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Effects of white-tailed deer herbivory on the growth and survival of seedlings in a coastal wetland forest

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EFFECTS OF WHITE-TAILED DEER HERBIVORY ON THE GROWTH AND SURVIVAL OF SEEDLINGS IN A COASTAL WETLAND FOREST

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ABSTRACT

Studies in upland forests of the northeastern and upper mid-western U.S. indicate that high densities of white-tailed deer can reduce vegetation abundance, survival, and richness through over-browsing. In the southern U.S., few studies have examined the effects of deer herbivory on vegetation, and even fewer have done so in forested wetlands. At Jean Lafitte National Park's Barataria Preserve in south Louisiana, managers were concerned that white-tailed deer were concentrating and limiting forest regeneration near a walking trail, where hunting is not allowed. An exclosure study was started there in December 2002 and was conducted through July 2004 to quantify the effects of white-tailed deer on forest regeneration. Differences in densities and heights of naturally occurring tree and woody shrub species $\geq$ 15 cm but < 200 cm in height were compared between six pairs of fenced and unfenced plots under the forest canopy. *Fraxinus pennsylvanica* and *Quercus nigra* juveniles also were planted in these plots, and survival and growth were compared between treatments. Naturally occurring shrub and juvenile tree abundance was compared among plots in treefall gaps and the paired plots under the forest canopy. White-tailed deer decreased the survival of planted *Fraxinus pennsylvanica* juveniles, but did not affect planted *Quercus nigra* juveniles or naturally occurring shrubs and juvenile trees. Juvenile trees were ten times more dense in treefall gaps than under the canopy because of the dominance of the exotic *Triadica sebifera* in gaps. Gap disturbances may be reducing diversity in these coastal wetland forests, rather than promoting diversity as they do in other forests. A more complete understanding of how deer modify the landscape may require future exclosure studies in treefall gaps.

INTRODUCTION

More than seventy-five percent of the wetlands in the southern U.S. are forested, including sixty-two percent in Louisiana (Shepard et al. 1998). Much of Louisiana's forested wetlands lie in the Mississippi river floodplain, thus they are influenced by the hydrologic and geomorphic processes of river systems. Major floodplains exist because upstream soils erode and are deposited downstream as the water loses turbidity (Kellison et al. 1998). This land building process causes the river to meander as it seeks the outlet of least resistance (Kellison et al. 1998). Major floodplains therefore are relatively flat (Kellison et al. 1998), but the subtle changes in topography, soil characteristics, soil drainage, etc., are associated with changes in the dominant plant communities (Patrick et al. 1981). Altering the hydrologic flows within these floodplains can have profound impacts on the structure and functions of these forested wetlands. Local hydrologic conditions (including flooding), soil properties, and light availability to seedlings (a result of structure) are some of the conditions that change when flow is altered.

Regeneration, which can be defined as the establishment, growth, and survival of seedlings into the sapling size class, is ultimately affected by flooding, soil, or light conditions. These three conditions can each affect the establishment, growth, or survival of seedlings in some way.

Flooding creates anaerobic soil conditions (Sharitz and Mitsch 1993). The duration, intensity, and timing of floods influence species composition and influence ecosystem structure and function (Wharton et al. 1982, Sharitz and Mitsch 1993). While winter and spring flooding from the river can promote forest productivity by depositing nutrient rich sediments, atypical floods during the growing season can have a greater

effect on species survival and ecosystem productivity (Sharitz and Mitsch 1993). Many tree species exhibit some degree of relative flood tolerance, but only a few species can live in swamps (Patrick et al. 1981). As stands become more permanently inundated, the number of tree and shrub species that can establish and grow will decrease (Patrick et al. 1981).

Flooding is increasing in coastal wetland forests in the Lower Mississippi Alluvial Valley (LMAV) because of recent landscape level and local changes in hydrologic conditions. Relative sea-level is increasing because of global sea-level rise (Titus et al. 1991) and coastal subsidence (Penland and Ramsey 1990). Subsidence rates are greater in coastal Louisiana's Barataria Basin (a part of the LMAV) than elsewhere in the United States, and the frequency, depth, and duration of its floods are increasing (Conner and Day 1988). This rapid subsidence has resulted in an apparent water level rise in the Barataria Basin of 8.5 mm/year from 1956 through 1986 (Conner and Day 1988).

Subsidence causes the plant communities in coastal bottomland hardwood forests to progress differently from the plant communities in other bottomland hardwood forests. In the Barataria Basin, evidence of increasing soil saturation can be seen in the presence of saplings of *Liquidambar styraciflua* L., *Quercus nigra* L., and *Ulmus americana* L. (botanical nomenclature follows Hardin et al. 2001 unless otherwise noted) at higher elevations than adult trees of the same species (Denslow and Battaglia 2002). Throughout the Barataria Basin, many tree species are suffering from severe flood stress (Conner et al. 1981, Conner and Day 1987). In a study by Conner and Day (1988), standing water prevented recruitment of new individuals into forested stands and allowed only *Cephalanthus occidentalis* L. (botanical nomenclature follows Godfrey and Wooten

1981) to germinate. The increasingly saturated soils in the Barataria Basin might be too stressful for most seedlings during the growing season.

Sharitz and Mitsch (1993) list aeration, texture, and nutrient content as some of the most important soil properties for bottomland hardwood forests. In relation to rooted plants, extended periods of anaerobic soil prevent some plants from functioning properly, while others have adaptations that enable survival under poor soil aeration (Sharitz and Mitsch 1993). Soil texture, which affects soil moisture, has been shown to influence species composition (Marks and Harcombe 1981). Nutrient content usually is high in southeastern floodplain soils because of higher organic matter content (Patrick 1981), higher clay content (Patrick 1981), and continual replenishment through riverine deposit (Sharitz and Mitsch 1993). With the Mississippi river no longer flooding much of the Barataria Basin, however, its soils may lack many of the nutrients that were historically deposited with sediments.

Sunlight could be another factor limiting the establishment, growth, and survival of seedlings in forests that have closed canopies. For example, Hall and Harcombe (1998) have shown that light availability influences the distribution of saplings within bottomland hardwood forests. The density of the canopy influences light availability. When branches or trees fall, light gaps are created in the canopy. Forest structure, forest composition, and flooding patterns are some of the factors that affect gap formation patterns in bottomland hardwood forests (King and Antrobus 2001).

Gaps promote tree species diversity in some forests by providing different light levels that benefit different tree species (Denslow 1980, Denslow 1987). Smaller gaps mostly benefit shade-tolerant species, but larger gaps allow shade-intolerant species to

persist in mature forests (Denslow 1987, Spies and Franklin 1989). For shade-intolerant species to establish and grow into saplings, canopy gaps larger than those caused by single tree mortality may be needed (Sharitz and Mitsch 1993).

Shade-tolerant species can become established and grow under closed canopies, but most ultimately need gaps to ensure their survival. These species grow very slowly as juveniles (Canham 1988, Streng et al. 1989, Jones and Sharitz 1998). They can endure as stunted seedlings or perennial rootstocks that survive while their tops are repeatedly killed back (Smith 1962). When the wind or other disturbances open the canopy above these slow growing juveniles, they are "released" to rapid height growth. These species are known as "advance-growth-dependent" species (Smith 1962). An example comes from Jones and Sharitz (1998), who noted the unlikeliness of juvenile trees in their study sites reaching the sapling size class without the presence of treefall gaps. Virtually all of the juvenile trees they measured were < 30 cm tall, even if they were > 8 years old. These "advance-growth-dependent" species can dominate the canopy replacement process of many mature forests (Smith 1962, Hartshorn 1978, Denslow 1980, Woods 1984, Connell 1989, Silvertown and Lovett Doust 1993). In mature coastal forests, disturbances such as hurricanes serve an important role in maintaining the canopy openness needed to release these species (Battaglia et al. 1999).

Some forests of the Barataria Basin have dense canopies and extensive understory palms, *Sabal minor* (Jacq.) Pers. (botanical nomenclature follows Godfrey and Wooten 1979), that shade the forest floor. When combined with flooding, low light conditions often are too severe for most species to tolerate (Menges and Waller 1983, Hall and

Harcombe 1998). The advance-growth species in the forests of the Barataria Basin may suffer from a combination of flood and shade stress.

Herbivory is another factor that can affect forest regeneration. Herbivores may impact the establishment of seedlings through seed consumption, but their effects on the growth and survival of juvenile trees is better documented (Russell et al. 2001). In bottomland hardwood forests throughout the southeast, the effects of white-tailed deer on the growth and survival of shrubs and juvenile trees are of increasing concern because of increasing deer densities (Castleberry et al. 2000), but no data are available to judge the magnitude of there effect (Russell et al. 2001).

The arrival of railroads and sawmills in Louisiana between 1890 and 1930 left most upland pine-hardwood and virgin cypress swamps stripped barren (St. Amant and Perkins 1953a). When combined with unregulated hunting, this reduction in habitat led to an all-time low estimate of 20,000 white-tailed deer in Louisiana between 1920 and 1925 (St. Amant and Perkins 1953a). Following this period, Louisiana's deer population increased as logged forests regenerated and more stringent conservation laws were enacted (St. Amant and Perkins 1953a). The state initiated a deer management program in the 1940's (Moreland 1996), and by 1952, the white-tailed deer population in Louisiana was estimated to be 72,000 (St. Amant and Perkins 1953a). The state also began a trapping and restocking program in the early 1950's, and by the 1960's legal hunting of males was reestablished (Moreland 1996). Even after populations increased and female harvest was resumed, many hunters were reluctant to harvest females, and by the 1970's many private lands were overpopulated (Moreland 1996). While St. Amant and Perkins (1953b) estimated Louisiana could support a maximum deer population in

excess of 230,000, the state's population less than 50 years later exploded to an estimated 750,000 to 1,000,000 white-tailed deer (Moreland 1996).

Other states from New York to Alabama have seen similar trends in white-tailed deer populations since the early 20th century (Trefethen 1970, McCabe and McCabe 1997, Rutberg 1997). McCabe and McCabe (1984) credit the large scale expansion of agriculture, the eradication of predators, and effective laws that govern hunting for the increased deer populations in the eastern U.S. Increased populations have resulted in higher densities of white-tailed deer in the early 2000's than during the 1880's and 1900's, and therefore the effects of white-tailed deer on forest ecosystems may be greater now than previously.

High densities of white-tailed deer can have profound negative effects on the deer themselves (Eve 1981), on other animal communities (deCalesta 1994, McShea and Rappole 1997), and on vegetative communities (Harlow and Downing 1970, Marquis and Brenneman 1981, Tilghman 1989, Alverson and Waller 1997, Healy 1997). Species richness and abundance of ground and intermediate canopy-nesting songbirds can decline where excessive deer browsing alters understory vegetation (deCalesta 1994, McShea and Rappole 1997) Russell et al. (2001) reviewed studies that relate deer with changes in tree species composition, reductions in plant growth and survival rates, and changes in plant morphology. Many studies, however, detected no effects of deer herbivory on plant survival and fecundity, or have found that these effects depend on the timing and intensity of tissue removal (Russell et al. 2001). Exclosure studies have shown that white-tailed deer can change the dominant tree species in the sapling layer, can decrease

tree regeneration, and can decrease species richness in the understory (Russell et al. 2001).

Most studies documenting the effects of deer on plant growth and survival were conducted in white pine-hemlock-northern hardwood forests, in maple-basswood forest fragments, and in old fields in Minnesota and Virginia; none have been conducted in coastal wetland forests (Figure 1). It is possible the additional stress of soil saturation in coastal wetland forests could make some tree and shrub species less tolerant of herbivory. To better understand the interactions between deer and vegetation, more studies need to be conducted in a variety of plant communities throughout the range of white-tailed deer (Russell et al. 2001).

At the Barataria Preserve, which is in the Barataria Basin, managers were concerned with forest regeneration in a portion of the park where hunting is prohibited. Few juvenile trees are present in the understory. Managers believe deer concentrate here during the winter because of hunting pressure in surrounding areas.

In most undisturbed bottomland hardwood forests, shade tolerant juvenile trees would be present throughout the understory in advance growth form (Smith 1962, Jones et al. 1994, Battaglia et al. 1999). At the Barataria Preserve, size-class distributions of tree species show many juvenile size classes are missing (Denslow and Battaglia 2002). Excessive deer browsing is one possible cause for this absence of juvenile size classes.

The objective of this experiment was to determine the effects of deer herbivory on the growth and survival of shrubs and juvenile trees at the Barataria Preserve. Although the growth and survival of shrubs and juvenile trees are most likely limited by a combination of the previously discussed biotic and abiotic factors, this experiment

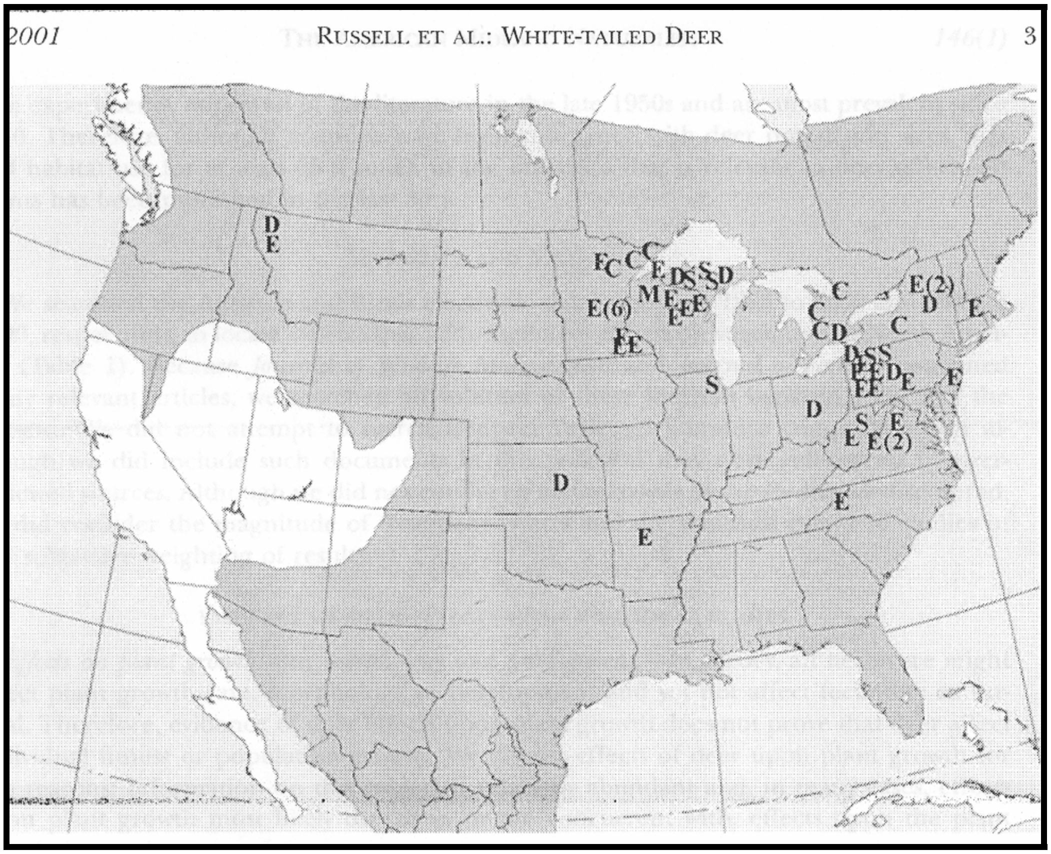

Figure 1. Locations of previous studies examining the effects of white-tailed deer on vegetation in the United States.

attempted to hold variables other than deer herbivory constant through careful study site selection and experimental design. I hypothesized deer herbivory would have no effect on the growth and survival of shrubs and juvenile trees at the Barataria Preserve. If white-tailed deer were impacting the growth and survival of shrubs and juvenile trees, reducing deer densities in the area could be a tool to increase regeneration.

STUDY AREA

The Barataria Preserve is 1 of 6 sites in south Louisiana that make up Jean Lafitte National Historical Park and Preserve. It borders Lake Salvador and Bayou Barataria, and encompasses more than 8000-ha of bottomland hardwood forest, cypress swamps, and freshwater marsh. The preserve lies on sediment deposited by the Mississippi River through Bayou de Familles. Bayou de Familles carried sediments until about 200 AD, but then flow naturally reduced (Muth 1991). Man-made levees built in the early 20^{th} century then prevented further sediment deposition from the river (Muth 1991). Local rainfall (1572 mm/yr in New Orleans), evapotranspiration, and drainage into Bayou Barataria determine the frequency, timing, depth, and duration of floods at the Barataria Preserve (Denslow and Battaglia 2002).

The study area is in Big Woods, an area of the Barataria Preserve where hunting is prohibited. Much of this area, which sits on the natural levee of Bayou de Families, was a sugarcane plantation until the early 1900s. Big Woods is now a mature bottomland hardwood forest. A cleared walking path known as the plantation trail provides visitors the opportunity to view Big Woods. The canopy is dense throughout most of the study area except in treefall gaps. Ridge-swale topography is prevalent throughout the landscape and underlies significant variation in plant communities. The tops of ridges are dominated by *Quercus virginiana* Mill. *Acer negundo* L., *Acer rubrum* L., *Carpinus caroliniana* Walt., *Celtis laevigata* Willd., *Fraxinus pennsylvanica* Marsh., *L. styraciflua, Q. nigra, Quercus texana* Buckley, *U. americana,* and others can be found in the overstory. *Craetagus sp.* L., *Ilex decidua* Walt. (botanical nomenclature follows Godfrey and Wooten 1981), and others can be found in the midstory. *S. minor*, an understory

palm, is abundant throughout Big Woods. Vegetation is changing, however, because of the rapid subsidence and subsequent increased flooding in the region (Denslow and Battaglia 2002). The exotic *Triadica sebifera* (L.) Small (botanical nomenclature follows Esser 1999) also appears to be invading the study area (Denslow and Battaglia 2002). Most of the soil in Big Woods is Sharkey clay (Very-fine, montmorillonitic, nonacid, thermic Vertic Haplaquepts) (USDA 1983).

METHODS

Six pairs of fenced and unfenced plots were used to isolate the effects of deer herbivory from other factors that affect the growth and survival of shrubs and juvenile trees. In September 2002, a vegetation survey was conducted at 2 random locations in Big Woods using nested plots to determine optimum plot size. Seedling densities were compared among five different plot sizes at each location (Figure 2). The results prompted the use of 100-m^2 plots (10m x 10m).

I established permanent study sites at six locations. To reduce variability among sites in hydrologic conditions, soil properties, light availability, and thus, vegetation, sites were located along a *Q. virginiana* ridge that stretches across the center of Big Woods (Figure 3). Because flooding is believed to be one of the major factors limiting seedling establishment, growth, and survival, it seemed logical to study the effects of herbivory on the natural ridge where flooding is least. I excluded swales from this study because I assumed shrub and juvenile tree densities there were limited by flooding rather than white-tailed deer herbivory.

To select the sites, 6 temporary points were randomly selected along the plantation trail. From each point, I walked perpendicular to the trail until I reached the ridge, as indicated by an abundance of *Q. virginiana*. Sites were established at these locations. A minimum of 100-m was afforded between each site.

At each site, the 4 corners of 2 plots were temporarily marked. The 2 plots were spaced approximately 15 m apart. On December 19, 2002, a coin toss determined which plot would be fenced at each site. The exclosures (fenced plots) were then constructed using eight feet tall game fence (Marquis and Brenneman 1981). The corners of

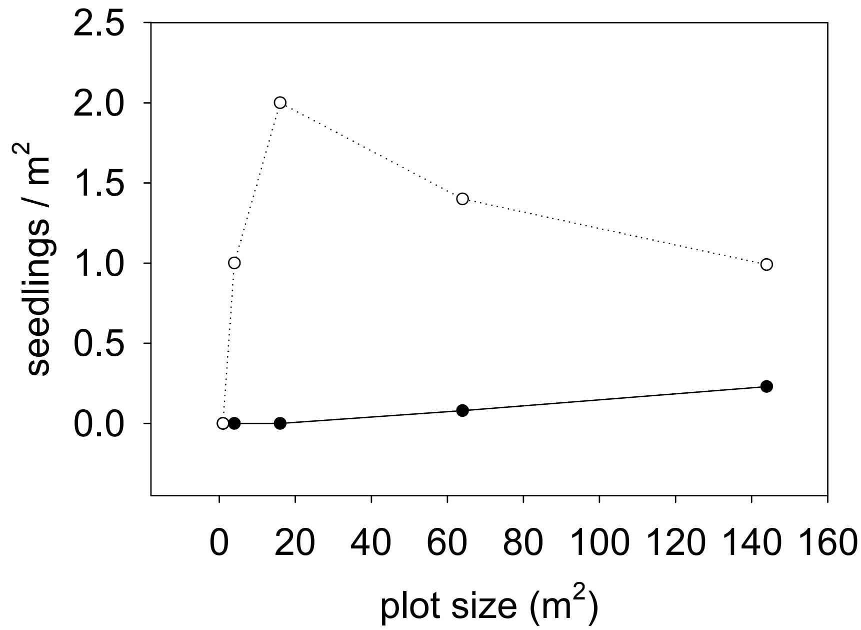

Figure 2. Relationship between plot size and woody seedling abundance (< 200 cm) in a coastal wetland forest at Jean Lafitte National Historical Park and Preserve's Barataria Preserve, Louisiana USA.

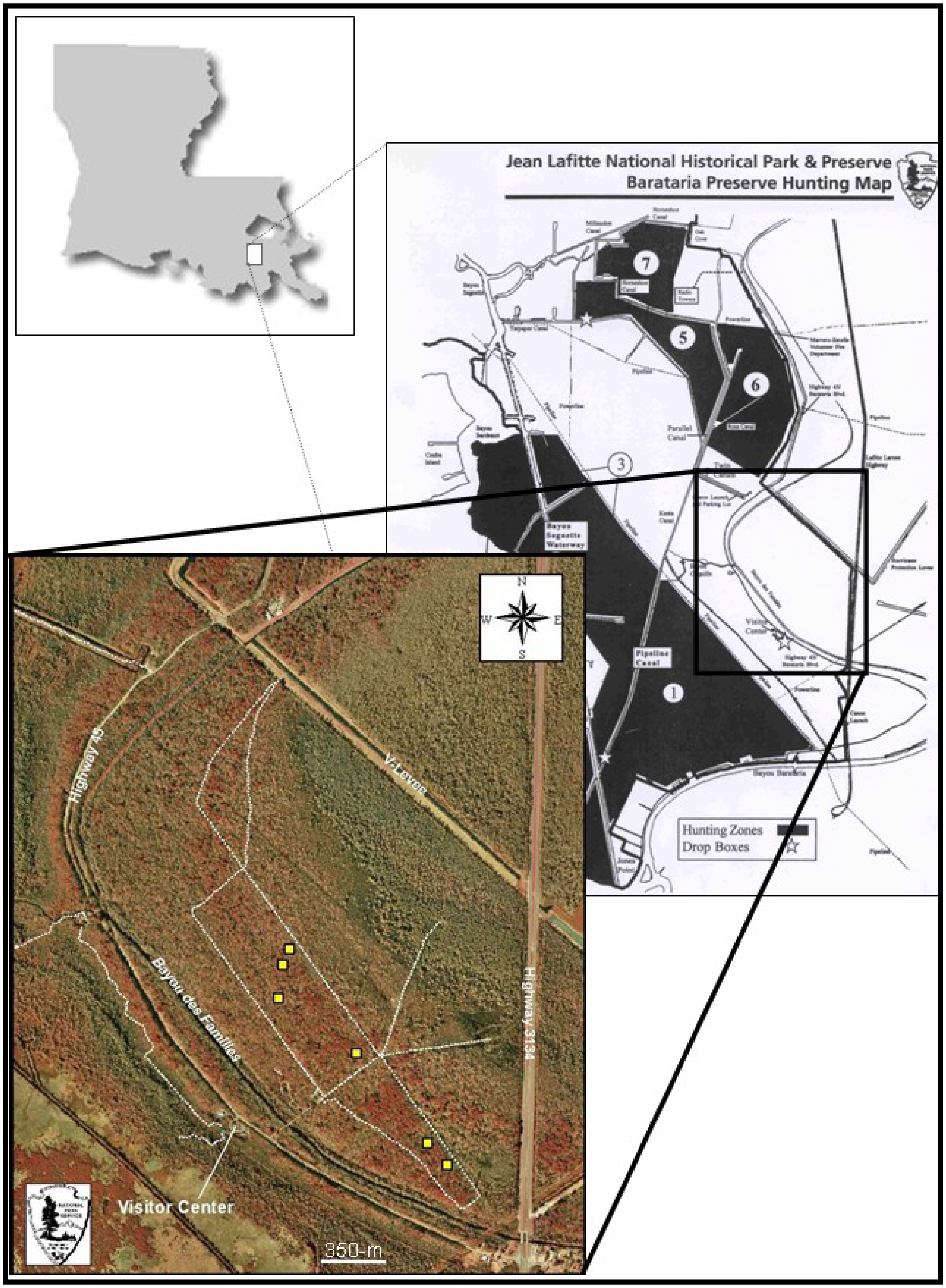

Figure 3. Maps showing locations of Jean Lafitte National Historical Park and Preserve's Barataria Preserve, hunting and non-hunting areas of the Barataria Preserve, and the location of six sites randomly selected for the study of the effects of white-tailed deer on the growth and survival of shrubs and juvenile trees.

unfenced plots were marked with gray p.v.c. pipe driven over iron rods, leaving approximately 18 inches above ground.

I attempted to analyze each of the following variables using a randomized block design, blocking by site, with repeated measures to determine if the variables changed differently over time between treatments. If the data did not meet the required assumptions for this test, then log, square root, and arcsin transformations were tried. If the transformations were unsuccessful, then the differences between paired plots were calculated for each site. I did this because differences between paired plots are often normally distributed when the underlying variables are not. I then used the randomized block design with repeated measures (blocking by site) to test for an effect of time on the differences. If the differences did not meet the assumptions of parametric statistics, and transformations were unsuccessful in achieving the assumptions, then nonparametric ANOVAs were used on the first and last time periods for that variable. If a variable contained too few observations for statistical analysis, then the data was visually analyzed.

Naturally Occurring Shrubs and Juvenile Trees (In Plots Under Canopy)

Naturally occurring trees and shrubs $\geq$ 15 cm but < 200 cm were surveyed in December 2002 (prior to construction), April 2003, July 2003, October 2003, January 2004, April 2004, and July 2004. The numbers and heights of individuals $\geq$ 15 cm but < 200 cm were recorded at each plot for each species. In December 2002, height was not recorded. All other surveys include numbers and heights. Also in July 2004, canopy cover was measured in fenced and unfenced plots using a densiometer (Spherical Densiometer Model-C, Robert E. Lemmon, Forest Densiometers, Bartlesville, OK). I

held the densiometer 12 to 18 inches in front of my body at elbow height (approximately 4 feet). I used the bubble to level the densiometer, made 4 readings per site facing N,S,E, and W, and calculated the average. The diameter (cm) at breast height also was recorded for trees $\geq$ 200 cm tall in July 2004.

I chose 15 cm as a minimum tree and shrub height because I was not interested in the effects of deer herbivory on establishment. Surveys of all 12 plots in April 2003 found an average of 688 individuals/plot < 15 cm in height (including herbaceous individuals). Most had just germinated and were very small; thus identification would have been difficult. I also assumed that if deer were having an effect on the growth or survival of individuals < 15 cm tall, a greater number of them would become 15 cm tall over time in the fenced plots than in the unfenced plots. Naturally occurring trees and shrubs that were < 200 cm when initially measured were included throughout the study.

All juvenile tree species were combined for analysis because of their low abundance. I used two independent Wilcoxon signed ranks tests (one for the first time period and one for the last time period) (Siegel and Castellan, Jr. 1988) to test for differences between treatments in the numbers of naturally occurring shrubs or juvenile trees per plot. The Bonferroni correction (Shaffer 1995) was used to determine the p-value (0.025) for the two tests because they tested one hypothesis: the number of juvenile tree species changed differently over time between treatments. Any differences between treatments in the numbers of trees or shrubs were then used to infer about differences in survival.

I used visual examination of data to determine if the average heights of shrubs or juvenile trees per plot changed differently over time between treatments. Any differences

between treatments in the average heights of trees or shrubs were then used to infer about differences in growth.

Planted Juvenile Trees (In Plots Under Canopy)

Although the nested plots contained numerous individuals < 15 cm tall, juvenile tree numbers appeared too low to provide an adequate sample size for statistical analysis. An equal number of juvenile trees therefore were planted in each plot to provide a larger sample size and to reduce variance among plots. I purchased *Q. nigra* and *F. pennsylvanica* bare-root juvenile trees from the Louisiana Department of Agriculture and Forestry. In February 2003, 25 *Q. nigra* and 25 *F. pennsylvanica* juveniles were planted in each fenced and unfenced plot in a one meter grid pattern, alternating species until all 50 juveniles were planted. All planted juveniles were marked with expandable plastic bird bands. Initial measurements of the juveniles' height were taken one month after they were planted.

In April 2003, July 2003, October 2003, January 2004, April 2004, and July 2004 the planted juveniles were surveyed. Juvenile heights and the number living were recorded at each plot for each species. The differences between treatments in the number of living juveniles and the average heights of juveniles were calculated for each species at each site. A randomized block design was used, blocking by site, to test for an effect of time on the differences between treatments in the number of living juveniles (by species) per plot and the average height of *Q. nigra* juveniles per plot. A log transformation was used to achieve homogeneity of variance for the test on the average height of *Q. nigra* juveniles per plot, but not for the average height of *F. pennsylvanica*. Two independent Wilcoxon signed ranks tests, nonparametric ANOVAs, were conducted (one for the first

time period and one for the last time period) (Siegel and Castellan, Jr. 1988) to determine if the differences between treatments in the average height of *F. pennsylvanica* juveniles per plot equaled zero (Freund and Wilson 2003:104-106). The Bonferroni correction was used to determine the p-value (0.025) for the two tests. The effect of treatments on the average height of the planted trees per plot was then used to infer about differences in the growth of the trees. The numbers of living juveniles per plot were used to calculate percent survival.

<u>Treefall Gaps</u>

A study was added to this project to determine the importance of treefall gaps on regeneration in Big Woods. In July 2004, the nearest treefall gap to each site was located. Most gaps were created by single tree mortality, usually adult *Q. nigra*. A 100-m^2 plot was laid out in the center of each gap. Naturally occurring tree and shrub species $\geq$ 15 cm but < 200 cm were counted and recorded. Juvenile tree species other than *T. sebifera* were combined for analysis because of their low abundance. The Friedman two-way analysis of variance by ranks (Siegel and Castellan, Jr. 1988) was used to test for differences among treatments (gaps, fenced plots, and unfenced plots) with blocking on sites in the number of shrubs, *T. sebifera*, and other juvenile trees per plot. The diameter (cm) at breast height was recorded for trees > 200 cm tall. Canopy cover was measured in the gaps using a densiometer.

RESULTS

The canopies above most fenced and unfenced plots were dominated by *Q. virginiana* (Table 1). Canopy cover in July 2004 averaged 88% above fenced plots, 89% above unfenced plots, and 71% above plots in treefall gaps. The canopy directly above the 100-m^2 plots in treefall gaps usually had 0% cover, but the large area visible on the densiometer's surface results in the 71% cover. The densiometer reading is probably a more accurate depiction of available sunlight to seedlings, however, so I am confident in the use of these percentages as an index of light availability. Details of the diameters at breast height (cm) and size-class distributions (stems/ha) of trees $\geq$ 200 cm tall in fenced, unfenced, and gap plots are provided (Tables 1-6, Appendix A, and Figures 1-5, Appendix B), but these data were not statistically analyzed.

Naturally Occurring Juvenile Trees (In Plots Under Canopy)

There were 84 observations (7 sampling periods and 12 plots) for the numbers of naturally occurring juvenile trees per plot. Of those, only 24 observations had juveniles, and only 14 had more than one juvenile. The species recorded were *C. laevigata* (3), *F. pennsylvanica* (1), *L. styraciflua* (18), *Quercus spp.* (13), *T. sebifera* (17), and unknowns (11). All unknowns were recorded in the first sampling period. There was no difference between fenced and unfenced plots in the number of naturally occurring juvenile trees per plot in the first or last time period. The number of juveniles per plot remained low throughout the study (Figure 4).

There were 72 observations (6 sampling periods and 12 plots) for the average heights of naturally occurring juvenile trees per plot. Of those, only 19 observations had juveniles, and only 12 had more than one juvenile. My conclusions regarding the average

Table 1. Mean number (standard error) and mean diameter at breast height in centimeters (standard error) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA.

	Fenced		Unfenced		Gaps	
Species	Mean number (stderr)	Mean DBH (stderr)	Mean number (stderr)	Mean DBH (stderr)	Mean number (stderr)	Mean DBH (stderr)
Acer negundo	0.17 (0.17)	15.60 (-)	0.33 (0.21)	15.85 (7.45)	0.17 (0.17)	16.30 (-)
Acer rubrum	0.50 (0.34)	6.80 (3.40)	0.83 (0.31)	8.53 (1.09)	0.67 (0.49)	12.88 (6.38)
Carpinus caroliniana	1.17 (0.83)	8.10 (0.60)	0.50 (0.34)	8.15 (3.25)	0.33 (0.21)	8.70 (1.50)
Celtis laevigata	1.17 (0.60)	23.00 (1.79)	0.67 (0.33)	15.15 (1.46)	0.00 (0.00)	(-) (-)
Crataegus sp.	1.83 (0.95)	4.40 (0.59)	0.33 (0.21)	4.70 (2.30)	0.83 (0.40)	4.80 (0.71)
Fraxinus pennsylvanica	0.00 (0.00)	(-) (-)	0.00 (0.00)	(-) (-)	0.50 (0.50)	5.77 (-)
Ilex decidua	1.50 (0.72)	3.18 (0.19)	2.00 (1.13)	3.65 (0.71)	2.50 (1.77)	5.18 (0.66)
Liquidambar styraciflua	0.50 (0.22)	36.23 (10.70)	1.00 (0.26)	35.19 (9.79)	0.33 (0.21)	15.00 (7.50)
Quercus laurifolia	0.17 (0.17)	11.30 (-)	0.50 (0.34)	14.35 (3.65)	0.00 (0.00)	(-) (-)
Quercus nigra	1.33 (0.42)	15.05 (2.65)	1.00 (0.52)	14.62 (4.69)	0.33 (0.21)	7.00 (0.40)
Quercus phellos	0.17 (0.17)	5.50 (-)	0.00 (0.00)	(-) (-)	0.00 (0.00)	(-) (-)
Quercus texana	0.17 (0.17)	7.90 (-)	0.00 (0.00)	(-) (-)	0.00 (0.00)	(-) (-)
Quercus virginiana	0.33 (0.21)	93.05 (13.05)	1.00 (0.45)	86.49 (13.33)	0.00 (0.00)	(-) (-)
Triadica sebifera	0.00 (0.00)	(-) (-)	0.00 (0.00)	(-) (-)	2.67 (1.86)	2.14 (0.56)
Ulmus americana	2.33 (1.17)	7.39 (1.74)	0.83 (0.40)	4.00 (0.44)	0.83 (0.54)	6.28 (1.78)

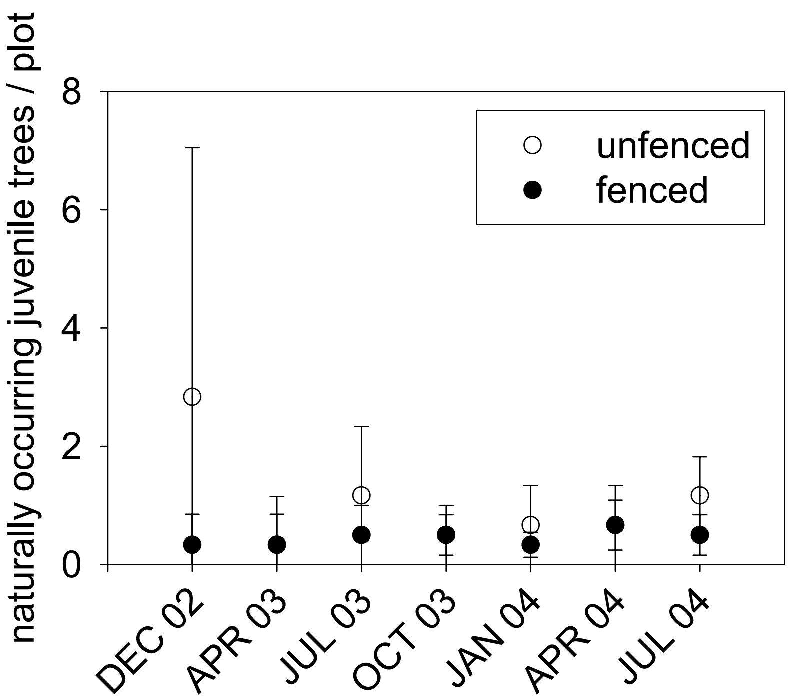

Figure 4. Changes in the number of naturally occurring juvenile trees ($\geq$ 15cm and < 200cm) per 100-m^2 plot, Jefferson Parish, LA.

heights of naturally occurring juvenile trees are not based on statistical analysis because the data did not meet the assumptions of parametric statistics and there were too few observations to use the nonparametric Wilcoxon signed ranks tests. After eliminating an outlier from the data set, visual examination of the data suggested that the average height of juveniles per plot did change differently over time between treatments (Figure 5). However, because the average heights were higher in unfenced plots, herbivory by white-tailed deer was eliminated as a possible cause of the difference in heights (Figure 5).

Naturally Occurring Shrubs (In Plots Under Canopy)

Callicarpa americana was the one shrub species found. *S. minor*, the abundant understory palm, was not analyzed because it was not a management concern. There were 84 observations (7 sampling periods and 12 plots) for the numbers of shrubs per plot. Of those, only 31 observations had shrubs, but 27 had more than one shrub. There was no difference between fenced and unfenced plots in the number of *C. americana* per plot in the first or last time period (Figure 6).

There were 72 observations (6 sampling periods and 12 plots) for the average height of shrubs per plot. Of those, only 27 observations had shrubs, but 26 had more than one shrub. My conclusions regarding the average heights of *C. americana* per plot are not based on statistical analysis because the data did not meet the assumptions of parametric statistics and there were too few observations to use the nonparametric ANOVAs. A plot of the means with standard error bars suggests no difference between fenced and unfenced plots in the average heights of *C. americana* per plot (Figure 7).

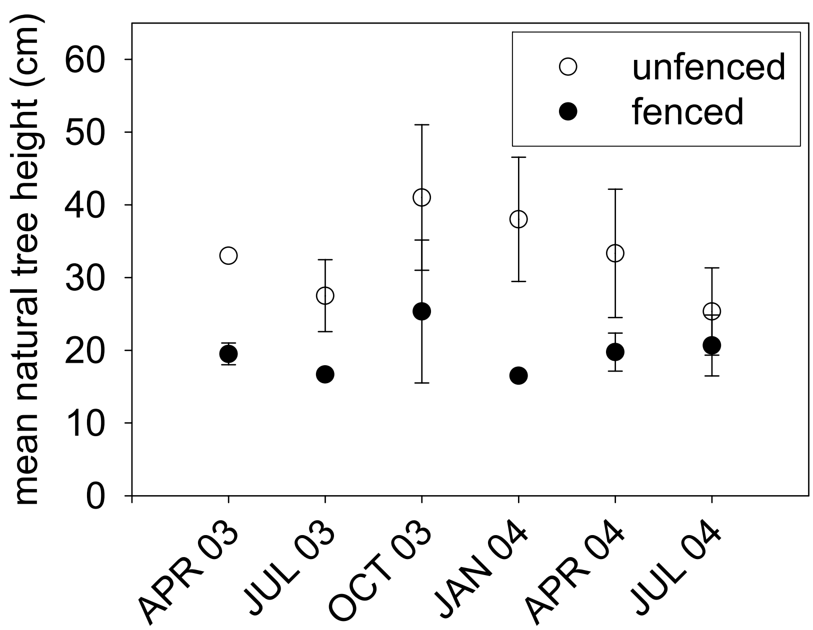

Figure 5. Changes in the mean height (cm) of naturally occurring juvenile trees per 100-m^2 plot, Jefferson Parish, LA.

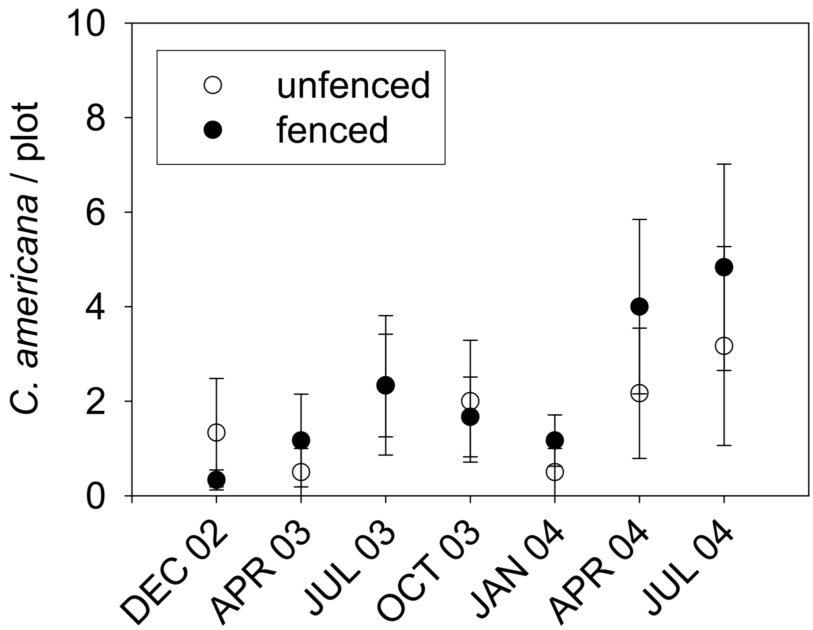

Figure 6. Changes in the number of *Callicarpa americana* ($\geq$ 15cm and < 200cm) per 100-m^2 plot, Jefferson Parish, LA.

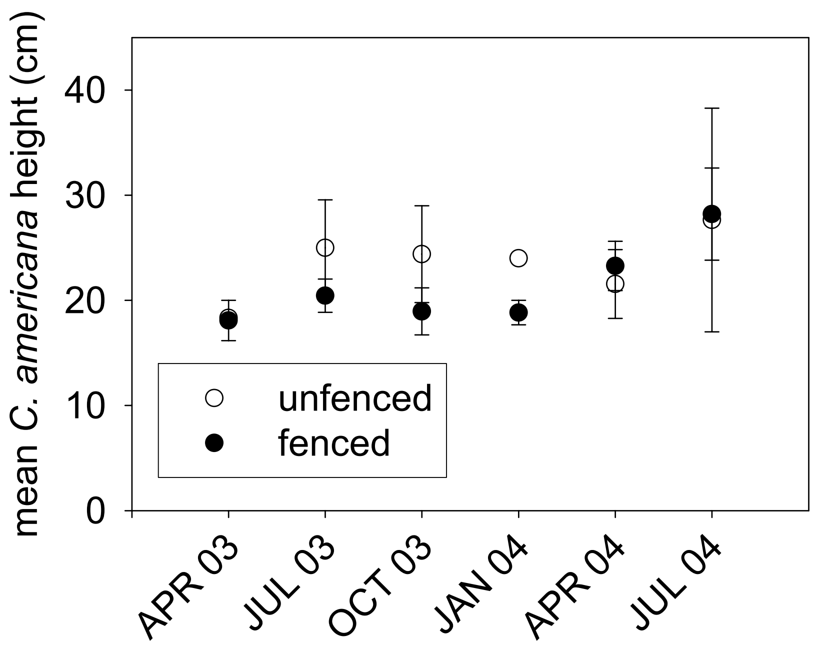

Figure 7. Changes in the mean height (cm) of *Callicarpa americana* per 100-m^2 plot, Jefferson Parish, LA.

Planted *Fraxinus pennsylvanica* Juveniles (In Plots Under Canopy)

In April 2003, the new leaves of many of the *F. pennsylvanica* juveniles had been browsed by white-tailed deer in the unfenced plots. Survival of *F. pennsylvanica* differed between fenced and unfenced plots ($p = <0.0001$); survival was greater in the fenced plots throughout the study (Figure 8). There was no statistical difference between treatments in the average height of *F. pennsylvanica* juveniles per plot in March 2003 or July 2004 ($p = 0.3846$, $p = 0.0494$) (Figure 9).

Planted *Quercus nigra* Juveniles (In Plots Under Canopy)

Exclosures did not affect the survival of *Q. nigra* juveniles or the average height of *Q. nigra* juveniles per plot. Small herbivores were able to move freely through openings (approximately 15 cm) in the game fence. The damage caused by these small herbivores was not confused with white-tailed deer damage, because the small herbivores would cut the main stem of juveniles at 45 degree angles without consuming any part of the tree. Most juvenile *Q. nigra* died in the fenced and unfenced plots by July 2003 (Figure 10). The average height of *Q. nigra* juveniles declined over time equally in the fenced and unfenced plots (Figure 11).

Treefall Gaps

The numbers of *T. sebifera*, "other trees", and *C. americana* per plot were analyzed separately because there was no correlation among them. "Other trees" consisted of *A. rubrum* (3)*, C. laevigata* (3)*, F. pennsylvanica* (7)*, and Ligustrum sp.* (1). There was no statistical difference among treatments in the number of "other trees" per plot (Figure 12) or in the number of *C. americana* per plot (Figure 13). There was a statistical difference among treatments in the number of *T. sebifera* per plot ($p = 0.0006$);

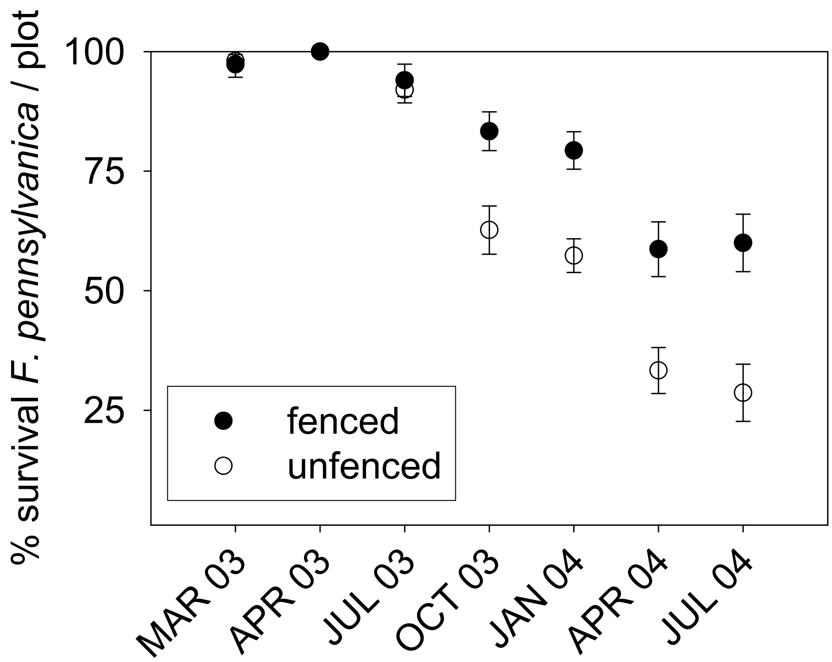

Figure 8. Percent survival of planted *Fraxinus pennsylvanica* juveniles per 100-m^2 plot, Jefferson Parish, LA.

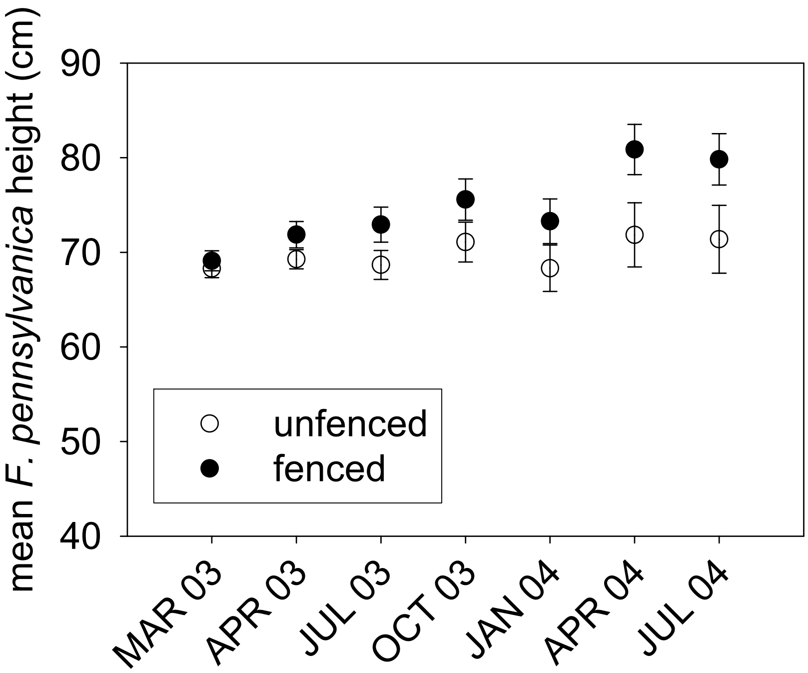

Figure 9. Changes in the mean height (cm) of planted *Fraxinus pennsylvanica* per 100-m^2 plot, Jefferson Parish, LA.

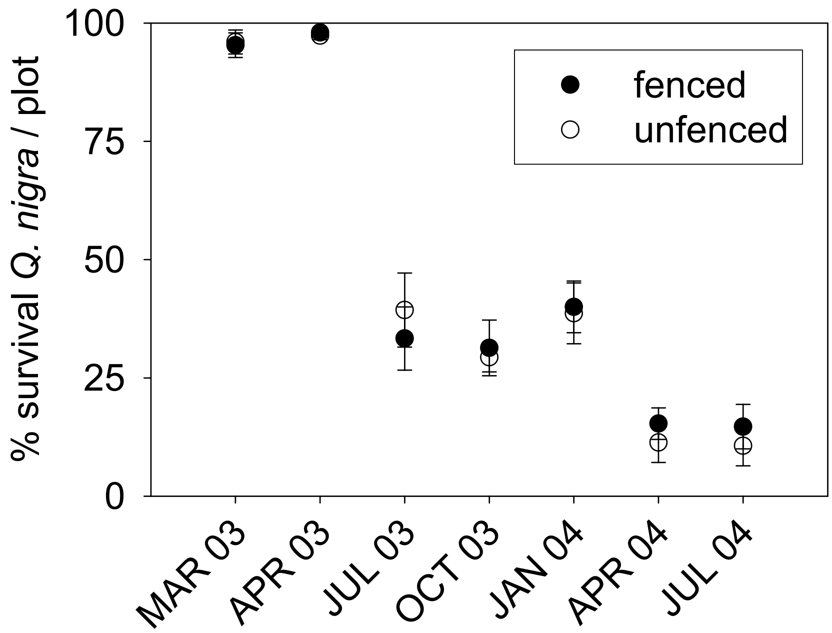

Figure 10. Percent survival of planted *Quercus nigra* juveniles per 100-m^2 plot, Jefferson Parish, LA.

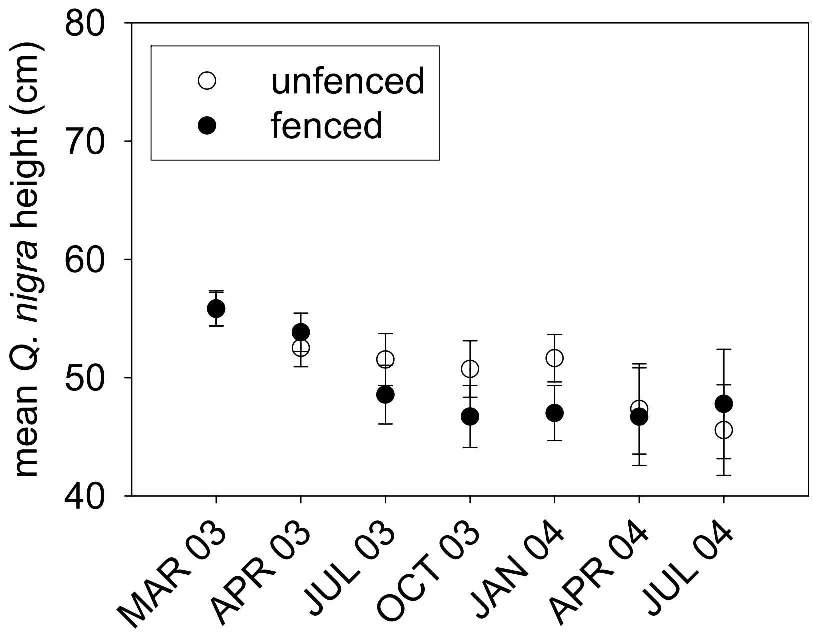

Figure 11. Changes in the mean height (cm) of planted *Quercus nigra* per 100-m^2 plot, Jefferson Parish, LA.

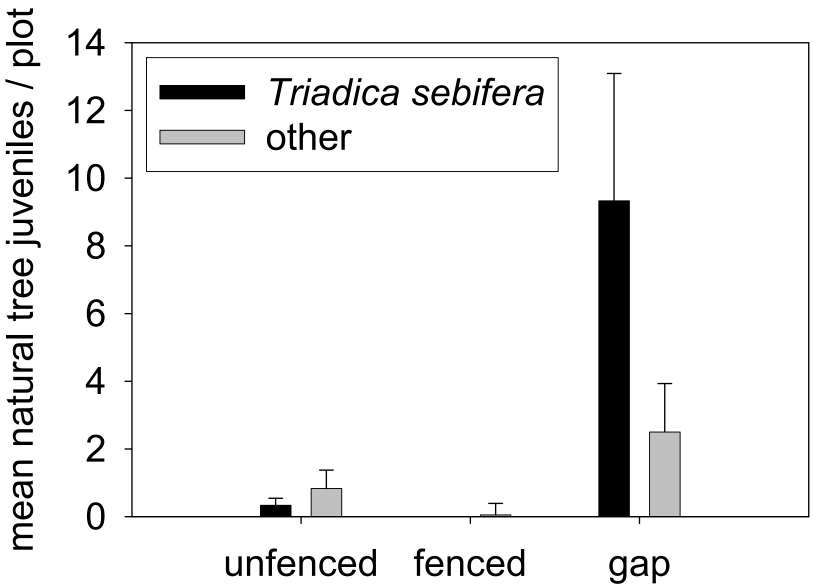

Figure 12. The mean number of naturally occurring juvenile trees (≥ 15cm and < 200cm) in 100-m^2 unfenced plots, fenced plots, and plots in treefall gaps in July 2004, Jefferson Parish, LA.

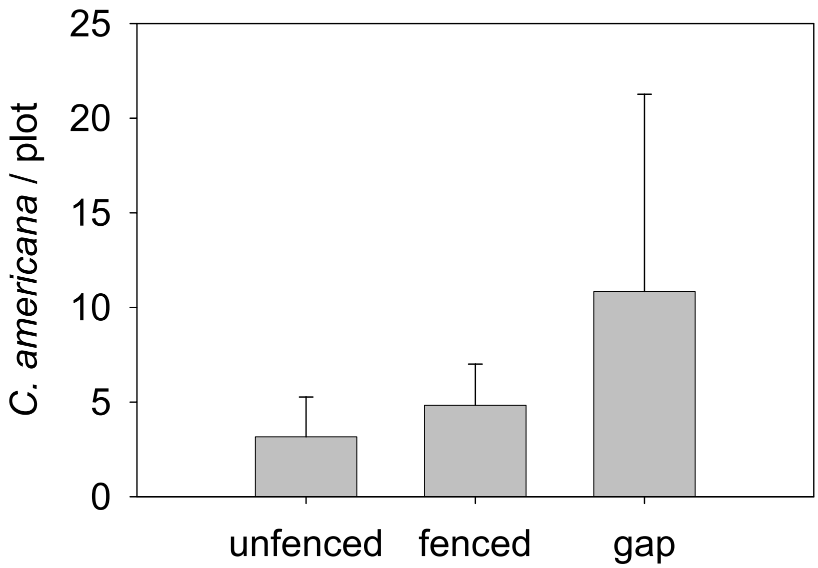

Figure 13. The mean number of *Callicarpa americana* (≥ 15cm and < 200cm) in 100-m^2 unfenced plots, fenced plots, and plots in treefall gaps in July 2004, Jefferson Parish, LA.

gaps contained more *T. sebifera* than the fenced or unfenced plots (Figure 12). Visual analysis of the diameter distributions of trees $\geq$ 200 cm tall also show more *T. sebifera* in gaps than in plots under the canopy (Figure 5, Appendix B).

DISCUSSION

The analysis of my three data sets suggests deer are affecting the survival and possibly growth of juveniles when juveniles are present, i.e. the planted juveniles. However, juveniles are virtually absent under the canopy in Big Woods. This raises new questions regarding the effects of deer in treefall gaps (where high concentrations of juveniles occur) and how those effects might modify the landscape. Studies of gaps in other forest types have suggested gap disturbances increase species diversity within a forest (Denslow 1980), while gap disturbances in other forests have no effect on diversity (Hubbell et al. 1999). At the Barataria Preserve, gap disturbances may be contributing to the forest becoming less diverse. The exotic *T. sebifera* accounted for 79% of the juvenile trees found in gaps, but only 27% under the canopy.

A logical conclusion from the naturally occurring shrubs and juvenile trees experiment is that factors other than deer herbivory were limiting the growth and survival of individuals under the canopy in Big Woods. It is unclear which factors are limiting the growth and survival of individuals in Big Woods, but in an east Texas floodplain forest many tree seedlings died because of flood stress or proximity to conspecific adults (Streng et al. 1989). Because ample seedlings were germinating in each fenced and unfenced plot, the small numbers of naturally occurring individuals $\geq$ 15 cm (which prevented statistical analysis of average heights) reinforced the conclusion that deer herbivory was not the limiting factor under the canopy. If deer herbivory was limiting growth or survival, one would have expected the average height and the number of individuals per plot $\geq$ 15 cm tall to become higher in fenced plots.

Rather than deer herbivory, shade probably limited seedling growth and survival and led to the lack of naturally occurring shrubs and juvenile trees under the canopy, which ultimately led to the lack of an effect by deer. One possible conclusion could have been that deer densities were too low to cause an effect, but the overall lack of naturally occurring individuals $\geq$ 15 cm tall in the fenced plots and the results from the planted juveniles experiment led me to reject this conclusion. Saunders and Puettmann (1999) also found that increased overstory competition (shade) reduced the growth and survival of *Pinus strobus* L. juveniles. It is likely shade had the same effect in our fenced and unfenced plots. This would explain why so many seedlings germinated and died under the canopy but survived in treefall gaps.

While management is not concerned about the welfare of *S. minor*, it is possible the dense palms could be negatively affecting regeneration of trees and shrubs. Most sunlight that penetrates the canopy is shaded from the forest floor by *S. minor*. The dense palms seem to have a lesser effect in treefall gaps, but their role is not well understood and warrants future investigation.

Where shade is not a factor, such as in treefall gaps, other factors such as herbivory, understory competition (Saunders and Puettmann 1999), or flood stress could limit the growth and survival of shrubs and juvenile trees. At the Barataria Preserve, flooding has already increased enough to affect species composition (Denslow and Battaglia 2002). The presence of *Q. nigra* and *U. americana* saplings in my fenced and unfenced plots on the ridge are consistent with Denslow and Battaglia (2002), who in this same forest found saplings of these two species at higher elevations than their adult

counterparts (Figures 4-5, Appendix B). If flooding continues to increase at the preserve, tree productivity will undoubtedly decrease (Megonigal et al. 1997).

The major finding from the planted juveniles experiment is that herbivory by white-tailed deer was high enough in Big Woods to reduce the survival of *F. pennsylvanica* and possibly other juvenile trees. This has been a common conclusion of deer studies in other forest types as well (Jacobs 1969, Boerner and Brinkman 1996, Strange and Shea 1998). This also agrees with conclusions from a review by Russell et al. (2001) that deer densities may be the primary predictor of deer effects, but the density of the plant consumed by deer (Augustine et al. 1998) also is an important predictor. I found no effect on natural juveniles where the densities were close to zero, but did find an effect on the planted juveniles that had higher densities.

I believe the decline in the number of living *F. pennsylvanica* juveniles by October 2003 is at least partially attributable to herbivory by white-tailed deer. Although the average height of *F. pennsylvanica* juveniles per plot did not differ between treatments in July 2004 ($p = 0.0494$), a difference will likely develop after another growing season (Figure 9). It is also possible I failed to detect a real difference in heights (as it appears in the figure) because I used the most conservative correction factor (Bonferroni). Almost every planted *Q. nigra* juvenile died in the fenced and unfenced plots. Shade, flood stress, and/or small herbivores probably killed these *Q. nigra* juveniles.

The study of treefall gaps suggests the invasion by *T. sebifera* will significantly alter the future composition of the Barataria Preserve. This mature forest seems to be replacing itself through gap-phase dynamics, but only 21% of the juvenile trees found in

gaps were native. The remaining juveniles found in gaps were *T. sebifera*. The abundance of *T. sebifera* juveniles in gaps, combined with their low occurrence in plots under the canopy, suggests they became established after the gaps formed at the Barataria Preserve. This has already been seen in Louisiana's Verrett Basin where *T. sebifera* invaded treefall gaps created by Hurricane Andrew (Conner et al. 2002). Diameter distribution graphs of trees $\geq$ 200 cm tall also show the dominance of *T. sebifera* in treefall gaps (Figure 5, Appendix B). Birds are the likely dispersers of most *T. sebifera* seeds in these forests (Renne et al. 2000).

Since the mid-1900's, *T. sebifera* has become established in bottomland hardwoods in the Coastal Plain from North Carolina to Texas (Bruce et al. 1997, Hardin et al. 2001). The rapid expansion of *T. sebifera* can partially be attributed to its ability to successfully compete under a wide range of conditions. Wall and Darwin (1999) report *T. sebifera* occurred at all elevations within a Louisiana bottomland hardwood forest except in standing water. *T. sebifera* has shown characteristics of shade tolerance by growing nearly 3 times taller than *Quercus pagoda* Raf. and *Platanus occidentalis* L. in the shade (Jones and McLeod 1989). Amazingly, it even grew at the same rate of *P. occidentalis* when grown under full sunlight (Jones and McLeod 1989). Some believe the success of *T. sebifera* over native species can be partly attributed to its lack of herbivores in its naturalized range (Jones and McLeod 1989, Rogers and Siemann 2004). Others suggest its remarkable tolerance of leaf damage and its rapid morphological and physiological compensation to herbivory give it a competitive advantage (Rogers et al. 2000). The long-term impacts of *T. sebifera* in Big Woods and in other forested wetlands throughout the Coastal Plain remain unknown.

Something else that remains unknown in Big Woods is the effect that white-tailed deer and other herbivores are having on species composition in treefall gaps. The results previously discussed show deer are capable of reducing juvenile survival in Big Woods, and gaps contain the highest concentrations of juveniles. It is possible that deer are consuming the more palatable native species at higher rates than the exotic *T. sebifera* in these gaps. The results from the planted juveniles suggest other small herbivores also damage juvenile trees in Big Woods. It is possible the small herbivores select and cut only certain species in gaps like they did *Q. nigra* juveniles in the fenced and unfenced plots. Additional research throughout the range of *T. sebifera* is needed to determine the effects of herbivore preference and density on *T. sebifera* success.

Future research on the effects of deer herbivory should first identify the areas of a forest where most regeneration is taking place. If the canopy is fairly open, trees and shrubs may be regenerating more evenly across the forest floor. Under these circumstances, one could conduct a study such as this by randomly selecting locations for paired plots. If the area of concern is a more mature forest that hasn't been thinned, gap-phase dynamics may be driving regeneration. Locating treefall gaps and building exclosures in half of them may be a more effective design for determining the effects of herbivory in that forest. The gaps should be stratified by hydroperiod, the primary factor driving early establishment in bottomland hardwood forests (Sharitz and Mitsch 1993).

In my study, the low survival of *Q. nigra* reduced my sample size. Future studies to determine the effects of white-tailed deer on juvenile trees in bottomland hardwoods might be more powerful if the planted species was flood and shade tolerant. Such a species would be a poor choice however, for studies designed to determine what factor

caused the number of juveniles to be so low. If the goal is to determine what factor is limiting regeneration, several species with different flood and shade tolerances could be planted. An example might be planting four different species: one that is flood and shade tolerant, one that is flood and shade intolerant, one that is flood tolerant and shade intolerant, and one that is flood intolerant and shade tolerant. This could help determine whether flooding and/or shade are stressing juveniles in that forest.

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APPENDIX A:
DIAMETERS OF TREES IN STUDY PLOTS

Appendix A Table 1. Diameter at breast height (cm) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 1 (UTM: Easting 15R 0779203, Northing 3299239)

species	dbh (cm)
- - - fenced - - -	
Celtis laevigata	9.9
Celtis laevigata	24.5
Celtis laevigata	27.3
Crateagus sp.	3.3
Crateagus sp.	3.6
Crateagus sp.	4.8
Crateagus sp.	4.8
Crateagus sp.	6.0
Crateagus sp.	6.5
Quercus nigra	6.0
Quercus nigra	41.5
Ulmus americana	12.5
- - - unfenced - - -	
Celtis laevigata	9.0
Celtis laevigata	19.5
Liquidambar styraciflua	48.6
Quercus laurifolia	18.0
Quercus nigra	12.8
Quercus nigra	24.8
Quercus nigra	26.5
- - - gap - - -	
Fraxinus pennsylvanica	3.0
Fraxinus pennsylvanica	5.3
Fraxinus pennsylvanica	9.0

Appendix A Table 2. Diameter at breast height (cm) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 2 (UTM: Easting 15R 0779997, Northing 3298159)

species	dbh (cm)
- - - fenced - - -	
Acer negundo	15.6
Acer rubrum	3.4
Carpinus caroliniana	6.7
Carpinus caroliniana	7.5
Carpinus caroliniana	9.4
Carpinus caroliniana	9.6
Carpinus caroliniana	10.3
Crataegus sp.	3.6
Liquidambar styraciflua	43.9
Quercus nigra	13.4
Ulmus americana	2.4
- - - unfenced - - -	
Carpinus caroliniana	10.7
Carpinus caroliniana	12.1
Quercus virginiana	64.4
Quercus virginiana	70.0
Quercus virginiana	95.6
- - - gap - - -	
Acer rubrum	15.2
Acer rubrum	20.4
Acer rubrum	22.2
Carpinus caroliniana	10.2

Appendix A Table 3. Diameter at breast height (cm) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 3 (UTM: Easting 15R 0779540, Northing 3298716)

species	dbh (cm)
- - - fenced - - -	
Celtis laevigata	15.0
Celtis laevigata	23.6
Celtis laevigata	27.2
Crataegus sp.	4.2
Crataegus sp.	4.7
Crataegus sp.	8.7
Ilex decidua	2.0
Ilex decidua	3.1
Ilex decidua	3.2
Ilex decidua	3.3
Liquidambar styraciflua	15.1
Ulmus americana	3.0
Ulmus americana	3.1
Ulmus americana	3.6
Ulmus americana	4.2
Ulmus americana	4.6
Ulmus americana	4.8
Ulmus americana	5.1
Ulmus americana	15.3
- - - unfenced - - -	
Acer rubrum	6.2
Acer rubrum	11.2
Celtis laevigata	18.0
Crataegus sp.	2.4
Ilex decidua	2.0
Ilex decidua	3.0
Ilex decidua	3.0
Ilex decidua	3.2
Ilex decidua	4.1
Ilex decidua	4.6
Liquidambar styraciflua	5.8
Quercus virginiana	101.1
Ulmus americana	2.6
Ulmus americana	7.0
- - - gap - - -	
Ilex decidua	2.3
Ilex decidua	2.4
Ilex decidua	2.7
Ilex decidua	2.9
Ilex decidua	4.1
Ilex decidua	4.2

Appendix A Table 3 (cont.).

Ilex decidua	4.3
Ilex decidua	6.6
Ilex decidua	8.2
Ilex decidua	8.6
Ilex decidua	12.0
Quercus nigra	6.6
Ulmus americana	3.4
Ulmus americana	4.3
Ulmus americana	5.8

Appendix A Table 4. Diameter at breast height (cm) of trees $\geq$ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 4 (UTM: Easting 15R 0779898, Northing 3298265)

species	dbh (cm)
- - - fenced - - -	
Acer rubrum	5.0
Acer rubrum	15.4
Carpinus caroliniana	7.0
Carpinus caroliniana	8.0
Quercus nigra	13.5
Quercus virginiana	80.0
Ulmus americana	5.7
Ulmus americana	7.9
- - - unfenced - - -	
Acer rubrum	10.2
Carpinus caroliniana	4.9
Liquidambar styraciflua	32.0
Liquidambar styraciflua	34.5
- - - gap - - -	
Acer negundo	16.3
Acer rubrum	6.5
Carpinus caroliniana	7.2
Crataegus sp.	5.3
Ilex decidua	5.8
Quercus nigra	7.4

Appendix A Table 5. Diameter at breast height (cm) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 5 (UTM: Easting 15R 0779168, Northing 3299161)

species	dbh (cm)
- - - fenced - - -	
Celtis laevigata	26.5
Crataegus sp.	3.0
Ilex decidua	2.5
Ilex decidua	4.6
Liquidambar styraciflua	49.7
Quercus nigra	17.0
Quercus virginiana	106.1
- - - unfenced - - -	
Acer negundo	23.3
Acer rubrum	5.4
Celtis laevigata	13.2
Crataegus sp.	7.0
Ilex decidua	5.0
Liquidambar styraciflua	63.0
Quercus laurifolia	9.6
Quercus laurifolia	11.8
Quercus nigra	14.7
Quercus nigra	19.1
Quercus virginiana	54.0
Ulmus americana	3.3
- - - gap - - -	
Crataegus sp.	3.3
Crataegus sp.	3.5
Liquidambar styraciflua	7.5
Triadica sebifera	1.1
Triadica sebifera	1.2
Triadica sebifera	1.6
Triadica sebifera	2.0
Triadica sebifera	2.0

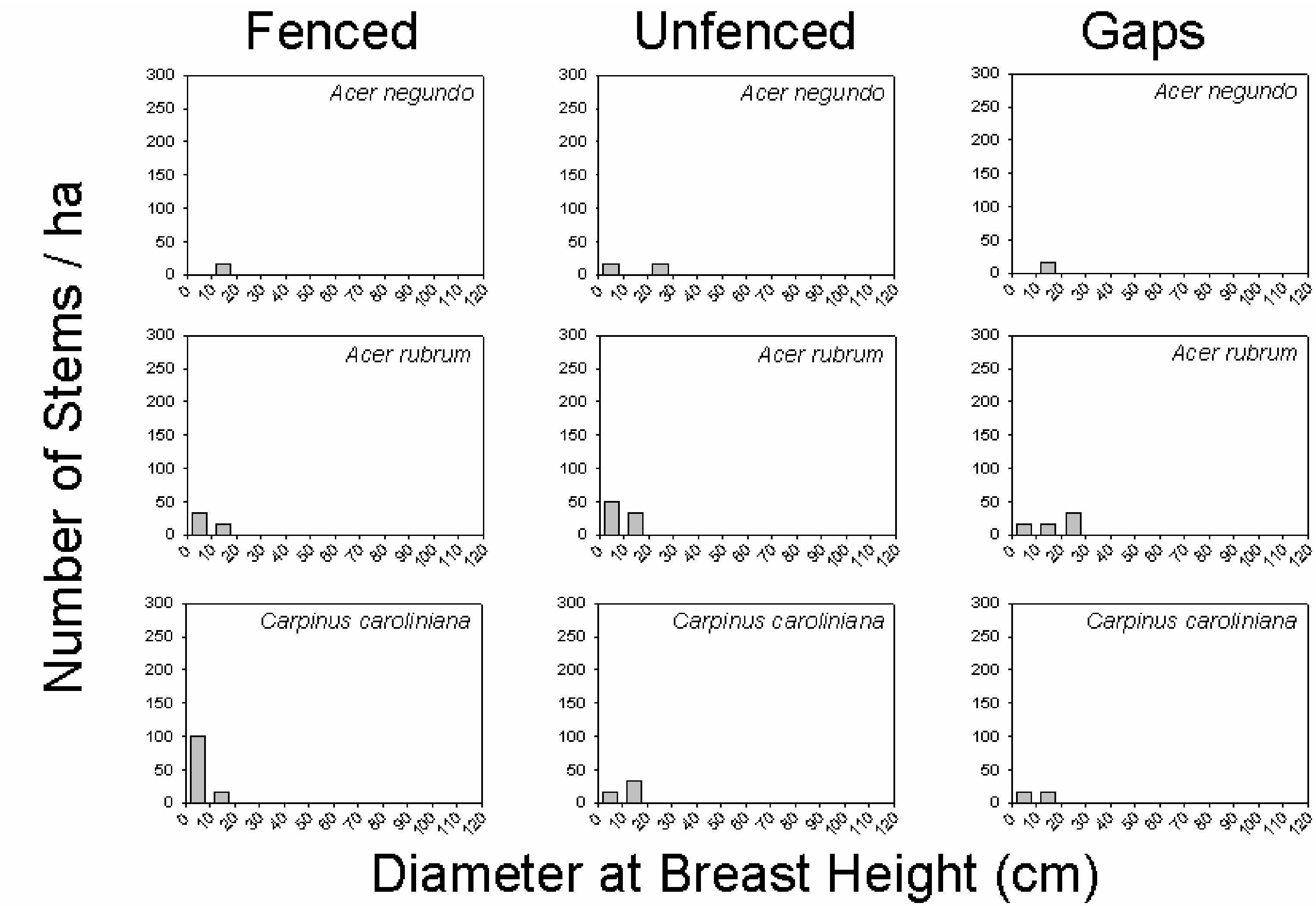
Appendix A Table 6. Diameter at breast height (cm) of trees ≥ 200 cm tall in 100-m^2 plots in July 2004, Jefferson Parish, LA. Site 6 (UTM: Easting 15R 0779149, Northing 3298991)

species	dbh (cm)
- - - fenced - - -	
Ilex decidua	2.9
Ilex decidua	3.0
Ilex decidua	3.4
Quercus laurifolia	11.3
Quercus nigra	4.9
Quercus nigra	5.9
Quercus nigra	12.0
Quercus phellos	5.5
Quercus texana	7.9
Ulmus americana	4.9
Ulmus americana	14.7
- - - unfenced - - -	
Acer negundo	8.4
Acer rubrum	9.8
Ilex decidua	2.1
Ilex decidua	2.1
Ilex decidua	2.7
Ilex decidua	2.8
Ilex decidua	3.4
Liquidambar styraciflua	25.3
Quercus nigra	5.6
Quercus virginiana	114.2
Ulmus americana	3.7
Ulmus americana	4.1
- - - gap - - -	
Crataegus sp.	5.3
Crataegus sp.	6.1
Ilex decidua	3.1
Ilex decidua	3.8
Ilex decidua	4.7
Liquidambar styraciflua	22.5
Triadica sebifera	1.1
Triadica sebifera	1.4
Triadica sebifera	1.7
Triadica sebifera	1.8
Triadica sebifera	2.4
Triadica sebifera	2.5
Triadica sebifera	3.0
Triadica sebifera	3.2
Triadica sebifera	3.8
Triadica sebifera	4.2

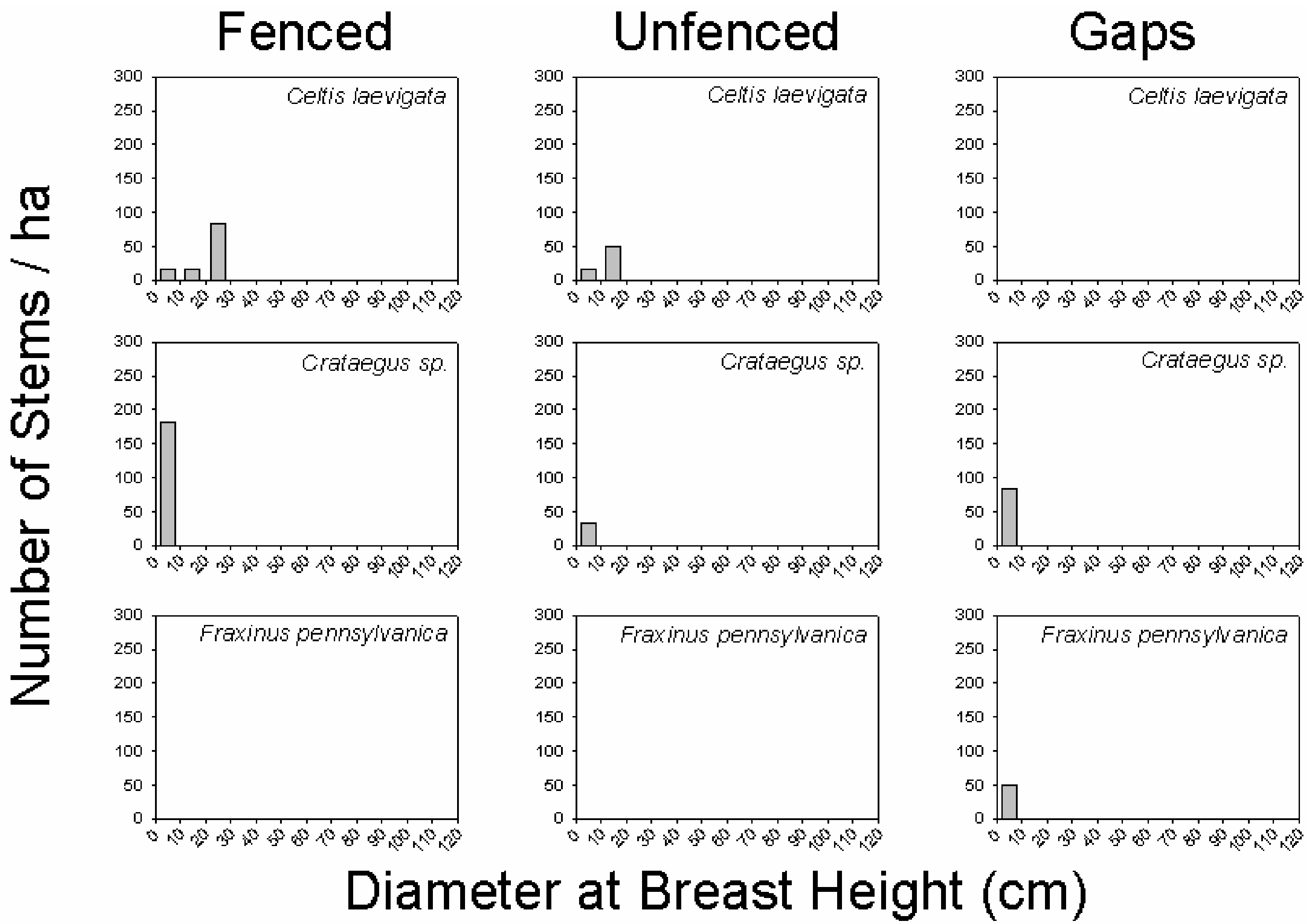
Appendix A Table 6 (cont.).

Triadica sebifera	4.5
Ulmus americana	7.7
Ulmus americana	8.4

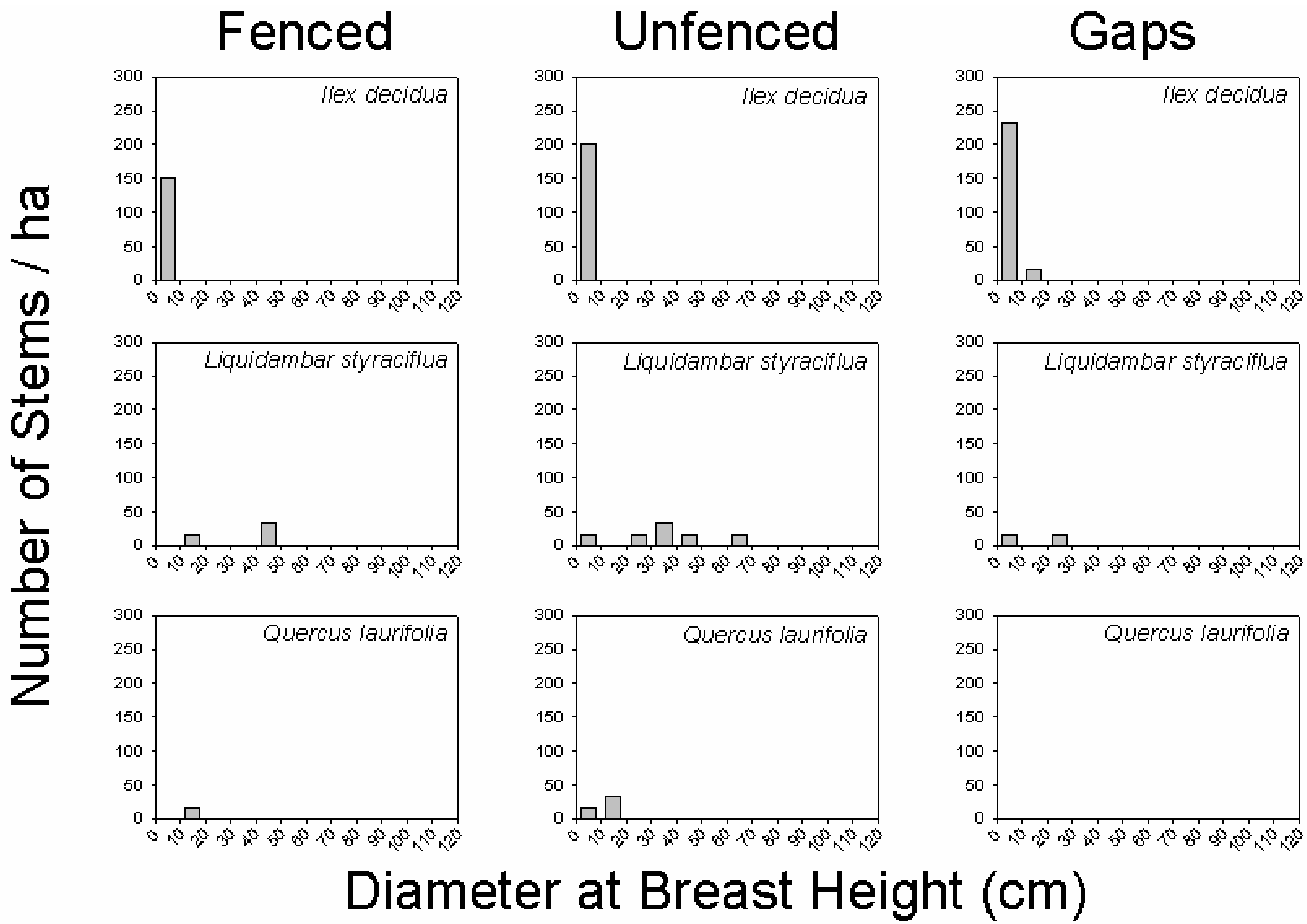
APPENDIX B:
DIAMETER DISTRIBUTIONS OF TREES IN STUDY PLOTS



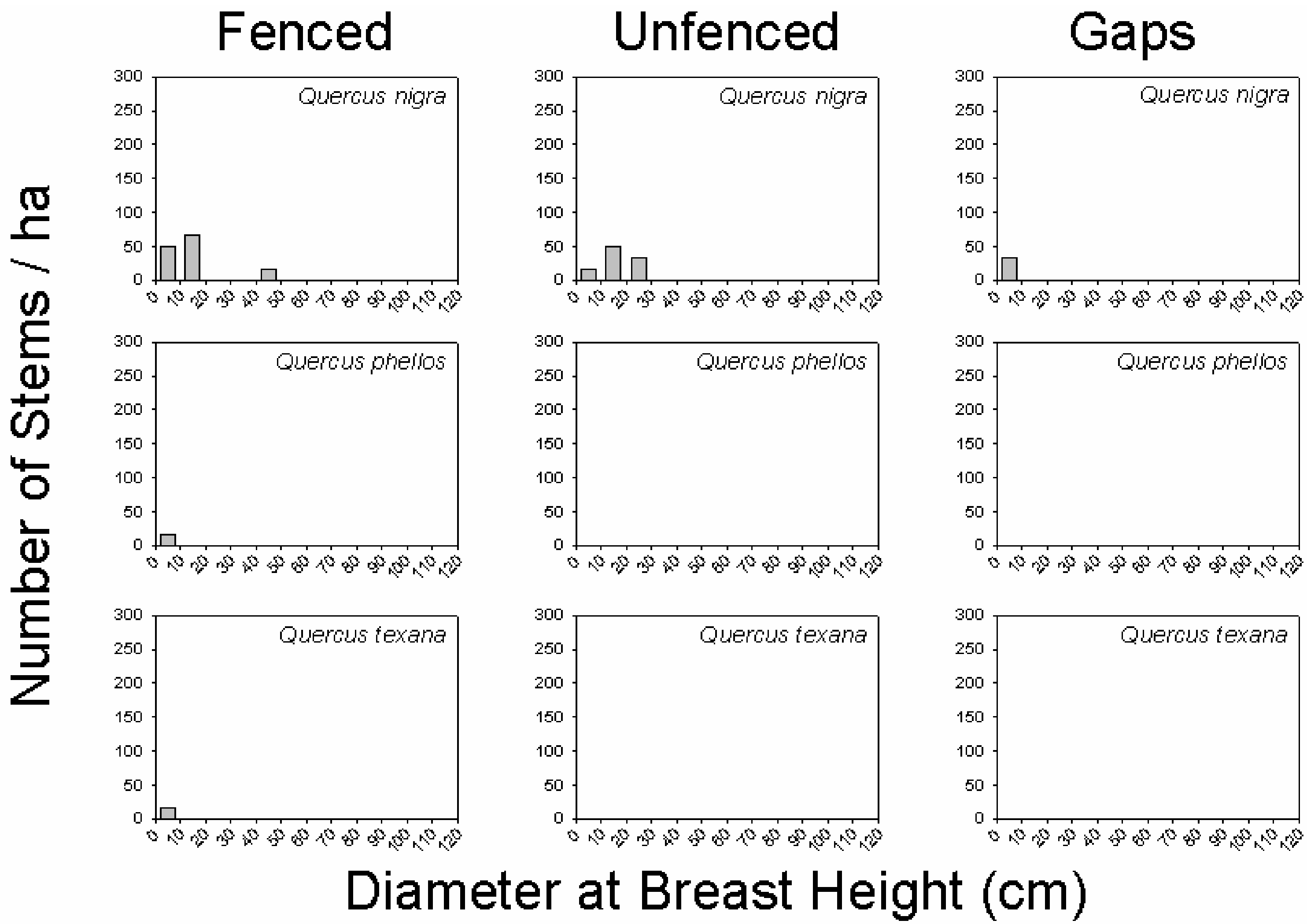
Appendix B Figure 1. Diameter distributions of trees ≥ 200 cm tall in six 100-m2 fenced plots, six 100-m2 unfenced plots, and six 100-m2 plots in gaps in July 2004, Jefferson Parish, LA.



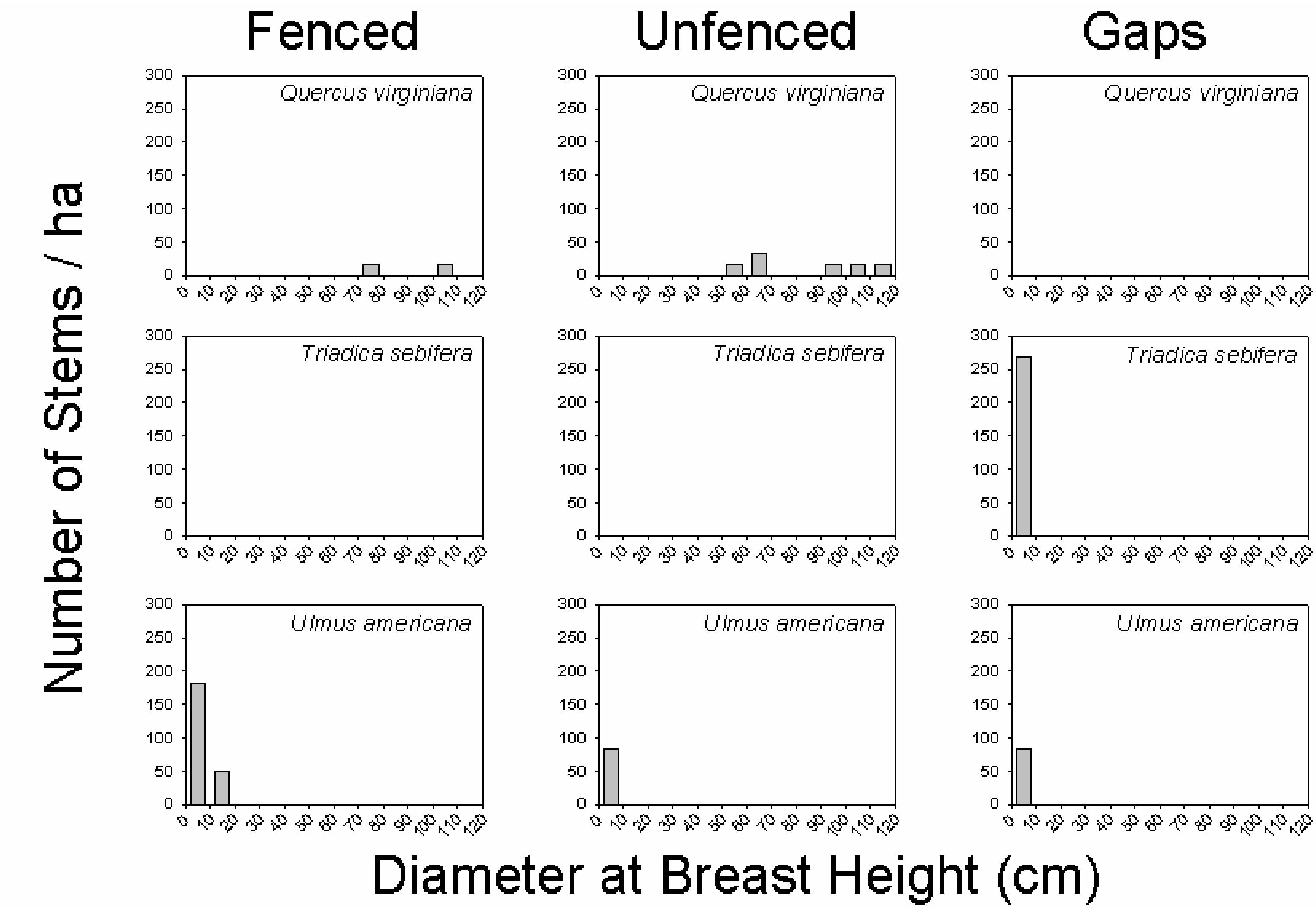
Appendix B Figure 2. Diameter distributions of trees $\geq$ 200 cm tall in six 100-m2 fenced plots, six 100-m2 unfenced plots, and six 100-m2 plots in gaps in July 2004, Jefferson Parish, LA.



Appendix B Figure 3. Diameter distributions of trees ≥ 200 cm tall in six 100-m2 fenced plots, six 100-m2 unfenced plots, and six 100-m2 plots in gaps in July 2004, Jefferson Parish, LA.



Appendix B Figure 4. Diameter distributions of trees ≥ 200 cm tall in six 100-m2 fenced plots, six 100-m2 unfenced plots, and six 100-m2 plots in gaps in July 2004, Jefferson Parish, LA.



Appendix B Figure 5. Diameter distributions of trees ≥ 200 cm tall in six 100-m2 fenced plots, six 100-m2 unfenced plots, and six 100-m2 plots in gaps in July 2004, Jefferson Parish, LA.

VITA

Seth T. Bordelon was born in Opelousas, Louisiana, on January 4, 1979, to Eddie and Sheila Bordelon. He grew up hunting and fishing with his father and brother in Avoyelles Parish. Seth graduated from Bunkie High School in 1997, and joined the Louisiana Army National Guard in 1999. He received his Bachelor of General Studies in the spring of 2002 from Louisiana State University. In the fall of 2002, Seth began studying the effects of white-tailed deer on shrubs and juvenile trees in a Louisiana coastal wetland forest. He will receive his Master of Science degree with a major in wildlife from Louisiana State University in December 2005. He resides in Baton Rouge with his wife, LaShaun, and daughter, Taylor.