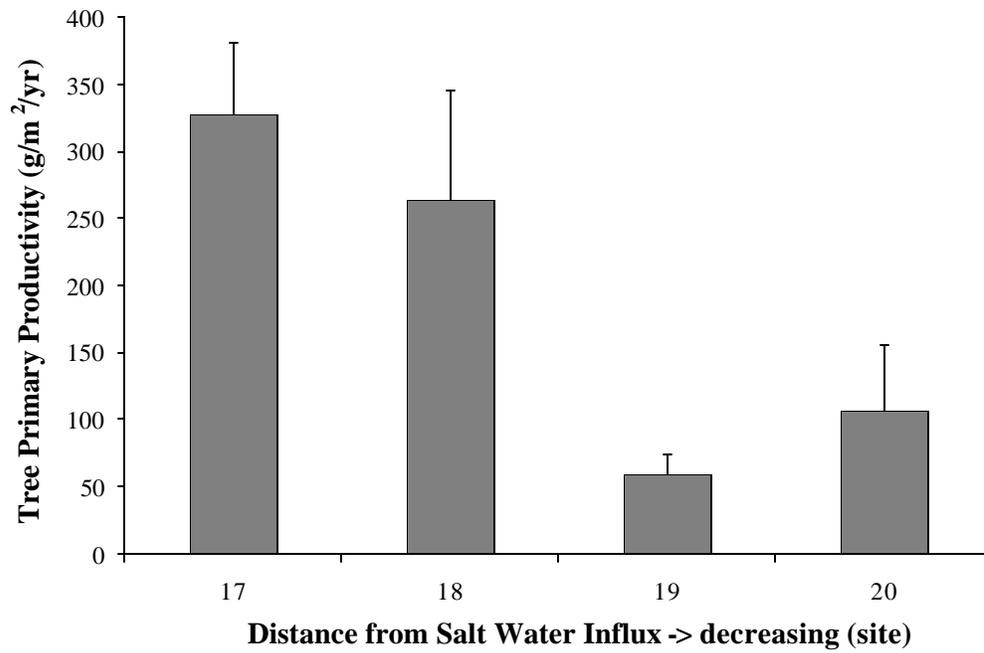


Figure 33. The effect of distance from salt water influx through Pass Manchac on total tree production (mean \pm standard error). Orthogonal polynomial contrast $F_{1,59} = 12.19$, $p < 0.00092$. Sites 17 is the Lake site farthest removed from Pass Manchac, site 20 is closest (see Figure 2).

Effect of Amite Flood-Relief Diversion Canal Total Tree Production

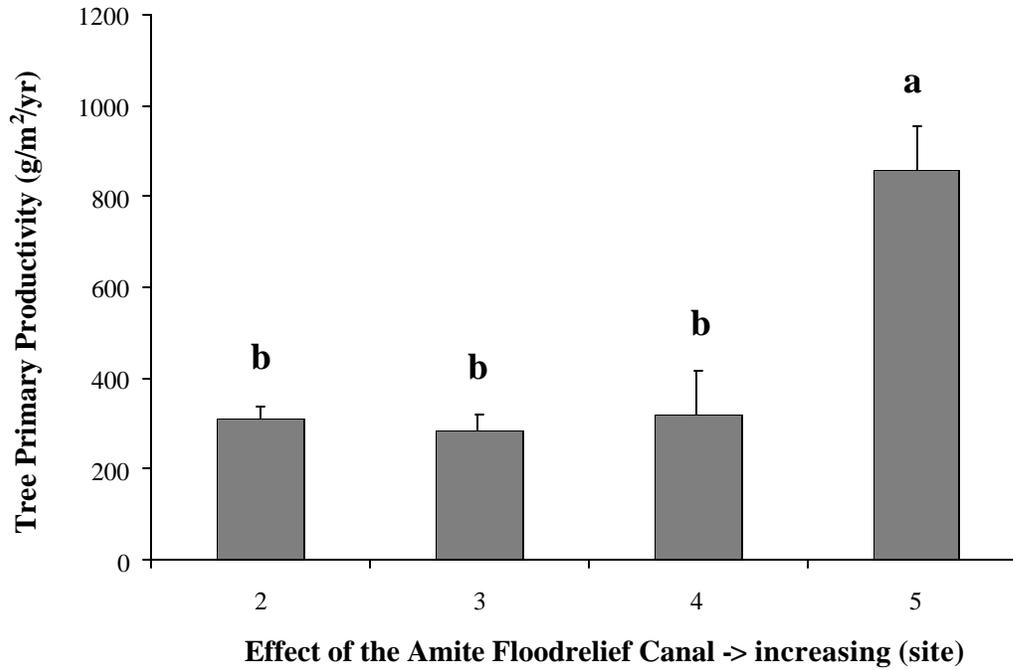


Figure 34. The effect of the Amite Flood-relief Diversion Canal on total tree production (mean \pm standard error). Linear contrast $F_{1,59} = 45.20$, $p < 0.000001$. Site 5 (Figure 2) has significantly higher tree primary productivity than any other site along Blind River.

Effect of Distance from Reserve Relief Canal on Tree Production

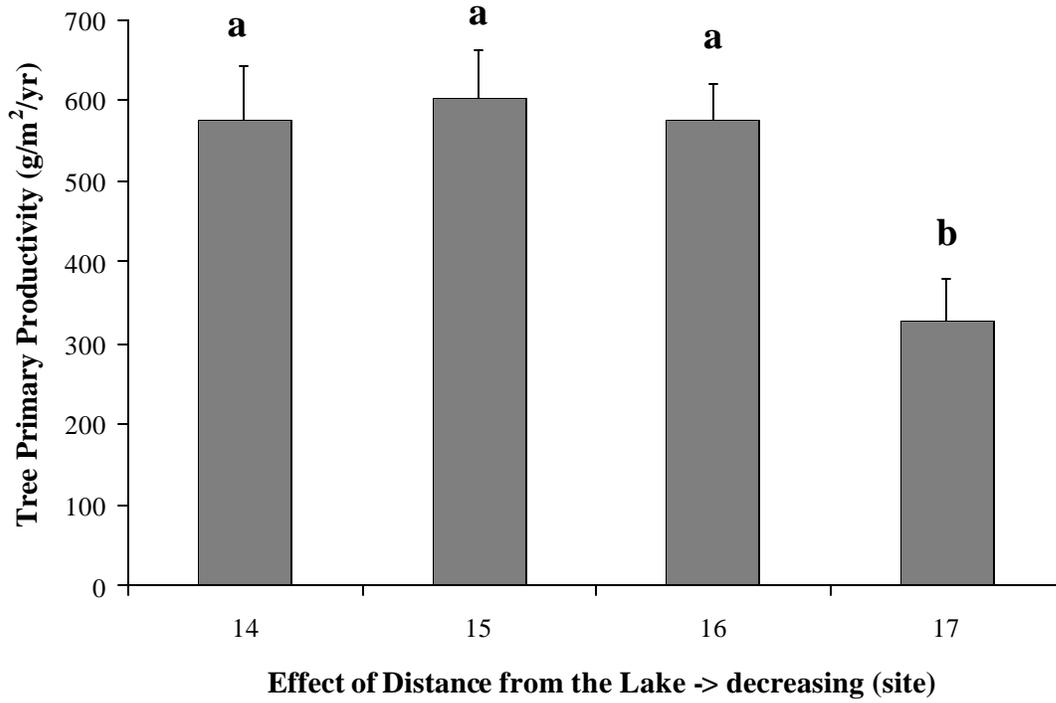


Figure 35. The effect of distance from Lake Maurepas on the Reserve Relief canal on total tree primary production (mean \pm standard error). Linear contrast $F_{1,59} = 15.95$, $p = 0.00018$. All sites along Reserve Relief canal are equally productive with the exception of site 17, which is closest to the Lake. Bars that share letters are not significantly different according to Bonferroni-adjusted multiple comparison tests.

Total Site Primary Production

Combining the total tree biomass production with herbaceous biomass production, Throughput sites are more productive than any other sites, and Lake sites are the least productive (Figure 36). Overall, most of the biomass produced during the year 2000 and 2001 was leaf litter from trees. Wood and herbaceous biomass produced were similar. An interaction between biomass type and group ($F_{6,106} = 25.44$, $p < 0.000001$; Figure 37) showed that although biomass production generally was dominated by litter, herbaceous biomass production was relatively high at Interior sites, and exceeded tree production at Lake sites during the year 2000 and 2002 (Figure 37). During fall of 2002, a large bloom of the exotic floating aquatic *Salvinia molesta* occurred at most of sites (Figure 37c). Presumably, this bloom was triggered by the nutrient- and sediment-laden waters churned up by the tropical storms of 2002.

Ordination of Species Occurrence

The common tree and herbaceous species found in the Maurepas swamp could be arranged along three major axes of environmental factors. Axis one (Figure 38) of the canonical correspondence analysis accounted for 18.6% of the total variance in species occurrence, axis two (Figure 39) accounted for another 11.4%, and axis three explained only 3.7%. Nitrate concentration ($r = -0.869$), well salinity ($r = -0.708$), and light ($r = -0.758$) correlated strongly with axis one, while axis two correlated positively with bulk density ($r = 0.749$). None of the environmental variables used in the analysis loaded on the third axes, which was thus non-interpretable. The trees most frequently found under low (ground-level) light, nitrate, and well salinity conditions were *Quercus obtusa* and *Fraxinus pennsylvanica*, although *Acer rubrum*, *Cephalanthus occidentalis*, *Nyssa sylvatica*, *Nyssa aquatica* and *Taxodium distichum* were usually found under similar conditions. Common herbaceous plants under these conditions

Corrected Total Site Production by Year and Site Type

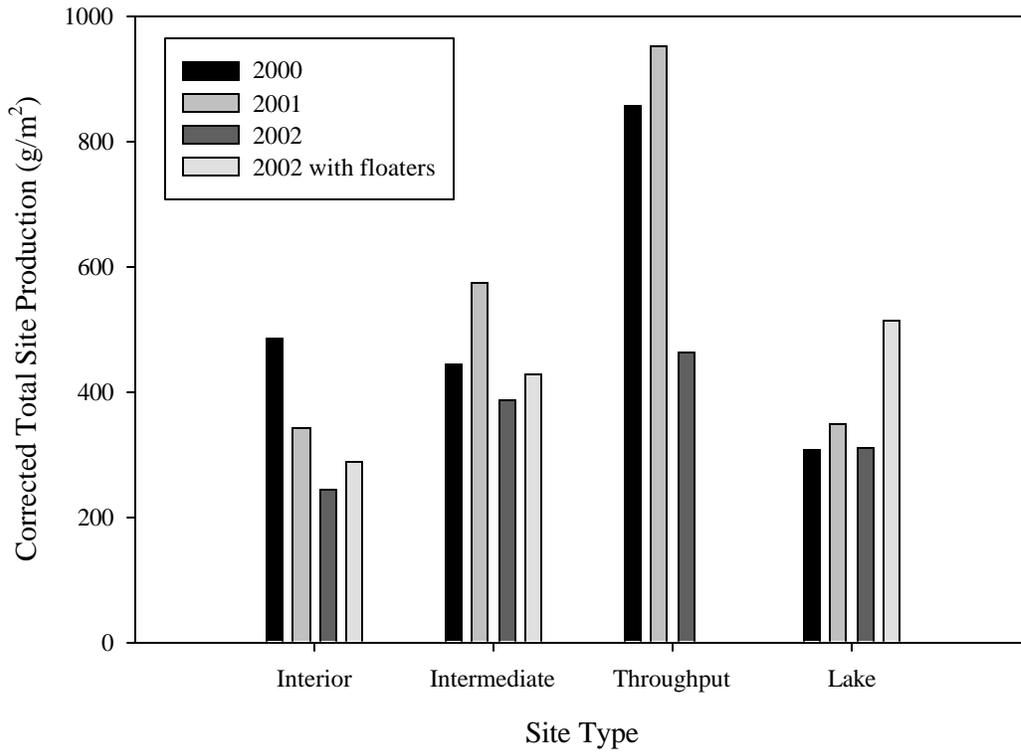


Figure 36. Total Site Production by Year and Site Type, including floating herbaceous communities dominated by *Salvinia*.

Total Site Production (g/m^2) by Site Type and Production Type

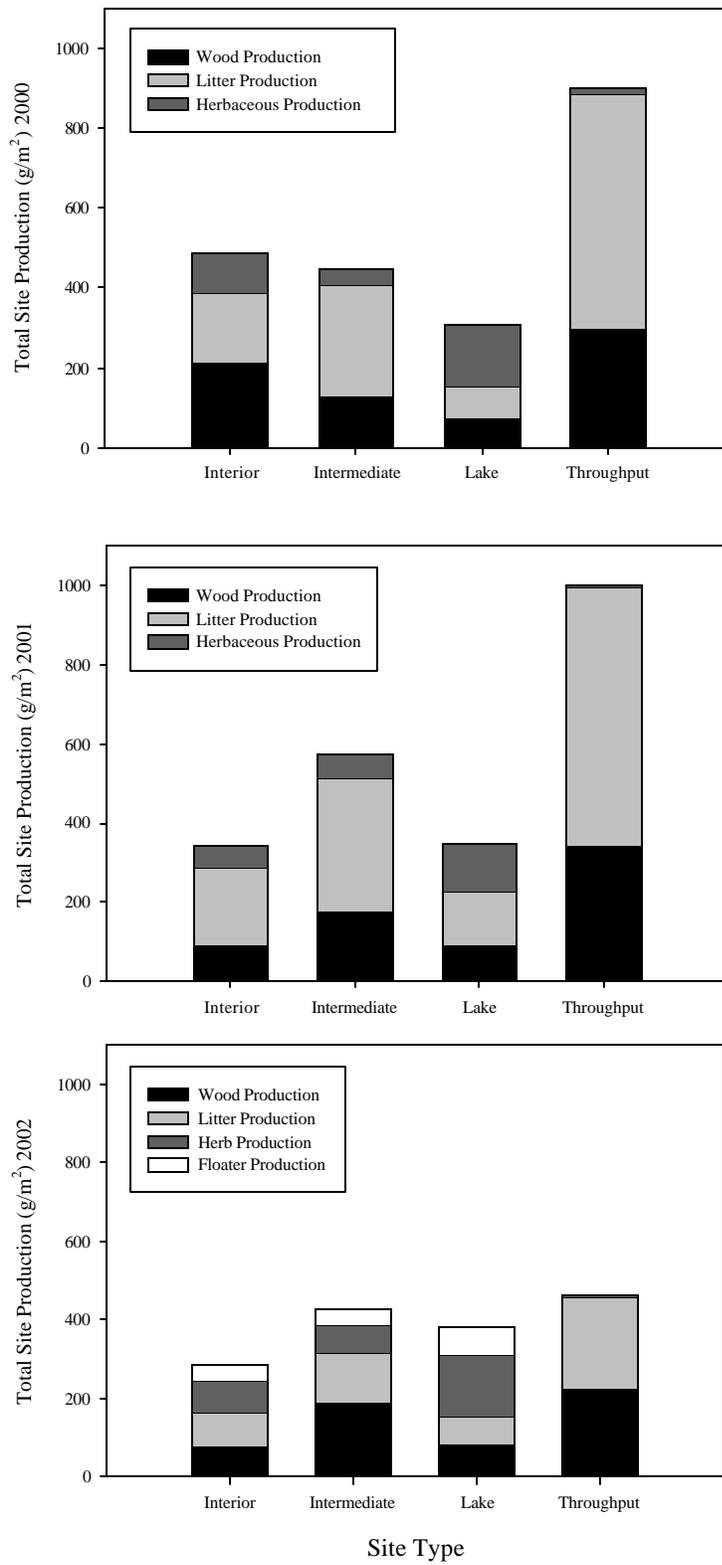


Figure 37. Total Site Production by Year, Site Type and Type of Biomass.

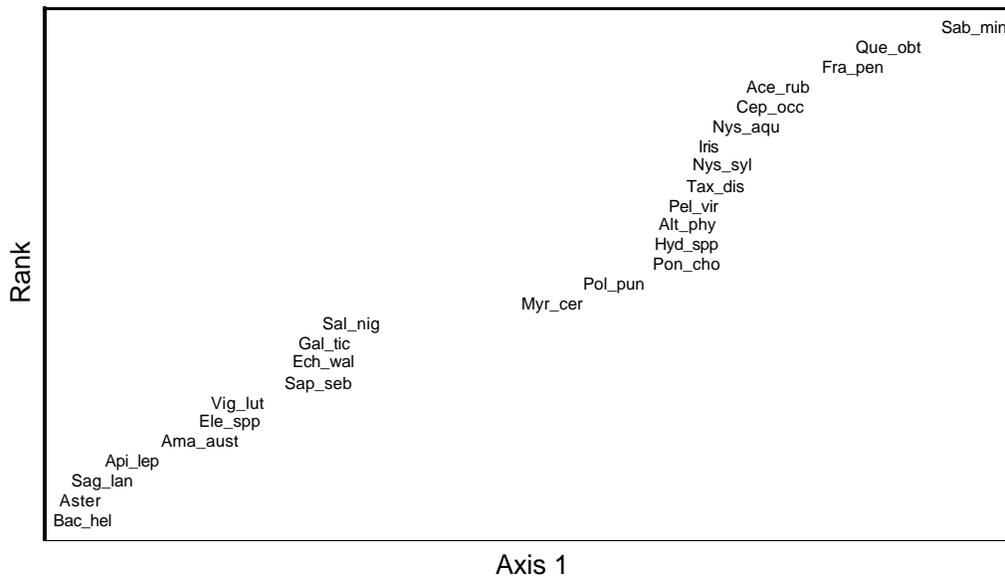


Figure 38. Results of the Canonical Correspondence Analysis of 16 herbaceous and 10 tree species occurrences in 2001 with four environmental variables. Axis one correlates negatively with nitrate concentrations ($r = -0.869$, min. = 0.1 ppm, max. > 250 ppm), well salinity ($r = -0.708$, min. = 0.3 ppt, max. = 3.22 ppt), and the amount of available light at ground level ($r = -0.758$, min. = 1%, max. = 96%). Species abbreviations are as follows:

Ace_rub	<i>Acer rubrum</i> var. <i>drummondii</i>
Alt_phi	<i>Alternanthera philoxeroides</i>
Ama_aus	<i>Amaranthus australis</i>
Api_lep	<i>Apium lepiphyllum</i>
Aster	<i>Aster</i> spp.
Bac_hel	<i>Baccharis helimifolia</i>
Cep_occ	<i>Cephalanthus occidentalis</i>
Ech_wal	<i>Echinochloa walterii</i>
Ele_spp	<i>Eleocharis</i> spp.
Fra_pen	<i>Fraxinus pennsylvanica</i>
Gal_tic	<i>Galium tictorium</i>
Hyd_spp	<i>Hydrocotyle</i> spp.
Iris	<i>Iris virginica</i>
Myr_cer	<i>Myrica cerifera</i>
Nys_aqu	<i>Nyssa aquatica</i>
Nys_syl	<i>Nyssa sylvatica</i> var. <i>biflora</i>
Pel_vir	<i>Peltandra virginica</i>
Pol_pun	<i>Polygonum punctatum</i>
Pon_cho	<i>Pontedaria chordata</i>
Que_obt	<i>Quercus obtusa</i>
Sab_min	<i>Sable minor</i>
Sag_lan	<i>Sagittaria lancifolia</i>
Sal_nig	<i>Salix nigra</i>
Sap_seb	<i>Sapium sebiferum</i>
Tax_dis	<i>Taxodium distichum</i>
Vig_lut	<i>Vigna luteola</i>

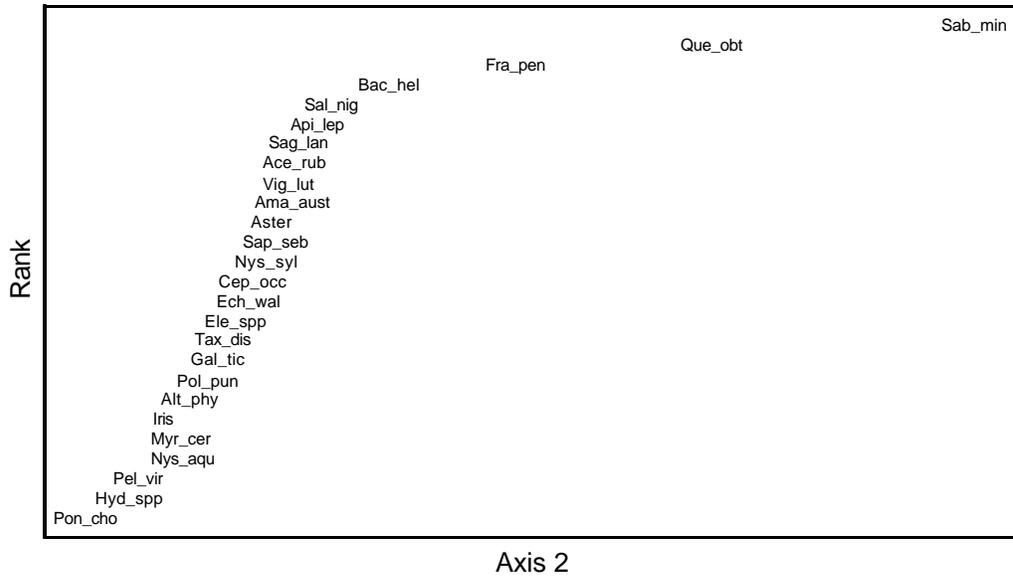


Figure 39. Results of the Canonical Correspondence Analysis of 16 herbaceous and 10 tree species occurrences in 2001 with four environmental variables. Axis two correlates positively with bulk density ($r = 0.749$, min. = 0.04 g/cm^3 , max. = 0.29 g/cm^3). Species abbreviations are as follows:

Ace_rub	<i>Acer rubrum</i> var. <i>drummondii</i>
Alt_phi	<i>Alternanthera philoxeroides</i>
Ama_aus	<i>Amaranthus australis</i>
Api_lep	<i>Apium lepiphyllum</i>
Aster	<i>Aster</i> spp.
Bac_hel	<i>Baccharis helimifolia</i>
Cep_occ	<i>Cephalanthus occidentalis</i>
Ech_wal	<i>Echinochloa walterii</i>
Ele_spp	<i>Eleocharis</i> spp.
Fra_pen	<i>Fraxinus pennsylvanica</i>
Gal_tic	<i>Galium tictorium</i>
Hyd_spp	<i>Hydrocotyle</i> spp.
Iris	<i>Iris virginica</i>
Myr_cer	<i>Myrica cerifera</i>
Nys_aqu	<i>Nyssa aquatica</i>
Nys_syl	<i>Nyssa sylvatica</i> var. <i>biflora</i>
Pel_vir	<i>Peltandra virginica</i>
Pol_pun	<i>Polygonum punctatum</i>
Pon_cho	<i>Pontedaria chordata</i>
Que_obt	<i>Quercus obtusa</i>
Sab_min	<i>Sable minor</i>
Sag_lan	<i>Sagittaria lancifolia</i>
Sal_nig	<i>Salix nigra</i>
Sap_seb	<i>Sapium sebiferum</i>
Tax_dis	<i>Taxodium distichum</i>
Vig_lut	<i>Vigna luteola</i>

included *Sabel minor*, *Iris virginica*, *Peltandra virginica*, *Alternanthera philoxeroides*, *Hydrocotyle* sp., *Pontedaria chordata*, and *Polygonum punctatum*. Increasing levels of salinity, nitrate and a corresponding open canopy favored shrubby trees such as *Myrica cerifera* and *Salix nigra* to establish as other trees became less abundant. Under these higher salinity, nitrate and light conditions, herbaceous plants commonly found were *Eleocharis* sp., *Amaranthus australis*, *Apium leptophyllum*, *Sagittaria lancifolia*, *Aster* sp. and, at the most saline end of the gradient, *Baccharis helimifolia*. When arranged along a bulk density gradient, *F. pennsylvanica* and *Q. obtusa* occurred more frequently in areas with higher bulk densities. Herbaceous species found under high bulk density conditions included *S. minor* and *B. helimifolia*. Herbaceous plants and trees were rather stacked at the low end of the bulk density gradient. However, of the trees *M. cerifera* and *N. aquatica* tend to occur with relatively great frequency at the low end of the bulk density gradient. Herbaceous plants found at low bulk densities include *P. virginica*, *Hydrocotyl* sp., and *P. chordata*.

DISCUSSION

Flooding has been reported to have doubled in the Manchac Wildlife Management Area adjacent to the Maurepas swamp since 1955 due to sea-level rise and subsidence (Thomson, et al. 2002). This trend has also occurred in the Maurepas swamps and is expected to be even greater because the elevations of the various swamp areas are lower (Shaffer, unpubl. data). Currently the Maurepas swamps are often lower in elevation than the Lake, rendering flooding semi-permanent. Furthermore, flood control levees and abandoned raised railroad tracks have impounded much of the remaining swamps, causing throughput to be low. These swamps have been cut off from the sustaining, spring floods of the Mississippi River for over a century and are in varying states of decline. Until this study was undertaken, the decline was evidenced by qualitative information such as dead and dying canopies of the predominant tupelogram trees. We now have quantitative information that allows us to compute the likely benefits of a future

with a diversion into Hope Canal in comparison to the continued demise of the swamp ecosystem in a future without such a project.

The Maurepas swamps are characterized by nutrient poor waters (see Day et al. 2000, Attachment F), soils of extremely low strength indicative of stress (see herein), and salt-water intrusions that occur during the late summer and fall seasons. The mean salinity of the lake water measured at the Manchac bridge also has increased gradually since 1951 (Thomson et al., 2002). Severe increases in salinity, like those experienced during the droughts in 1999 and 2000, may be prevented or greatly ameliorated by the increased fresh water throughput that the proposed diversion would bring. It is likely that the influences of freshening would be felt in areas as remotely located as Jones Island and the Manchac landbridge, because the proposed diversion could replace all of the water in Lake Maurepas roughly twice each year. Pass Manchac and North Pass (adjacent to Jones Island and Manchac) are the only two direct conduits that will allow the additional fresh water to eventually reach Lake Pontchartrain.

The soil characteristics at the majority of the study sites are indicative of a lack of riverine influence (lack of sediment input and throughput) as evidenced by high soil organic matter content and low bulk density values (DeLaune et al., 1979; Hatton, 1981; Messina and Conner, 1998). With the exception of Throughput sites (sites influenced by the present Amite River Diversion Canal or Hope Canal), soil bulk density values are low, and in the range of those typically found in fresh and intermediate marshes (Hatton, 1981). Mineral sediment input is only apparent at the Throughput sites where soil bulk density values are the highest recorded in this study (approximately 0.22 g cm^{-3}). Correspondingly, soil organic matter content was lower at the Throughput sites (approximately 48%) and Intermediate sites (44%), compared to 63% at the Lake sites and to 74% at the Interior sites. Together, these soil values are very representative of non-forested, herbaceous fresh and intermediate wetlands that are located interior of potential streamside hydrology effects (Hatton, 1981). This agrees with the observed mortality of many trees and the conversion to a more herbaceous plant community in the Lake sites, as discussed below. The Interior sites had the highest percent organic matter and the

lowest soil bulk density values recorded in the study ($\sim 0.07 \text{ g cm}^{-3}$), which again is indicative of hydrologic isolation and a lack of sediment input.

The pH values measured throughout the Maurepas swamp are slightly acidic and thus indicative of organic soils. Nutrient levels in soil water (above) and surface water (Day et al. 2001) indicated that the system is nutrient poor. Nitrate levels were less than 1% of those found in Mississippi River water (Day et al. 2001). Ammonium levels were also lower than those in the Mississippi River, but not as dramatically. Total nitrogen in the Maurepas swamp was almost comparable to total nitrogen levels in river water, which indicates the presence of high concentrations of organic nitrogen in the Maurepas swamp (Day et al. 2001). Phosphate and total phosphorous levels in the swamp were comparable to those found in the river (Day et al. 2001). Surface water ratios of nitrate/ammonium to phosphate were close to 2:1, an indication that the Maurepas swamp is definitely nitrogen limited (according to the Redfield Ratio of 16:1; Mitsch and Gosselink 2000).

The difference in forest structure among different areas in the Maurepas swamp is also an indication of the health and future of these sites. Overall, the overstory is dominated by either *N. aquatica*, *T. distichum* or both, while the midstory is largely dominated by large numbers of smaller *A. rubrum* and *F. pennsylvanica*, both of which are more shade tolerant than either of the dominants (Fowells 1965). Similar observations have been made in comparable swamps in the Barataria Basin (Conner and Day 1976). *M. cerifera*, *S. sebiferum*, and *S. nigra* dominate the mid-story in areas of disturbance that were characterized by more open canopies and measurable salt-water intrusion effects. Shrub-scrub habitats are often observed on the transitional edges between marshes and forested wetlands or uplands (White 1983, Barras et al. 1994). *Q. obtusa* and *F. pennsylvanica* were found in greater abundance at sites characterized by higher bulk densities, which were indicative of increased throughput and generally less flooding. These observations support similar findings from wetland plant ordinations by White (1983) in the Pearl River, Louisiana, and Rheinhardt (1998) in the forested riverine wetlands of the inner coastal plain of North Carolina. As *Sapium sebiferum* has been found to be more

shade, flooding and salt tolerant (Jones et al. 1989, Conner and Askew 1993) than several other, native wetland tree species, this invasive tree may become more dominant in the coastal wetlands of the southeastern United States (Conner and Askew 1993).

Stem densities at Throughput sites are similar to densities reported for impounded (Conner et al. 1981, Conner and Day 1992) or continuously flooded (Dicke and Toliver 1990) swamps throughout Louisiana, whereas stem densities at Interior and Intermediate sites are less than those reported for impounded swamps (Table 2). Average stem densities at Lake sites are not even half of those reported for impounded swamp sites, most likely due to the fact that neither *A. rubrum*, *N. aquatica*, nor *F. pennsylvanica* have salt tolerances that could withstand the chronic salinity conditions of 2-5 ppt found at these sites. Conner et al. (1997) and Pezeshki et al. (1989) reported that *N. aquatica*, *F. pennsylvanica*, and *A. rubrum* showed signs of stress and reduced growth even at salinities as low as 2-3 ppt. *N. sylvatica* var. *biflora* seedlings experienced 100% mortality when exposed to chronic flooding with 2 ppt (McCarron et al. 1998). Likewise, the lower stem densities observed at the Interior swamp sites are primarily the result of the decreased abundance of *F. pennsylvanica* and *A. rubrum* in the impounded and stagnant hydrologic regimes characteristic of these sites.

Similar to stem densities, basal areas for Throughput sites in the Maurepas swamp compare favorably with basal areas of the most productive, naturally flooded cypress-tupelo swamps (Conner et al. 1981, Dicke and Toliver 1990, Conner and Day 1992), while Interior and Intermediate sites have basal areas in the range of sewage enriched pure cypress stands (Nessel et al. 1978) and impounded swamps (Conner et al. 1981, Conner and Day 1992; Table 2). Again, average Lake site basal areas are below any reported in the literature, although these areas were largely forested as recently as the late 1950s (Barras et al. 1994). Integrating the information from the size class distributions of the dominant tree species at the Maurepas swamp sites, it becomes apparent that the recruitment of small trees is minimal throughout the swamp, an effect that is especially pronounced at Interior and Lake sites. Even at the healthiest

throughput sites, the recruitment of young *N. aquatica* and *T. distichum* is extremely limited relative to older size classes.

In terms of above-ground net primary productivity, only the most productive sites of the Maurepas swamp compare well with natural, periodically flooded cypress-tupelo swamps (Carter et al. 1973, Conner and Day 1976, Conner et al. 1981, Megonigal et al. 1997) - and then only during years of normal precipitation. The vast majority of the Maurepas swamp, including Interior, Intermediate and Lake sites, range in total productivity (including herbaceous productivity) between swamps that have been identified as either nutrient-poor and stagnant (Schlesinger 1978), just stagnant (Taylor 1985, Mitsch et al. 1991), or near-continuously flooded cypress swamps (Megonigal et al. 1997). Alligator Island (site 5, Figure 1) had higher tree primary production than comparable sites along the Blind River, again an indication that this site receives freshwater, nutrient, and/or sediment subsidies from the Amite Flood-relief Diversion Canal. Spatial patterns of differences in productivity among individual sites within the study area are remarkably comparable to the spatial patterns of mean yearly salinity, indicating that salt stress (Pezeshki et al. 1989, Conner et al. 1997, McCarron et al. 1998) is a major factor in influencing tree productivity at all but the swamp Interior sites. The increasing proportions of herbaceous productivity at Interior sites and more emphatically at Lake sites are indications that these forested wetlands are converting to marshes (Barras et al. 1994).

Because the severe drought during the year 2000 was accompanied by relatively low water stages throughout the Maurepas swamp, primary production may have been considerably higher than normal during the first half of the growing season. Indeed, the soils were sufficiently less flooded (i.e., less reduced) at this time as evidenced by the relatively high soil redox potentials measured and the lack of detectable sulfides at all sites. Contrarily, the late-season primary productivity was most likely greatly suppressed as salt water penetrated deep into the Interior sites. As a result, tree primary productivity throughout the Maurepas swamp was reduced by roughly 20-40% during the drought year when compared to a year of normal precipitation. Tree primary productivity at the Interior sites was not as strongly impacted by the

drought as other sites, however, and remained low throughout the study period. This is an indication that these sites are chronically stressed, most likely by near-constant, stagnant flooding.

Overall, *T. distichum* was the most productive species in the Maurepas, while *N. aquatica* was the second most productive. This finding agrees with the observation that these two species are the canopy dominants, make up the majority of the basal area found at each site, and are the most flood-tolerant tree species in this system (Hook 1984). Furthermore, the higher biomass production of *T. distichum* also agrees with several studies that reported *T. distichum* seedlings to be more tolerant of low salinity and permanent flooding than *N. aquatica*, *A. rubrum*, and *F. pennsylvanica* (Dickson and Broyer 1972, Pezeshki et al. 1989, Keeland and Sharitz 1995, Conner et al. 1997). Only at Interior swamp sites was *N. aquatica* more productive than *T. distichum*, although this observation is likely due to the greater abundance of *N. aquatica* in these areas.

A larger proportion of biomass production was generally allocated to leaf litter than to stem wood production, a finding that is in agreement with the biomass values reported by Megonigal et al. (1997) and Conner and Day (1976) for forested wetlands under a variety of hydrological regimes. Other studies, however, found that biomass was allocated to a larger part into stem growth (Carter et al. 1973, Schlesinger 1978, Conner et al. 1981, Mitsch et al. 1991). Decreased biomass allocation into leaf litter has been noted as a reliable measure for detecting stress in forests (Brown 1981, Megonigal et al. 1997), as reductions in photosynthetic capacity are manifested in lower leaf biomass ratios rather than lower rates of photosynthesis per se. Further greenhouse and field investigations have shown that continuously flooded *T. distichum* saplings allocated more carbon to boles (Megonigal and Day 1992), while mature *T. distichum* had a greater proportion of wood production under continuously flooded conditions (Young et al. 1995). The overall low tree biomass production at Interior sites, the relatively low allocation of biomass into leaf litter at these sites, and for *N. aquatica* throughout the study, indicate chronic flooding stress. In addition, we speculate that the relatively low ratio of leaf litter to

wood production during 2002 may be a result of export during the two tropical storms. An alternative explanation is that the trees were under prolonged (10 continuous weeks) of high water during the late summer and fall of 2002 and this additional stress may have triggered a decrease in leaf production relative to wood production.

All of these observations indicate that the majority of the Maurepas swamps may be relic stands. They are continuously flooded and largely impounded, which prevents seed germination and recruitment of *T. distichum* and *N. aquatica* (DuBarry 1963, Harms 1973, Conner and Day 1976, Williston et al. 1980, Conner and Day 1988, Myers et al. 1995, Souther and Shaffer 2000) and may signify that these swamps are relic stands. Modeling efforts by Conner and Brody (1989) have shown that even though *T. distichum* and *N. aquatica* are flood-tolerant (Carter et al 1973, Brown 1981, Conner et al. 1981, Mitsch and Rust 1984), the total basal area of both will decline if water levels continue to rise. Thus, continuous flooding, though not immediately detrimental to cypress-tupelo swamps, will lead to their gradual death over time (Harms et al. 1980, Mitsch and Rust 1984, Conner and Day 1988, Conner and Brody 1989, Conner and Day 1992). The different regions identified within the Maurepas swamp appear to be in various stages along this trajectory of swamp decline, ranging from the continuously flooded but productive Throughput sites to the impounded, flooding and/or salinity stressed Interior and Lake sites, respectively.

The degree of salt-water intrusion into the soils of the Lake sites, particularly, was sufficient to kill hundreds of *A. rubrum*, *N. aquatica*, and *F. pennsylvanica* trees, as evidenced by tagged trees that were alive in spring 2000 and dead at the time of the fall 2001 diameter measurements. As the 1999-2000 drought was only broken by a severe rainfall event during June 2001, the continued high mortality rates (during 2001 and 2002) at sites impacted by the drought-precipitated salt-water intrusions possibly remain stressed by factors associated with the drought. If these mortality rates continue into the future, some areas along the southern shore of Lake Maurepas may be completely deforested within the next 2-5 years (at sites 20 and 19, respectively).

Table 2. Measurements of swamp primary production.

Forest Type (State)	Tree Standing Biomass (kg/m ²)	Litterfall (g·m ⁻² ·yr ⁻¹)	Stem Growth (g·m ⁻² ·yr ⁻¹)	Above-Ground NPP ^a (g·m ⁻² ·yr ⁻¹)	Reference
Reference Locations					
Cypress - Tupelo (LA)	37.5 ^b	620	500	1,120	Conner and Day (1976)
Impounded managed swamp (LA)	32.8 ^{b,c}	550	1,230	1,780	Conner et al. (1981)
Impounded stagnant swamp (LA)	15.9 ^{b,c}	330	560	890	Ibid.
Tupelo stand (LA)	36.2 ^b	379	---	---	Conner and Day (1982)
Cypress stand (LA)	27.8 ^b	562	---	---	Ibid.
Nutrient-poor Cypress Swamp (GA)	30.7 ^e	328	353	681	Schlesinger (1978)
Stagnant Cypress Swamp (KY)	9.4	63	142	205	Taylor (1985), Mitsch et al. (1991)
Sewage enriched cypress strand (FL)	28.6	650	640	1,290	Nessel (1978)
Near-continuously flooded Cypress-Ash swamp (LA)	---	553 ^d	443 ^c	996	Megonigal et al. (1997) ^f
Near-continuously flooded riverine Cypress-Tupelo swamp (SC)	---	438 ^d	216 ^c	654	Megonigal et al. (1997) ^f
Naturally flooded swamp (LA)	---	487 ^d	338 ^c	825	Megonigal et al. (1997) ^f
Periodically flooded riverine swamp (LA)	---	725 ^d	430 ^c	1155	Megonigal et al. (1997) ^f
Frequently flooded swamp (SC)	---	---	---	1887	Muzika et al. (1987)
Maurepas Swamp locations (this study)					
Interior Sites	12.11 ^g	185.0 ⁱ	128.3 ^{e,g,h}	325.4 ^g	
Intermediate Sites	11.25 ^g	309.2 ⁱ	164.9 ^{e,g,h}	485.3 ^g	
Lake Sites	4.0 ^g	106.8 ⁱ	83.9 ^{e,g,h}	194.7 ^g	
Throughput Sites	23.26 ^g	618.4 ⁱ	290.1 ^{e,g,h}	931.76 ^g	
Total Average	12.65 ^g	304.9 ⁱ	166.8 ^{e,g,h}	484.35 ^g	

^a NPP = net primary productivity = litterfall + stemgrowth

^b Trees defined as > 2.54 cm DBH (diameter at breast height)

^c Cypress, Tupelo, Ash only

^d Litterfall does not include woody litter

^e Trees defined as > 10 cm DBH

^f All values are presented as averages of two replicate plots in two consecutive years

^g Averages of 3-6 sites with 2 sub-stations each

^h Cypress, Tupelo, Ash, Maple, and Blackgum, where present

ⁱ All litterfall values exclude 2002, to correct for litter loss due to tropical storms.

Our study indicates that the herbaceous and woody vegetation in the Maurepas swamps is nutrient limited. The dramatic increases in herbaceous standing crop with increased nutrient loading were only evidenced in caged plots. Fertilizing during spring and again during summer resulted in nearly a 300% increase in standing crop, when compared with non-caged plots. Applying $118 \text{ g}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$ of timed-release fertilizer emulates a diversion of 3,000 cfs. It appears that doubling the currently proposed diversion capacity (to 3,000 cfs) would greatly benefit the swamp in terms of primary and secondary production.

During the growing season of 2003 we will convert the 'A' plots at 'A' stations with cages to a four-fold application of fertilizer to determine if additional increases in herbaceous standing crop occur with a nitrogen augmentation emulating a 6,000 cfs diversion. With regard to increases in tree production, several previous studies conducted over the last decade have demonstrated that nutrient augmentation to baldcypress seedlings at least doubles growth rates in the Manchac/Maurepas area (Boshart, 1997; Forder, 1995; Greene, 1994; Myers et al., 1995).

Lake sites are dominated by herbaceous production, followed by Interior sites. Both of these groups of sites are converting to marsh and open water, apparently for very different reasons, although both reasons appear to be tied to greatly hindered riverine influence. Salt stress appears to be killing trees at the Lake sites, whereas stagnant, standing water and nutrient deprivation appear to be the largest stressors at the Interior sites. If subsidence continues in the Maurepas swamps, a direct conversion from marsh/swamp to open water seems likely.

In summary, at present salinity seems to be an important stressor in the Maurepas swamps. However, degradation of tupelogum trees is evident in trees of widely varying age, from less than a decade old to as old as nearly a century. It is clear that this degradation has been occurring for decades and is almost certainly primarily due to altered hydrology and lack of throughput. Low soil bulk densities and high soil organic matter contents throughout much of this swamp are indicative of a lack of riverine influence. In short, all measures of ecosystem health collected thus far in the southern Lake Maurepas region indicate that these swamps are

highly degraded and would benefit from a substantial infusion of nutrients and freshwater from the Mississippi River.

Land conversion observations on the Manchac land-bridge and Jones Island (Barras et al. 1994) demonstrate what is expected in the Maurepas swamps in the coming decades, if a diversion from the Mississippi River is not implemented. In 1956, most of the area of the Manchac land-bridge was dominated by second-growth swamp. By 1978, much of this swamp had converted to marsh, and by 1990 the marsh had begun to break up and to convert to open water. Comparing the 1990 marsh coverage with current cover further demonstrates conversion to open water (Shaffer, unpublished data).

Conclusions

Most of the Maurepas swamp appears to be converting to marsh and open water primarily due to the lack of riverine input. Salt stress is killing trees at Lake sites, whereas stagnant standing water and nutrient deprivation appear to be the largest stressors at Interior sites. Furthermore, as increasing periods of flooding have been found to decrease the allocation of carbon to the root system (Powell and Day 1991), sites with stagnant standing water such as Interior swamp sites are expected to show a greater rate of subsidence than sites that are only seasonally flooded.

Severe increases in salinity, like those experienced during the drought in 1999-2000, however, may be prevented or greatly ameliorated by the increased fresh water throughput that the proposed diversion would bring. It is likely that the influences of freshening would be felt in areas as remotely located as Jones Island (site 20, [Figure 1](#)) and the Manchac landbridge, as the proposed diversion could replace all of the water in Lake Maurepas roughly twice each year, and Pass Manchac and North Pass (adjacent to Jones Island and Manchac) are the only two direct conduits that will allow the additional fresh water to eventually reach Lake Pontchartrain.

Besides decreasing the detrimental effects of salinity throughout the Maurepas swamp, the proposed diversion would also increase the sediment load and nutrient supply to these

wetlands. Hydrologic modeling showed that due to the low water-holding capacity of Hope Canal, most of the diversion water is likely to flow as sheet-flow through the interior Maurepas swamps (Kemp et al. 2001). The resulting, evenly distributed influx of sediments is expected to strengthen the highly organic soils of the Maurepas swamp and to increase elevation in certain areas sufficiently to make the natural regeneration of several wetland forest species possible. The potentially negative impacts of lake eutrophication due to the increase in nutrient loading to the swamp are unlikely to occur, as nutrient models indicate high nutrient retention in the swamp with nutrient removal efficiencies of 94-99% from the time diversion water enters the swamp until it reaches Lake Maurepas (Day et al. 2001). Experimental nutrient augmentation enhanced biomass production of the herbaceous vegetation by up to 300%. Furthermore, several studies conducted over the last decade have demonstrated that nutrient augmentation to *T. distichum* seedlings doubles growth rates in the Manchac/Maurepas area (Greene 1994, Forder 1995, Myers et al. 1995, Boshart 1997). Swamps as nutrient poor, stagnant, and impounded as the Interior Maurepas swamps would be expected to at least double their rates of production if they received an infusion of freshwater and nutrients from the Mississippi River. This enhanced productivity is essential for subsiding coastal wetlands to offset RSLR, as roots may contribute as much as 60% of the annual increment to soil organic matter (Meronigal and Day 1988). According to Keddy (2000), the exact duration and depth to cause the transition from swamp to marsh remains an interesting unresolved mystery. Without a diversion from the Mississippi River, however, the Maurepas swamp may soon resolve this issue all too clearly.

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