-Draft Report-

APPENDIX E

Characterization of Ecosystem Health of the Maurepas Swamp, Lake Pontchartrain Basin, Louisiana: Feasibility and Projected Benefits of a Freshwater Diversion

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Introduction

The modern Mississippi River Delta and its associated delta lobes encompasses an area that accounts for roughly 40% of the coastal wetlands found in the 48 continuous 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 undergone many major geologic changes, such as ice ages and continental drift, since then. 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 in the natural deltaic processes (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, which extends to the continental shelf, is the most recent. While most of these sediments are deposited in deep Gulf waters and do not construct new land, these sediments historically maintained old deltas and slowed their natural degradation and subsidence (Coleman et al., 1998; Day et al., 2000). However, the process of deltaic deterioration has sped up manifold (Pezeshki and DeLaune, 1995) since the construction of levees and channels, which limit or prevent this natural distribution of freshwater and sediments, alter natural hydrology, and cause 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 km^2 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 the freshwaters and any sediments they carry are distributed through the receiving area to most effectively maintain existing wetlands, and/or 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, 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, to define benefits that would 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. We then evaluate the health and rates of primary production of the woody and herbaceous components of the vegetation at these sites.

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.

Wetland Subsidence Rates

The effects of a freshwater diversion on the rate of wetland subsidence is still 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. The lakeshore sites are characterized by highly organic soils and do not appear to be impacted by suspended sediments from the Lake. Since 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.

Woody Vegetation

We tested the hypotheses that there are 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. Hope Canal 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 herbaceaous 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 appeared to be following the same trend as occurred on the Manchac landbridge and Jones Island, both just east of the Maurepas system. Since the mid-1950s these two areas have undergone transition from forested swamp to open marsh, and are transitioning to open water at present. We expect these trends in the Maurepas swamp to be lagged behind Manchac and Jones Island because it is slightly more buffered from Gulf influences.

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 expected to find a shift in species composition towards more 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

The study sites: To characterize the Maurepas swamp as accurately as possible, twenty study sites, each with two 625 m^2 permanent stations, 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 Maurepas swamp. Of these sites, sites 1 (Red Top), 2 (Cher Bayou), 6 (Potato Run), 7 (Peter's Run), 9 (Interior Mississippi Bayou), and 13 (Black Lake) were the most interior swamp (hereafter referred to as "Interior"), located remotely from any direct water exchange with Lake Maurepas and only accessible by airboat. Sites 3 (Blind River/Amite River Diversion Canal confluence), 4 (Lil' Chene Blanc), and 5 (Alligator Island) are located below the confluence of the Blind River and the Amite River Diversion Canal and were chosen to capture any potential benefits that this flood-control diversion might introduce into the swamp system. These sites will, hereafter, be referred to as "Diversion" sites. Sites 10 (Tent Bayou/Hope Canal), 11 (middle of Hope Canal) and 12 (near “headwaters” of Hope Canal) will be referred to as "Hope Canal" sites hereafter. These sites are all located along Hope Canal, a manmade canal used for storm drainage (that crosses beneath I-10) and are far enough removed from Lake Maurepas to make water exchange with the lake minimal. Sites 8 (Dutch Bayou), 14 (Reserve Relief Canal near I-10), 15 (Reserve Relief Canal west), and 16 (Reserve Relief Canal east), hereafter referred to as "Average" 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, are located near Lake Maurepas and are more likely to be influenced by Pass Manchac, the main waterway between Lake Maurepas and Lake Ponchartrain. To validate the ecological groupings above, the groups were subjected to a discriminant function analysis (Hair et al., 1998) using all abiotic and biotic metrics as potential predictor variables (for details, see Statistical Analysis section below).

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. These sites will be monitored in the future as part of the “Manchac Project” (Keddy et al., 2001).

Sediment elevation tables: Two SETs were installed at twelve representative study sites throughout the Maurepas swamp. Each permanent SET setup consists of a 5 m long aluminum pipe (8 cm in diameter) that was driven into the ground to “refusal” (i.e. when vertical movement ceases despite repeated pounding). 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 reading of these SET tables occurred during the months of October and November 2000. The difference in elevation between this reading and a second reading, that will be taken during the months of October and November 2001, will provide 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 second SET readings will be taken, frozen soil cores will be 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 will 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 assess the effects of salinity, sediment deprivation, and prolonged flooding, well water salinity and soil variables such as interstitial salinity, pH, bulk density, redox potential (Eh), soil moisture, sulfide concentrations, and concentrations of nitrate, ammonia, phosphorous and selected elements were monitored at each of the 40 stations. Two 1-m (6 cm diameter) PVC wells were inserted 0.75 m into the ground at each of the 40 stations. Below ground slits every 2 cm (from a depth of 5 cm to 70 cm) enabled ground water to enter each well. The wells were evacuated before each measurement.

Sample collection: Soil cores for interstitial water were collected from plots 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, so as not to lose either sediment or interstitial water, and then extruding 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 in an ice chest and were refrigerated upon returning from the field and processed within 24 hours.

Soil cores for bulk density analysis were collected during the summer sampling period using a soil corer with a 2.2-cm inner diameter. Samples were collected by coring to a depth of 6.5 cm, carefully removing the soil corer and then extruding cores into sample bags. All soil cores for bulk density analysis were placed into a ventilated drying oven set at 65° C upon returning from the field and dried until constant weight was achieved.

Soil reduction: The degree of soil reduction was measured at the surface (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). Probes were allowed to come into equilibrium with the soil (for a minimum of 15 minutes) and readings were then taken. Surface and deep measurements were each replicated three times and then averaged within each plot to reduce within plot variation.

Soil chemical properties: Because of the unusual drought conditions that occurred in the Lake Maurepas area during the summer 2000 sampling period, there was insufficient water available in soil cores for interstitial water extraction and analysis. Therefore a dry extraction (i.e., a drying of the sample and grinding, followed by subsequent rehydration with distilled water) was employed to achieve an estimate of interstitial soil characteristics (see Soil and Plant Analysis Council Inc. 1999 for discussion of method). Samples were dried in a ventilated drying oven set to 65° C until a constant weight was achieved. Samples were then homogenized using a mortar and pestle. A subsample (between 5 g and 15 g depending on the total amount of sample collected) was then placed into a centrifuge bottle in a 1:5 ratio with distilled water. Samples were shaken for one hour on a shaker table and then centrifuged. Centrifuge time was 15 minutes at 5000 RPM with temperature set to ambient (between 20°C and 25°C). Aliquots of the extracted interstitial water were then placed into acid-washed containers and the suspended sediments were allowed to drop out of solution. The remaining supernatant was then filtered with 2µm syringe filters. Sulfide analysis was not performed on these samples because any sulfides present would have oxidized during the drying process. Because of the limited volume of extracted water, these samples were used for determination of pH and salinity and then partitioned into aliquots for ICP (elemental) and autoanalyzer (nitrate-nitrite and ammonium) analyses. To maintain sample integrity all probes used were rinsed with distilled water six times between measurements. Solution pH was determined using a Jenco pH/mV meter and salinity was determined using a YSI salinity/conductivity/temperature meter. 10-ml aliquots were then placed into acid-washed scintillation vials and preserved with two drops of concentrated HNO_3 for ICP analysis. 5-ml aliquots were then placed into autoanalyzer vials for autoanalyzer analysis and then frozen. ICP samples were kept refrigerated and autoanalyzer samples were kept frozen until transport to LSU for analysis.

Fall soil samples were processed within 24 hours after collection. Samples were removed from the refrigerator, dried of any external condensation, and then weighed. Samples were then centrifuged at 4° C, for 15 minutes at 5000 RPM. Interstitial soil water was then partitioned into aliquots for sulfide analysis, autoanalyzer analysis, ICP analysis, and for salinity and pH determination. Samples for sulfide analysis were immediately filtered and placed into an antioxidant buffer. Samples for autoanalyzer and ICP analysis were placed into acid-washed vials refrigerated while the suspended sediments dropped out of suspension. The remaining supernatant was then filtered using a 2-µm syringe filter into approximately 10-ml samples for ICP analysis and two approximately 5-ml samples for autoanalyzer analysis. Samples for ICP analysis were placed into acid-washed scintillation vials and preserved with 2 drops of concentrated HNO_3 and then refrigerated until analysis. Equal portions of samples for autoanalyzer analysis were placed into autoanalyzer vials and frozen. These samples were later transported to LSU for analysis. Sulfide concentration was determined using a Jenco pH/mV meter, LAZAR sulfide selective electrode, and reference electrode (see McKee et al., 1998 for details). Samples were analyzed the same day as extracted. Samples for pH and salinity determination were allowed to warm to room temperature. pH was measured using a calibrated Jenco pH/mV meter and salinity was measured using YSI salinity/conductivity/temperature meter.

The dry extraction technique was applied to a set of replicate soil cores taken during the Fall 2000 sampling period to determine if a predictable relationship existed between dried extraction and wet interstitial water extraction results. These soil cores were collected as normal for interstitial soil cores except that they were not purged with nitrogen gas and placed in 1-gallon sampling bags. After returning from the field, these samples were immediately placed into a ventilated drying oven set at 65°C and dried to constant weight. These samples were processed in the same fashion as described in the Summer Dry Extraction section above.

Soil physical properties: Soil percent moisture was determined from soil cores taken for interstitial water analysis by subtracting the dried soil core weight from the wet soil core weight and then dividing by the wet soil core weight. Soil bulk density was determined by dividing the dry soil weight by the volume of the soil corer. See Parent and Garon (1993) for discussion of method and considerations. Percent organic matter was determined by heating 2-g portions of pre-weighed dry samples in a muffle furnace at 550 ° C for a minimum of 5 hours. Samples were then cooled to room temperature and reweighed. Samples were then returned to the oven for another 5-hour period, cooled to room temperature and again reweighed. This process was continued until a constant sample weight was achieved. The weight of the remaining sample was divided by the starting weight of the dried sample yielding the percent mineral matter. This was then subtracted from 100 to yield the percent organic matter.

Herbaceous Vegetation

Within each of the forty 25 m x 25 m (625 m^2) permanent stations, four 4m x 4m (16 m^2) permanent herbaceous plots were established at a diagonal distance of 5 m in from each of the corners of each station. A 4-m^2 plot was established in the center of each 16-m^2 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. 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 Blanuet cover method (Braun-Blanquet 1932).

Nutrient augmentation: Half of the 16-m^2 plots were fertilized with 236 g of Osmocote 18-6-12 timed-release fertilizer. This dosage emulates a loading rate of 11.25 g N m^2/year.

Annual production: 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). 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 late summer/early 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. Fortunately, we had micro-mapped each species location in all 160 plots during early season cover value estimates. 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.

Woody Vegetation

Tree health and primary productivity were determined through the collection of annual litter. Annual tree diameter growth was used to estimate wood production and calculate basal wood area per hectare of swamp (Brown, 1981; Conner and Day, 1992; Mitsch and Ewel, 1979).

<u>Wood production</u>: The majority of all trees within each of the two 625 m^2 plots at each of 20 study sites in the Maurepas swamp were tagged using 8-penny galvanized nails and pre-numbered 5 cm aluminum ID tags in February and March 2000. All baldcypress (*Taxodium distichum* (L) Rich.) and tupelogum (*Nyssa aquatica* Wang) in the plot were tagged, as well as other woody species > 5 cm diameter, including pumpkin ash (*Fraxinus profunda* Bush), green ash (*Fraxinus pennsylavanica* Marshall var. *subintegerrima*), swamp red maple (*Acer rubrum* (Vahl) Fernald var. *drummondii*), black gum (*Nyssa sylvatica* Walter var. *biflora*), black willow (*Salix nigra* Marshall), Chinese tallow (*Sapium sebiferum* (L.) Roxb), and southern waxmyrtle (*Myrica cerifera* L.). Trees were tagged at breast height, unless the fluting bases of baldcypress and tupelogum or the complex branching structure of other trees required the tags to be somewhat higher. Initial tree diameters of the approximately 1800 tagged trees throughout the study sites were then measured in February and March of 2000 using metric diameter tapes at the bottom of the freely hanging metal tags. To be able to locate trees during subsequent field sampling events, hand drawn maps of each plot, with the location of tagged trees within them, were prepared for all 40 stations. At the time of measuring the first set of tree diameters, the percent defoliation (due to forest tent and leaf roller caterpillars) of cypress and tupelo trees was, furthermore, estimated and noted for a subset of these sites for the evaluation of the effect of caterpillar herbivory on annual production. During October 2000, a second diameter measure was taken of every tree previously tagged. All tree diameter data was entered into SYSTAT 10. Tree diameter growth was then calculated from the difference in initial and final diameters with the assumption that no tree shrinkage occurred. The yearly tree diameter growth data was then used to calculate the amount of wood production per tree for the dominant tree species ("Cypress", "Tupelo", and "Other", consisting of maple, ash, and blackgum) using the wood production regression formulas of Clark et al., (1985), Muzika et al., (1987), and Scott et al., (1985), summed by species per plot and analyzed using an ANOVA model as detailed below.

Basal wood: The measured tree diameters in each plot were then converted into basal wood area, summed and multiplied by 16 (since there are sixteen 625-m^2 plots to a hectare) to estimate the total area of basal wood per hectare for each plot for each of the 40 stations.

Litter production: Five litter traps were randomly installed at each of two plots at 40 stations throughout the Maurepas swamp (Figure 1), to yield a total of 200 litter traps deployed. Each of these litter traps was 0.25 m^2 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 time). 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, since the five traps are pseudo-replicates. For this study, we use the term litter for both woody and non-woody (leaves, flowers, fruits, and seeds) litter. Collected litter was then dried to constant mass at 60 ^{o}C. After drying, the litter was sorted into cypress, tupelo and other litter. This enabled us to monitor productivity at the species level for the two most dominant tree species in the swamp. All litter data was then entered into SYSTAT 10 (Wilkenson, 2000). Total litter per site was estimated from these components and analyzed using an ANOVA model with the site groupings (locations) and the three "species" categories as the independent variables and bulk density and interstitial salinity as potential covariables.

Statistical Analysis

All statistical analyses were performed using SYSTAT 10.0 (Wilkenson, 2000). 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 and salinity 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 m^2 per year, again with bulk density and interstitial salinity as potential covariables. Finally, herbaceous, wood, and litter production were combined for total primary production and analyzed for site grouping differences, also using bulk density and salinity as potential covariables. Unless otherwise noted, all statistical findings were significant at a protected $\alpha = 0.05$ level (Zar, 1996).

Results

Wetland Subsidence

Since SET measurements were taken as recently as October and December, 2000, further SET readings will not be taken until the fall of 2001, as it seems likely that these SET readings will not yet convey any reliable information. Similarly, the feldspar marker horizons will be sampled during the fall of 2001 in concurrence with the SET readings taken at that time (Keddy et al., 2001).

Site Groupings

Evaluation of a *priori* site groupings using discriminant analysis indicated that 67% to 100% (Table 1) of sites were correctly classified using the variables bulk density, interstitial salinity, and leaf litter productivity in the discriminant classification. Further evaluation of discriminant functions with principle components analysis confirmed that the canonical discriminant functions used to discern groupings correspond to salinity (interstitial salinity), mineral input (bulk density), and aboveground productivity (leaf litter; analysis not shown). Salinity best separated the Lake sites from the Interior and Hope Canal sites (Figure 2). Bulk density best separated the Diversion sites, and productivity best separated Hope Canal and Interior sites. None of the predictor variables were able to separate the Average sites alone, yet together the discriminant function correctly classified 100% of the Lake sites (Table 1).

Table 1. Jacknifed classification matrix from discriminant function analysis (Hair et al., 1998) classifying sites based on salinity, bulk density, and leaf litter. Numbers in matrix indicate how the sites were classified and last column is the percent correctly classified.

	Average	Diversion	Hope	Interior	Lake	% Correct
Average	8	0	0	0	0	100
Diversion	0	4	0	2	0	67
Hope	0	0	5	1	0	83
Interior	1	1	0	10	0	83
Lake	1	0	0	0	7	88
Total	10	5	5	13	7	85

Abiotic Characterization

Well water salinity: The analysis of the well water salinity data revealed that the salinity differed significantly between site groupings ($F_{4,35} = 24.64$, $P << 0.0001$; Figure 3). Salinity levels at Lake sites exhibited the highest summer salinities, ranging from 3.0 to 5.8 ppt (plot averages) with a mean of 4.4 ± 0.4 ppt (group average). Average study plots had higher well water salinity levels (mean = 2.9 ± 0.2 ppt) than all other sites, except Lake sites. No significant well water salinity differences were found between either the Diversion, Hope Canal, or Interior sites.

Interstitial salinity: Interstitial salinities also differed among the site groupings ($F_{4,35} = 3.62$, $P = 0.014$; Figure 4), though the pattern in this data set was not as clearly defined as in the well water salinity data. Again, sites located near the lake had the highest interstitial salinites (mean = 3.0 ± 0.5 ppt). The lowest interstitial salinities occurred at the Hope Canal sites and the Interior sites (mean = 1.6 ± 0.1 ppt and mean = 1.8 ± 0.1 ppt, respectively), while Amite River Diversion sites and Average sites were not significantly different from either of these two extremes (mean = 2.4 ± 0.5 ppt and mean = 2.4 ± 0.1 ppt, respectively).

Bulk density: Bulk densities differed significantly among site groupings ($F_{4,35} = 12.26$, $P < 0.0001$; Figure 5). The highest bulk densities were found at the Diversion sites (mean = 0.231 ± 0.039 g/cm^3). All other sites, regardless of the biological characterization of the system, had significantly lower bulk densities than the Diversion sites, ranging from mean = 0.074 ± 0.011 g/cm^3 at the Interior sites to mean = 0.125 ± 0.014 g/cm^3 at the Hope Canal sites, and did not differ significantly from one another.

Percentage organic matter: No significant difference between fertilized and unfertilized herbaceous plots was detected in percentage organic matter levels ($F_{1,70} = 0.049$, $P = 0.826$; Figure 6). As a group, Diversion sites had lower percentage organic matter than Hope Canal sites ($F_{1,70} = 7.041$, $P = 0.010$). No significant difference in percentage organic matter was detected between the Hope Canal and Diversion sites compared with the Average, Interior, and Lake sites ($F_{1,70} = 0.002$, $P = 0.965$). The Lake sites were not different in percentage organic matter levels compared with other sites ($F_{1,70} = 2.907$, $P = 0.093$). Similarly, no difference in percentage organic matter was detected between Interior sites and Average sites ($F_{1,70} = 1.596$, $P = 0.211$; Figure 6).

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.

Fall interstitial ammonium levels: Interstitial ammonium levels were significantly higher in fertilized plots than unfertilized plots ($F_{1,70} = 6.164$, $P = 0.015$; Figure 7). No significant difference was detected between Diversion sites and Hope Canal sites ($F_{1,70} = 0.428$, $P = 0.525$) in interstitial ammonium levels. The Hope Canal and Diversion sites had significantly less interstitial ammonium levels compared with the Average, Interior, and Lake sites ($F_{1,70} = 4.067$, $P = 0.048$; Figure 7). The Lake sites had significantly greater interstitial ammonium levels compared with all other sites ($F_{1,70}$ = 13.124, P= 0.001). The Average and Interior sites were not significantly different in interstitial ammonium concentration for fall 2000 wet extractions ($F_{1,70}$ = 0.603, P= 0.440).

Fall interstitial nitrate levels: Fall interstitial nitrate levels were significantly higher in fertilized plots than unfertilized plots ($F_{1,70}$= 2.791, P=0.033; Figure 8). No significant difference was detected between Diversion sites and Hope Canal sites ($F_{1,70}$ = 1.787, P= 0.186) in fall interstitial nitrate levels. In contrast to the ammonium results, the Hope Canal and Diversion sites had significantly greater interstitial nitrate concentrations for fall 2000 compared with the Average, Interior, and Lake sites ($F_{1,70}$= 4.067, P= 0.048; Figure 8). Also in contrast to the ammonium results, the Lake sites were not significantly different in interstitial nitrate levels compared with all other sites ($F_{1,70}$ = 0.079, P= 0.780). The Average and Interior sites were not significantly different in interstitial nitrate concentration for fall 2000 ($F_{1,70}$ = 0.102, P= 0.751). There was no significant interaction of plot and location for interstitial nitrate levels in fall of 2000 ($F_{4,70}$ = 2.153, P= 0.083).

Fall interstitial phosphorus levels: No significant differences were detected with regard to phosphorus for plot, location (including *a priori* linear contrasts), or plot x location interaction (Figure 9).

Interstitial sulfide levels: All interstitial sulfide readings were below detectable levels for fall 2000.

Soil redox: The Hope Canal 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 10). 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 10). 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 10). The Lake and Diversion 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 the lack of detectable sulfides as sulfide production does not occur until soil reduction reaches -75 mV (Mitsch and Gosselink 2000).

pH: Interstitial pH levels were significantly lower in fertilized plots than unfertilized plots ($F_{1,70}$ = 6.197, P <0.001; Figure 11). Diversion sites had a lower pH than Hope Canal sites (F= 9.647, P= 0.003). Taken together, Hope Canal and Diversion sites had significantly lower interstitial pH levels than the Average, Interior, and Lake sites ($F_{1,70}$ = 5.808, P= 0.019). The Lake sites were not significantly different in interstitial pH levels than all other sites ($F_{1,70}$= 2.556, P= 0.115). Interior sites had significantly greater pH than Average sites ($F_{1,70}$ = 6.94, P= 0.011; Figure 11).

Herbaceous Vegetation

Cover values: Early season cover value estimates of the 15 dominant species that represent 97% of the totalcover revealed that the Lake sites have the highest species richness and the Hope Canal sites the lowest (Figure 12). The relatively low species richness of the Hope Canal sites is almost certainly due to dense overstory cover and mid-story woody species. Alligatorweed (*Alternanthera philoxeroides*), smartweed (*Polygomum punctatum*), and arrow arum (*Peltandra virginica*) appeared to be the most ubiquitous species as they were present in all habitat types. *P. virginica* and pickerelweed (*Pontederia chordata*) decrease in abundance as habitats become more degraded and may serve as an indicator species of ecosystem health, as may palmetto (*Sabal minor*) which only occurs at the Alligator Island Diversion sites (Figure 12) which have the highest soil strength (bulk density) of the 40 stations.

Annual production: Annual herbaceous standing crop was significantly higher early in the growing season than during the late season (Figure 13a). Herbaceous production was highest at the Lake sites, followed by Interior sites, and did not differ for Diversion and Average sites, which had significantly higher production than the Hope Canal sites (Figure 13a).

Nutrient augmentation: Timed-release fertilizer increased growth early in the season, but had no effect on late season biomass production which produced a significant interaction between nutrient augmentation and season (Figure 13b). Nutrient Augmentation resulted in as much as a six-fold increase at Average sites, whereas it had little to no effect at Diversion sites and Interior sites (Figure 14).

Woody Vegetation

Basal area: Basal areas per hectare differed significantly among site groupings ($F_{4,34}$ = 10.23, P << 0.001; Figure 15). Well Water salinity was a significant covariable in the model ($F_{1,34}$ = 4.745, P = 0.036; Figure 3), indicating that sites with small basal areas also had a higher well water salinity than sites with larger basal areas. The greatest basal areas were found at the Hope Canal sites (mean = 54.32 ± 5.31 m^2/ha). All other sites, had significantly lower basal areas than the Hope Canal sites, ranging from mean = 9.98 ± 2.79 m^2/ha at the Lake sites to mean = 28.70 ± 2.39 m^2/ha at the Interior sites, and did not differ significantly from one another (Figure 15). Basal areas of forested species are low to moderate at all sites except Hope Canal.

Wood production: Total wood production differed significantly among site groupings ($F_{4,34}$ = 6.356, P < 0.001; Figure 16). Salinity was a significant covariable in the model ($F_{1,34}$ = 4.320, P = 0.045), indicating that sites with high rates of wood production (all species combined) had a lower interstitial water salinity than sites with less wood production. The highest rates of total wood production were found at the Hope Canal sites (mean = 201.28 ± 25.67 $g{\cdot}m^{-2}{\cdot}yr^{-1}$), followed by similar rates at the Interior sites (mean = 140.73 ± 10.55 $g{\cdot}m^{-2}{\cdot}yr^{-1}$). Total wood production was lower at the Diversion and Average sites, though these were similar to the Interior sites. The lowest rates were found at the Lake sites, where wood production only reached a rate of mean = 42.60 ± 17.90 $g{\cdot}m^{-2}{\cdot}yr^{-1}$. Interior and Average sites combined were significantly more productive than Lake sites by *a priori* contrast ($F_{1,34}$ = 5.235, P = 0.028). A separate ANOVA model was run to separate out the wood production of the different tree species from one another, where tree species were selected as Cypress, Tupelo and Other, and other was a combination pumpkin and green ash (*Fraxinus profunda and Fraxinus pennsylvanica*, respectively), swamp red maple (*Acer rubrum* var. *drummondii*), and swamp black gum (*Nyssa sylvatica* var. *biflora*). Again, there was a significant group effect ($F_{4,99}$ = 5.82, P << 0.001), indicating that the Hope Canal sites produced significantly more wood than Diversion, Average, and Lake sites. Interior sites were not significantly different in wood production from any other site. The species effect was insignificant and the interaction was only marginally significant (F=xx, P=0.07?) and was due to tupelo having the highest wood production only at interior sites (Figure 17). Of the tree species examined, cypress and tupelo overall were similarly productive and each significantly more productive than other, less abundant woody species ($F_{2,133}$ = 34.69, P < 0.0001; Figure 17) and accounted for more than 90% of all the wood production measured.

Litter production: Total litter production differed significantly among site groupings ($F_{4,34}$ = 18.92, P < 0.0001; Figure 18). Salinity was a significant covariable in the model ($F_{1,34}$ = 6.726, P = 0.014), indicating that sites with high rates of litter production (all species combined) had a lower interstitial water salinity than sites with less litter production. The highest rates of total litter production were found at the Hope Canal sites (mean = 566.82 ± 75.89 $g{\cdot}m^{-2}{\cdot}yr^{-1}$). The next most productive sites in terms of litter production were Diversion (mean = 262.77 ± 72.83 $g{\cdot}m^{-2}{\cdot}yr^{-1}$), Average (mean = 308.32 ± 30.92 $g{\cdot}m^{-2}{\cdot}yr^{-1}$), and Interior (mean = 181.19 ± 14.39 $g{\cdot}m^{-2}{\cdot}yr^{-1}$) sites, though Interior sites did not significantly differ from the litter production of the Lake sites (μ = 82.19 ± 23.99 $g{\cdot}m^{-2}{\cdot}yr^{-1}$), which had by far the lowest rates of litter production (Figure 18). Again, baldcypress was the most productive tree species in terms of litter production, whereas litter production in tupelogum and all other woody species was similar and significantly lower ($F_{2,117}$ = 13.44, P < 0.0001, Figure 19). Examined separately by species, litterfall production was dominated (~ 45-70% by group average) by cypress at all sites, followed by tupelo (~10-40%), though tupelo was generally more abundant by stem count than cypress at all sites except the Lake sites. Hope Canal and Interior sites combined were more productive in cypress, tupelo and other litter production than all other sites by *a priori* contrast based on observations of salinity differences ($F_{1,33}$ = 11.01, P = 0.002 for cypress, $F_{1,34}$ = 19.22, P < 0.0001 for tupelo, $F_{1,34}$ = 8.03, P = 0.0077 for other litter; Figure 19). Litter production in the Maurepas swamp was higher in every site grouping as well as overall than wood production (compare Figures 16 and 18).

Total Aboveground Primary Productivity

Total primary production (the average of the sum of the herbaceous, wood, and litter production per site) was highest (mean = 800.90 ± 67.27 $g{\cdot}m^{-2}{\cdot}yr^{-1}$) at the Hope Canal sites and similar at all other sites, ranging from (mean = 571.47 ± 53.12 $g{\cdot}m^{-2}{\cdot}yr^{-1}$) at Interior sites to (mean = 441.13 ± 34.06 $g{\cdot}m^{-2}{\cdot}yr^{-1}$) at Average sites ($F_{4,34}$ = 9.86, P < 0.0001; Figure 20). Salinity was a marginally non-significant covariable in this model ($F_{1,34}$ = 3.163, P = 0.084) and was retained in the model as it substantially increased statistical power. Hope Canal, Diversion, and Average sites were clearly dominated by litter and wood production, while Interior and Lake sites were dominated by herbaceous production (Figure 20). After, building the model with the 20 individual sites as the independent variable, rather than the five groups of sites, an *a priori* linear contrast showed, furthermore, that total primary production was significantly higher at the Alligator Island site (located close to the confluence of the Amite River Diversion Canal and Blind River) than at sites 3 and 4 (located more interior to the Diversion).

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, 2000). 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 such that throughput is 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 tupelogum trees. We now have quantitative information that allows us to compute the likely benefits of a future with a diversion into Hope Canal vs. 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., Appendix F), soils indicative of stress and extremely low strength (see herein), and salt-water intrusions that occur mostly during the fall. The mean salinity of the lake water measured at the Manchac Bridge also has increased gradually since 1951 (Thomson, 2000). 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 land-bridge, since 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.

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 the Diversion sites (sites influenced by the present Amite River Diversion 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 Diversion 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 Diversion sites (approximately 29%) compared to 38% to 42% at the Hope, Interior, and Average sites. The Lake sites had soil organic matter contents similar to the Diversion sites (29%), but relatively low soil bulk densities (~0.09 g cm^{-3}). 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 lowest soil bulk density values recorded in the study (~0.07 g cm^{-3}), which again is indicative of hydrologic isolation and a lack of sediment input.

Because the severe drought this past year (2000) was accompanied by relatively low water stages, we believe that 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 (Figures 10 and 11) and lack of detectable sulfides at all sites. Contrarily, we expect that late season primary production was greatly suppressed as salt water penetrated deep into the Interior sites. The degree of salt water penetration into the soils of the Lake sites was sufficient to kill hundreds of swamp red maple, tupelogum, and ash, as evidenced by tagged trees that were alive in the spring and dead at the time of the fall diameter measurements (Figure 21).

As a result, the Lake sites have extremely low basal areas. These areas were largely forested as recently as the late 1950s. The Hope Canal sites are the only sites in the southern Maurepas with basal areas approximating a "healthy wetland forest." In terms of annual aboveground net primary production, however, even these (Hope Canal) sites compare most closely to that of an impounded stagnant swamp elsewhere in Louisiana (Conner et al., 1981; Table 2), whereas most other sites range between swamps that have been identified as either nutrient-poor (Schlesinger, 1978) or stagnant (Taylor, 1985; Mitsch et al., 1991; Table 2). However, if these swamps received an infusion of freshwater and nutrients from the Mississippi they would be expected to increase in primary production by an estimated three-fold or greater (Table 2).

The differences observed in wood production between Hope Canal and Interior sites, with significantly higher rates of cypress and tupelo wood production than all other sites, and Lake sites, with significantly lower rates of tupelo wood production than all other sites, seem to map well onto the salinity differences found at these sites. Thus, there seems to be an observable negative effect of increased salinity on the rate of wood production in cypress, which was even more pronounced in tupelo.

The sites grouped as “Diversion” sites were not as high in productivity as expected. It is very likely that four of these stations received little influence from the Amite River (flood-control) Diversion Canal during the study period because of the severe drought conditions. Due to low water stages, almost all of the water from the diversion canal bypassed the swamp and flowed straight to the mouth of Blind River into Lake Maurepas. Because this water completely surrounds Alligator Island (Figure 1) this site did realize benefits from the diversion canal.

Our study indicates that the herbaceous and woody vegetation in the Maurepas swamps may be nutrient limited. Nutrient augmentation significantly enhanced (by about 1/3) biomass production of herbaceous vegetation during 2000. Furthermore, several studies conducted over the last decade have demonstrated that nutrient augmentation to baldcypress seedlings doubles growth rates in the Manchac/Maurepas area (Boshart, 1997; Forder, 1995; Greene, 1994; Myers et al., 1995; Figure 22). Interstitial soil water nutrients in fertilized plots were not detectably higher than those in non-fertilized plots, suggesting that the vegetation assimilated all available nutrients and that the assimilative capacity of the swamp system therefore was not exceeded.

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.

Table 2. Various measurements of swamp primary production for comparison with this study.

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[e]	996	Megonigal et al. (1997) [f]
Near-continuously flooded riverine Cypress-Tupelo swamp (SC)	---	438[d]	216[e]	654	Megonigal et al. (1997) [f]
Naturally flooded swamp (LA)	---	487[d]	338[e]	825	Megonigal et al. (1997) [f]

Periodically flooded riverine swamp (LA)	---	725[d]	430[e]	1155	Megonigal et al. (1997) [f]
Frequently flooded swamp (SC)	---	---	---	1887	Muzika et al. (1987)
<u>Maurepas Swamp locations (this study)</u>					
Amite River Diversion Sites	10.0[g]	262.8[g]	130.8[e,g,h]	393.6[g]	
Hope Canal Sites	25.5[g]	566.8[g]	322.0[e,g,h]	888.8[g]	
Interior Sites	12.1[g]	181.2[g]	225.2[e,g,h]	406.4[g]	
Average Sites	11.2[g]	308.3[g]	130.3[e,g,h]	438.6[g]	
Lake Sites	3.9[g]	82.2[g]	68.2[e,g,h]	150.4[g]	
Total Average	12.0[g]	256.9[g]	175.2[e,g,h]	432.1[g]	

[a] NPP = net primary productivity = litterfall + stem growth

[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 two 625m^2 replicates each

[h] Cypress, Tupelo, Ash, Maple, and Blackgum, where present

Land conversion observations on the Manchac land-bridge and Jones Island (Figure 23) demonstrate what is expected in the Maurepas swamps in the coming decades, if a diversion 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 would demonstrate further conversion to open water (Shaffer, unpubl. data).

Conclusions

The results of this study have provided information on the feasibility of the proposed freshwater diversion into the Maurepas swamp. Due to the large spatial scale of the study (about 160 km^2), the results should provide information necessary for the selection of the best site and the most effective scale of a freshwater diversion into the swamp ecosystem to maximize the benefits of this restoration method. Results from the experimental fertilization component of the study, combined with the soil nutrient concentration monitoring, may, furthermore, be used in predicting the nutrient assimilation capabilities of the cypress-tupelo swamp and the effects a potential freshwater diversion would have on the water quality of the system, helping to ensure that waters entering the lake will not trigger algae blooms. The information from this study may further be useful in identifying the major stress factors in these degraded wetlands and to predict more accurately and/or prevent future wetland loss.

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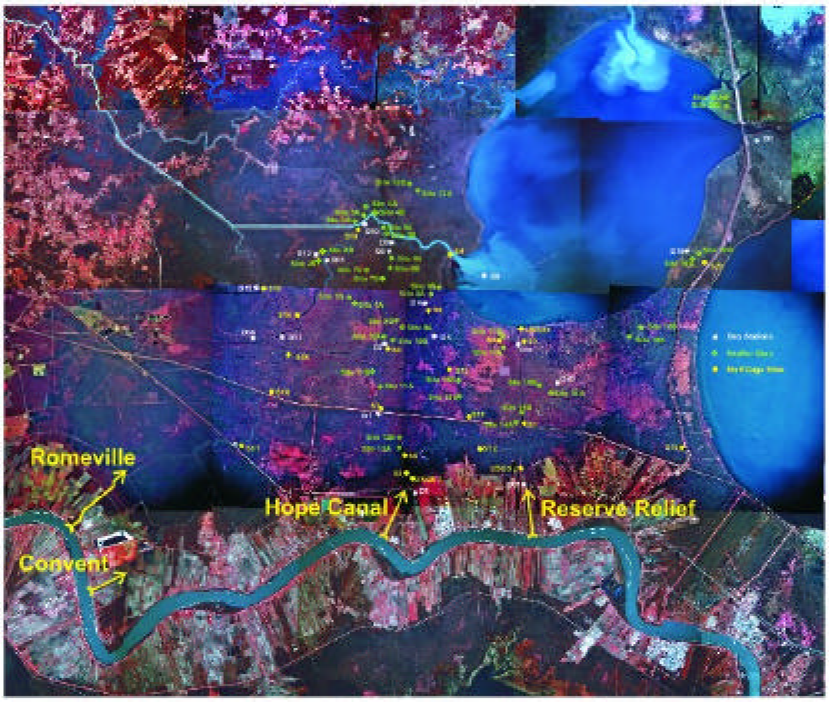

Figure 1. Composite aerial photographic map of the southern Lake Maurepas study area, showing the sampling and measurement locations for various study components, as well as the four candidate diversion locations reviewed.

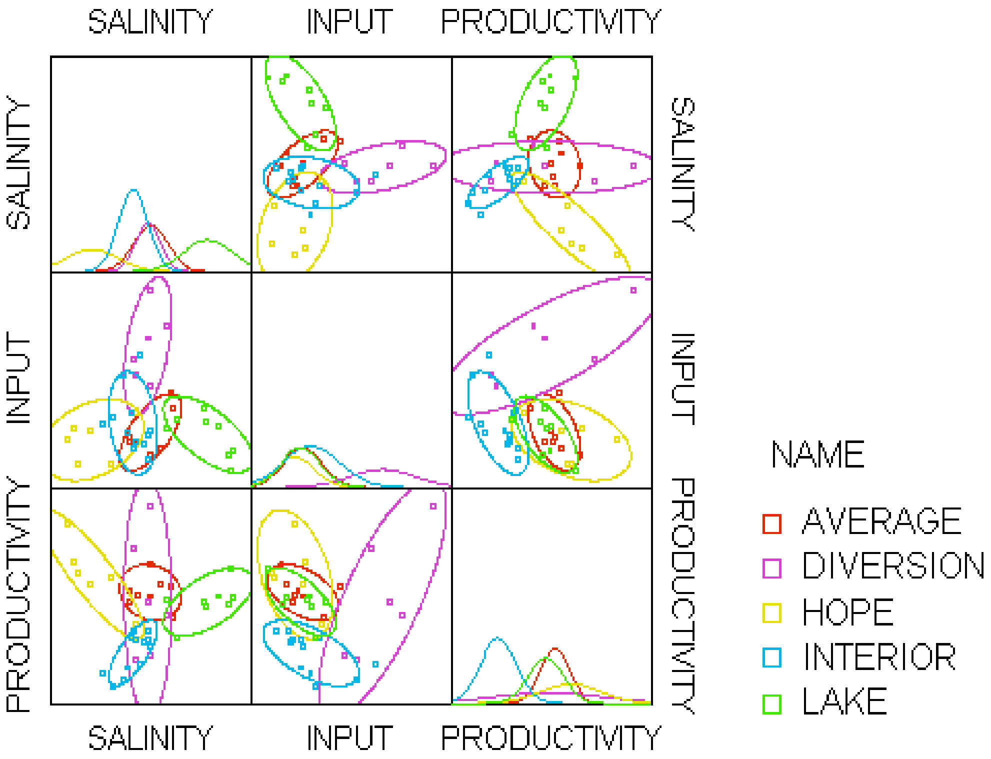

Figure 2. Results of discriminant analysis of *a priori* categorization. Shown are 95% confidence ellipses demonstrating degree of site separation that the discriminant functions render. The first function was dominated by salinity, the second by bulk density, (i.e. sediment INPUT), and the third variate by litterfall (PRODUCTIVITY).

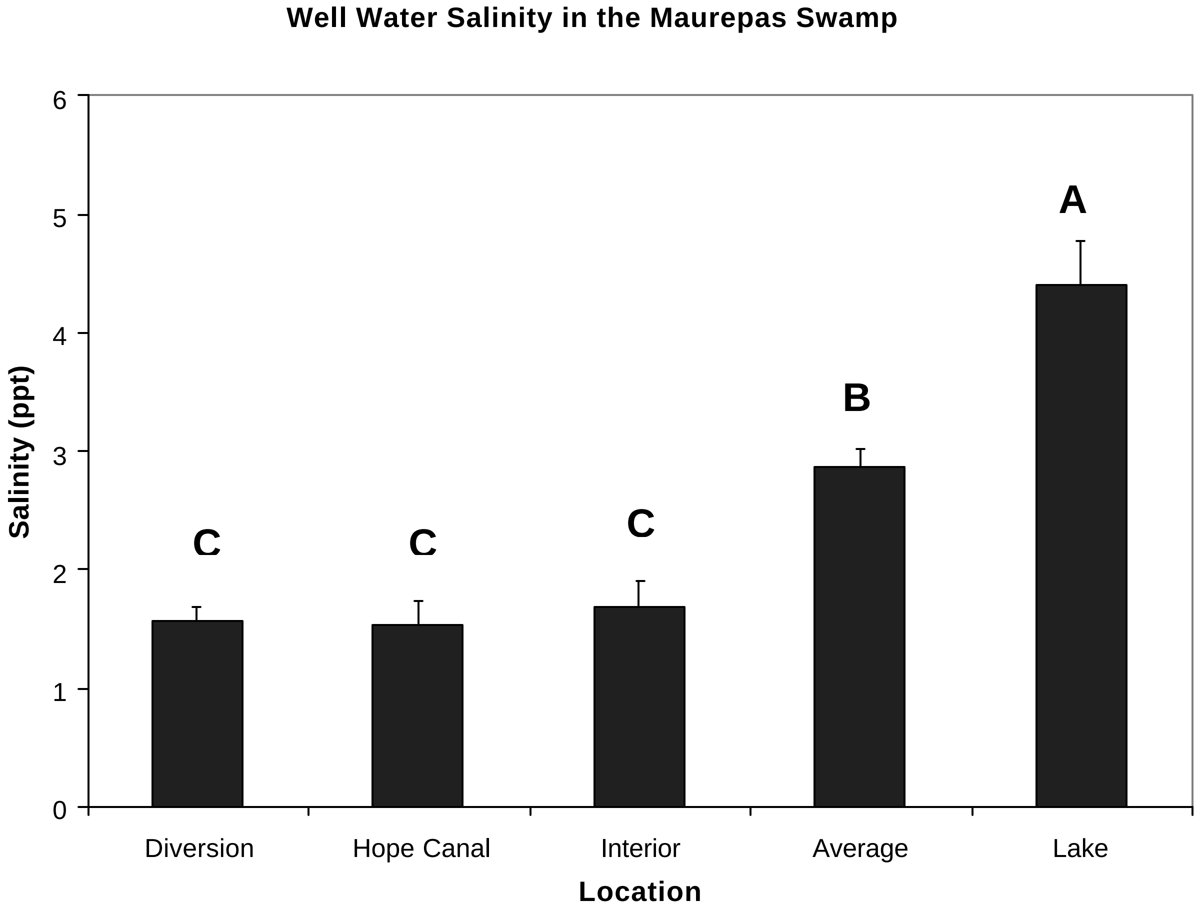

Figure 3. Summer well-water salinities at the different habitat types located in southern Lake Maurepas. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

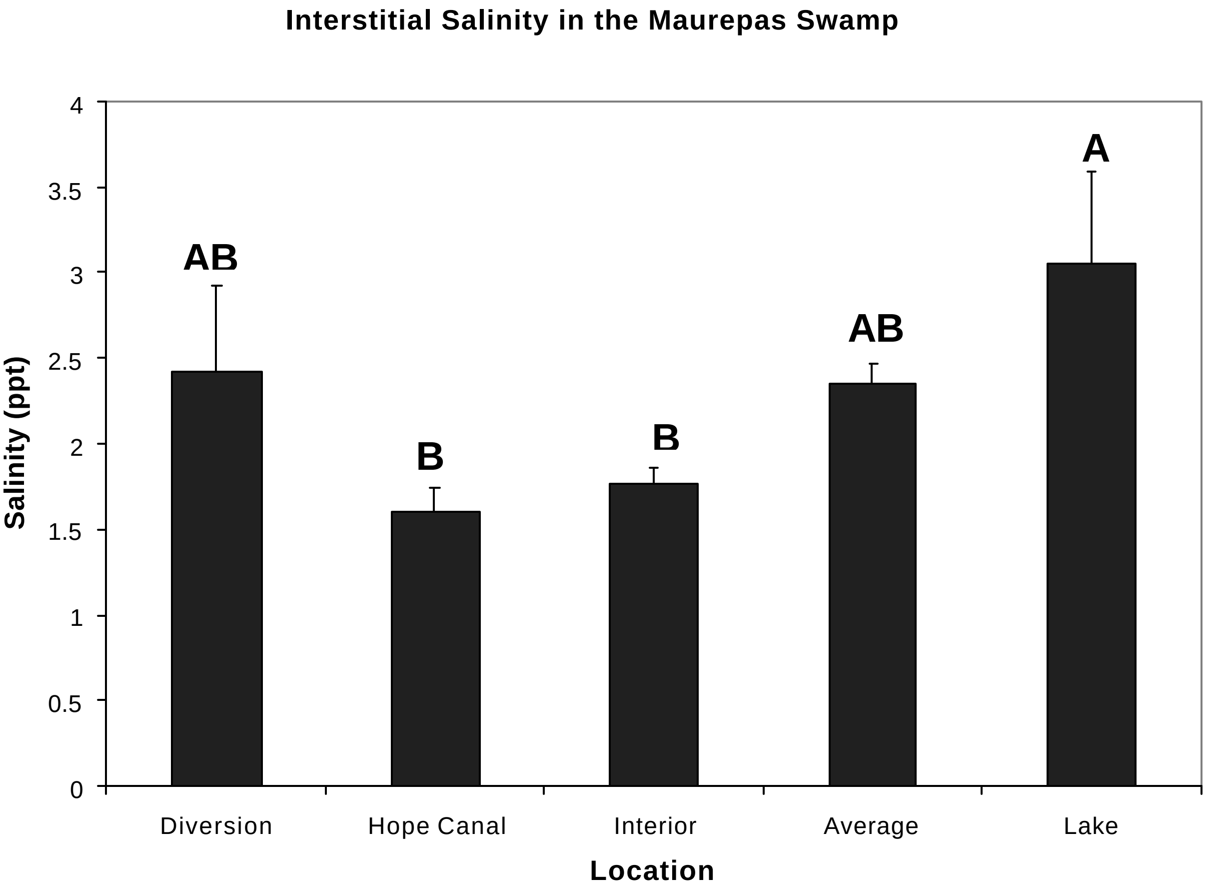

Figure 4. Summer interstitial soil water salinities at the different habitat types located in southern Lake Maurepas. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

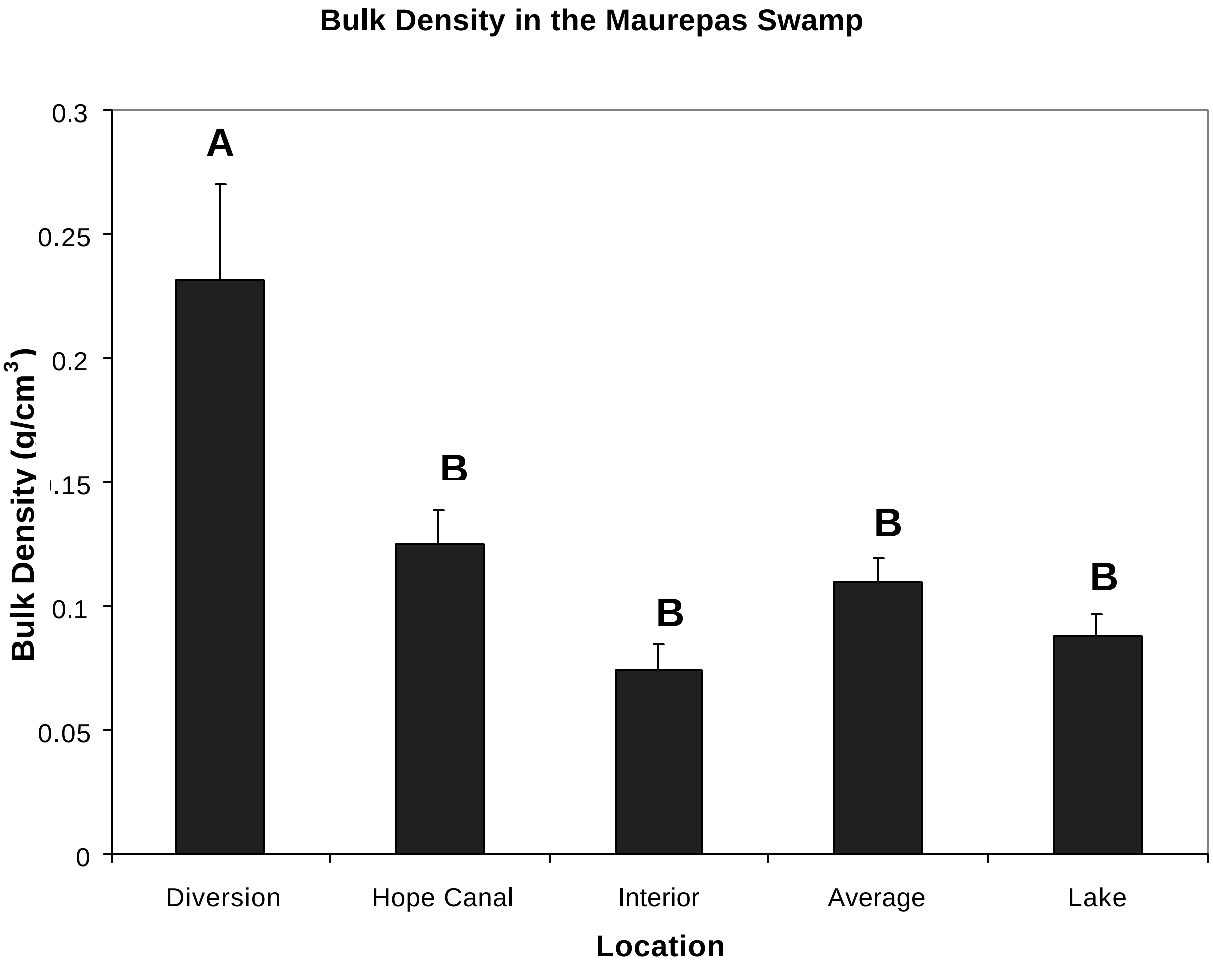

Figure 5. Bulk densities at the different habitat types located in southern Lake Maurepas. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

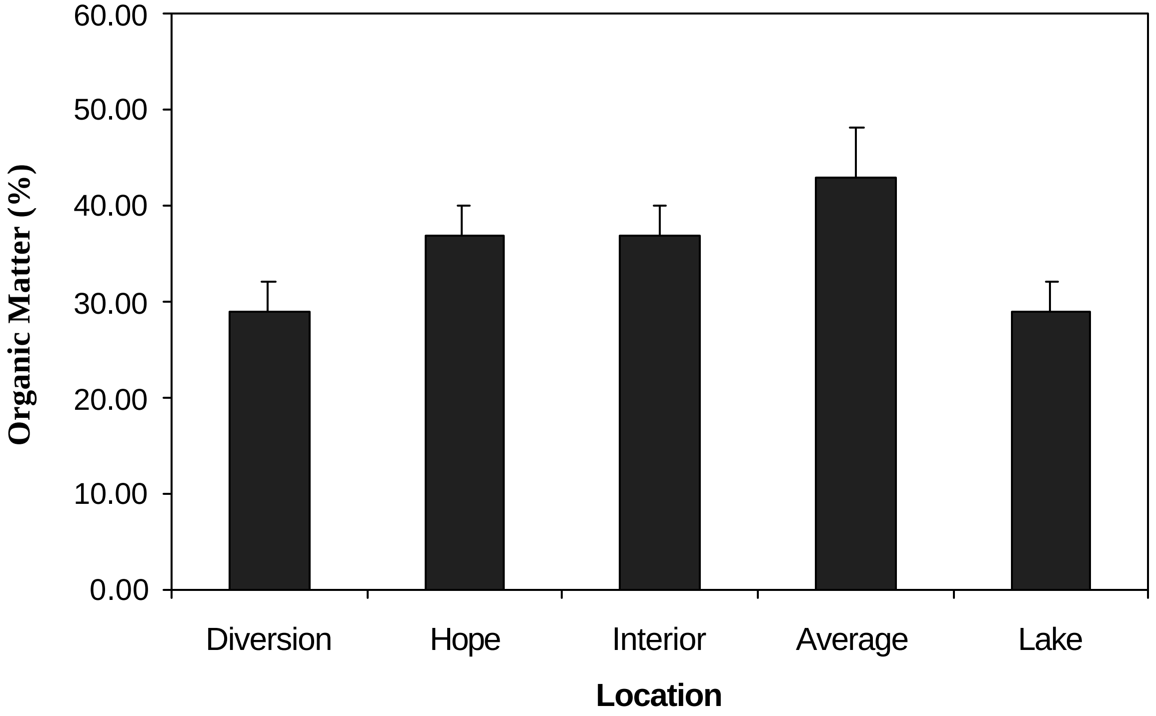

Figure 6. Effect of location on percent organic matter. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

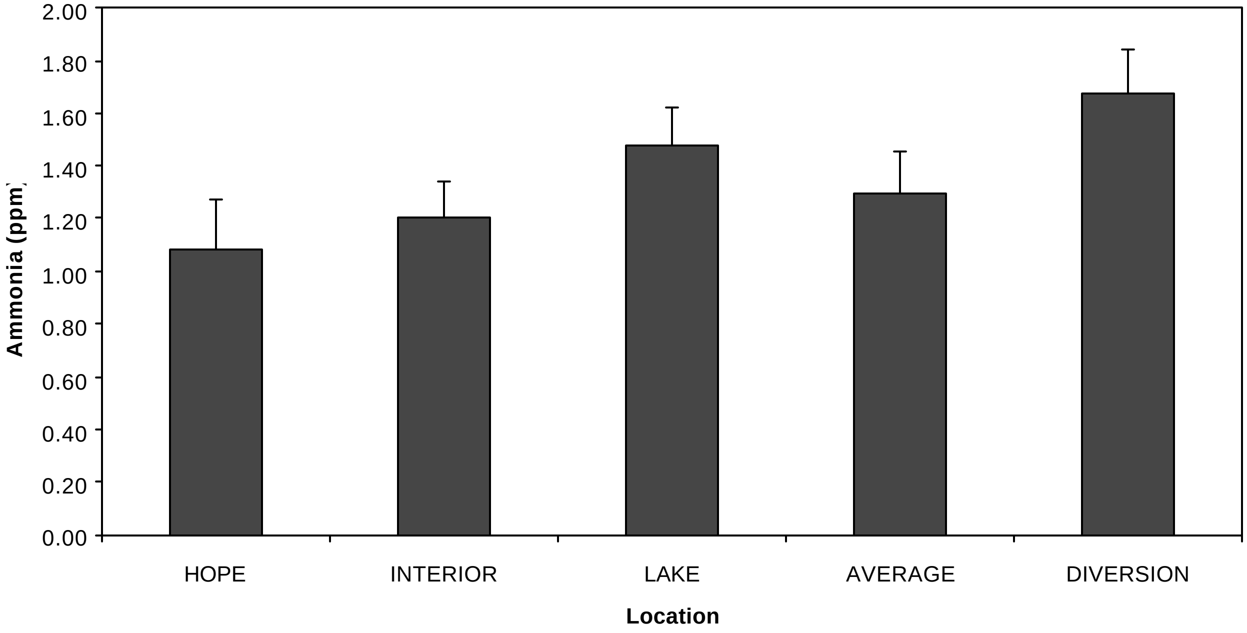

Figure 7. The effect of location on interstitial ammonium (mean +/- standard error) averaged across plot.

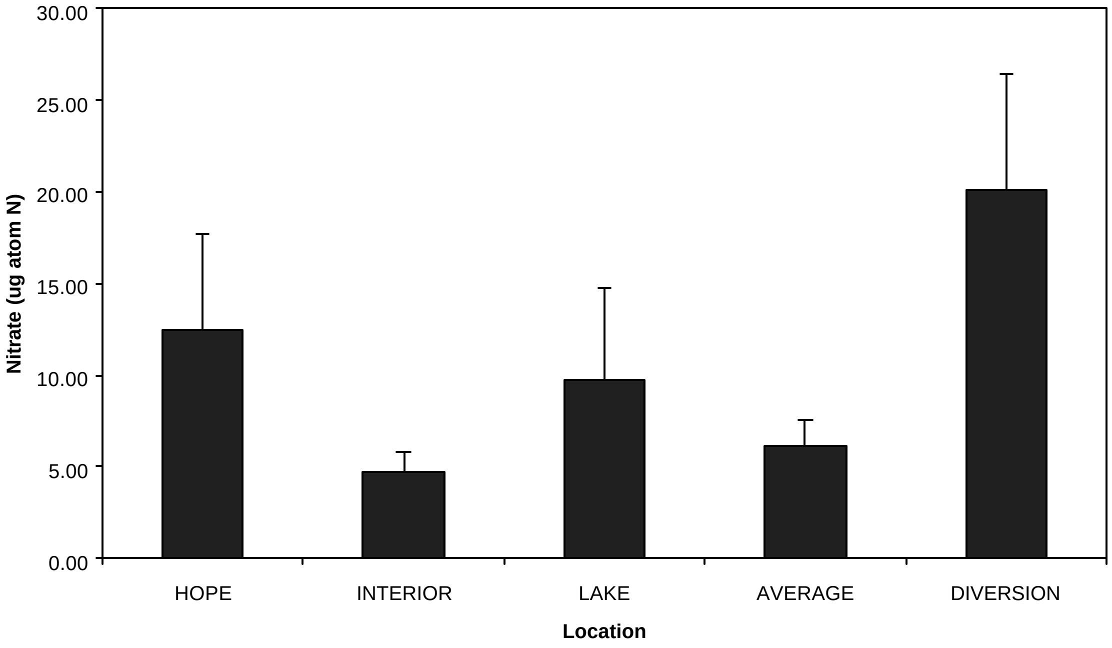

Figure 8. The effect of location on interstitial nitrate (mean +/- standard error) averaged across plot. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

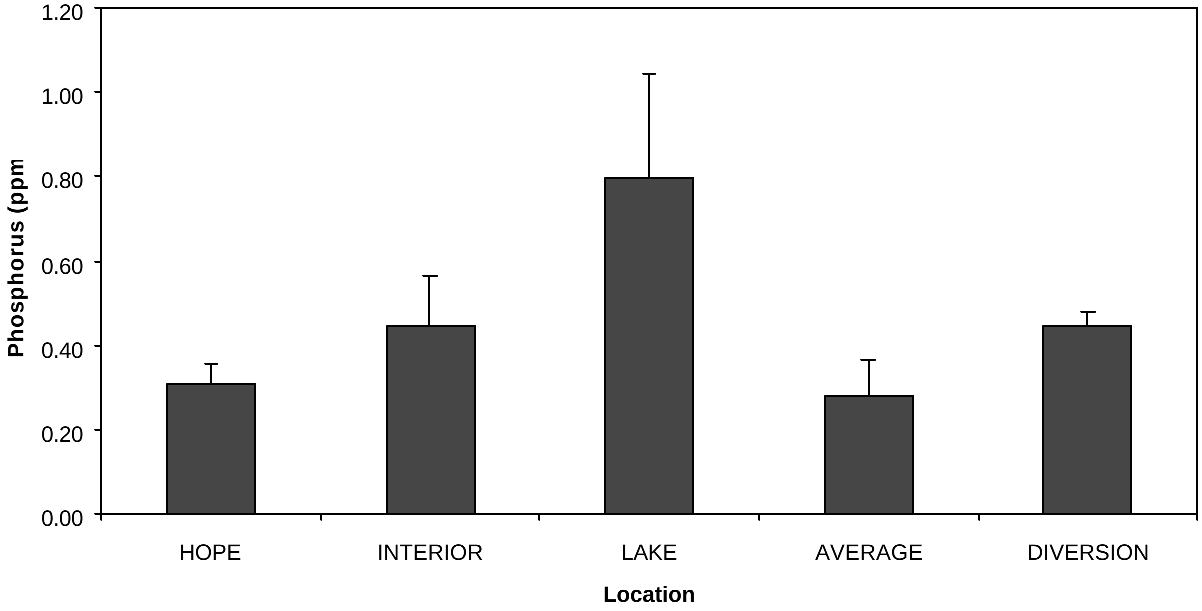

Figure 9. The effect of location on interstitial phosphorus (mean +/- standard error) averaged across plot.

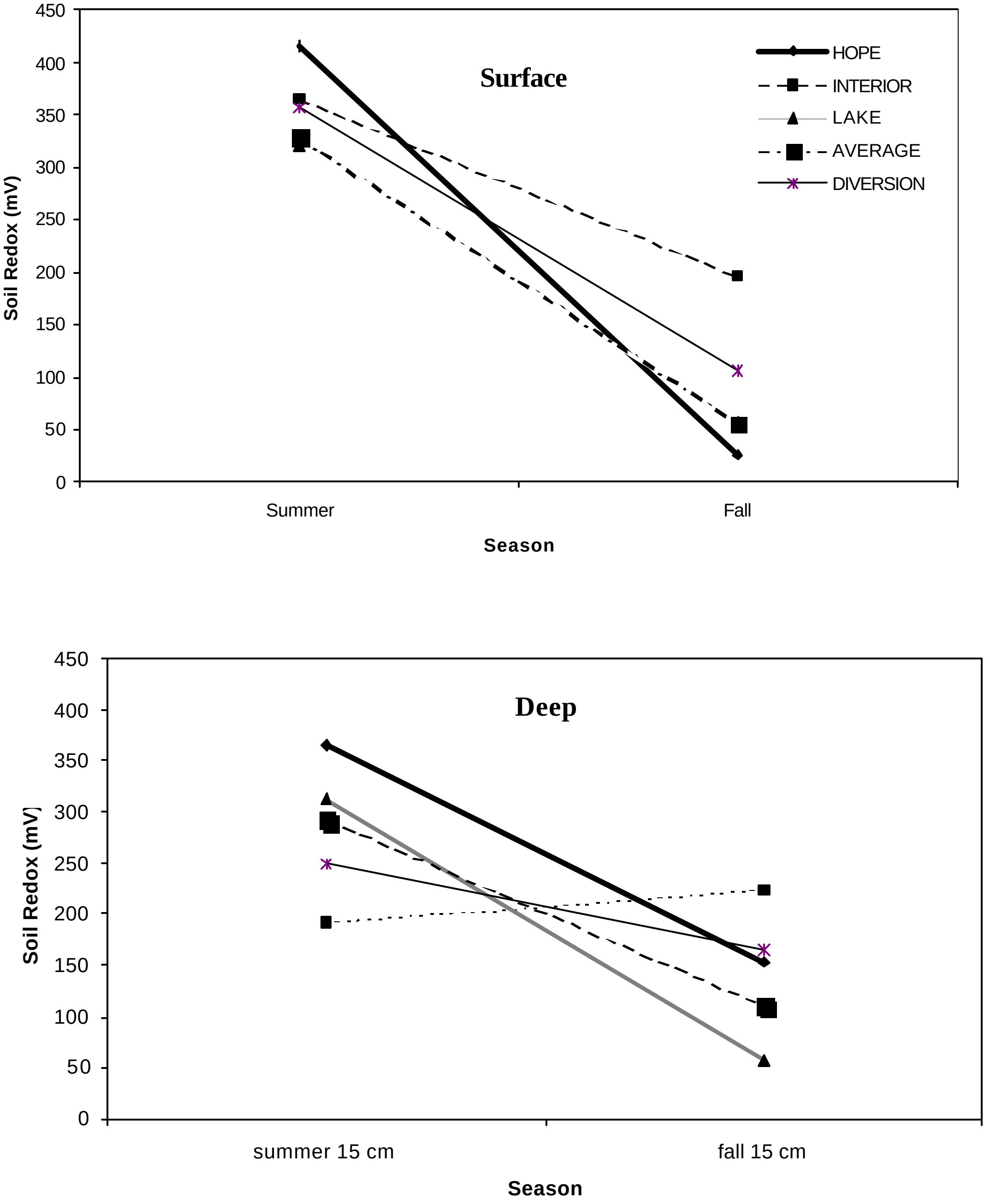

Figure 10. The effect of season and location on soil redox averaged across plot. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

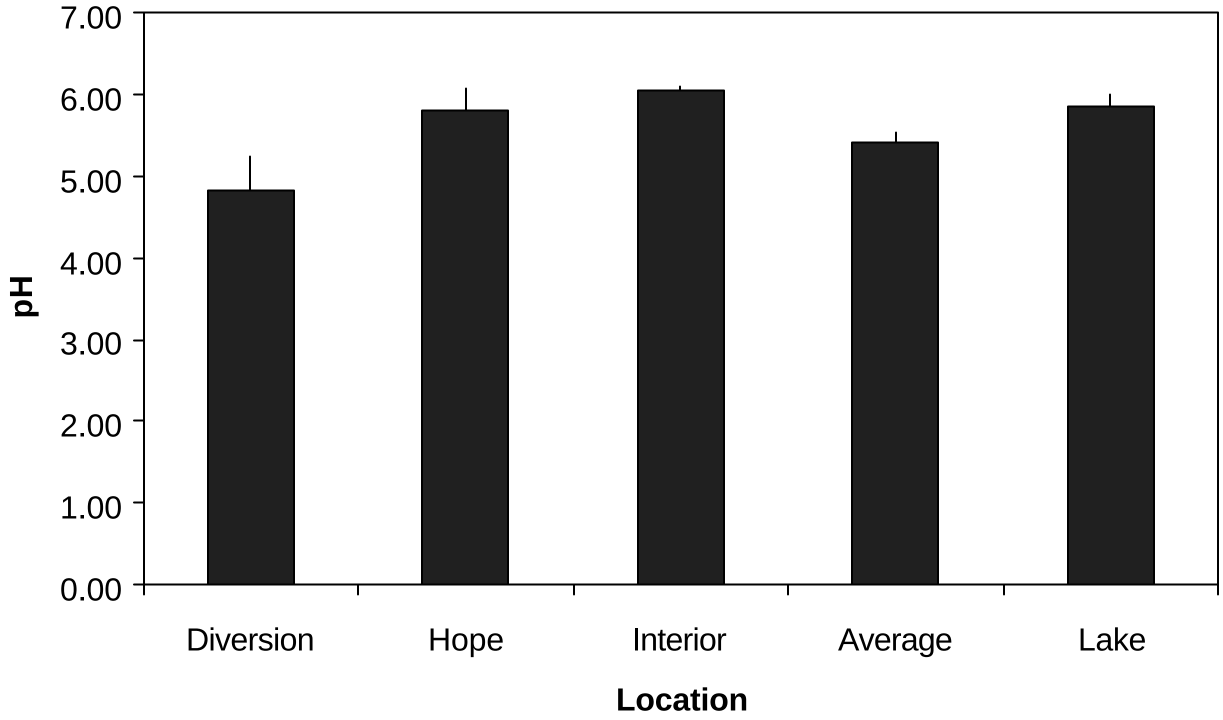

Figure 11. The effect of location on fall interstitial pH (mean +/- standard error) averaged across plot. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

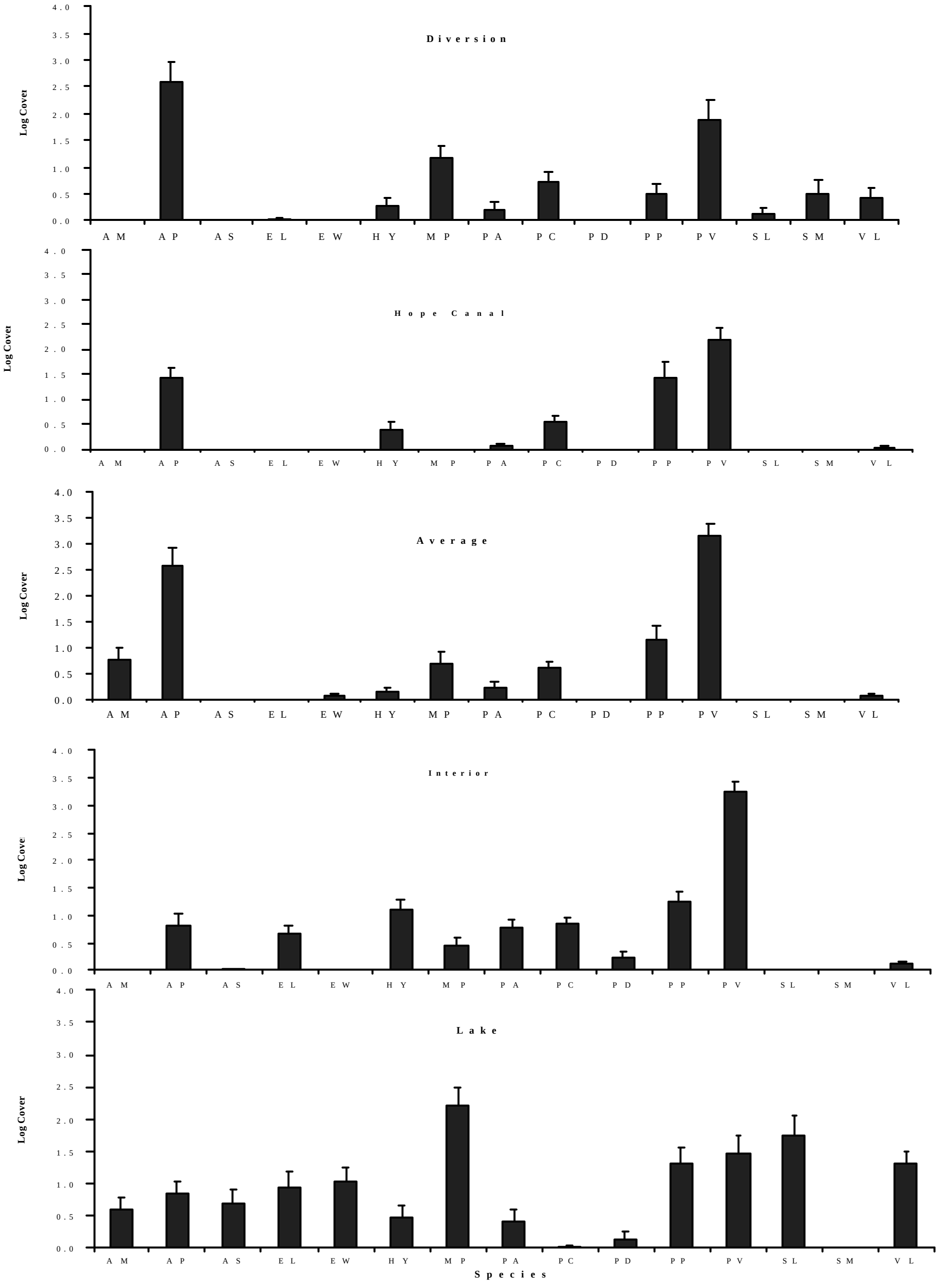

Figure 12. Early season cover values for the dominant herbaceous species (AM = *Amaranthus spp.*, AP = *Alternanthera philoxeroides*, AS = *Aster spp.*, EL = *Eleocharis spp*., EW = *Echinocloa walterii*, HY = *Hyrdocotle spp.*, MP = Marsh Parsely, PA = *Panicum spp*, PC = *Pontedaria cordata*, PD = *Panicum distichum*, PP = *Polygonum punctatum*, PV = *Peltandra virginica*, SL = *Sagittaria lancifolia*, SM = *Sabal minor*, VL = *Vigna luteola*. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

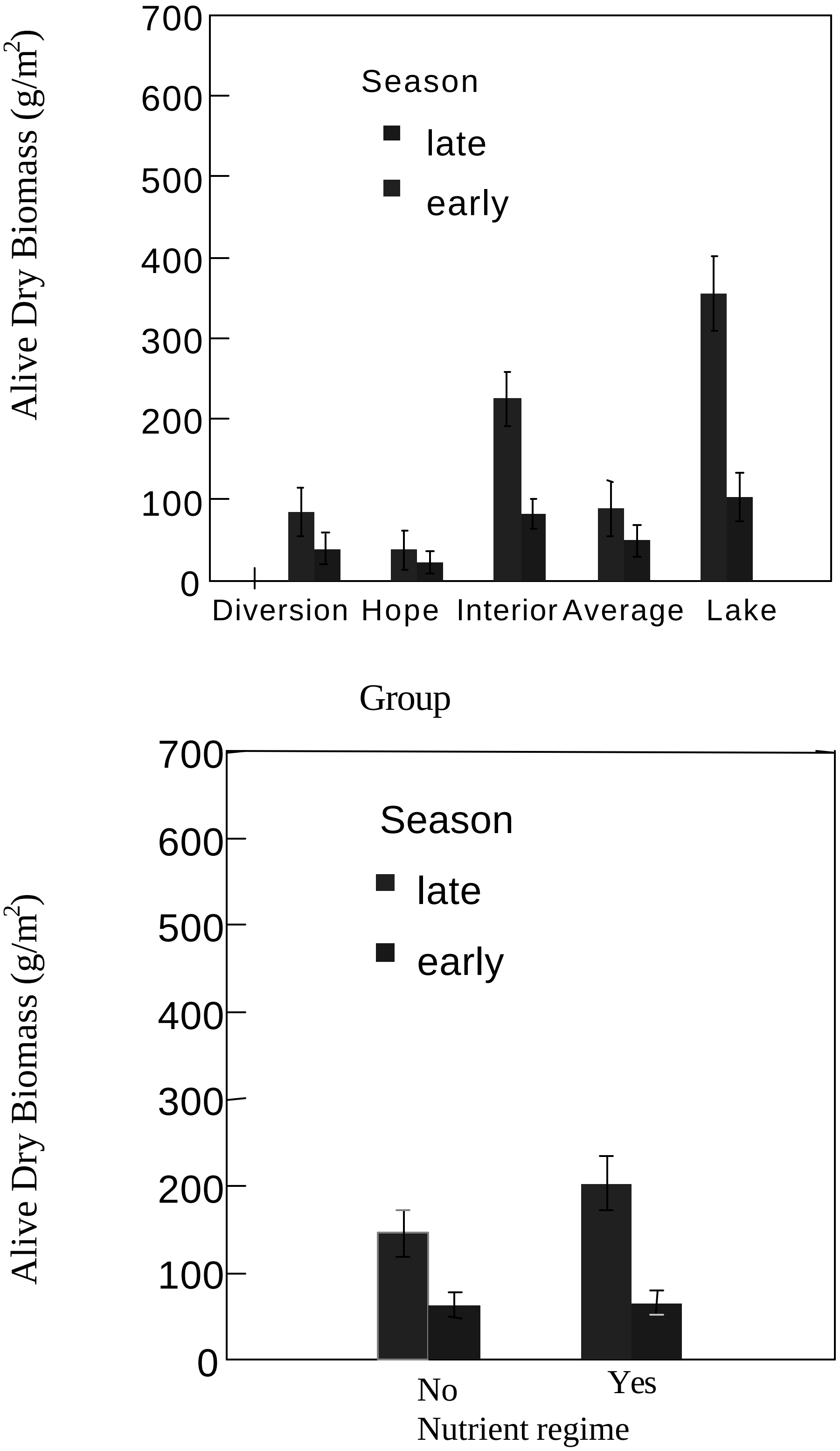

Figure 13. (a) Early and late season alive dry biomass of herbaceous vegetation collected at the 40 permanent stations grouped into five functional types. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

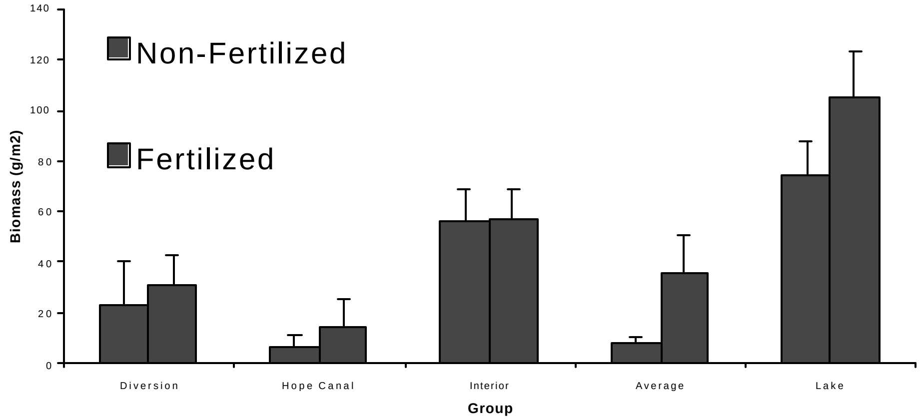

Figure 14. Interaction between nutrient augmentation and site grouping. Biomass increased up to six-fold at Average sites, whereas little effect occurred at Diversion and Interior sites. Groups on the X-axis are comprised of: Diversion (site 3, 4, 5), Hope (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 13), Average (sites 8, 14, 15, 16), and Lake (sites 17, 18, 19, 20).

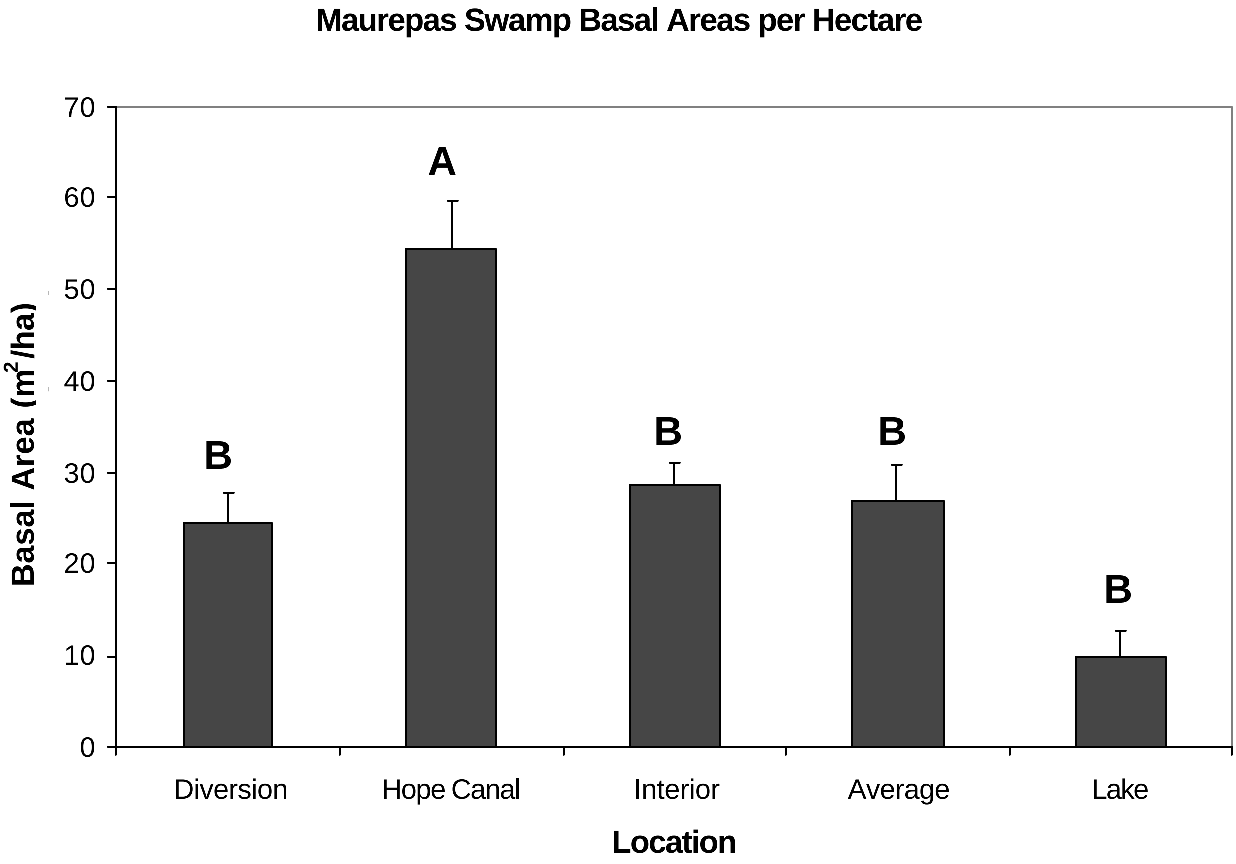

Figure 15. Basal area for all tree species combined. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

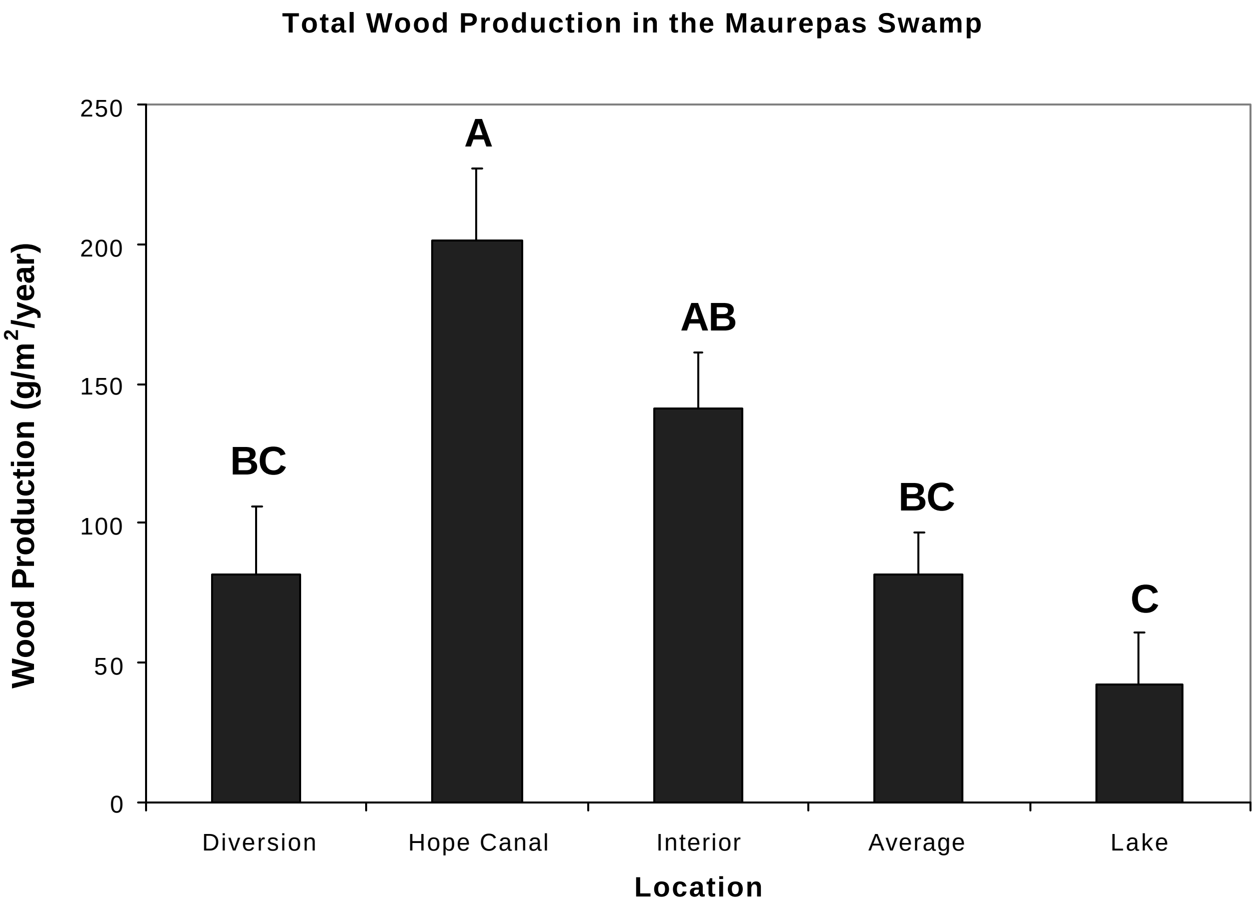

Figure 16. Wood production for all tree species combined. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

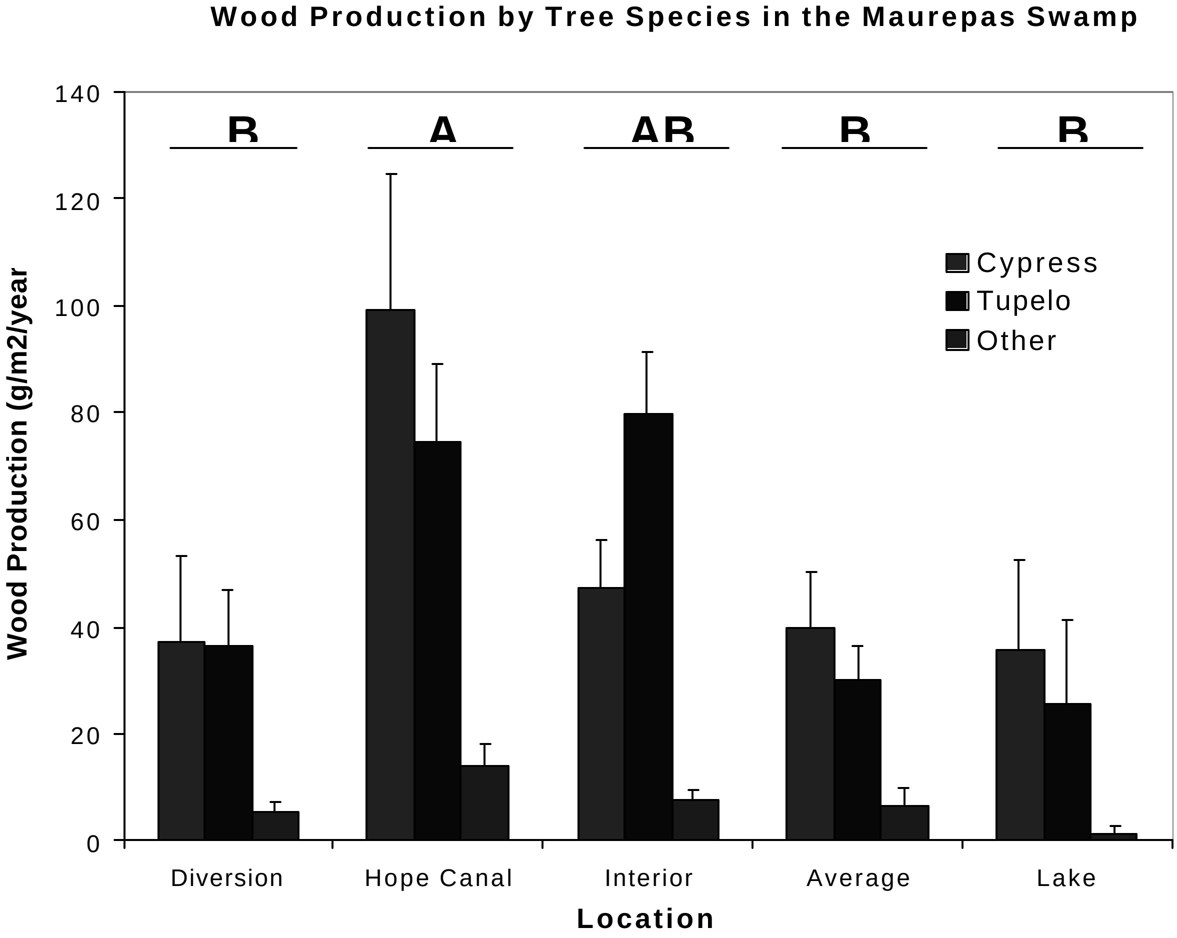

Figure 17. Annual wood production for each tree species (baldcypress, tupelogum, and "other"). Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

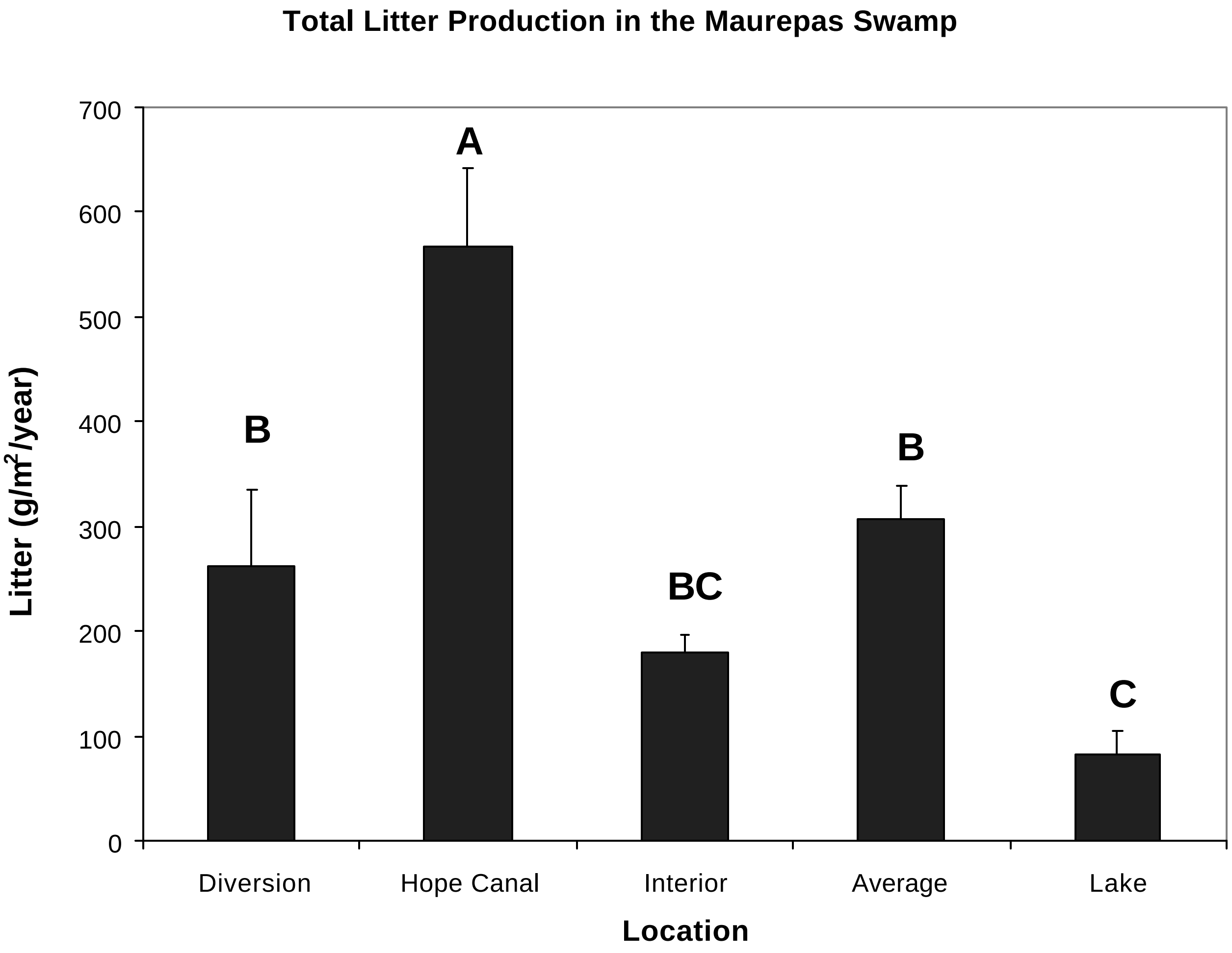

Figure 18. Total litterfall production for all tree species combined. Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

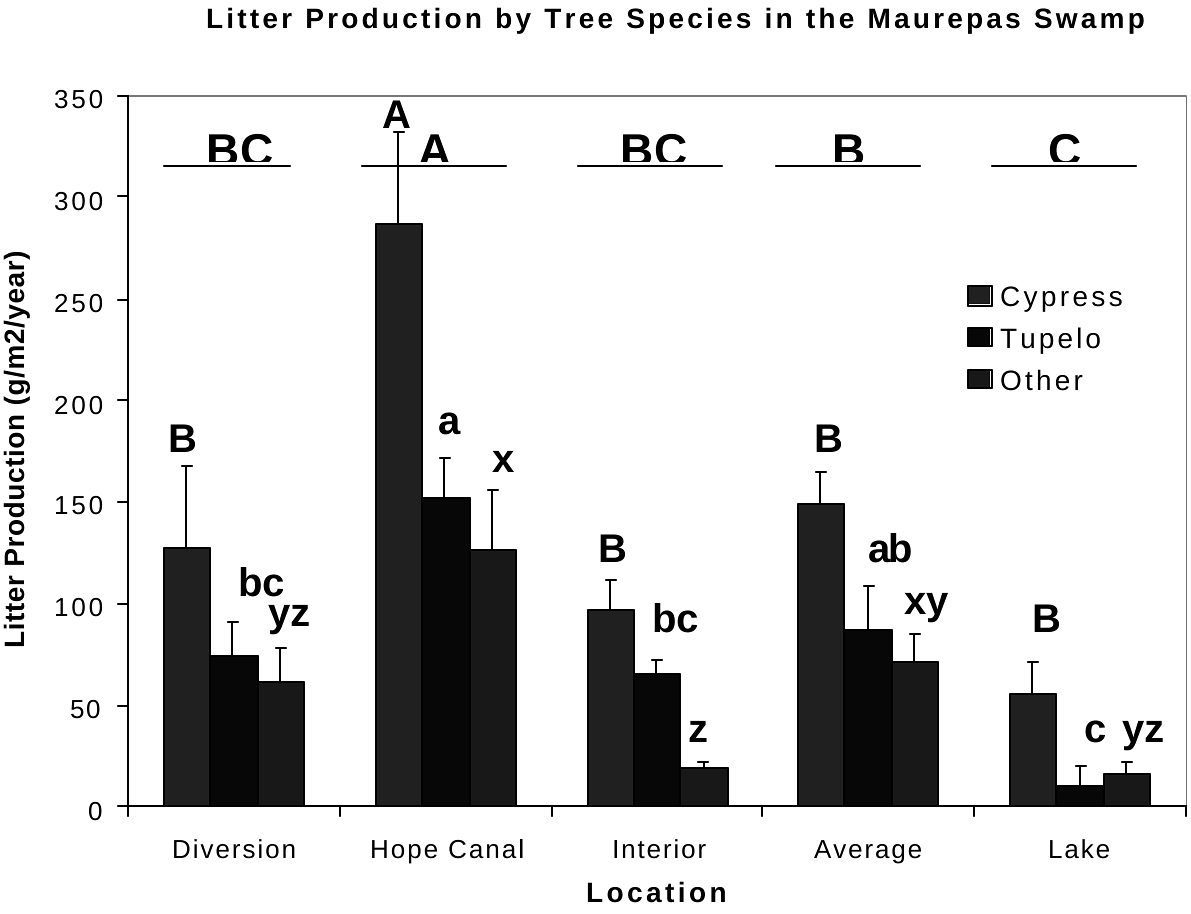

Figure 19. Annual litter production for each tree species (baldcypress, tupelogum, and "other"). Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

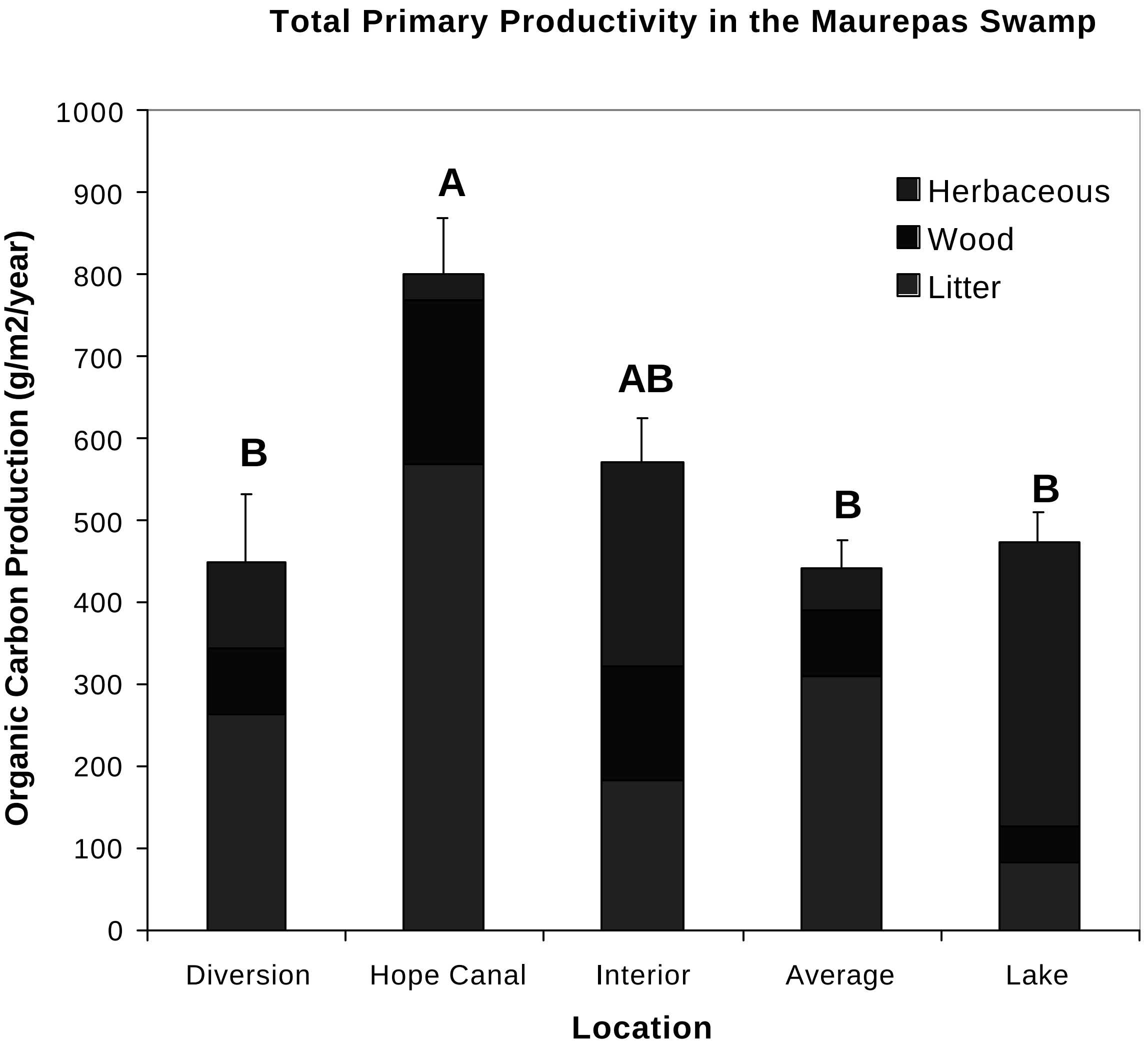

Figure 20. Annual dry weight production for each component of vegetation (herbaceous, woody, and tree litter). Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20). Bars that share letters are not significantly different from one another according to a Bonferroni-adjusted multiple comparison test.

Total Number of Trees Dying Between March and November 2000

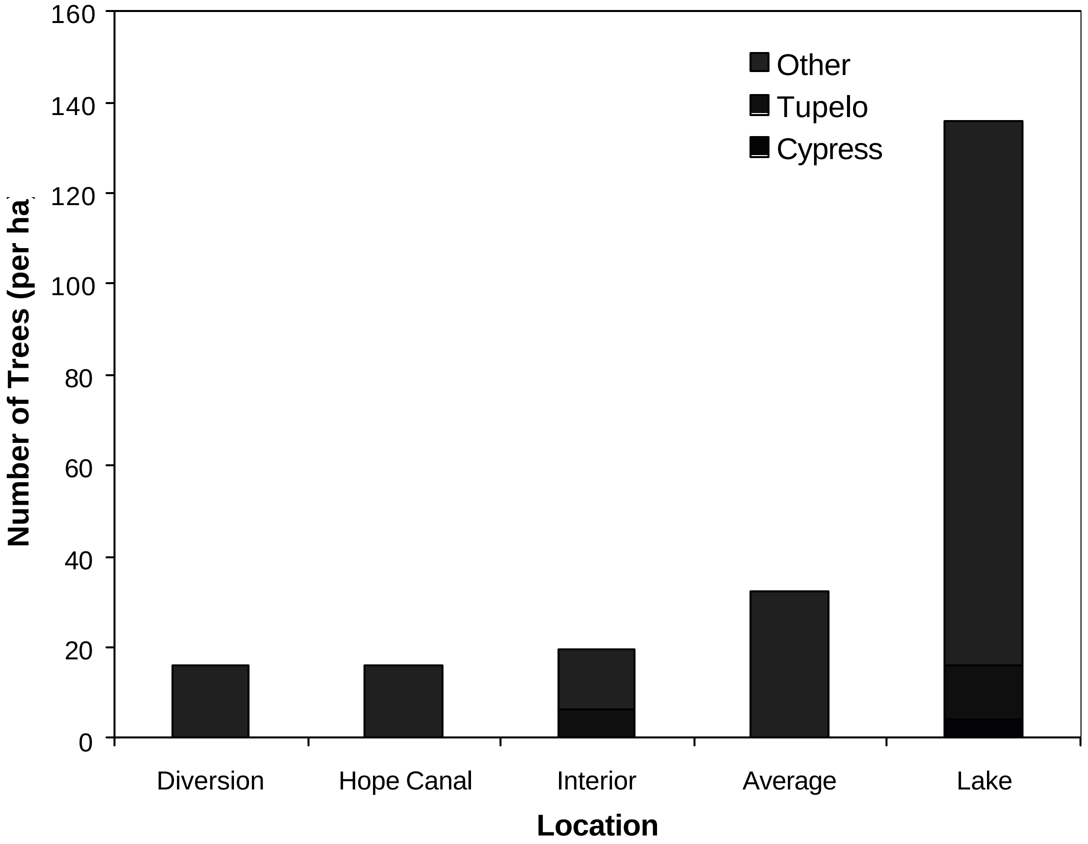

Figure 21. Total number of trees from each of the three species categories (baldcypress, tupelogum, and other). Groups on the X-axis are comprised of: Diversion (sites 3, 4, 5), Hope Canal (sites 10, 11, 12), Interior (sites 1, 2, 6, 7, 9, 13), Average (8, 14, 15, 16), and Lake (17, 18, 19, 20).

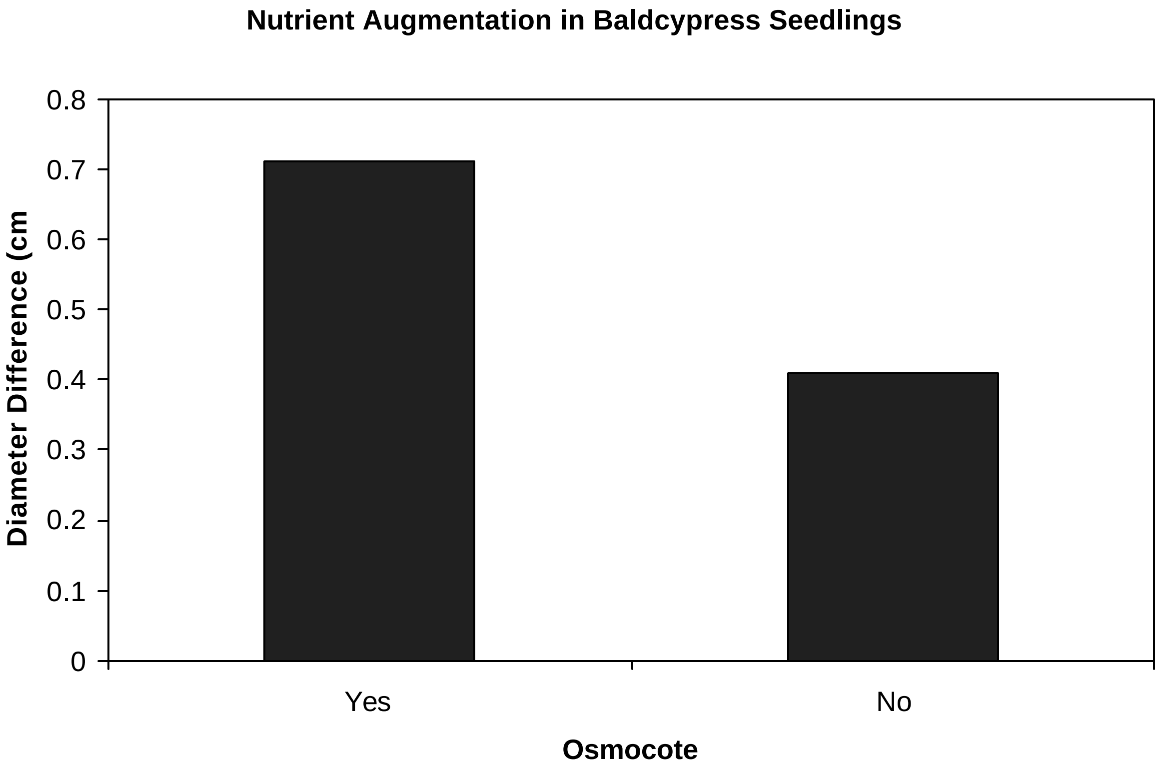

Figure 22. Diameter difference after one growing season of baldcypress seedlings fertilized (yes) and not fertilized with Osmocote timed-release 18-6-12 fertilizer (redrawn from Myers et al. 1995). Experiment conducted on the Manchac landbridge in Southeastern Louisiana.

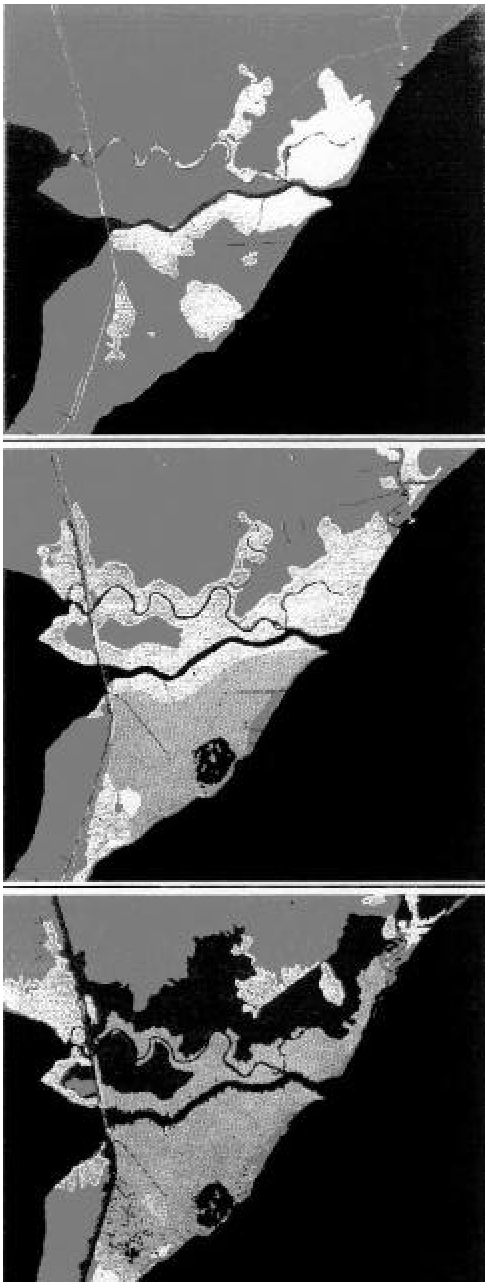

Figure 23. Chronology of the habitat change of the Manchac landbridge between 1956 and 1990 (redrawn from Barras et al., 1994). Top frame is during 1956, middle frame is from 1978, and bottom frame is from 1990. Black is open water (except for 1990 where some inland shrub/scrub is also black), dark gray is swamp, intermediate gray is intermediate marsh, and light gray is fresh marsh.