

**STUDIES ON BOTTOMLAND HARDWOOD FOREST RESTORATION AND  
TEACHING WITH GEOGRAPHIC INFORMATION SYSTEMS (GIS) IN  
ECOLOGY LABS**

A Dissertation

by

MATTHEW EARL SIMMONS

Submitted to the Office of Graduate Studies of  
Texas A&M University  
in partial fulfillment of the requirements for the degree of  
DOCTOR OF PHILOSOPHY

May 2007

Major Subject: Rangeland Ecology and Management

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Approved by:

Co-Chairs of Committee,	X. Ben Wu Steven G. Whisenant
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## ABSTRACT

Studies on Bottomland Hardwood Forest Restoration and Teaching with Geographic Information Systems (GIS) in Ecology Labs. (May 2007)

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The development of graduate students in the sciences preparing for careers in academia has long included elements to advance students as science researchers, but recent emphasis is being placed on developing students as instructors and education researchers as well. As such, objectives of this study included assessments of seedling responses to hydrology typical of floods in urban settings, the role of created microtopography in community development of a bottomland hardwood forest, and the influence of geographic information systems (GIS) on student motivation and conceptual knowledge.

Substantial losses of bottomlands in Texas necessitate restoration to regain the ecosystem services that they provide. Restoration of proper hydrology is the most important aspect of wetland restoration, but this can prove difficult in urbanizing environments where hydrology has been irreversibly altered. Microtopography has been shown to be an important component of bottomland hardwood forests, and its restoration may aid in hydrologic restoration as gradients are created that support a diverse community.

Tree seedlings were subjected to experimental flooding regimes typical of floodplain forests in rural and urban settings. Growth rates of seedlings varied over time and differed depending on species and treatment. Created microtopography resulted in a spatially heterogeneous system similar to that of natural bottomlands and strongly influenced hydrology, soil properties, survival of planted seedlings, and abundance and distribution of colonizing species. Proper bottomland restoration in urbanizing

environments should include species selection based on current and potential future hydrologic conditions. In addition, restoring microtopography may improve survival of a variety of species introduced during restoration, as well as enhance colonization of a diverse plant community under changing hydrologic regimes.

Trends indicated a slight improvement in attitude and performance for students that used GIS. More important, the authenticity of the experience appeared to affect student attitude. The effective use of GIS in teaching may be scale-dependent. Small-scale phenomena may be assessed as easily in a field exercise as with GIS. Using GIS to assess large-scale, complex patterns may have a substantial impact on student understanding. Further studies are needed to determine direct benefits of teaching with GIS in undergraduate ecology classrooms.

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## CHAPTER 1

### INTRODUCTION

Bottomland hardwood forests provide many services and functions that benefit humans and promote ecosystem integrity. These functions include particle and contaminant filtration that improve water quality, hydrologic energy dissipation for flood abatement, providing wildlife habitat, cycling nutrients, and affording recreational opportunities (Brinson et al. 1995, Mitsch and Gosselink 2000). However, less than 37% of bottomland hardwood forests remain in east Texas, and less than 22% still exist in the Lower Mississippi Alluvial Valley (Frye 1987). These losses are largely due to agricultural development and urbanization (Frye 1987). As forested wetlands are lost or degraded, the vital functions they perform are also lost or minimized. To regain these functions it is necessary to restore or create bottomland hardwood forests with similar structure and function as those being lost. However, this has proven difficult because of the complexity of ecosystem processes (Zedler and Callaway 1999).

Proper hydrologic restoration is key to successful wetland restoration. Most wetland restoration failures have been attributed to failures to restore proper hydrologic regimes (Mitsch and Gosselink 2000, Stolt et al. 2000). Restoring hydrology in urbanizing floodplains can be particularly difficult because of drastically altered stream hydrology (Schueler 1992). Restoration efforts resulting in wetlands becoming excessively flooded may limit above ground production of bottomland species, while those resulting in dry conditions may not support wetland species (Niswander and Mitsch 1995).

Three chapters of this dissertation directly address issues of bottomland restoration in urbanizing environments. The objective of the first chapter was to determine responses of bottomland hardwood seedlings' productivity to experimental flooding regimes typical of urban environments, specifically altered flooding frequency and duration. Studies have shown differences in plant communities between rural and

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This dissertation follows the style of Ecology.

urban floodplain forests (Ehrenfeld and Schneider 1993, Burton et al. 2005), but as yet, the magnitude and trajectory of these changes remains unpredictable. In order to increase understanding of direct impacts of urban flooding regimes on species productivity, such as stem and leaf growth, *Cephalanthus occidentalis*, *Fraxinus pennsylvanica*, and *Quercus shumardii* seedlings were subjected to experimental flooding regimes typical of floodplain forests in rural and urban settings.

The objective of the second chapter was to determine responses of planted and colonizing species to created microtopography and its influence on soil physiochemical development within a mound-and-pool complex. Spatial and temporal heterogeneity are thought to promote diverse communities (Huston 1994), and topographic heterogeneity has long been recognized as an important component of bottomland hardwood forests (Oosting 1942, Wikum and Wali 1974, Buchholz 1981, Hardin and Wistendahl 1983). Despite its importance and influence on bottomland forest structure and function, microtopography has rarely been considered in bottomland restoration, and those that have typically only assess planted species survival. This study assessed the role of microtopography in influencing hydrology, soil properties, and planted and colonizing species survival and distribution.

The objective of the third chapter was to assess biotic and abiotic responses to topographic heterogeneity in combination with various planting mixtures and soil treatments as part of bottomland forest restoration. Topographic heterogeneity may buffer some of the effects of urban hydrology when the previous hydrologic regime is unknown or unable to be restored. Large-scale topographic heterogeneity is a result of the formation of floodplains and includes drainages, meander scrolls, levees, and swales (Buchholz 1981, Mitsch and Gosselink 2000). Species associations and distinct communities often occur at different levels along hydrologic gradients created by topographic variations (Buchholz 1981, Keddy and Ellis 1984). As such, restoring topography in urban floodplain forests may create hydrologic gradients conducive to supporting a diverse community of wetland species when exact hydrologic signatures are unpredictable.

In addition to assessing bottomland forest restoration in urban environments, this dissertation includes research conducted on learning and motivation of undergraduate students in an ecology laboratory. Because the primary role of academics is education, increasing emphasis for ecology academicians is being placed on preparing graduate students to succeed in teaching in the classroom (Campbell and Fuller 2005, Fleet and Rosser 2006). One method to accomplish this is to perform education research and action research, i.e., conducting research in the classroom. Action research is “continual professional development” (Calhoun 2002) that allows educators to identify learning challenges, evaluate methods of overcoming these challenges, and disseminate results through publication or other outlets (Smith and Knight 1997). This is much the same process that ecologists follow when addressing ecological questions. Additionally, results of action research can be applied directly to the classroom, thereby improving instruction and enhancing understanding of ecological concepts for learners. As such, action research can play a significant role in the development of a PhD student preparing for a career in academia.

The fourth chapter presents results of education research conducted in an undergraduate ecology lab. Geographic information systems (GIS) have long been used by researchers as tools to store, manage, analyze, and display spatial data, but they are also increasingly being used by educators in K-16 classrooms as a means to support problem-based learning. However, relatively little research has assessed impacts of GIS on student motivation and knowledge. The objective of this chapter was to determine if GIS could be used to enhance knowledge of ecological concepts and improve attitudes of students enrolled in an introductory, field-based undergraduate ecology lab.

## CHAPTER II

### BOTTOMLAND HARDWOOD FOREST SPECIES RESPONSES TO FLOODING REGIMES ALONG AN URBANIZATION GRADIENT\*

#### Introduction

Hydrology is considered one of the most influential factors regulating the structure and function of wetlands (Bedford 1996, Mitsch and Gosselink 2000). The hydrologic regime largely determines species composition and productivity, soil biochemistry, and nutrient cycling (Mitsch and Gosselink 2000, Ahn et al. 2004, Schilling and Lockaby 2005). Altering wetland hydrology results in changes in germination and sedimentation rates, species productivity and composition, and water and habitat quality (Toner and Keddy 1997, Keddy 2000, Paul and Meyer 2001, Ahn et al. 2004, Walls et al. 2005). Such changes make the conservation and restoration of wetlands a difficult process (Tremolieres et al. 1998, Zedler 2000).

Urbanization alters stream hydrology in a number of ways. As the area of impervious surface cover increases through the process of urbanization, surface runoff increases (Chow et al. 1988, Arnold and Gibbons 1996). This results in increased stream velocity, flooding magnitude, and flooding frequency (Schueler 1992). Flood duration typically decreases as the time from peak to base flow discharge is reduced (Paul and Meyer 2001); however, duration may increase in watersheds with a patchy distribution of impervious surface cover or those in which impervious surface cover is concentrated lower in the catchment, which results in two peak flows (Hirsch et al. 1990, Paul and Meyer 2001, Sheeder et al. 2002). Flood duration, therefore, appears to depend on the degree and location of urbanization within watershed. Because of the drastic changes to stream hydrology caused by urbanization, floodplain wetlands such as bottomland hardwood forests are particularly susceptible to altered urban flooding regimes.

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\*Reprinted from *Ecological Engineering*, M.E. Simmons, X.B. Wu, and S.G. Whisenant, "Bottomland hardwood forest species responses to flooding regimes along an urbanization gradient" doi:10.1016/j.ecoeng.2006.07005, Copyright 2007, with permission from Elsevier.



Due to the importance of hydrology in determining vegetative species survival, productivity, and composition, many studies have focused on determining the flood tolerance of a variety of bottomland hardwood species (Hook and Brown 1973, Hook 1984, Theriot 1992, McLeod 2000). Historically, these studies consisted of flooding seedlings for a specified amount of time while recording information on productivity, vigor, and survival throughout the flooding period, thereby resulting in flood tolerance ratings (Hook and Brown 1973, Gomes and Kozlowski 1980, Hook 1984, Gravatt and Kirby 1998). More recent studies include interactions between flooding and other environmental factors such as salinity, drought, competition, and sedimentation (Nash and Graves 1993, Conner et al. 1998, McLeod et al. 2000, Walls et al. 2005). These flood-tolerance ratings can be used to indicate a species' degree of adaptation to continuous flooding and can be useful in selecting species to be used in wetland management, revegetation, and restoration efforts (McLeod 2000).

Flood-tolerance studies have great importance in many ecological and restoration realms; however, they do not answer questions regarding species' flooding tolerance under various flooding regimes. Studies have addressed species and community responses to regulated rivers (Toner and Keddy 1997, Robertson et al. 2001) and characterized vegetative zonation within floodplains (Metzler and Damman 1985, Blom et al. 1990, Siebel and Blom 1998), but few have explicitly tested responses of bottomland species to the effects of altered hydrology caused by urbanization (Nash and Graves 1993). Studies have shown differences in plant communities between rural and urban floodplain forests (Ehrenfeld and Schneider 1993, Burton et al. 2005), but as yet, the magnitude and trajectory of these changes remain unpredictable. As urbanization continues to encroach upon wetlands, the need to understand species and community responses to changing hydrology continues to increase, as does the need to restore urban wetlands.

The objectives of this study were to (1) determine responses of bottomland hardwood seedlings' productivity to experimental urban flooding regimes, specifically altered flooding frequency and duration, and (2) determine whether urban flooding

regimes impact species from various wetland indicator groups differently. Productivity of bottomland hardwood species exposed to a ‘typical’ flooding regime from a rural setting was compared with productivity of species exposed to flooding regimes from two different urban settings. The experimental urban flooding regimes included an “Urban-short” regime, with double the flooding frequency of the Rural regime but with decreased flood duration, and an “Urban-long” regime, with double the flooding frequency and a longer flood duration than the Rural regime.

## **Methods**

### *Species selection*

Three woody species, *Cephalanthus occidentalis* L. (buttonbush), *Fraxinus pennsylvanica* Marsh. (green ash), and *Quercus shumardii* Buckl. (shumard oak), were selected to examine the impacts of urban hydrology on species productivity. These species were chosen because of their prevalence in bottomland hardwood forests, their importance in bottomland restoration, and to test the effects of flooding on species with various flood tolerance levels (Allen 1990, King 1996, Kruse and Groninger 2003). As such, these species represent three different wetland indicator groups in Region 6 of the Natural Resources Conservation Service, which consists of Oklahoma and Texas, USA. *Cephalanthus occidentalis* is a highly flood-tolerant, obligate wetland shrub that is common along riparian areas that are inundated for part of the year (Hook 1984, Reed 1988). *Fraxinus pennsylvanica* is a facultative wetland tree adapted to fertile, moist, well-drained soils, but can survive in a variety of soil types and soil moisture regimes (Wright 1959, Reed 1988). *Quercus shumardii* is a facultative tree that grows best on terraces of moist, well-drained loamy soils (Reed 1988, Burns and Honkala 1990) and is only weakly tolerant of flooding (Hook 1984).

### *Experiment setup*

Nine 1135-liter, round (diameter = 1.7 m) Rubbermaid® stock tanks were set up outdoors in the Ecology and Natural Resource Teaching Area of Texas A&M University

in College Station, Texas, for the flooding experiment. The stock tanks were fitted with drain plugs on the bottom, which were used to control the duration of flooding. The round tanks were filled with 12 to 15 cm of coarse gravel and separated into three equal wedge-shaped sections with polystyrene insulation. The polystyrene was used to prevent different species from competing for belowground resources. Approximately 50 to 55 cm of soil was packed over the gravel in each of the sections. The soil, which had been collected from a floodplain, was mixed with sand by a local vendor. Ten bare-root seedlings of each species were then planted in one randomly selected section of each tank. The seedlings were allowed to grow for approximately two months (from March to mid-May) following which five average-sized individuals of each species in each section were chosen for the study. The remaining individuals were removed from the tanks. This resulted in a sample size of 15 individuals for *C. occidentalis* and *F. pennsylvanica* per treatment. However, of the 90 *Q. shumardii* initially planted only 38 survived until the start of the experiment. Poor survival of *Quercus* seedlings is not uncommon (Kruse and Groninger 2003). To maximize sample size, all surviving *Q. shumardii* individuals were retained for use in the experiment. This resulted in potential error, as initial heights of *Q. shumardii* seedlings varied substantially from 10 cm to 100 cm. Data collected from *Q. shumardii* seedlings that died during the experiment were not included in the final analyses. Final *Q. shumardii* sample size was 11 individuals per treatment.

### *Flooding regimes*

Flooding treatments included one rural and two urban flooding regimes. Three stock tanks were randomly chosen as replicates for each of the three flooding treatments. The Rural flooding treatment consisted of three floods between May and August 2002 with a duration of seven days per flood. The Urban-short treatment consisted of six floods each lasting four days. The Urban-long treatment also had a flooding frequency of six floods but with a duration of ten days per flood. The six flooding cycles were initiated on 21 May, 9 June, 27 June, 15 July, 2 August, and 20 August 2002. The

Urban-short and Urban-long treatments were flooded during each flooding cycle. The Rural treatment was flooded on the first, third, and fifth flooding cycles. On the first day of the second, fourth, and sixth cycles, the Rural tanks were filled and immediately drained to simulate a precipitation event but without the effects of flooding. The stock tanks were filled with tap water to a depth of 5 cm above the soil surface during the flooding cycles. This level was maintained for the duration of each flood.

Frequency and duration of the rural flooding regime were based on information from generalized flooding regimes for bottomland hardwood forests (Hodges 1998) and from data collected from a USGS stream gage station located on Rowlett Creek in Dallas County, Texas (unpublished data). These sources indicated that a typical bottomland hardwood forest may be flooded three times during the growing season. Actual flooding duration within bottomlands is a result of the length of time the stream exceeds overbank flow and topographic features such as ridges and sloughs that may retain surface water. Therefore, flooding duration cannot be determined by stream-gage data alone. Few studies have precisely reported the duration of water inundation in bottomlands (Metzler and Damman 1985, Baker et al. 2001), but based on available literature, a typical flood can last approximately seven days (Hodges 1998). This average was used in this study for the Rural flooding treatment.

The regimes for the two urban flooding treatments were based on data from the literature and the stream gage station. The watershed upstream from this gage station has experienced a high degree of urbanization with 5.1% impervious surface cover in 1972 and 35.1% impervious surface cover in 1995 (unpublished data). Data from the gage station indicated that flooding frequency doubled from the 1970s to the 1990s; therefore, the frequency of floods for the two urban flood treatments were double that of the Rural treatment. An Urban-short treatment with a flooding frequency of six floods and a duration of four days per flood was used to represent the flooding regime of a bottomland hardwood forest with high impervious surface cover that would reduce the time from peak to base-flow discharge compared with a rural flooding regime. An

Urban-long treatment was used to represent a watershed with a patchy distribution of impervious surface cover, which would increase flood duration.

#### *Data collection and statistical analyses*

Stem length, leaf area, and total leaf, stem, and root biomass were used as response variables to determine the impact of flooding regimes on the seedlings. Stem length and leaf area were measured immediately prior to and following each flooding cycle. When stem dieback occurred, stem lengths were measured to the point where stems turned brown or black. Leaf area was determined at each sampling period by counting the total numbers of leaves and categorizing each as small, medium, or large. The areas of ten representative leaves from each species in each size category were measured using a Licor® leaf area meter. The averaged size was multiplied by the total number of leaves counted at each sampling date to determine total leaf area for each seedling. Stem length and leaf area data were normalized by using percent change from pre-flood measurements (Smit 1988). Multivariate analysis of variance (MANOVA) was used to evaluate the effects of flooding treatments on stem length and leaf area over the sampling periods (referred to as ‘time’). The initial sampling period was not included as values for this period were set to zero. Contrasts were used to reveal differences between treatments within the MANOVA. Analysis of variance (ANOVA) was then used on stem length and leaf area for each sampling period. Tukey’s test adjusted for multiple comparisons was used to compare treatment means where significant differences were revealed by ANOVA. MANOVA was again used for specific time periods when ANOVA results indicated temporal patterns in species’ responses.

Aboveground biomass was determined following the final flooding cycle by harvesting the leaves and stems. Leaves were dried at 75 C for 68 hours, and stems were dried at 60 C for 117 hours. An estimate of root biomass was calculated by collecting three 790-cm<sup>3</sup> soil cores from each wedge-shaped section in each stock tank. Soil cores were washed with a Hydropneumatic Elutriation System (Gillison's Variety Fabrication, Benzonia, MI, USA) equipped with a 410um sieve. Root materials were then dried at 65

C for 72 hours. Analysis of covariance (ANCOVA) was used to test for treatment differences in aboveground biomass with initial stem diameter used as the covariate to correct for variation in initial plant biomass (Siebel et al. 1998). ANOVA was used for root biomass, and Duncan's multiple-range test was used to determine differences between treatments for both aboveground and root biomass.

## Results

### *Stem length*

The effects of time ( $F = 39.55$ ,  $P < 0.05$ ) and the interaction between time and flooding treatment ( $F = 6.24$ ,  $P < 0.05$ ) had significant impacts on *C. occidentalis* stem length as revealed by MANOVA. Contrasts indicated differences between the time-by-Rural versus Urban-short interaction ( $F = 21.73$ ,  $P < 0.05$ ), as well as marginal differences between the time-by-Rural versus Urban-long interaction ( $F = 9.25$ ,  $P < 0.10$ ). Changes in stem growth rates among treatments between sampling periods two and three, and three and four most likely caused these differences. Following sampling period four, only the effect of time ( $F = 27.38$ ,  $P < 0.05$ ) was significant, with the time-by-Urban-short versus Urban-long contrast being marginally significant ( $F = 4.92$ ,  $P < 0.10$ ). Although MANOVA revealed differences between treatments over time, there were no differences in mean stem lengths between treatments for any sampling period as revealed by ANOVA (Fig 2.1a).

MANOVA indicated a time effect for *F. pennsylvanica* stem lengths ( $F = 15.40$ ,  $P < 0.10$ ) with a slight time-by-treatment interaction ( $F = 3.92$ ,  $P < 0.10$ ). The initial response of stem growth was similar between treatments, but between sampling dates three and seven MANOVA revealed significant effects of time ( $F = 26.20$ ,  $P < 0.05$ ), contrast of time-by-Rural versus Urban-long ( $F = 11.85$ ,  $P < 0.05$ ), and contrast of time-by-Rural versus Urban-short ( $F = 13.54$ ,  $P < 0.05$ ). ANOVA results for each sampling date correspond with MANOVA results in that most of the differences occurred during

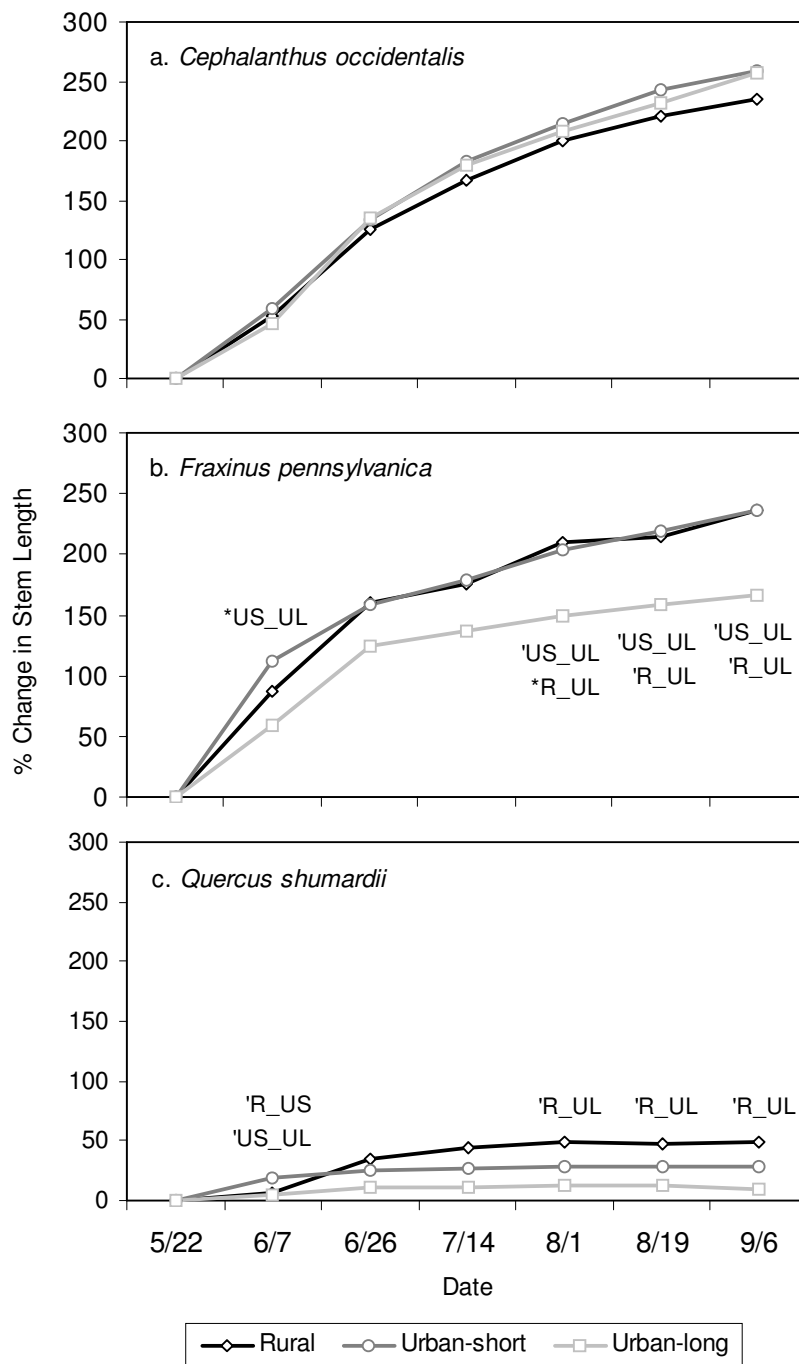


Figure 2.1. Percent change in stem lengths between 22 May and 6 September for (a) *Cephalanthus occidentalis*, (b) *Fraxinus pennsylvanica*, and (c) *Quercus shumardii*. The letters (R = Rural, US = Urban-short, UL = Urban-long treatments) indicate significant differences between treatments at each sampling date. A prime (') before the treatments indicates a significance level of  $P < 0.10$ . An asterisk (\*) indicates significance level of  $P < 0.05$ .

the later sampling dates (Fig. 2.1b); however, differences between the Urban-short and Urban-long treatments initially occurred prior to the second flooding cycle and continued during the fifth, sixth, and seventh sampling periods (Fig. 2.1b). Stem growth in the Rural treatment was also significantly greater than the Urban-long treatment during the final three sampling periods (Fig. 2.1b). There were no significant differences at any time period between the Rural and Urban-short treatments.

Percent change in *Q. shumardii* stem length showed a similar pattern with reduced stem growth in the Urban-long treatment starting prior to the second flooding cycle with differences between the Rural and Urban-long treatments occurring during the second, fifth, sixth, and seventh sampling periods (Fig. 2.1c). MANOVA results also indicated strong effects of time ( $F = 38.73, P < 0.05$ ), marginal effects for the contrast of time-by-Rural versus Urban-long ( $F = 11.40, P < 0.10$ ), and marginal effects for the contrast of time-by-Rural versus Urban-short ( $F = 11.02, P < 0.10$ ).

Final percent change in stem lengths for *C. occidentalis* and *F. pennsylvanica* were comparable in the Rural treatment with increases of 235% and 236%, respectively. The Urban-short treatment also produced similar results in *C. occidentalis* and *F. pennsylvanica* with increases of 259% and 236%, respectively. However, differences in stem length were pronounced in the Urban-long treatment with *C. occidentalis* increasing 257% and *F. pennsylvanica* increasing 167% by the end of the study. Final *Q. shumardii* stem length increased by 49% in the Rural treatment, 29% in the Urban-short treatment, and 9% in the Urban-long treatment.

### *Leaf area*

Results of MANOVA and ANOVA on percent change in leaf area revealed differences between treatments for all three species, with reduced leaf-area growth in the Urban-long treatment. There was a significant time effect on *C. occidentalis* leaf area ( $F = 190.09, P < 0.05$ ) but no time-by-treatment interactions as revealed by MANOVA. Results of ANOVA, however, indicated differences between Urban-short and Urban-long treatments during all but the third sampling period (Fig. 2.2a). This seeming



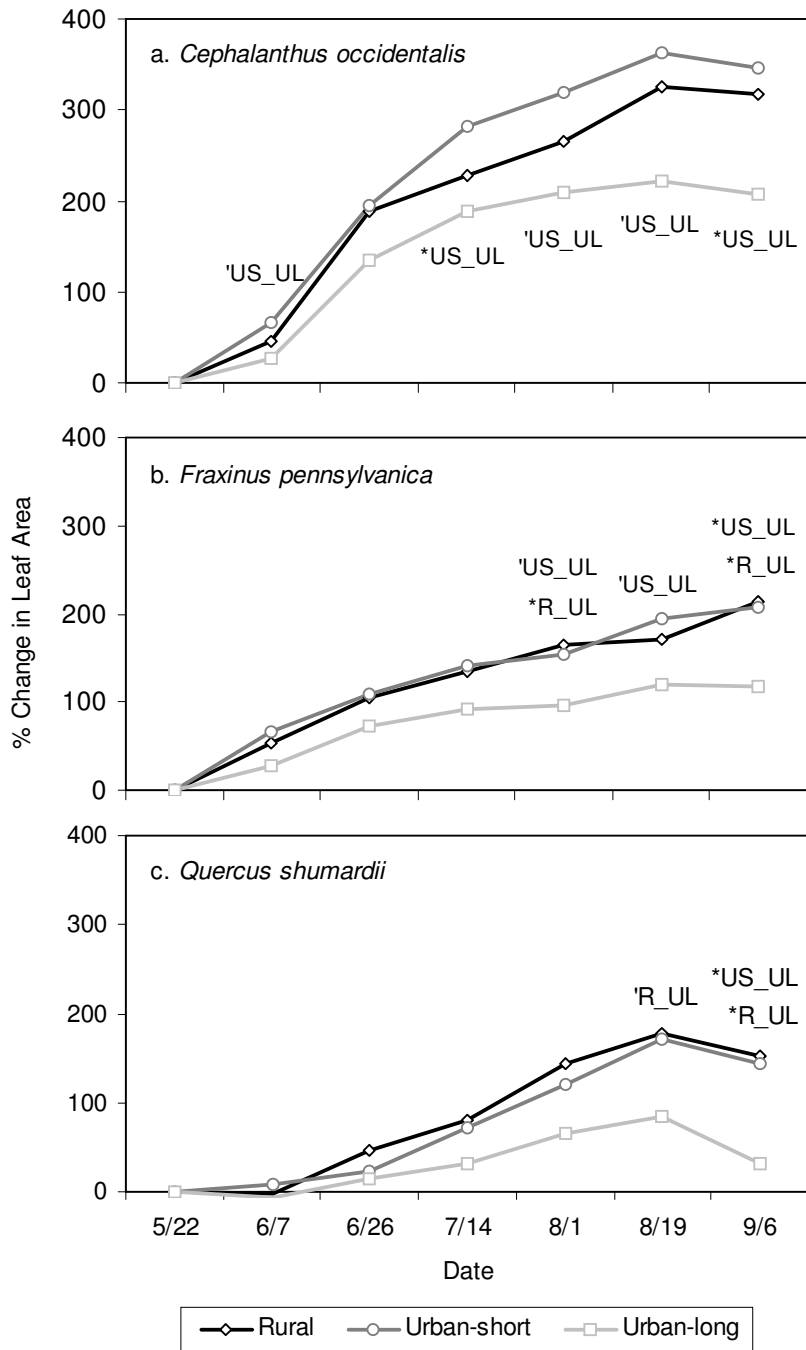


Figure 2.2. Percent change in leaf area between 22 May and 6 September for (a) *Cephalanthus occidentalis*, (b) *Fraxinus pennsylvanica*, and (c) *Quercus shumardii*. The letters (R = Rural, US = Urban-short, UL = Urban-long treatments) indicate significant differences between treatments at each sampling date. A prime (') before the treatments indicates a significance level of  $P < 0.10$ . An asterisk (\*) indicates significance level of  $P < 0.05$ .

discrepancy can be accounted for as the pattern of leaf area change was similar between treatments, resulting in no differences in MANOVA, but the magnitude of the changes were different, as indicated by ANOVA.

There was a significant time effect on *F. pennsylvanica* leaf area ( $F = 73.95$ ,  $P < 0.05$ ) but no overall time-by-treatment interactions; however, MANOVA results for the fourth through the seventh sampling dates indicate marginal differences occurring in the contrast of time-by-Rural versus Urban-long ( $F = 4.77$ ,  $P < 0.10$ ) and contrast of time-by-Urban-short versus Urban-long ( $F = 5.80$ ,  $P < 0.10$ ). ANOVAs corroborate these results as leaf area in the Urban-long treatment was significantly less than the Urban-short treatment during the fifth, sixth, and seventh sampling dates (Fig. 2.2b). The Urban-long treatment was also significantly lower than the Rural treatment during the fifth and seventh sampling dates. By the end of the study period, leaf area in Rural and Urban-short treatments had increased by 213% and 207%, respectively, compared with only 118% in the Urban-long treatment. There were no significant differences between the Rural and Urban-short treatments.

MANOVA revealed a significant effect of time on *Q. shumardii* leaf area ( $F = 59.13$ ,  $P < 0.05$ ), as well as in the contrasts of time-by-Rural versus Urban-short ( $F = 29.32$ ,  $P < 0.05$ ) and time-by-Urban-short versus Urban-long ( $F = 31.52$ ,  $P < 0.05$ ). Change in leaf area was significantly lower in the Urban-long treatment compared with the Rural treatment during the sixth and seventh sampling dates (Fig. 2.2c). The Urban-short treatment also had higher leaf area than the Urban-long treatment at the final sampling date. Final results indicate a net gain in leaf area of 152%, 144%, and 32%, for the Rural, Urban-short, and Urban-long treatments, respectively.

#### *Aboveground biomass*

Results of ANCOVA revealed differences between treatments in *C. occidentalis* leaf biomass ( $F = 9.04$ ,  $P < 0.05$ , Fig. 2.3a). Duncan's multiple-range test revealed significantly lower leaf biomass in the Urban-long treatment (12.7 g) than the Rural (17.3 g) and Urban-short (18.1 g) treatments ( $P < 0.05$ , Fig. 2.3a). There were no

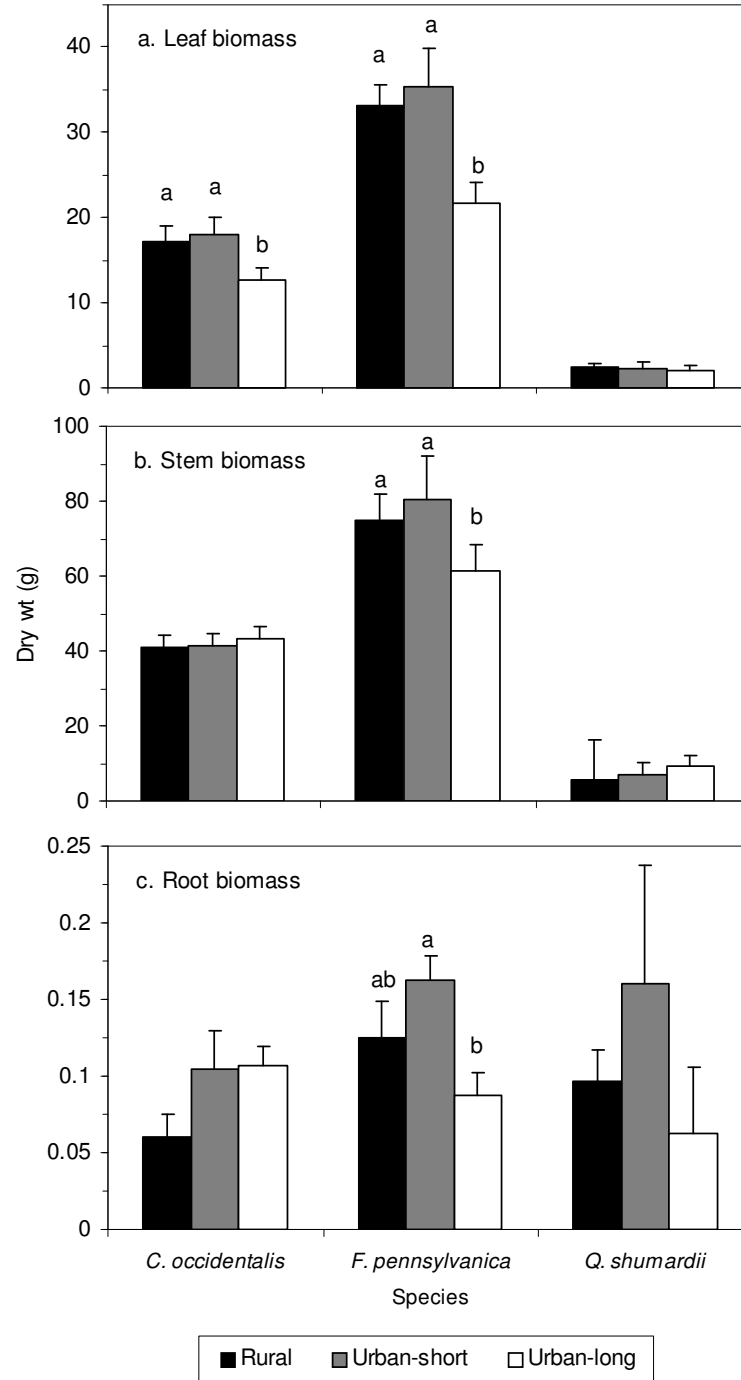


Figure 2.3. Results of (a) leaf, (b) stem, and (c) root biomass for *Cephalanthus occidentalis*, *Fraxinus pennsylvanica*, and *Quercus shumardii*. Bars specify standard error, and different letters indicate significant differences between treatments at  $P < 0.05$ .

differences in stem biomass ( $P = 0.80$ , Fig. 2.3b). Results of both *F. pennsylvanica* leaf and stem biomass (ANCOVA  $F = 90.55$ ,  $P < 0.05$  and  $F = 12.17$ ,  $P < 0.05$ , respectively) were significantly lower in the Urban-long treatment than the Urban-short and Rural treatments (Duncan  $P < 0.05$ , Figs. 2.3a and 2.3b). There were no treatment differences for *Q. shumardii* in either leaf or stem biomass.

#### *Root biomass*

There were no significant differences in root biomass between treatments for *C. occidentalis* or *Q. shumardii* (ANOVA  $F = 2.00$ ,  $P = 0.22$  and  $F = 1.07$ ,  $P = 0.40$ , respectively, Fig. 2.3c). Large standard errors in *Q. shumardii* might have been a result of small sample size and/or lack of uniformity in initial plant sizes. ANOVA revealed weak differences in *F. pennsylvanica* root biomass at  $P < 0.08$ , while Duncan's multiple-range test indicated less biomass in the Urban-long treatment than the Urban-short treatment ( $P < 0.05$ , Fig. 2.3c).

#### **Discussion**

Urban flooding regimes change as impervious surface cover increases due to increased urban growth. Typically, an increase in impervious surface cover will cause increased flooding frequency and magnitude (Schueler 1992), but flood duration may increase or decrease depending on the degree of urbanization and the spatial arrangement and location of impervious surface cover within the basin (Hirsch et al. 1990, Paul and Meyer 2001). The results of this study show that experimental flooding regimes used to mimic those altered by various stages and spatial arrangements of urbanization within a basin have direct impacts on bottomland species productivity when grown in a monoculture, and these impacts vary for species of different functional groups. These results also illustrate the importance of the temporal pattern of flooding on species' responses, as opposed to the many studies conducted to measure species' tolerances to prolonged continuous flooding.

### *Species' responses*

In this study, urban flooding had various effects on three bottomland species with the least impact occurring in *C. occidentalis*, the largest occurring in *F. pennsylvanica*, and moderate impacts occurring in *Q. shumardii*. Productivity in *C. occidentalis*, which was chosen to represent wetland obligates, was only mildly reduced by urban flooding regimes. The only impacts were seen in the Urban-long treatment, which reduced growth of leaf area compared with the Urban-short treatment and final leaf biomass compared with the Rural and Urban-short treatments. Differences in MANOVA stem length were due to changes in growth patterns over time and not a reduction in growth. As a wetland obligate, *C. occidentalis* is highly tolerant of flooding (Hook 1984, Faber-Langendoen and Maycock 1989) and appears to be well adapted to the rural and urban flooding regimes used in this study.

The Urban-long treatment greatly impacted *F. pennsylvanica*, a facultative wetland species, by reducing stem length and leaf area, as well as reducing final leaf, stem, and root biomass; however, there were no differences in any of the response variables between the Rural and Urban-short treatments as indicated by ANOVA. *Fraxinus pennsylvanica* is moderately tolerant of flooding and has been documented to remain healthy in flooded conditions up to 40% of the growing season (Wright 1959, Hook and Brown 1973). Flooding in the Rural and Urban-short treatments occurred for approximately 10% and 12% of the growing season, respectively, while flooding in the Urban-long treatment occurred for approximately 29% of the growing season. Although the temporal pattern of flooding was different between the Rural and Urban-short treatments, total flooding duration was similar between the two and well below 40%, which may explain why differences were seen in MANOVA results between sampling dates 3 and 7 but not in the ANOVA.

Responses to the urban flooding regimes were not as strong in *Q. shumardii*, a facultative species, as those in *F. pennsylvanica* as revealed by ANOVA. Percent change in stem length in the Urban-long treatment was reduced compared with the Rural treatment during four of the sampling periods, but the Urban-long treatment differed

from the Urban-short treatment during only one sampling period. Leaf area was also significantly lower in the Urban-long treatment than the Rural treatment during the sixth and seventh sampling dates and the Urban-short treatment during the seventh sampling date. However, there were no differences in final leaf, stem, or root biomass.

Differences between treatments of *Q. shumardii* may not have been as strong as expected because of small sample size and large initial variation due to low initial seedling survival and oak borer herbivory. Because of low initial survival, all surviving individuals were used in the study, as opposed to selecting for average-sized individuals of *C. occidentalis* and *F. pennsylvanica*. This resulted in wide variations in initial stem lengths and numbers of leaves between individuals. Also, individuals that died during the experiment were not included in the analyses because it was unknown if mortality was a direct result of treatment effects, poor seedling quality, or improper seedling handling or planting. Finally, in July, unidentified oak borers were discovered in stems of several of the oaks. Four trees in both the Urban-short and Urban-long treatments were infested, and seven trees in the Rural treatment were infested. The borers were removed, but some of the trees may have been negatively affected by reduced productivity (Dajoz 2000). Because these factors reduced sample size and potentially influenced plant growth, actual treatment effects might have been obscured (see standard error bars in Fig. 2.3b and 2.3c). Although the trends for *Q. shumardii* productivity indicate differences between Rural and Urban-long treatments, further experiments need to be conducted in order to discount these potentially confounding factors.

Although not explicitly tested in this study, flood duration appears to have had a greater impact on species productivity than did flooding frequency as evidenced by reduced productivity in the Urban-long treatment compared with the other treatments. Total flood duration for the length of the study was 28, 24, and 60 days for the Rural, Urban-short, and Urban-long treatments, respectively. The Urban-long treatment most likely suppressed productivity because of the large number of cumulative days of inundation during the growing season (Megonigal et al. 1997, Ahn et al. 2004). Although the Urban-short treatment had twice the floods as the Rural treatment, there

were few differences between these treatments in any species. This implies that if flooding frequency increases but total flood duration remains unchanged in floodplain forests undergoing urbanization, there may be little impact on productivity of some species from altered hydrology. However, results of MANOVA do indicate differences between the Rural and Urban-short treatments in some of the cases based on changes in growth patterns over the study period, even when ANOVA revealed no differences between means at any one sampling date. Also, the effects of urbanization on hydrology are varied, including increased flooding magnitude and erosive forces, changes in sedimentation rates, species establishment, and nutrient cycling (Schueler 1992, Keddy 2000, Hughes et al. 2001). Further studies should incorporate these other factors to more accurately predict vegetation responses to urban hydrology.

#### *Implications for species composition changes*

Although the current study is too small in scale to extrapolate to ecosystem-level responses, results do show a direct link between altered flooding regimes and changes in species productivity. While there were few differences in productivity of *C. occidentalis* between treatments, differences in *F. pennsylvanica* and *Q. shumardii* were more pronounced. As such, altered hydrology can lead to changes in species composition when productivity of some species is suppressed while productivity of other species is unaffected or enhanced (Bledsoe and Shear 2000, Robertson et al. 2001, Townsend 2001). That being said, seedlings are less flood tolerant than mature plants (Kozlowski 1984a). Changes in woody species composition in floodplain forests may take many years to occur only after mature trees die and younger surviving trees of different species replace them.

In addition to flooding frequency and duration, the temporal flooding pattern can influence species productivity and ultimately composition (Gough and Grace 1998, Townsend 2001, Ahn et al. 2004). In this study, changes in productivity occurred within a short time, i.e., one growing season, and for some species, significant differences between flooding regimes developed after the first flood. For example, although the

difference in flood duration between treatments was relatively short, six days for Urban-long and Urban-short treatments and three days for Urban-long and Rural treatments, differences in *F. pennsylvanica* and *Q. shumardii* stem length and *C. occidentalis* leaf area emerged between treatments after the first flood. Additionally, the first two floods in the Urban-long treatment seem to have set the trajectory of growth for *Q. shumardii* stem length and leaf area, with very little growth occurring after the second flood. In contrast, plants in the Rural treatment were not flooded during the second flooding cycle during which time stem length and leaf area seem to have rebounded for all species as indicated by high percent growth. The two urban treatments, however, suppressed growth during the second flooding cycle. In some cases, differences were not seen until later floods, such as *F. pennsylvanica* leaf area between Rural and Urban-long treatments. The differences in response variables between species suggests that temporal flooding patterns affect different species in different ways, which may ultimately bring about changes in species composition (Toner and Keddy 1997, Siebel and Blom 1998).

#### *Implications for wetland restoration*

In a study on the status of United States wetlands from 1986 to 1997, Dahl (2000) found that forested wetlands experienced higher losses than any other wetland type, and 30% of all wetland loss was due to urbanization. As such, the need for forested wetland restoration, particularly in urban and urbanizing environments is great. Efforts to restore wetlands have proven difficult (Zedler and Callaway 1999, Stolt et al. 2000, Zedler 2000), and those in urbanizing environments are particularly difficult to restore because of modified hydrology, in which cases predisturbance conditions may be impractical or impossible to restore. In order to improve successful urban wetland restoration, proper steps should be taken to overcome the hydrologic limitations imposed by urbanization.

One way in which to reduce the effects of urban hydrologic regimes is to provide robustness within the system that would allow it to adapt to future changes. This can be



accomplished through the selection of species adapted to the hydrologic signature, including frequency, duration and temporal flooding pattern, of the site at the time of restoration (King and Keeland 1999, Bledsoe and Shear 2000, McLeod 2000, Johansson and Nilsson 2002). However, in urbanizing watersheds hydrology may continue to change as impervious surface cover increases. In such cases, it may be necessary to include species with wide ecological amplitudes with respect to flooding, i.e., include species that are adapted to current and potential future hydrologic conditions based on the degree of current and predicted future urbanization. Alternatively, species with narrow ecological niches can be selected if an adaptive management approach is used. This would require continued site evaluation with successive plant introductions based on the changing conditions (Walters and Holling 1990).

In this study, *C. occidentalis* was well adapted to both urban hydrologic regimes tested, making it potentially useful for urban wetland restoration in a variety of environments. *Fraxinus pennsylvanica* and *Q. shumardii* were also well adapted to the Urban-short flooding regime, which implies that these species may be used in restoration efforts in urbanizing environments that create floods of greater frequency but shorter duration. Expansions of the current study may prove beneficial for developing restoration strategies in species composition with respect to specific hydrologic regimes. By testing responses of a variety of species and species assemblages to specific urban flooding regimes, those species that are most adapted to current hydrologic conditions can be selected. If future hydrologic conditions can be predicted based on predictions of urbanization (Sheeder et al. 2002), studies such as this can provide insight into appropriate species assemblages for both current and future conditions.

Creating microtopography within a restored wetland may also compensate for changing urban hydrologic regimes. Microtopography creates hydrologic gradients along which a variety of species become established, resulting in high-diversity floodplain systems (Titus 1990). The presence of microtopography can allow vegetation to adjust their distribution along the elevation gradient as hydrology within the floodplain changes (Leyer 2005). Even slight microtopographic variations can result in

large changes in flooding dynamics. Bledsoe and Shear (2000) found that a 10-cm difference in surface elevation resulted in a 20% change in the frequency of surface flooding. Combining microtopography with a variety of species with both wide and narrow ecological amplitudes may provide enough robustness within the system to be able to adapt to continued hydrologic modifications.

Concepts and methods of reference site selection to improve restoration efforts have been covered extensively in the literature (Hobbs and Norton 1996, Hobbs and Harris 2001, Brooks et al. 2005, Ruiz-Jaen and Mitchell 2005). The hydrogeomorphic approach to defining reference standards based on reference wetlands (Smith et al. 1995), as well as the approach proposed by Ehrenfeld (2000) in urban watersheds rely heavily on reference site selection based on hydrologic conditions. The results of this study support the recommendation to base reference site selection in urbanizing environments on the urban hydrologic regime of the restoration project. Smith et al. (1995) suggest creating reference standards from data collected at reference sites that encompass a range of variability in both natural processes and anthropogenic disturbances within a given domain, or geographic area. In urban and urbanizing environments, altered hydrology may be the largest and most important anthropogenic disturbance. In these environments suitable reference wetlands with similar hydrologic regimes may be difficult to locate. By testing responses of species and species assemblages to urban hydrology, studies such as this may prove beneficial in species selection when the availability of reference sites is limited.

## **Conclusions**

This study has shown that urban flooding regimes can have direct effects on the productivity of bottomland species. Specifically, an urban flood with higher frequency and longer duration reduced growth in all three species tested. However, specific responses varied between species from different wetland indicator classes. *Cephalanthus occidentalis*, a wetland obligate, exhibited few changes with decreased leaf area and leaf biomass in the Urban-long treatment compared with Rural and Urban-

short treatments. Stem length, leaf area, and stem, leaf, and root biomass of *Fraxinus pennsylvanica*, a facultative wetland species, were reduced in the Urban-long treatment. Stem growth of *Quercus shumardii*, a facultative species, was reduced in the Urban-long treatment compared with the Rural treatment but not the Urban-short treatment, and differences in leaf area occurred only at the end of the growing season. Differences between flooding treatments occurred rapidly, at times after the first flooding cycle, and remained throughout much of the study. The implications of altered productivity as a result of urbanization-induced changes in flooding regimes include possible shifts in species composition, the importance of proper species selection in wetland restoration based on hydrologic conditions of the site, and the significance of selecting appropriate reference sites and reference standards.

### CHAPTER III

## SPECIES RESPONSES TO CREATED MOUND-AND-POOL TOPOGRAPHY AND SOIL TREATMENTS

### Introduction

Spatial and temporal heterogeneity are thought to promote diverse communities (Huston 1994), and topographic heterogeneity has long been recognized as an important component of bottomland hardwood forests (Oosting 1942, Wikum and Wali 1974, Buchholz 1981, Hardin and Wistendahl 1983). Microtopographic heterogeneity influences community structure through differential seedling recruitment along hydrologic gradients (Keddy and Ellis 1984), species responses to litter accumulation in depressions (Todd et al. 2000), variation in plant growth and mortality (Eldridge et al. 1991), and differences in biogeochemical cycling (Darke and Walbridge 2000). Microtopographic variations such as tip up mounds and depressions are often created through disturbances such as tree fall, sediment accumulation, erosion following flooding, and animal activities (Ehrenfeld 1995).

Although an important and natural part of bottomland hardwood forests, recreating topographic heterogeneity during restoration efforts has been largely absent (Stolt et al. 2000). However, restoring microtopography in wetland restoration may create edaphic and hydrologic variations that would support the establishment, colonization, and survival of a diverse community (Beatty 1984, Todd et al. 2000), thereby accelerating the development of wetland species composition and function (Bruland and Richardson 2005).

A common practice in wetland restoration is to introduce specific species to produce a desirable overstory community. However, high mortality rates often result due to poor suitability of site conditions to the species, particularly inappropriate hydrology (Whittecar and Daniels 1999, Patterson and Adams 2003). As hydrology is the most important factor in wetland ecology and restoration, proper species selection should reflect adaptation to hydrologic regimes encountered at the site (Simmons et al.

2007). For example, Bledsoe and Shear (2000) found that a 10 cm difference in surface elevation resulted in a 20% change in surface flooding frequency, and Vivian-Smith (1997) reported significant differences in plant community structure due to a 1 to 3 cm change in microtopography. Planting a mix of species adapted to a wide range of flooding conditions along microtopographic gradients may produce a diverse community by improving survival of planted species and promoting colonization of species adapted to particular hydrologic zones.

A few studies have assessed colonizing (Bruland and Richardson 2005) or planted (Patterson and Adams 2003) species responses to restored microtopography, but none have addressed the effects of restored or created microtopography on both planted woody and colonizing herbaceous species. Assessing both planted and colonizing community responses to created microtopography can provide a better framework from which to analyze restoration efforts. The objective of the current study was to determine the influences of created microtopography and soil treatments on (1) planted and colonizing species survival, distribution, and abundance and (2) soil physiochemical development. Specific research questions included: Will the creation of microtopography lead to spatial heterogeneity in soil physiochemical development that would promote differential colonization and planted species survival? Will straw-based erosion control mats differentially effect species colonization based on topographic position?

## **Methods**

### *Study site*

A 30-ha borrow pit at the Castle Drive Landfill, located in Garland, Dallas County, Texas (32° 93' N 96° 58' W), provided an opportunity to evaluate topoedaphic features, and hydrologic and vegetation dynamics of bottomland hardwood forest restoration. The borrow pit is adjacent to Rowlett Creek. Flooding from the creek periodically flooded the borrow pit until 2003, when landfill managers reinforced a levee separating the creek from the pit to enable construction of the site. Currently, water

inputs consist of precipitation, ground water, and runoff from the landfill during large precipitation events. The long-term average annual precipitation (30-year) as recorded in Rockwall, TX, which is approximately 12 km from Garland, is 999 mm, which occurs in a bimodal distribution with peaks in May (134 mm) and October (116 mm) (NOAA 2002).

Open since the 1970s, the landfill was capped in 2002. During the 30 years in operation, soil was removed from the borrow pit to be used in the landfill. By 2002, the top 5 m of soil had been excavated from the borrow pit. In 2004, part of the borrow pit was contoured to include a mound-and-pool complex (Fig. 3.1) following the design by Barry et al. (1996). A bulldozer with a 3-m wide blade was used to create a checker-board series of mounds and pools so that mound tops were approximately 60 to 100 cm higher in elevation than pool bottoms. Mounds were approximately 30 to 50 cm above and pools 30 to 50 cm below surrounding flat areas.

#### *Treatment Interactions*

A factorial design was implemented to evaluate the interactions between tree seedling planting mixes (three levels) and soil treatments (two levels) on mound tops, pool bottoms, and transitional side slopes between mound tops and pool bottoms. Each treatment combination was replicated five times totaling 30 plots.

The three levels of planting mixes included a pioneer mix, a mature mix, and no planting. All trees were bare-root seedlings and were planted across the mound-and-pool complex in February 2004. Holes for the seedlings were drilled with hand augers to a depth of 30–40 cm. A 10-g fertilizer tablet (20-10-5) was placed in each hole along with the seedling. Holes were then backfilled with sand. Species selected for planting were native to the Dallas region, adapted to clay soils as are found at the site, and ranged from weakly to highly tolerant of flooding during the growing season. The pioneer mix was composed of early successional, fast growing tree species including *Cephalanthus occidentalis* L. (buttonbush), *Fraxinus pennsylvanica* Marsh. (green ash), *Populus deltoides* Bart. ex Marsh. (eastern cottonwood), and *Salix nigra* Marsh. (black willow).

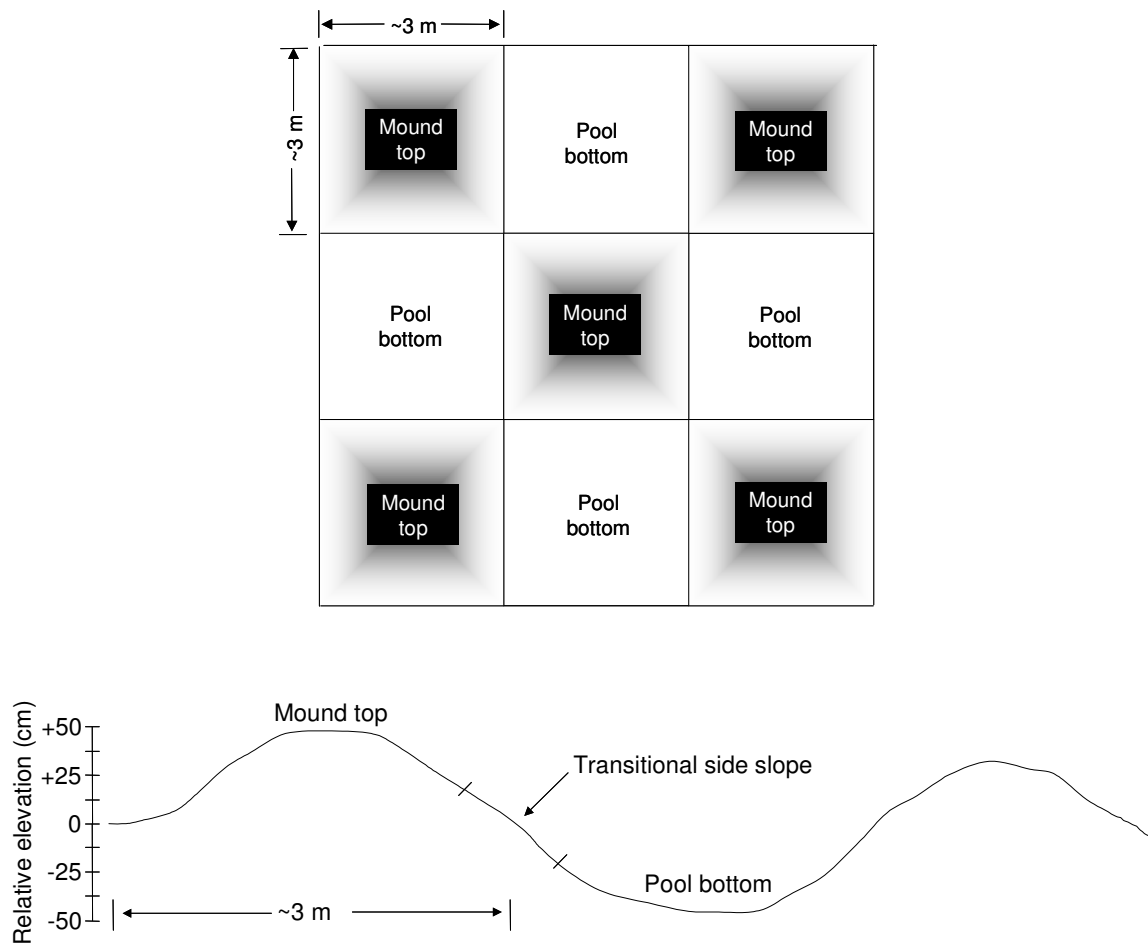


Figure 3.1. Aerial view and cross-section of mound-and-pool complex.

The mature mix was composed of species representative of mature forested wetlands in the region including *Acer negundo* L. (box elder), *Carya aquatica* (Michx. f.) Nutt. (water hickory), *C. illinoensis* (Wang.) K. Koch (pecan), *Celtis laevigata* Willd. (sugarberry), *Forestiera acuminata* (Michx.) Poir. (swamp privet), *Platanus occidentalis* L. (sycamore), *Quercus nigra* L. (water oak), *Quercus shumardii* Buckl. (shumard oak), and *Ulmus americana* L. (American elm). Each plot measured 12-m by 12-m, and trees were spaced 1.5 m apart making nine rows of plants per plot. One additional row was planted around each plot to serve as a buffer. Species were randomly located within each row of each plot.

The original design called for shredded wood mulch to be incorporated into the soil on half of the plots. However, prior to the time of planting, the soil was too wet to use machinery to incorporate the mulch into the soil. Instead of wood mulch, soil treatments included straw-based erosion control mats and no erosion control mats. The erosion control mats were purchased from North American Green and used as a substitute for mulch to provide a source of organic matter. Specifications for the mats included a thickness of 8.38 mm, mass of 218 g/m<sup>2</sup>, and light penetration of 11%. The mats were designed to have a functional longevity of up to 12 months. The straw fibers were enclosed within a lightweight photodegradable polypropylene net with a mesh size of 1.27 cm by 1.27 cm. The erosion control mats were applied to half the plots.

#### *Vegetation sampling*

Data collection occurred over a two-year period. Planted tree seedling growth, survival, and basal diameter were assessed in Spring and early Fall of 2004 and 2005. Because of high water levels early in 2005, seedling sampling in Spring 2005 was performed in June after water levels receded. Topographic position for each seedling was noted as mound top, intermediate side slope, or pool bottom. Herbaceous species composition was analyzed for three randomly selected mound tops, side slopes, and pool bottoms within each plot. Species composition was estimated within 1-m<sup>2</sup> quadrats centered on each topographic position. Herbaceous sampling occurred in late August and early September of each year during the time of maximum production. Numbers of colonizing woody species located within each quadrat were noted, and in 2005, percent litter cover within quadrats was estimated.

#### *Soil sampling and analysis*

Soil samples were collected in October 2005. A 2.5-cm soil probe was used to collect soil to a depth of 30 cm on mound tops and pool bottoms. Samples were collected within the same 1-m<sup>2</sup> quadrats used for herbaceous sampling. Three additional samples were collected for each topographic position from three randomly selected



locations totaling six samples per plot for each topographic position. A composite was made from the six samples for analysis. A more intensive sampling was conducted in plots planted with pioneer species. In these plots, samples were collected on intermediate side slopes in addition to mound tops and pool bottoms. Soil texture was determined via the hydrometer method (Gavlak et al. 2003), and plant-available potassium, calcium, magnesium, sodium, sulfur, and phosphorous were determined with the Mehlich 3 method (Mehlich 1984). A 1 N KCl solution was used to extract nitrate-nitrogen (Keeney and Nelson 1982), and dry combustion was used to determine total nitrogen (Bremner 1996). Loss on ignition was used to determine soil organic matter by heating soil samples for two hours at 150 C to remove hygroscopic water, after which the temperature was raised to 360 C for an additional two hours (Gavlak et al. 2003). Soil pH and electrical conductivity were determined in a 1:2 soil:water mixture using deionized water. Samples were stirred and allowed to equilibrate for at least 30 minutes. A hydrogen selective electrode was then used to determine soil pH, and a conductivity probe was used to determine electrical conductivity (Schofield and Taylor 1955, Rhoades 1982).

Because measures of soil moisture are difficult to perform in clay soils, a visual estimate was used to assess levels of water inundation and soil saturation in each of the pools every three weeks from 31 May 2005 through 8 September 2005. Categories from one to six were used to describe the level of inundation in the pools and level of soil saturation. Categories were based on percent inundation estimated as depth of water from pool bottoms to mound tops. A level of one indicated that less than 20% of the pool was filled, a two indicated 20-40% inundation, a three indicated 40-60% inundation, a four indicated 60- 80%, and a five indicated that the mounds and pools were inundated from 80% to the top of the mound. A six indicated water inundation over the top of the mound.

### *Data analysis*

All analyses were performed based on an incomplete block design. A mixed-model approach was used in which blocks and blocks nested within replicates were random variables (Federer and Wolfinger 1998). In this manner, interblock information was retained. Multivariate analysis of variance (MANOVA) was used to determine effects of time on tree seedling survival. Mixed model analyses were then used to determine differences in survival between topographic position, soil treatment, and the interaction between topographic position and soil treatment for each time period. Relative seedling growth rates were determined for all seedlings that survived until the final measurement in Fall 2005 ( $(\text{Diameter}_{\text{Fall2005}} - \text{Diameter}_{\text{Spring2004}}) / \text{Diameter}_{\text{Spring2004}} \times 100$ ). Soil nutrient values were log transformed and seedling growth rates were log transformed prior to analysis in order to better conform to normality assumptions of ANOVA. Results presented in figures and text are displayed in their original units.

## **Results**

### *Hydrology*

When tree seedlings were planted in February 2004, many of the pools were inundated up to 50%, but most pools were free of standing water by June. Precipitation in June and July 2004 was approximately twice the 30-year average but did not result in substantial inundation in the pools. On 2-3 January 2005, a large precipitation event resulted in complete inundation of the mounds and pools, which continued through much of the growing season. Because of extended water inundation during 2005, water levels within the pools were recorded between 31 May and 9 September. Water level depths were variable between treatments on 31 May from an average low of 38% inundation in plots without erosion control mats planted with pioneer species to 71% in plots with erosion control mats planted with mature species (Table 3.1). Plots in this latter treatment had higher inundation than any other treatment. Although inundation continued through much of the growing season, all treatments were free from water inundation by 8 September.

Table 3.1. Percent water inundation by depth in pools in 2005 for each soil/seedling treatment combination.

Soil treatment	Plant mix	31 May	9 June	29 June	21 July	16 Aug	8 Sept
No mat	None	40	41	14	4	0	0
	Pioneer	38	30	20	13	1	0
	Mature	52	49	18	6	1	0
Mat	None	65	60	24	7	0	0
	Pioneer	46	42	5	0	0	0
	Mature	71	64	27	14	6	0

### *Soil properties*

Soil texture was consistent between mounds, slopes, and pools. Mean sand, silt, and clay content were 18%, 33%, and 49%, respectively. Several differences existed between mound tops and pool bottoms for other soil properties (Table 3.2). Nitrate-nitrogen was ~83% greater in pools than on mounds, and total nitrogen and sulfur were also significantly higher in pools. Plots without erosion control mats had higher pH, total nitrogen, phosphorus, potassium, and sodium, but calcium was greater in plots with erosion control mats than without mats. There was also a slightly significant treatment interaction between topographic position and soil treatment for total nitrogen ( $F = 3.31$ ,  $P < 0.10$ ).

A more detailed assessment in which transitional slopes were sampled in addition to mound tops and pool bottoms revealed that while nitrate-nitrogen was greater in pools than on mounds, concentrations were also greater in pools than on slopes (Table 3.3). Sulfur exhibited a gradual increase from mound tops to transitional slopes to pool bottoms, while sodium was lower in pools than both mounds and slopes. Sulfur was also significantly higher in plots with erosion control mats than those without mats (Table 3.3). There was significantly more litter on mounds and slopes than in pools (41%, 38% and 24% for mounds, slopes, and pools, respectively,  $P < 0.01$ ) and in pioneer plots than in mature plots (37% and 30%, respectively,  $P < 0.05$ ). There were no differences in litter between soil treatments ( $P = 0.91$ ).

Table 3.2. Soil properties based on microtopographic position and soil treatment for unplanted plots and plots planted with the mature species mix. Values are means  $\pm$  one SE. *P* values in bold indicate probabilities considered to represent significant differences.

	Topographic position			Soil treatment		
	Mound	Pool	<i>P</i>	Mat	No Mat	<i>P</i>
pH	8.2 $\pm$ 0.04	8.1 $\pm$ 0.05	<b>0.060</b>	8.1 $\pm$ 0.04	8.3 $\pm$ 0.05	<b>0.004</b>
Nitrate-N (ppm)	2.9 $\pm$ 0.25	5.3 $\pm$ 0.52	<b>&lt;0.001</b>	4.1 $\pm$ 0.5	4.1 $\pm$ 0.5	0.689
Total N (%)	0.039 $\pm$ 0.002	0.045 $\pm$ 0.002	<b>0.040</b>	0.040 $\pm$ 0.002	0.044 $\pm$ 0.002	<b>0.085</b>
P (ppm)	4.9 $\pm$ 0.73	5.4 $\pm$ 0.80	0.671	4.0 $\pm$ 0.6	6.6 $\pm$ 0.8	<b>0.033</b>
K (ppm)	5.2 $\pm$ 0.06	5.3 $\pm$ 0.6	0.413	5.2 $\pm$ 0.05	5.3 $\pm$ 0.06	<b>0.084</b>
OM (%)	1.4 $\pm$ 0.06	1.5 $\pm$ 0.07	0.570	1.4 $\pm$ 0.08	1.5 $\pm$ 0.03	0.284
S (ppm)	83 $\pm$ 7	131 $\pm$ 27	<b>0.036</b>	125 $\pm$ 26	88 $\pm$ 9	0.131
Na (ppm)	531 $\pm$ 49	443 $\pm$ 37	<b>0.005</b>	429 $\pm$ 39	552 $\pm$ 46	<b>0.034</b>
Ca (ppm)	35453 $\pm$ 2457	36165 $\pm$ 2254	0.650	39541 $\pm$ 1910	31663 $\pm$ 2449	<b>0.036</b>
EC (umho/cm)	542 $\pm$ 30	629 $\pm$ 57	0.199	610 $\pm$ 55	558 $\pm$ 32	0.645

Table 3.3. Soil properties based on microtopographic position and soil treatment for plots planted with the pioneer species mix. Values are means  $\pm$  one SE. *P* values in bold indicate probabilities considered to represent significant differences. Different letters indicate significant differences between treatments.

	Topographic position				Soil treatment		
	Mound	Slope	Pool	<i>P</i>	Mat	No Mat	<i>P</i>
pH	8.3 $\pm$ 0.07	8.3 $\pm$ 0.05	8.2 $\pm$ 0.06	0.241	8.2 $\pm$ 0.05	8.3 $\pm$ 0.04	<b>0.033</b>
Nitrate-N (ppm)	2.9 $\pm$ 0.3 b	2.6 $\pm$ 0.18 b	4.9 $\pm$ 0.56 a	<b>&lt;0.001</b>	3.3 $\pm$ 0.25	3.7 $\pm$ 0.59	0.521
Total N (%)	0.039 $\pm$ 0.002	0.038 $\pm$ 0.002	0.040 $\pm$ 0.002	0.782	0.040 $\pm$ 0.001	0.038 $\pm$ 0.001	0.120
P (ppm)	4.9 $\pm$ 1.3	4.7 $\pm$ 1.2	3.8 $\pm$ 1.0	0.586	4.2 $\pm$ 0.85	4.8 $\pm$ 1.1	0.194
K (ppm)	188 $\pm$ 17	186 $\pm$ 17	188 $\pm$ 15	0.858	182 $\pm$ 12	193 $\pm$ 13	<b>0.035</b>
OM (%)	1.47 $\pm$ 0.10	1.55 $\pm$ 0.13	1.53 $\pm$ 0.09	0.552	1.5 $\pm$ 0.08	1.48 $\pm$ 0.10	0.599
S (ppm)	73 $\pm$ 8 b	97 $\pm$ 11 ab	133 $\pm$ 27 a	<b>0.056</b>	114 $\pm$ 17	85 $\pm$ 11	<b>0.088</b>
Na (ppm)	552 $\pm$ 68 a	559 $\pm$ 89 a	461 $\pm$ 64 b	<b>0.002</b>	465 $\pm$ 44	598 $\pm$ 74	0.436
Ca (ppm)	37,622 $\pm$ 3461	36,489 $\pm$ 3803	39,856 $\pm$ 3394	0.538	38,710 $\pm$ 2441	37,087 $\pm$ 3390	0.154
EC (umho/cm)	489 $\pm$ 41	607 $\pm$ 57	671 $\pm$ 75	0.154	632 $\pm$ 47	536 $\pm$ 54	0.120

### Seedling survival and growth

Initial tree seedling survival was high for most species following planting in March 2004 (Figs. 3.2 and 3.3), but survival significantly decreased for all species over time as indicated by MANOVA ( $P < 0.0001$ ). Neither topographic position nor soil treatment strongly influenced survival of the pioneer species *C. occidentalis*, *F. pennsylvanica*, or *S. nigra*, but survival of *P. deltoides* was significantly lower in pools than on mounds in Spring and Fall 2005 (Fig. 3.2). MANOVA also indicated a significant interaction between time and topographic position for *P. deltoides* ( $P < 0.05$ ). Further, in Fall 2005, *P. deltoides* had higher survival in plots without erosion control mats than with mats (33% and 22%, respectively,  $P = 0.057$ ). Final survival of pioneer species in Fall 2005 averaged 40–65% but varied from a low of 14.5% for *P. deltoides* in pools to a high of 74% for *C. occidentalis* in pools (Fig. 3.2).

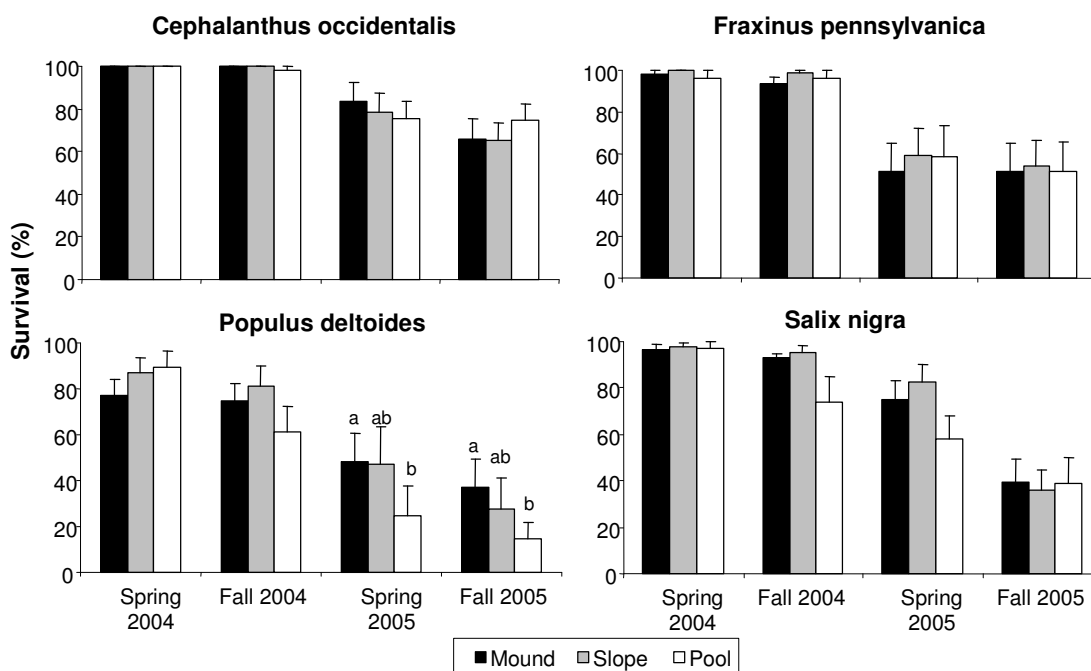


Figure 3.2. Survival of planted pioneer tree species in mounds, slopes, and pools from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicate significant differences between microtopographic zones at  $P < 0.10$ .

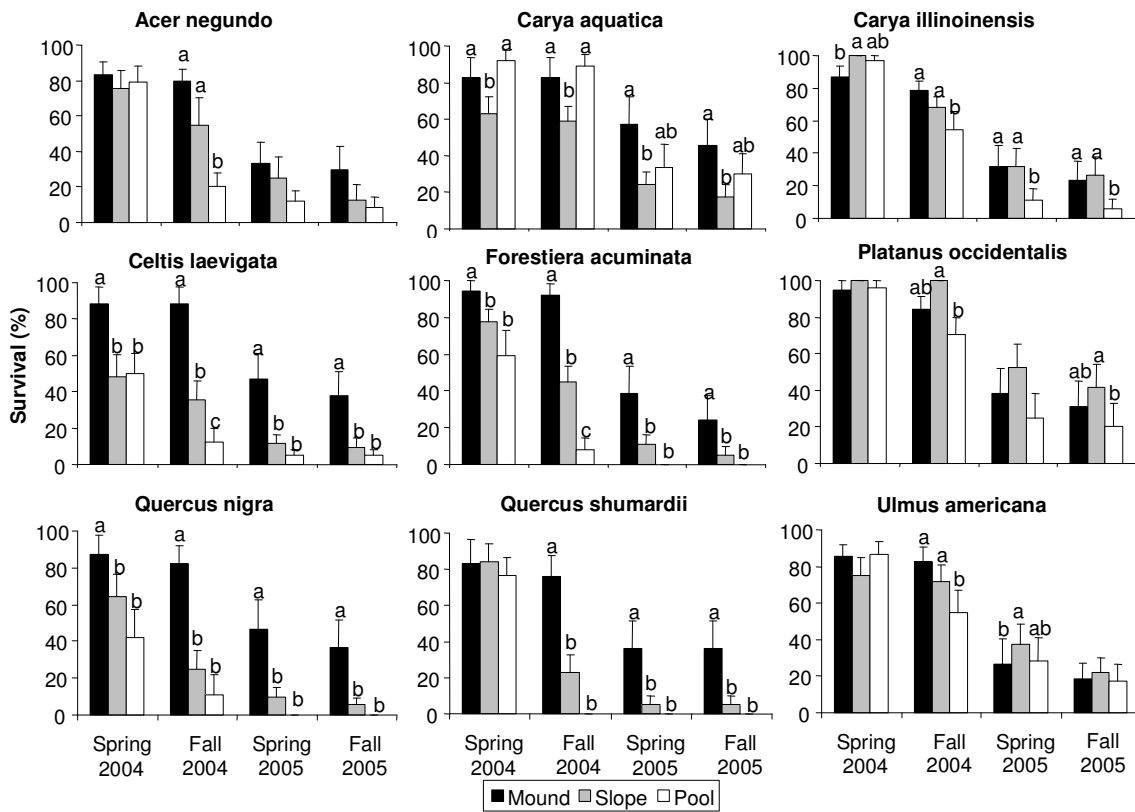


Figure 3.3. Survival of planted mature tree species in mounds, slopes, and pools from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicate significant differences between microtopographic zones at  $P < 0.10$ .

Topographic position affected survival of all species in the mature mix. Most species had significantly higher survival on mound tops than in pool bottoms (Fig. 3.3). Mortality of *F. acuminata*, *Q. nigra*, and *Q. shumardii* in pools reached 100% by Spring 2005, but survival on mounds in Fall 2005 was 24%, 36%, and 36%, respectively for each species. MANOVA also revealed significant interactions between time and topographic position in *C. laevigata* ( $P < 0.10$ ), *F. acuminata* ( $P < 0.01$ ), *U. americana* ( $P < 0.05$ ), and *Q. shumardii* ( $P < 0.01$ ). There were also significant differences between soil treatments for *C. illinoensis*, *C. laevigata*, *F. acuminata*, and *Q. shumardii* (Fig. 3.4). Survival of these species was typically higher in plots without erosion control mats.

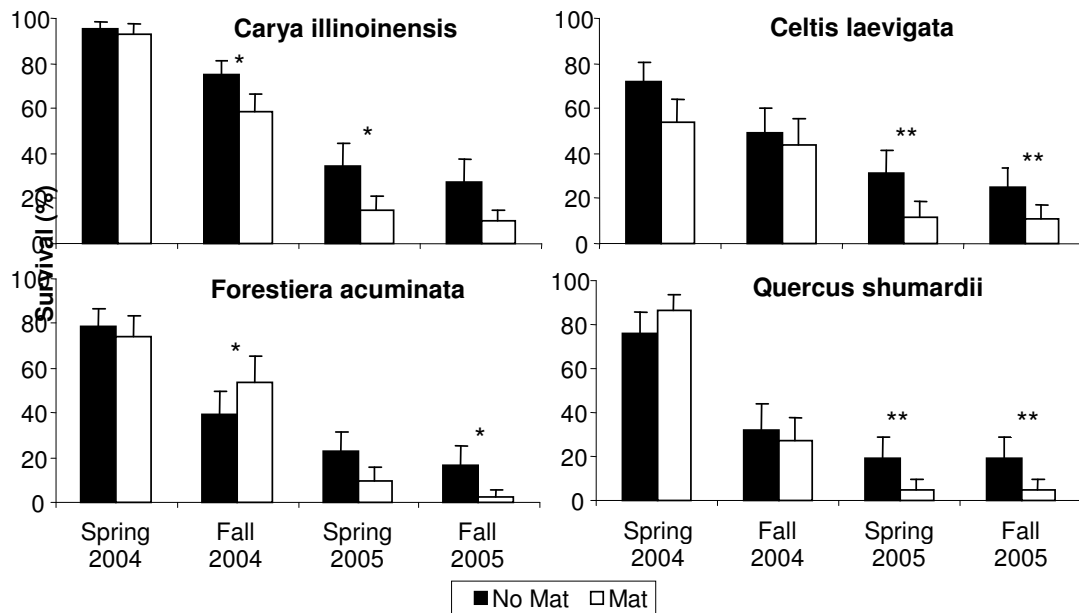


Figure 3.4. Survival of planted seedlings based on soil treatment. Bars on columns represent one SE, and asterisks indicated significant differences between treatments at  $P < 0.10$  (\*) and  $P < 0.05$  (\*\*).

Seedling growth from Spring 2004 to Fall 2005 was highly variable between species and between topographic positions within species. *S. nigra* planted in pools without erosion control mats had the highest relative growth rate increasing by 224%. Growth rates of *C. occidentalis* and *F. pennsylvanica* were highest in pools while *Q. shumardii*, *F. acuminata*, and *P. occidentalis* were highest on mound tops (Figs. 3.5 and 3.6). *P. deltooides*, *C. illinoensis*, and *Q. nigra* had similar growth rates between mounds and slopes, but these species grew slower in pool bottoms. Topographic position had no effect on growth rates of *S. nigra*, *U. americana*, *A. negundo*, *C. laevigata*, and *C. aquatica*. *F. pennsylvanica* grew ~53% more with erosion control mats than without mats ( $P < 0.10$ ), and *C. laevigata* grew ~393% more with mats than without mats ( $P < 0.10$ ). *C. illinoensis*, however, had slower relative growth rates in plots with erosion control mats than without mats (16% and 23%, respectively,  $P < 0.10$ ).

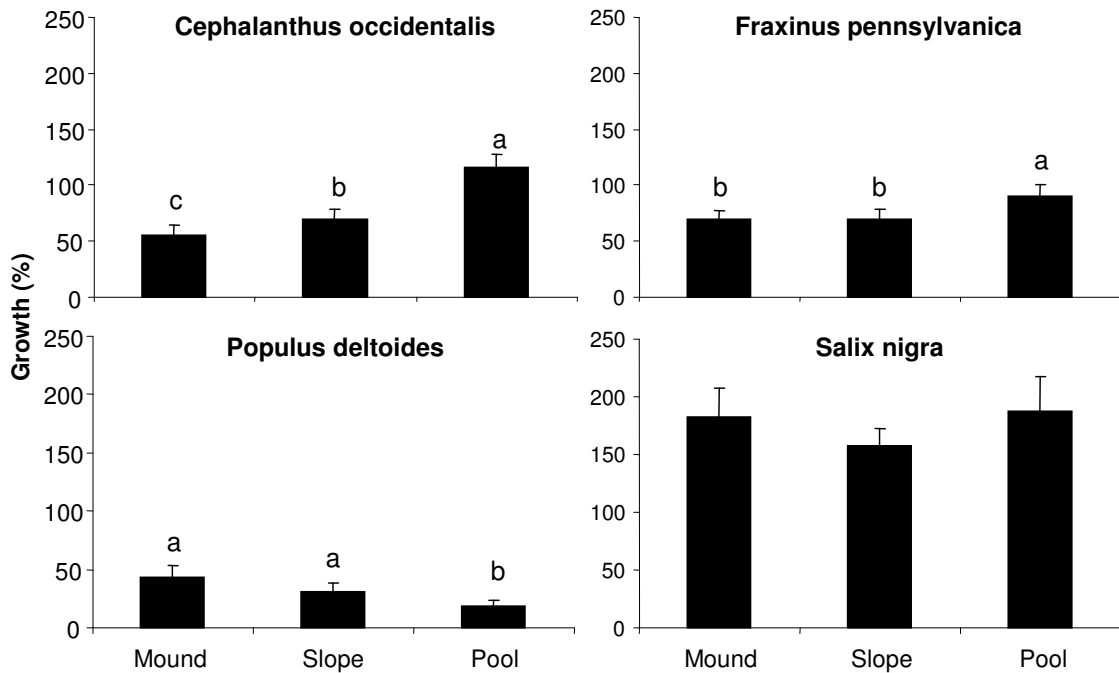


Figure 3.5. Growth of planted pioneer tree species from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicated differences at  $P < 0.10$ .

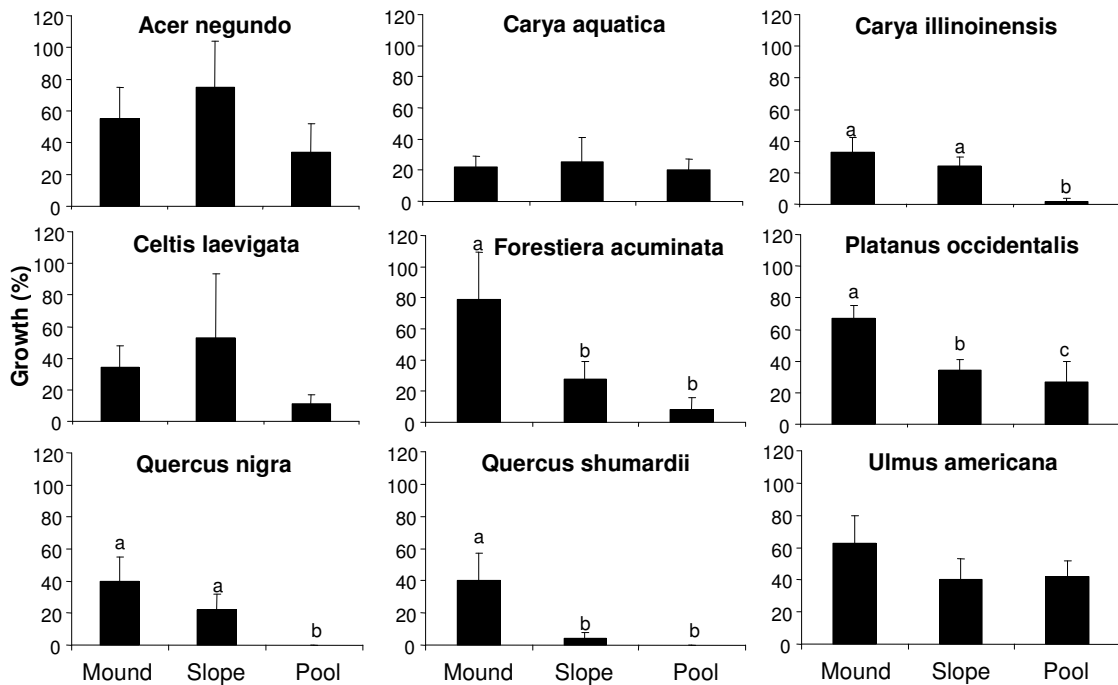


Figure 3.6. Growth of planted mature tree species from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicated differences at  $P < 0.10$ .



### *Colonizing species composition*

In 2004, while sampling colonizing species composition, a total of 40 colonizing herbaceous species were sampled with 33 species encountered on mounds, 28 on transitional slopes, and 22 in pools. Average species richness was lower in pools, with three species, than on mounds and slopes, with 11 and 13 species, respectively ( $P < 0.001$ ). There were no differences in richness between soil treatments. In 2005, 39 colonizing herbaceous species were sampled with 36 encountered on mounds, 31 on slopes, and 24 in pools. Average species richness was again significantly lower in pools (five species) than on mounds and slopes, each with 10 species ( $P < 0.001$ ). There were no differences between soil treatments. Of the 40 species sampled in 2004, only five were dominant (contributed 5% or more to total species composition) in at least one topographic position, whereas nine of the 39 sampled in 2005 were dominant (Table 3.4). The eleven species characterized as dominants in either 2004 or 2005 contributed a high percentage to total species composition at each topographic position. For mound tops, nine species contributed 72% of species composition in 2004, while eight species contributed 88% of species composition in 2005 (Table 3.4). Due to difficulty in distinguishing between species in the genera *Polygonum*, *Sesbania*, *Cyperus*, and *Echinochloa* prior to flowering or seed set, species within each of these genera were lumped together. As a result, actual species richness is likely higher than numbers reported here.

With the exception of *Melilotus officinalis*, an introduced annual and *Leersia oryzoides*, a native perennial, most of the remaining dominant colonizing species were native annuals. However, the genera *Polygonum* and *Cyperus* were composed of both native and introduced, and annual and perennial species. Although identification of *Sesbania* spp. was difficult because of lack of flowers during sampling, they were later identified as *S. vesicaria* and *S. macrocarpa*, both of which are native, annual, nitrogen fixers that are facultative and facultative wetland species, respectively. The annuals *Chamaecrista fasciculata* and *M. officinalis* are also nitrogen-fixing facultative upland species that were most dominant on mound tops (Table 3.4). The two most dominant

wetland obligate species, *L. oryzoides* and *Ammannia coccinea*, first appeared in 2005 and were more dominant in pools and slopes than mounds.

Table 3.4. Percent species composition on mounds, transitional side slopes, and pools in 2004 and 2005. Percent composition in the final row reflects contribution of the species listed in the table to total community composition.

Species	2004			2005		
	Mound	Slope	Pool	Mound	Slope	Pool
<i>Echinochloa</i> spp.	18	51	69	14	19	54
<i>Chamaecrista fasciculata</i>	21	8	1	22	12	1
<i>Iva annua</i>	9	6	1	18	16	2
<i>Polygonum</i> spp.	9	5	0	0	0	0
<i>Panicum dichotomiflorum</i>	5	7	1	0	0	0
<i>Cyperus</i> spp.	2	4	1	14	7	2
<i>Ambrosia trifida</i>	4	1	0	2	0	0
<i>Sesbania</i> spp.	3	0	1	6	6	4
<i>Melilotus officinalis</i>	1	0	0	8	4	0
<i>Leersia oryzoides</i>	0	0	0	0	8	0
<i>Ammannia coccinea</i>	0	0	0	4	14	23
% Composition	72	82	74	88	86	86

*Echinochloa* spp. were the most dominant species overall in both years followed by *C. fasciculata* and *Iva annua*. *Echinochloa* spp. dominated slopes and pools in 2004 and pools in 2005. *C. fasciculata* and *I. annua* were dominant on mounds and slopes during both years. Although *Polygonum* spp. and *Panicum dichotomiflorum* were dominant species on mounds and slopes in 2004, they were absent from all topographic zones in 2005. Alternatively, *Cyperus* spp. and *M. officinalis* were minor components in all zones in 2004, but increased in dominance in 2005. Some species expanded to colonize additional topographic positions in 2005. In 2004, *Sesbania* spp. were primarily restricted to mound tops, but in 2005 their relative composition was similar in each topographic zone.

In 2004, percent species composition was higher in plots without erosion control mats for *P. aviculare* ( $P < 0.05$ ) and *Helianthus annuus* ( $P < 0.10$ ), but percent

composition was higher for *I. annua* in plots with erosion control mats ( $P < 0.10$ ). There were no differences between soil treatments for any species in 2005.

In 2004, total colonizing species cover was highest on side slopes, intermediate on mound tops, and lowest in pool bottoms ( $P < 0.01$ ); however, in 2005, cover was highest on mounds, intermediate on slopes, and lowest in pool bottoms ( $P < 0.01$ ). MANOVA revealed significant effects of time with increasing cover for all topographic zones between years ( $P < 0.01$ ). From 2004 to 2005, cover on mounds increased from 32% to 76%, slopes increased from 40% to 58%, and pools increased from 13% to 41%. There were no differences in cover between soil treatments.

In 2004, three colonizing *Ulmus* individuals, nine *Salix* individuals, and 35 *P. deltoides* were encountered during sampling. Most of these woody species colonized transitional side slopes as all but three *Salix* and one *Ulmus* were located within this zone. Pools remained largely uncolonized by woody species, as only one *Salix* individual was located there. Numbers of woody colonizers were similar in 2005, but the predominant topographic location of colonization shifted to mound tops. All eight *Ulmus*, nine of the 11 *Salix*, and 18 of 33 *P. deltoides* were located on mound tops. The remaining two *Salix* and eight of the remaining 15 *P. deltoides* were located on slopes.

## Discussion

Variations in microtopography have been shown to influence belowground biomass (Jones et al. 1996), litterfall production (Schilling and Lockaby 2005), aboveground productivity (Wassen et al. 2002), and species distribution (Titus 1990). In this study, creating microtopography strongly influenced hydrology, soil properties, planted seedling survival and growth, and colonizing species abundance and distribution. Microtopography created spatial and temporal heterogeneity, which has significant implications for bottomland forest restoration.

*Hydrology and soil properties*

Prolonged water inundation is typical in depressions of floodplain forests. Flood duration in the created pools in this study was similar to flooding duration in natural hummocks in Alabama (Jones et al. 1996). Flooding patterns were similar in 2004 and 2005 in that pools remained inundated during most of the summer months. Flood duration and depth, however, appeared to be greater in 2005 than 2004. Except for complete inundation following the 2 January 2005 precipitation event, mound tops generally extended above the water level. Depth and duration of inundation also differed between plots over time (Table 3.1).

Differences in flood duration between mounds and pools and between plots likely caused variations in anaerobic conditions and soil nutrient concentrations which created a spatially diverse nutrient environment. Flooding dynamics have been shown to create spatial patterns of nutrient availability depending on duration of inundation, direction of water movement, and responses of plants to the hydrologic regime (Fennessy and Mitsch 2001). Nutrient levels have also been shown to vary between topographic zones in bottomland hardwood forests (Jones et al. 1996) as anaerobic conditions present during flooding at lower elevations typically induce a reduction of nitrate and sulfate (Whitmire and Hamilton 2005). In this study, however, inundated pools had higher levels of nitrate and sulfur than mounds. Nutrient loading in pools may have been caused by inundation of nutrient-rich water (Mitsch and Rust 1984). During large precipitation events, such as on 2 January 2005, water inputs into the mounds and pools included direct precipitation, runoff from the closed landfill, and ground water from nearby sloughs. Elevated nutrient levels in urban streams and groundwater are common (Hey 2002), and these water sources may have introduced nutrient-rich sediments that settled in pool bottoms. In contrast, sodium concentrations were higher on mound tops and slopes than pool bottoms. This may be due to concentration of salts on the soil surface as water evaporated from mound tops (Ponnameruma 1984).

Because of a lack of oxygen needed for respiration in flooded soils, decomposition rates are lower in anaerobic conditions than aerobic conditions, resulting

in accumulation of soil organic matter in flooded soils. In this study, levels of soil organic matter did not differ between mounds and pools. These results were similar to those from a study in North Carolina in which the creation of hummocks and hollows did not result in significant organic matter accumulation in hollows after three growing seasons (Bruland and Richardson 2005). In another study, Mitsch et al. (2005) reported a 63% increase in soil organic matter to an average of 8.6% ten years after wetland creation. Because these pedogenic processes are slow, accumulation of organic matter in pools may require more than two growing seasons as was assessed in this study.

Although there were no differences in soil organic matter between topographic zones, litter cover was higher on mounds and side slopes than in pools. This is most likely due to higher cover of colonizing species on mounds and slopes. Surprisingly, there were no differences in litter cover between soil treatments after two growing seasons. This indicates that either the straw-based erosion control mats decomposed within two growing seasons, or litter production in plots without erosion control mats was equivalent to litter cover provided by erosion control mats and vegetation after two growing seasons.

#### *Responses of planted seedlings*

Woody species' growth, survival, and distribution in floodplains are dependent on a number of factors including flooding frequency, duration, and timing, as well as nutrient levels and type of substrate. Continuously flooded conditions result in anaerobic conditions that cause physiologic stress in woody species. This stress results in limited shoot, root, and leaf growth and may ultimately lead to mortality (Kozlowski 1984b). In bottomlands, establishment and distribution of woody species have been shown to occur nonrandomly and correlate strongly with elevation (Titus 1990). Elevation is an indicator of hydroperiod as species adapted to flooded conditions become established in lower elevations and those not suited to flooding become established on higher elevations. In this study, colonization of woody species in pools was extremely limited as only 8% of colonizing woody species were located in pools. These results are

similar to those reported by Titus (1990), who found few woody species located in frequently flooded depressions in a hardwood swamp in Florida.

Many of the planted seedlings also showed reduced growth and survival in pools compared to mounds. As pools were flooded during part of the growing season in 2004 and much of the 2005 growing season, species establishment varied based to some degree on species' adaptation to prolonged flooding conditions. Survival of all species in the mature mix was affected by topography with significantly lower survival in pools. The only exception was *C. aquatica*, which had highest survival on mounds, lowest on slopes, and intermediate survival in pools. High survival in pools is not unexpected as *C. aquatica* is moderately adapted to flooding and has been classified as an obligate wetland species (Reed 1988, Allen et al. 2001).

Compared to species in the mature mix, survival of pioneer species was less dependent on topographic position. Survival of three pioneer species, *C. occidentalis*, *F. pennsylvanica*, and *S. nigra*, did not differ between topographic zones. *C. occidentalis* had higher survival than most species and survived well regardless of topographic zone, soil treatment, or time. Likewise, relative growth rate of this species was highest in pools and decreased on slopes and mound tops. As a wetland obligate, this species has been shown to be well adapted to long periods of flooding (Hook 1984). As the least flood-tolerant pioneer species in this study (Allen et al. 2001), *P. deltoides* was the only pioneer species to exhibit differences in survival between topographic positions with higher survival on mounds than in pools.

Differences in seedling survival not only occurred between topographic position, but variation in survival also occurred over time. MANOVA analysis indicated significant reductions in species survival between Spring 2004 and Fall 2005. Although not explicitly tested, survival of pioneer species appeared to be similar between Spring 2004 and Fall 2004 and again between Spring 2005 and Fall 2005. Highest mortality for most species occurred between Fall 2004 and Spring 2005. Similar patterns were observed in some of the mature species. This temporal heterogeneity in species survival most likely was a result of extended inundation into the growing season of 2005.

Survival from Spring 2005 to Fall 2005 was probably similar because of rapid decreases in water inundation after 7 June 2005 (Table 3.1).

In addition to hydroperiod, substrate type and soil nutrient levels also determine productivity and distribution of woody species (Barko and Smart 1983, Titus 1990). Flooding in bottomlands often introduces nutrient-rich waters and sediments that can be utilized after flood levels subside (Mitsch and Rust 1984). While tree growth has been positively correlated with available nutrients and growing season length, it is negatively correlated with competition and flood duration during the growing season (Mitsch and Rust 1984). While flooding in pools introduced nutrients that might have contributed to plant growth, continuous flooding most likely inhibited germination of colonizing species and created physiological stress in planted species, thereby reducing survival and growth.

Soil treatment had little effect on growth or survival of most seedlings, but some differences were observed. *F. pennsylvanica* and *C. laevigata* grew more with erosion control mats than without, while *P. deltoides* had higher survival and *C. illinoensis* grew more without mats. Likewise, species in the mature mix that exhibited differences in survival between soil treatments generally had higher survival without erosion control mats (Fig. 3.4). While presence of erosion control mats may have effected growth and survival of some species, the differences seen between soil treatments are more likely to be due to hydrologic dynamics. Due to variability in elevation across the borrow pit, plots with erosion control mats that were planted with the mature mix were flooded higher and longer than most other plots (Table 3.1). Decreased survival of *C. laevigata*, *F. acuminata*, and *Q. shumardii* and growth and survival of *C. illinoensis* in plots with erosion control mats are most likely explained by prolonged flooding within these plots. Likewise, the pioneer plots without erosion control mats had longer inundation than pioneer plots without mats. This would also explain why *F. pennsylvanica* and *C. laevigata* also had higher growth rates in plots with erosion control mats.

### *Colonizing species*

Although most dominant colonizing herbaceous species were native annuals that commonly grow in disturbed areas, they represented a diverse community in terms of adaptations to environmental variables. *Echinochloa* spp. were the most dominant species across all topographic positions. These species are pioneers that readily colonize moist to wet soils and are prolific seed producers (Ahn et al. 2004). Several nitrogen-fixing species colonized the mounds and pools, including *Chamaecrista fasciculata*, *Melilotus officinalis*, and *Sesbania* spp., and colonizing species were also diverse with respect to flood tolerance with wetland obligates such as *Ammannia coccinea* and *Leersia oryzoides* and facultative upland species such as *Melilotus officinalis* and *Chamaecrista fasciculata* (Reed 1988).

Several changes in dominant species occurred between years. Although *Polygonum* spp. and *Panicum dichotomiflorum* occupied less than 10% of species composition in 2004, they were completely absent from plots in 2005. Alternatively, *Cyperus* spp., *Iva annua*, *Melilotus officinalis*, *Sesbania* spp., and *Ammannia coccinea* all increased in abundance in 2005. Because these species exhibit wide variations in adaptation to flooding and soil nutrient requirements, it is difficult to speculate why changes in species composition occurred. Increases in species abundance were likely simply due to expansion of range following initial establishment in 2004.

### *Implications for restoration*

Bottomland hardwood forests are diverse systems, and much of the diversity in soil attributes, litter cover, and species distribution has been correlated with relative elevation and microtopography (Buchholz 1981, Vivian-Smith 1997, Grell et al. 2005). Despite the known influence of microtopography on many features in bottomlands, efforts have only recently begun to incorporate microtopography within restoration designs (Barry et al. 1996, Tweedy and Evans 2001, Patterson and Adams 2003, Bruland and Richardson 2005). Although studies involving restored microtopography are new, results are promising and include improved survival and growth of planted species



(Patterson and Adams 2003), a more tempered hydrology (Tweedy and Evans 2001), and more diverse soil nutrient distributions (Bruland and Richardson 2005).

Creating mounds and pools in this study resulted in a spatially and temporally heterogeneous system that appears to reflect variations in natural bottomlands. Compared to mounds, pools had longer hydroperiods, lower colonizing species richness and cover, higher soil nutrient concentrations, and mixed responses of planted species survival and growth. Transitional slopes often exhibited characteristics of both mound tops and pool bottoms, or responses were intermediate between the two.

By recreating microtopography in bottomland restoration, natural structures are mimicked that might take decades to form naturally. There are several benefits for including microtopography in restoration designs. It has been suggested that doing so could initiate wetland functions that would otherwise develop only after microtopography developed on its own (Bruland and Richardson 2005). Also, if the hydrologic regime is altered in the future, hydrologic conditions may still be suitable to support a diverse community given a high degree of topographic relief. In this study, year-to-year variation in flooding will continue to influence seedling growth and survival, but unless mound tops are permanently flooded, elevation differences should allow a variety of species to persist. Finally, microtopography provides diverse microsite characteristics which may improve chances for the establishment of a diverse community (Titus 1990).

Bottomland restoration often requires the introduction or reintroduction of woody species. Many early restoration and afforestation efforts were limited to the introduction of more desirable mast-producing species that would provide valuable resources for wildlife (Schoenholtz et al. 2001). While *Quercus* spp. are still the most prevalent species used, several other species are beginning to be introduced as part of bottomland restoration (King and Keeland 1999). Although developing a quick canopy cover of desirable species is often the goal, it is imperative that realistic goals are set for a particular site and that species selections match site characteristics (Zedler 2000).

The introduction of several species in this study appears to be important for the continued development of a diverse community. The majority of trees chosen in the mature mix represented those that would be considered desirable for wildlife habitat and aesthetics. However, many of these species exhibited poor survival by the end of the study or reached 100% mortality in certain topographic zones. The pioneer species, while less desirable, had higher survival and were less influenced by topographic position. These results illustrate the need to introduce several types of species to ensure successful seedling establishment when hydrology is highly variable, unknown, or unpredictable. In addition, some pioneer species that have high initial survival and growth rates may be used as nurse crops to ameliorate harsh hydrologic and soil conditions to better aid in the establishment of latter successional species (Dulohery et al. 2000, McLeod et al. 2001).

## **Conclusions**

Microtopography has been shown to be an important component of bottomland hardwood forests as it influences hydrologic regimes, nutrient dynamics, and species distribution. Despite its ecological significance, creation of microtopography is rarely included in bottomland restoration efforts. The objective of this study was to determine the responses of planted and colonizing species to created microtopography and its influence on soil physiochemical development within a mound-and-pool complex. Microtopography created spatial heterogeneity in terms of hydrology, soil nutrient concentrations, planted seedling growth and survival, and colonizing species distribution. Although nutrient levels were generally higher in pool bottoms, differences in hydroperiod between pool bottoms and mound tops during the summer months likely had the greatest influence on ecosystem development. Planted seedling survival and growth and herbaceous colonizing species distribution differed across topographic zones and depended on flood tolerance of each species.

The results of this study provide support for the recommendation to include microtopography in bottomland forest restoration. Providing microtopographic relief

may have several benefits. The spatial heterogeneity that resulted from the creation of mounds and pools appeared to be similar to that in natural bottomlands. In addition, creating topographic heterogeneity may improve survival of a variety of species introduced during restoration, as well as enhance colonization of a diverse plant community under changing hydrologic regimes. In contrast, erosion control mats had no effect on colonizing species richness and little effect on percent composition of individual species. Erosion control mats also did not improve growth or survival of most planted species, and, therefore, their use does not appear to be a beneficial restoration strategy in the mound and pool complex of this study.

**CHAPTER IV**

**EFFECTS OF LARGE-SCALE TOPOGRAPHIC HETEROGENEITY,  
PLANTING MIXTURES, AND SOIL TREATMENTS ON BOTTOMLAND  
HARDWOOD FOREST RESTORATION**

**Introduction**

The natural development of topographic heterogeneity within a bottomland forest can be a complex process that requires long time periods and differs at various spatial scales. As stream hydrology changes through time, stream channels move laterally through a floodplain resulting in a number of topographic features such as oxbow lakes, meander scrolls, flats, terraces, and levees (Hodges 1997, Mitsch and Gosselink 2000). This large-scale topographic heterogeneity creates hydrologic gradients based on current stream flooding dynamics. Lower elevations have the potential to be inundated for long periods of time, while higher ridges may be rarely inundated (Allen et al. 2001). In addition, during overbank flow water velocity decreases and sedimentation occurs throughout the floodplain. Coarse sediments are typically deposited on levees and ridges near stream channels, and silts and clays fill backwaters and low-lying areas further from the channel. Although stream channel hydrology creates topographic heterogeneity at the scale of floodplains, smaller-scale microtopography is created by tree fall, through animal manipulation, and local erosion and sedimentation processes (Ehrenfeld 1995). These variations in topography interact with flooding dynamics to create variations in biogeochemical processes and vegetation community composition (Darke and Walbridge 2000, Xiong et al. 2003).

Despite its importance in the development of bottomland structure and function, the creation of large-scale topography is rarely part of restoration efforts. Recent bottomland restoration efforts have largely focused on hydrologic restoration (Kolka et al. 2000, Middleton 2002). Since hydrology is the most important factor determining wetland structure and function (Mitsch and Gosselink 2000), this focus is justified.

However, in bottomland forests, the hydrologic regime and relative surface elevation due to topographic variations are inextricably connected. By restoring topographic heterogeneity, hydrologic conditions might also be reestablished (Bruland and Richardson 2005).

In addition to hydrology, most bottomland restoration and afforestation efforts involve the reintroduction of woody species. Historically, these introductions have been limited to relatively few species, most of which are mast producing and slow growing such as *Quercus* spp. and *Carya* spp. (Twedt 2006). The reason behind this is that emphasis has been placed on establishing desired species with less concern for or knowledge about interactions between species, soil, and hydrology (Patterson and Adams 2003). As a result, many plantings fail or forest stands develop limited vertical structure (Twedt 2006). Alternatively, planting a mix of faster and slower growing species has been successful in establishing a complex forest structure and increasing species diversity (Stanturf et al. 2000, Twedt 2006). However, careful consideration is needed to identify species adaptations to hydrologic and edaphic conditions of the particular site in order to improve species survival and community development.

In drastically disturbed sites it may be necessary to address edaphic limitations to species colonization and survival. Improving soil conditions through the use of organic matter or fertilizer, and increasing the retention of nutrients through proper species introductions brings about autogenic repair processes that promote continued self-repair (Whisenant 1999). For example, pioneer species are generally the first to colonize a site following disturbance. They typically have r-selected life history strategies that are short lived, fast growing species that are adapted to variable climatic conditions (Pianka 1970). Some pioneer species have been shown to ameliorate site conditions to promote the establishment of later successional species (McNamara et al. 2006). These facilitory processes come about as pioneer species improve soil development through the trapping of sediments, decreasing soil bulk density through root expansion, and increasing organic matter content (Wali 1999, Whisenant 1999). Initial plantings of pioneer species in bottomland restoration may be advisable over planting more desirable species if the

pioneer species can achieve a quick canopy cover and regulate hydrology to improve establishment of later successional species (Stanturf et al. 2001).

The objective of this study was to assess biotic and abiotic responses to topographic heterogeneity in combination with various planting mixtures and soil treatments as part of bottomland forest restoration. Specific research questions included: Are pioneer or later successional species more adapted to current highly disturbed site conditions? Can the addition of mulch enhance site conditions to improve seedling survival and growth? Will topographic heterogeneity improve planted seedling survival in a widely varying hydrologic regime? Will created topographic heterogeneity create gradients producing distinct species assemblages within two growing seasons?

## **Methods**

### *Study site*

A 30-ha borrow pit at the Castle Drive Landfill, located in Garland, Dallas County, Texas (32° 93' N 96° 58' W), provided an opportunity to evaluate topographic features, and hydrologic and vegetation dynamics of bottomland hardwood forest restoration. The borrow pit is adjacent to Rowlett Creek. Flooding from the creek periodically flooded the borrow pit until 2003, when landfill managers reinforced a levee separating the creek from the pit to enable construction of the site. Currently, water inputs consist of precipitation, ground water in sloughs, and runoff from the landfill during large precipitation events. The long-term average annual precipitation (30-year) is 999 mm, which occurs in a bimodal distribution with peaks in May (134 mm) and October (116 mm).

Open since the 1970s, the landfill was capped in 2002. During the 30 years in operation, soil was removed from the borrow pit to be used in the landfill. By 2002, the top 5 m of soil had been excavated from the borrow pit. In 2004, part of the borrow pit was contoured to include a mound-and-pool complex following the design by Barry et al. (1996) and a series of ridges. The mound and pool microtopography was constructed to replicate smaller-scale features such as tip up mounds and hummocks and hollows,

while the ridges were constructed to resemble meander scrolls. A bulldozer with a 3-m wide blade was used to create a checker-board series of mounds and pools so that mound tops were approximately 60 to 100 cm higher in elevation than pool bottoms. Mounds were approximately 30 to 50 cm above and pools 30 to 50 cm below surrounding level areas. Ridges were constructed to be approximately 1.8 m in height above the surrounding flats with average lengths of 100 m (Fig. 4.1). Sides were contoured to have 3:1 to 4:1 slopes (33% and 25%, respectively). A series of sloughs were also created adjacent to the ridges. Sloughs were approximately 1.5–2 m deep and 75 m long. An oxbow lake was constructed along the northern and eastern perimeter of the borrow pit.

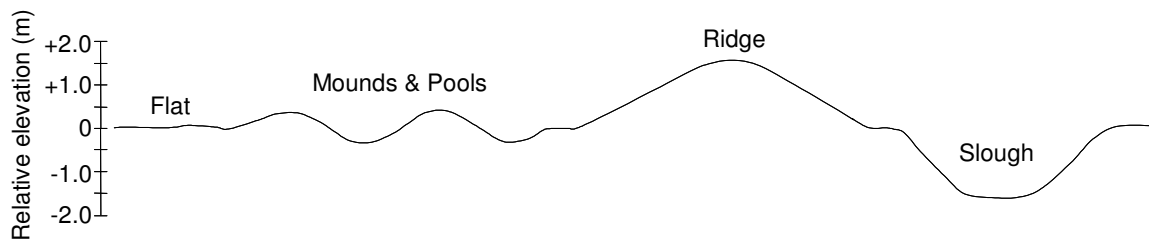


Figure 4.1. Cross section view of flats, mounds and pools, ridges, and sloughs with relative elevations.

#### *Treatment interactions*

A factorial design was implemented to evaluate the interactions between topography (three levels), tree seedling planting mixes (three levels), and soil treatments (two levels). Each treatment combination was replicated five times totaling 90 plots.

The three levels of topography included ridges, mounds and pools, and flats and were conducted at the plot scale. The three levels of planting mixes included a pioneer mix, a mature mix, and no planting. All trees were bare-root seedlings and were planted in February 2004. Holes for the seedlings were drilled with hand augers to a depth of 30–40 cm. A 10-g fertilizer tablet (20-10-5) was placed in each hole along with the

seedling. Holes were then backfilled with sand. Species selected for planting were native to the Dallas region, adapted to clay soils, and ranged from weakly to highly tolerant of flooding during the growing season. The pioneer mix was composed of early successional, fast growing tree species including *Cephalanthus occidentalis* L. (buttonbush), *Fraxinus pennsylvanica* Marsh. (green ash), *Populus deltoides* Bart. ex Marsh. (eastern cottonwood), and *Salix nigra* Marsh. (black willow). The mature mix was composed of species representative of mature forested wetlands in the region including *Acer negundo* L. (box elder), *Carya aquatica* (Michx. f.) Nutt. (water hickory), *C. illinoensis* (Wang.) K. Koch (pecan), *Celtis laevigata* Willd. (sugarberry), *Forestiera acuminata* (Michx.) Poir. (swamp privet), *Platanus occidentalis* L. (sycamore), *Quercus nigra* L. (water oak), *Quercus shumardii* Buckl. (shumard oak), and *Ulmus americana* L. (American elm). Each plot measured 12-m by 12-m, and trees were spaced 1.5 m apart making nine rows of plants per plot. One additional row was planted around each plot to serve as a buffer. Species were randomly located within each row of each plot.

The original design called for shredded wood mulch to be incorporated into the soil on half of the plots. However, prior to the time of planting, the soil was too wet to use machinery to incorporate the mulch into the soil. Instead of wood mulch, soil treatments included straw-based erosion control mats and no erosion control mats. The erosion control mats were purchased from North American Green and used as a substitute for mulch to provide a source of organic matter. Specifications for the mats included a thickness of 8.38 mm, mass of 218 g/m<sup>2</sup>, and light penetration of 11%. The mats were designed to have a functional longevity of up to 12 months. The straw fibers were enclosed within a lightweight photodegradable polypropylene net with a mesh size of 1.27 cm by 1.27 cm. The erosion control mats were applied to half the plots.

### *Vegetation sampling*

Effects of topography, planting mixtures, and soil treatments on system development were determined from the factorial experiment in 2004 and 2005. Planted



tree seedling survival and diameter of stems at the soil surface were assessed in Spring and early Fall of 2004 and 2005. Because of high water levels early in 2005, seedling sampling in Spring 2005 was performed in June after water levels receded. Herbaceous species composition was analyzed from four randomly located 1-m<sup>2</sup> quadrats. Herbaceous sampling occurred in late August and early September of each year during the time of maximum production. Numbers of colonizing woody species located within each quadrat were noted, and in 2005, percent litter cover within quadrats was estimated.

*Community self-organization/species sorting along topographic/hydrologic gradient*

To analyze the effect of the topographic and associated hydrologic gradient on ridges, two 1-m wide belt transects were placed parallel with the topographic gradient for each planting regime and soil treatment assemblage. Herbaceous species composition was assessed in 1-m<sup>2</sup> quadrats at the highest, middle, and lowest elevation of each transect, corresponding to the top, middle, and bottom of ridges.

*Soil sampling and analysis*

Soil samples were collected in October 2005. A soil corer with a diameter of 2.5-cm was used to collect soil to a depth of 30 cm from each plot. Samples were collected within the same 1-m<sup>2</sup> quadrats used for herbaceous sampling. On ridges and flats, two additional cores were collected from randomly selected locations within each plot totaling six samples per plot. On mounds and pools, three samples were collected from randomly selected pool bottoms, and three samples were collected from randomly selected mound tops. A composite was made from the six samples for analysis. Soil texture was determined via the hydrometer method (Gavlak et al. 2003), and plant-available potassium, calcium, magnesium, sodium, sulfur, and phosphorous were analyzed in the Mehlich 3 extract (Mehlich 1984). A 1 N KCl solution was used to extract nitrate-nitrogen (Keeney and Nelson 1982), and dry combustion was used to determine total nitrogen (Bremner 1996). Loss on ignition was used to determine soil organic matter by heating soil samples for two hours at 150 C to remove hydroscopic

water, after which the temperature was raised to 360 C for an additional two hours (Gavlak et al. 2003). A 1:2 soil:water mixture was used to determine soil pH with a hydrogen selective electrode and electrical conductivity with a conductivity probe (Schofield and Taylor 1955, Rhoades 1982).

### *Data analysis*

All analyses were performed based on an incomplete block design. A mixed-model approach was used in which blocks and blocks nested within replicates were random variables (Federer and Wolfinger 1998). In this manner, interblock information was retained. Multivariate analysis of variance (MANOVA) was used to determine effects of time on tree seedling survival. Mixed model analyses were then used to determine differences in survival between topographic treatment, soil treatment, and the interaction between topographic and soil treatments for each time period. Seedling growth was determined for all seedlings that survived until the final measurement in Fall 2005 ( $(\text{Diameter}_{\text{Fall2005}} - \text{Diameter}_{\text{Spring2004}}) / \text{Diameter}_{\text{Spring2004}} \times 100$ ). Mixed model analysis was also performed to determine differences in species composition between topography, soil treatments, and the interaction between topography and soil treatments. MANOVA was used to determine changes in composition of species between years.

## **Results**

### *Hydrology*

When tree seedlings were planted in February 2004, flats and ridges were free of standing water, but many of the pools were inundated up to 50%. Most pools were free of standing water by June 2004, and all treatments remained in that condition until January 2005. On 2-3 January 2005, a large precipitation event resulted in complete inundation of the mounds and pools, and flats were submerged to approximately 60 cm. Because of extended water inundation during 2005, water levels were monitored between 21 April and 9 September. Pools continued to be inundated through much of the summer months, while surface water on most flats had dissipated by mid June.

### Soil properties

Soil properties varied greatly between topographic treatments (Table 4.1). Soil texture was sandier on ridges with less clay than flats or mounds and pools. Ridges had lowest sulfur, sodium, and conductivity, while nitrate was highest on mounds and pools, intermediate on ridges, and lowest on flats. Total nitrogen and calcium was also lowest on mounds and pools. Organic matter was the same on ridges and mounds and pools but was significantly lower on flats. There were no differences in soil properties between soil treatments, but there were significant interactions between topography and soil treatments in potassium ( $F = 3.43$ ,  $P < 0.05$ ), sodium ( $F = 2.81$ ,  $P < 0.10$ ), and calcium ( $F = 2.51$ ,  $P < 0.10$ ).

Table 4.1. Soil properties on ridges, mounds and pools, and flats. Values are means  $\pm$  one SE.  $P$  values in bold indicate probabilities considered to represent significant differences.

	Ridge	MP*	Flat	$P$
pH	8.31 $\pm$ 0.03 a	8.19 $\pm$ 0.03 b	8.21 $\pm$ 0.03 b	<b>0.029</b>
Nitrate-N (ppm)	2.3 $\pm$ 0.18 b	4.0 $\pm$ 0.26 a	1.5 $\pm$ 0.13 c	<b>&lt;0.001</b>
Total N (%)	0.046 $\pm$ 0.001 a	0.041 $\pm$ 0.001 b	0.046 $\pm$ 0.001 a	<b>0.003</b>
P (ppm)	4.3 $\pm$ 0.79	4.9 $\pm$ 0.45	4.6 $\pm$ 0.55	0.863
K (ppm)	188 $\pm$ 9.7	193 $\pm$ 6	204 $\pm$ 8	0.303
S (ppm)	54 $\pm$ 5 b	106 $\pm$ 11 a	128 $\pm$ 24 a	<b>0.007</b>
Na (ppm)	419 $\pm$ 49 c	493 $\pm$ 26 b	604 $\pm$ 54 a	<b>0.001</b>
Ca (ppm)	41,195 $\pm$ 1906 a	36,750 $\pm$ 1352 b	38,164 $\pm$ 1950 ab	<b>0.054</b>
Mg (ppm)	289 $\pm$ 14 b	308 $\pm$ 7 b	332 $\pm$ 15 a	<b>0.011</b>
OM	1.37 $\pm$ 0.06 a	1.46 $\pm$ 0.04 a	1.10 $\pm$ 0.04 b	<b>&lt;0.001</b>
EC (umho/cm)	364 $\pm$ 24 b	583 $\pm$ 26 a	596 $\pm$ 57 a	<b>&lt;0.001</b>
Sand (%)	24.4 $\pm$ 1.8 a	18.0 $\pm$ 1.1 b	18.9 $\pm$ 1.0 b	<b>0.002</b>
Silt (%)	31.7 $\pm$ 1.0	32.9 $\pm$ 0.9	30.8 $\pm$ 0.6	0.247
Clay (%)	43.9 $\pm$ 1.2 b	49.0 $\pm$ 1.1 a	50.3 $\pm$ 0.9 a	<b>&lt;0.001</b>

\*Mounds and pools are abbreviated as MP.

In 2005, there was significantly more litter on ridges and flats than on mounds and pools (53%, 52% and 36% for ridges, flats, and mounds and pools, respectively,  $P < 0.001$ ). Litter cover was also significantly higher in plots with erosion control mats than

without mats on ridges ( $P < 0.001$ ) and flats ( $P < 0.10$ ), but there were no differences in litter between soil treatments on mounds and pools ( $P = 0.52$ ).

#### *Seedling survival and growth*

Initial planted seedling survival was high for most species regardless of topographic treatment (Figs. 4.2 and 4.3). However, survival of all species declined significantly over time as revealed by MANOVA ( $P < 0.01$ ). Of the pioneer species, only *F. pennsylvanica* showed significant effects of time by topography (MANOVA,  $P < 0.05$ ) and time by soil treatment ( $P < 0.05$ ), indicating that survival over the four sampling dates differed between topography and soil treatments. All of the mature species showed a significant effect of time by topography (MANOVA,  $P < 0.05$ ) except *A. negundo*, *C. aquatica*, and *C. laevigata*.

Initial and final survival of pioneer species was not effected by topographic treatments (Fig. 4.2). *C. occidentalis* had highest final survival in Fall 2005 of over 70% for most topographic treatments. *F. pennsylvanica* also had high final survival with a low of 53% on mounds and pools to a high of 77% on ridges. *P. deltoides* and *S. nigra* had lowest final survival of the pioneer species with less than 43% survival for all topographic treatments.

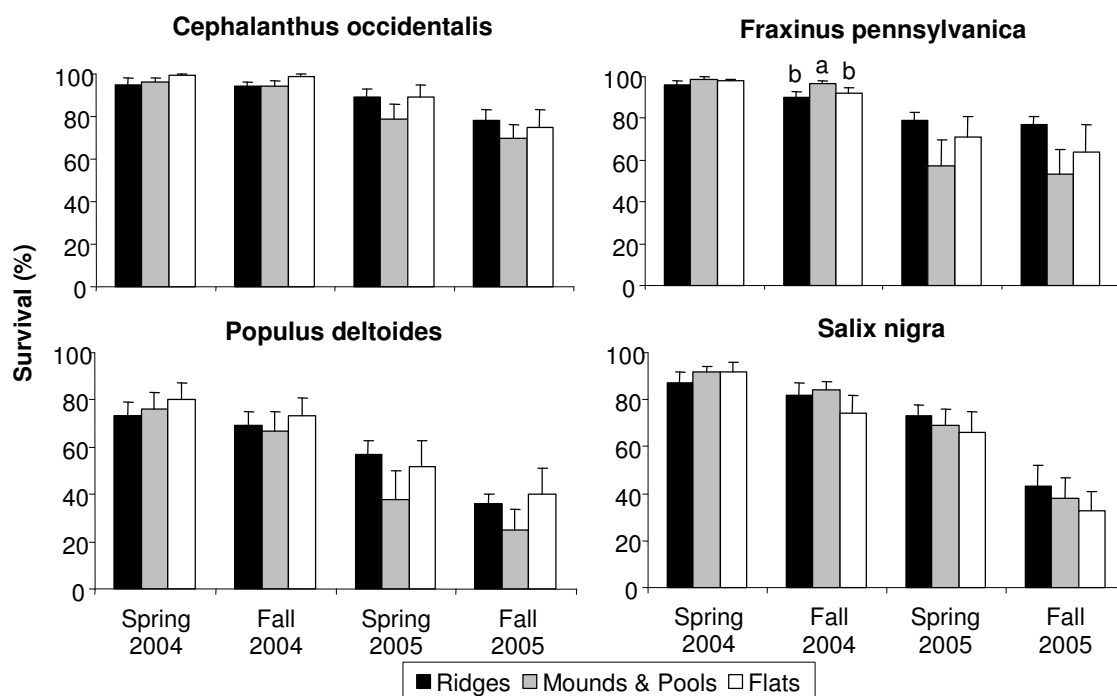


Figure 4.2. Survival of planted pioneer tree species on ridges, mounds and pools, and flats from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicate significant differences between topographic zones at  $P < 0.10$ .

Topographic treatments had significant effects on survival of all mature species during most sampling periods (Fig. 4.3). Survival in Fall 2005 was consistently higher on ridges than on mounds and pools, and survival of all species except *C. illinoensis* and *C. laevigata* was higher on ridges than on flats. Final survival was the same between flats and mounds and pools for *C. aquatica*, *C. illinoensis*, *F. acuminata*, *P. occidentalis*, *Q. nigra*, and *U. americana*. *F. acuminata* and *Q. shumardii* showed strong differences between topographic treatments with final survival of nearly 80% on ridges and less than 15% survival on mounds and pools. Other species such as *C. illinoensis* and *C. aquatica* showed less distinction between topographic treatments. There were also differences between soil treatments for *F. acuminata* in Fall 2004 ( $P < 0.05$ ) and *P. occidentalis* in Fall 2005 ( $P < 0.10$ ). Survival for both species was significantly higher in plots with erosion control mats than without mats.

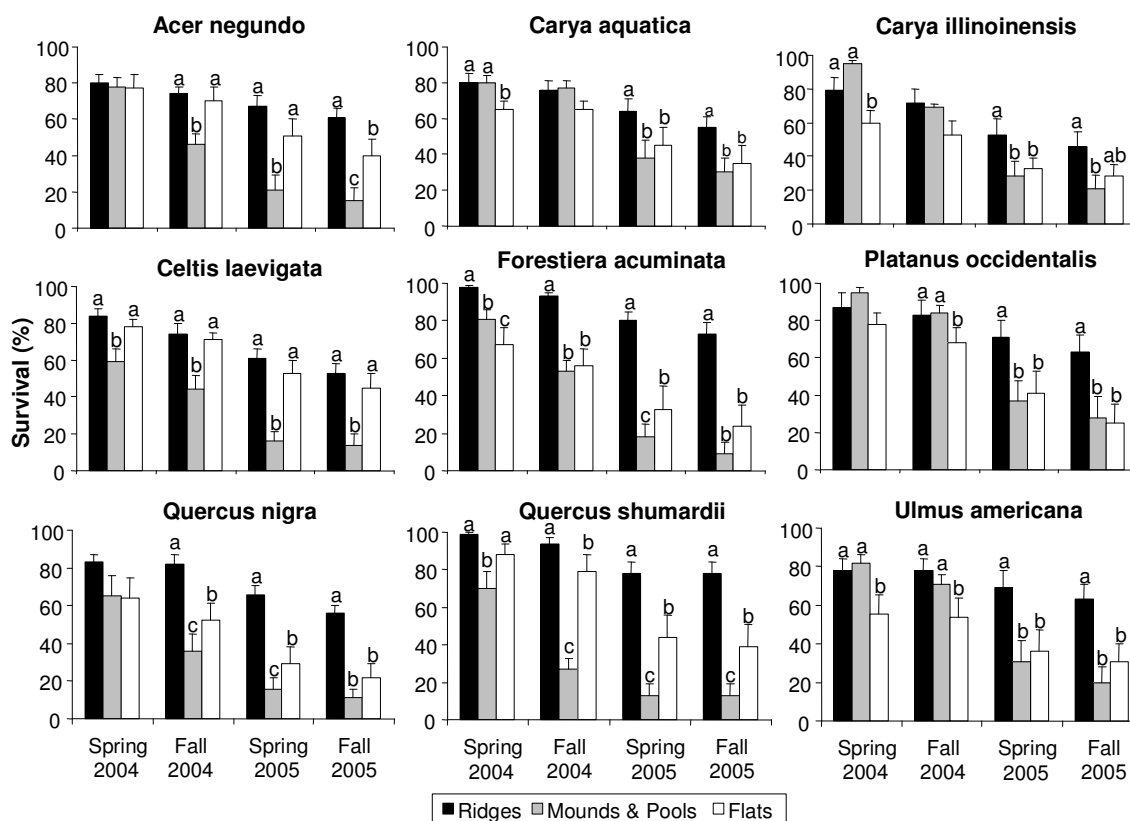


Figure 4.3. Survival of planted mature tree species on ridges, mounds and pools, and flats from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicate significant differences between topographic zones at  $P < 0.10$ .

Seedling growth from Spring 2004 to Fall 2005 was highly variable between species and between topographic positions within species (Figs. 4.4 and 4.5). The pioneer species *S. nigra* and *C. occidentalis* exhibited higher growth on ridges and mounds and pools than on flats, while *F. pennsylvanica* had highest growth on mounds and pools, followed by ridges, and then flats (Fig. 4.4). *S. nigra* had the highest growth rate with over 150% on ridges and mounds and pools. Species in the mature mix, *A. negundo*, *F. acuminata*, and *P. occidentalis*, grew over 100% on ridges, but most other species had far lower growth rates (Fig. 4.5). Topographic treatments had no effect on growth of *P. deltoides*, *C. aquatica*, *C. illinoensis*, and *Q. nigra*.

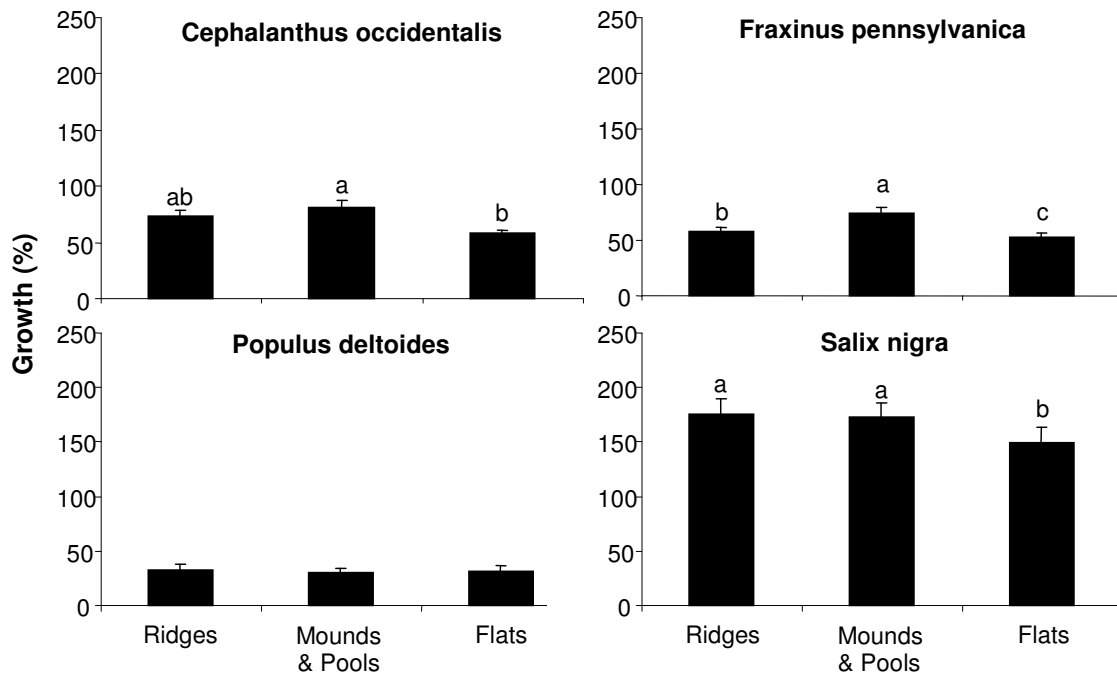


Figure 4.4. Growth of planted pioneer tree species on ridges, mounds and pools, and flats from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicated differences at  $P < 0.10$ .

In addition to topography, soil treatments effected growth of *F. pennsylvanica*, *C. occidentalis*, and *C. laevigata*. These species had higher growth rates with erosion control mats than without mats ( $P < 0.05$ ). There were also significant interactions between topography and soil treatments for *S. nigra* and *C. laevigata*. *S. nigra* on flats had higher growth rates without erosion control mats than with mats, but on ridges growth was higher with mats ( $P < 0.01$ ). *C. laevigata* had higher growth rates with mats on ridges and mounds and pools, but there was no difference between soil treatments on flats ( $P < 0.10$ ).

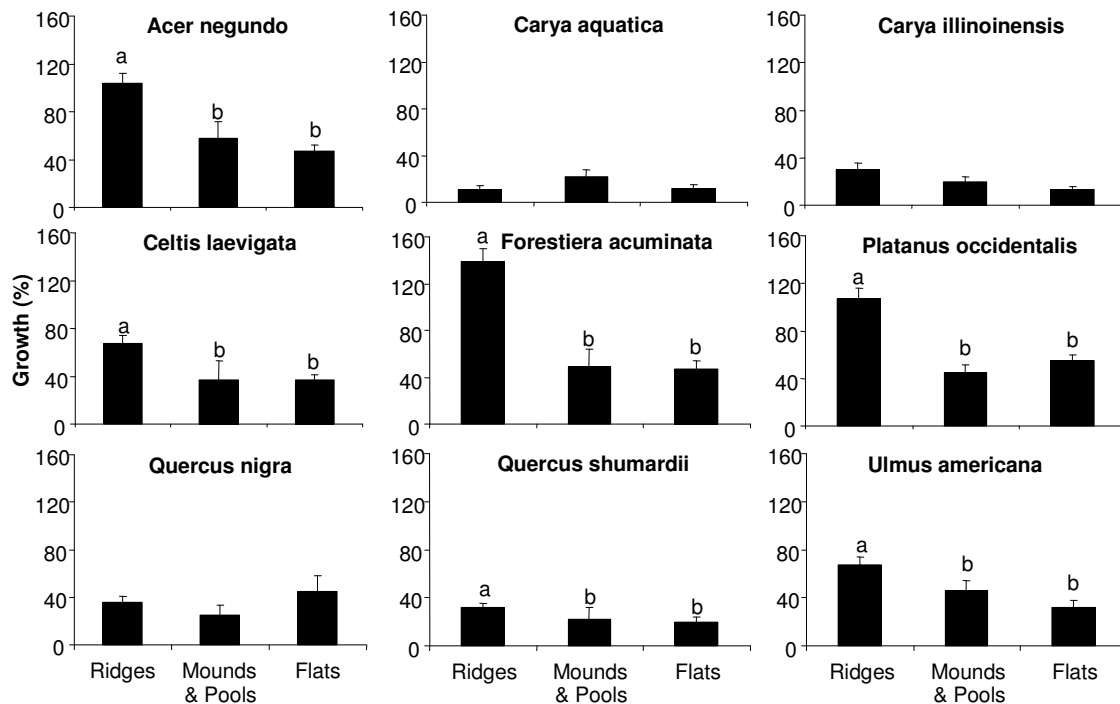


Figure 4.5. Growth of planted mature tree species on ridges, mounds and pools, and flats from Spring 2004 to Fall 2005. Bars on columns represent one SE, and different letters indicated differences at  $P < 0.10$ .

#### *Colonizing species composition*

Total numbers of species varied slightly between topography treatments and between years with a low of 26 species encountered on ridges in 2004 to a high of 35 species on ridges in 2005 (Table 4.2). The ten most dominant species (contributed 5% or more to total species composition) together comprised over 80% of the total species composition (Table 4.2). In 2004, ridges had lowest average species richness with 9.5 species, and mounds and pools had highest richness in 2005 with 11.4 species (Fig. 4.6). Also, species richness declined on flats from 2004 to 2005 (MANOVA,  $P < 0.01$ ). There were no significant differences in species richness between soil treatments, and there were no significant interactions between topography and soil treatments.

Six or fewer species were dominant in each topographic treatment (Table 4.2). *Echinochloa* spp., *Chamaecrista fasciculata*, and *Iva annua* were among the most



dominant colonizing herbaceous species in both 2004 and 2005. In 2004, the most dominant species on ridges were *C. fasciculata*, *Polygonum* spp., *Echinochloa* spp., *I. annua*, and *Ambrosia trifida*. In 2005, dominant species on ridges also included *Melilotus officinalis* and *Sesbania* spp, while *Polygonum* spp. and *Echinochloa* spp. declined. Mounds and pools were dominated by *Echinochloa* spp. during both years, and *Sesbania* spp., *Ammannia coccinea*, and *Cyperus* spp. all increased in dominance from 2004 to 2005. Flats were also dominated by *Echinochloa* spp., *C. fasciculata*, and *I. annua*.

Table 4.2. Percent species composition on ridges, mounds and pools (M & P), and flats in 2004 and 2005. Percent composition in the final row reflects contribution of the species listed in the table to total community composition.

Species	2004			2005		
	Ridges	M & P	Flats	Ridges	M & P	Flats
<i>Echinochloa</i> spp.	12	44	24	0.6	30	20
<i>Chamaecrista fasciculata</i>	27	12	21	27	12	21
<i>Iva annua</i>	10	6	14	21	12	23
<i>Polygonum</i> spp.	19	7	7	0.1	0.2	0.4
<i>Polygonum aviculare</i>	5	4	8	0.1	0.4	0.1
<i>Ambrosia trifida</i>	8	2	5	9	0.1	0.2
<i>Melilotus officinalis</i>	3	3	4	20	3	2
<i>Sesbania</i> spp.	0.02	2	0.3	7	5	7
<i>Ammannia coccinea</i>	0	0	0	0.3	12	10
<i>Cyperus</i> spp.	1	4	0.4	3	8	2
% Composition	85	84	84	88	83	86
Total species richness	26	34	29	35	29	32

Soil treatments resulted in differential species colonization. In 2004, percent composition of *M. officinalis* and *Cyperus* spp. was higher on flats with erosion control mats than without mats ( $P < 0.10$ , 4.9% and 2.2%, respectively for *M. officinalis*, and 0.7% and 0.1%, respectively for *Cyperus* spp.). In contrast, percent composition of *P. ramosissimum* in mounds and pools was higher in plots without erosion control mats ( $P < 0.05$ , 1.8% and 0.7%, respectively). In 2005, percent composition of *M. officinalis* was higher on ridges without mats than with mats ( $P < 0.01$ , 26% and 14%,

respectively). Percent composition of *I. annua* was higher on flats with mats than without mats ( $P < 0.01$ , 31% and 15%, respectively), and *Typha latifolia* was higher on mounds and pools with mats than without mats ( $P < 0.10$ , 4.5% and 2.6%, respectively).

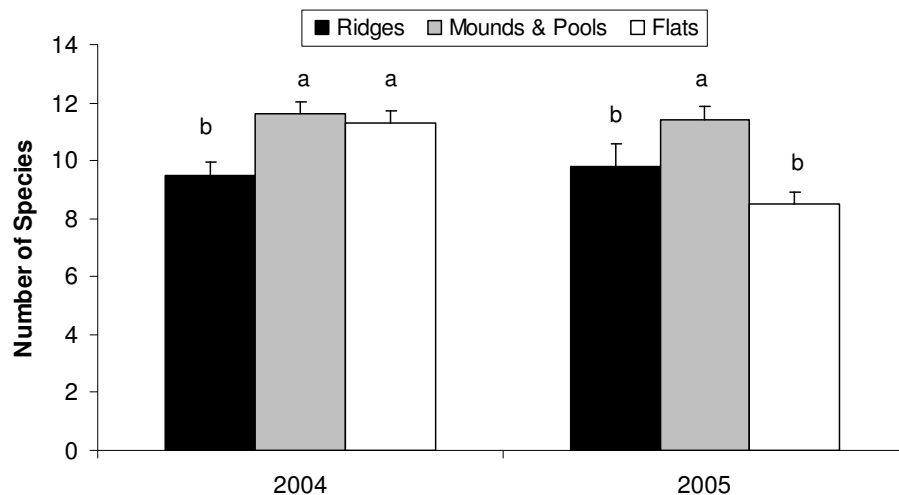


Figure 4.6. Average species richness on ridges, mounds and pools, and flats in 2004 and 2005. Bars on columns represent one SE, and different letters indicated differences at  $P < 0.05$ .

Three colonizing woody species were encountered in 2004, and five were encountered in 2005 (Table 4.3). *Salix* spp. were the most prevalent colonizing woody species in 2004. Numbers of *P. deltoides* and *Salix* spp. remained similar between years, but numbers of *Ulmus* spp. increased from 17 to 100, with 99 individuals occurring on ridges.

Table 4.3. Total number of wooding colonizing species in 2004 and 2005 based on topography.

	2004			2005		
	Ridges	Mounds & Pools	Flats	Ridges	Mounds & Pools	Flats
<i>Acer negundo</i>	0	0	0	2	0	0
<i>Celtis</i> spp.	0	0	0	6	0	0
<i>Populus deltoides</i>	1	11	5	4	16	2
<i>Salix</i> spp.	41	5	6	36	3	1
<i>Ulmus</i> spp.	9	2	6	99	1	0

Detailed analysis of species composition on ridges revealed differential species colonization based on elevation. In 2004, species composition of *Cyperus* spp., *Echinochloa* spp., and *M. officinalis* was highest at the bottom of ridges and lowest at the top (Table 4.4). The opposite trend was observed in *P. ramosissimum*. In 2005, nine species displayed differential colonization. All species except for *A. trifida* had higher composition on ridge bottoms than tops. In addition, *Leersia oryzoides*, *Ludwigia* spp., and *T. latifolia* only colonized the bottoms of ridges, and *Sesbania* spp. only colonized the bottom and middle elevations. Total species richness was also highest at the base of ridges during both years, and in 2005, litter cover was lowest at the bottom of ridges (Table 4.4).

Table 4.4. Percent species composition  $\pm$  one SE on ridges for selected species. Top, middle, and bottom refer to relative elevation on ridges. Different letters indicate significant differences between elevations within the same year.

	2004				2005			
	Top	Middle	Bottom	<i>P</i>	Top	Middle	Bottom	<i>P</i>
<i>Ambrosia trifida</i>					6.1 $\pm$ 1.1a	6.1 $\pm$ 1.1a	2.3 $\pm$ 0.6b	0.008
<i>Ammannia coccinea</i>					0 $\pm$ 0b	0 $\pm$ 0b	0.83 $\pm$ 0.3a	0.001
<i>Chamaecrista fasciculata</i>					14.2 $\pm$ 2.1b	22.3 $\pm$ 3.0a	22 $\pm$ 3.5a	0.08
<i>Cyperus spp.</i>	0.13 $\pm$ 0.06b	0.31 $\pm$ 0.09ab	0.42 $\pm$ 0.16a	0.04	0.10 $\pm$ 0.05b	0.15 $\pm$ 0.08b	8.2 $\pm$ 1.7a	0.001
<i>Echinochloa spp.</i>	1.4 $\pm$ 0.24c	2.7 $\pm$ 0.48b	5.4 $\pm$ 0.89a	0.001	0.004 $\pm$ 0.004b	0.03 $\pm$ 0.02b	1.1 $\pm$ 0.2a	0.001
<i>Leersia oryzoides</i>					0 $\pm$ 0b	0 $\pm$ 0b	0.21 $\pm$ 0.10a	0.02
<i>Ludwigia spp.</i>					0 $\pm$ 0b	0 $\pm$ 0b	2.1 $\pm$ 0.4a	0.001
<i>Melilotus officinalis</i>	0.29 $\pm$ 0.07b	0.62 $\pm$ 0.17a	0.47 $\pm$ 0.11a	0.08				
<i>Polygonum spp.</i>								
<i>P. ramosissimum</i>	1.3 $\pm$ 0.28ab	2.1 $\pm$ 0.61a	0.6 $\pm$ 0.11b	0.03				
<i>Sesbania spp.</i>					0 $\pm$ 0b	1.6 $\pm$ 0.8b	15 $\pm$ 3.2a	0.001
<i>Sorghum halepense</i>								
<i>Typha latifolia</i>					0 $\pm$ b	0 $\pm$ b	0.8 $\pm$ 0.20a	0.001
Total Richness	4.9 $\pm$ 0.3c	5.4 $\pm$ 0.3 b	6.0 $\pm$ 0.3 a	0.003	4.5 $\pm$ 0.18b	4.6 $\pm$ 0.20b	6.4 $\pm$ 0.30a	0.001
Litter (%)					54 $\pm$ 3.2b	60 $\pm$ 2.9a	46 $\pm$ 3.9c	0.001

## **Discussion**

Community structure in bottomland forests results from interactions between topography and hydrology, nutrient dynamics, and inter- and intraspecific interactions (Xiong et al. 2003). In natural bottomlands, these interactions create differences in flood frequency and duration (Koponen et al. 2004), pH and soil nutrient levels (Beatty 1984), litter accumulation (Schilling and Lockaby 2005), and species composition (Titus 1990) between topographic positions. Results of this study indicate that creating topographic features of floodplain forests as part of a restoration effort creates gradients and patterns and initiates processes that may be similar to those found in naturally formed bottomland forests.

### *Soil properties*

Many of the soil physiochemical attributes of ridges were significantly different than flats and mounds and pools. In naturally created floodplains, ridges are typically created as coarse sediments are deposited and build up over time (Hodges 1997). In this study, ridges were constructed by excavating and contouring deeper subsoil material. Texture on ridges was significantly sandier than on flats and mounds and pools. These deeper subsoils may have had higher sand content than materials closer to the surface that were used to create flats and mounds and pools. Alternatively, finer clays may have been deposited on flats and mounds and pools during inundation or been removed from ridges through weathering.

Significantly higher sand content on ridges may have produced many of the differences in soil chemical concentrations seen in this study. More permeable soils have higher potential for leaching, which may explain lower sulfur, sodium, and magnesium concentrations on ridges. Electrical conductivity was also lower on ridges, which was similar to results of another study that reported higher conductivity in soils with higher clay content (Noon 1996).

Soil pH was lower on flats and mounds and pools than on ridges, and nitrate was highest on mounds and pools. These results may reflect hydrologic conditions of the

study site. Soil pH typically declines in saturated soils as carbon dioxide accumulates (Ponnameruma 1984). However, nitrate also generally declines in flooded conditions, which is opposite of results of this study. The high nitrate concentration in mounds and pools may have accumulated from inundation by eutrophic water (Mitsch and Rust 1984). During large precipitation events, water inputs included direct precipitation, runoff from the closed landfill, and ground water from nearby created sloughs. Elevated nutrient levels in urban streams and groundwater are common (Hey 2002), and these water sources may have introduced nutrient-rich sediments that settled in pool bottoms.

#### *Responses of planted seedlings*

In bottomland hardwood forests, woody species are closely associated with topographic variations due to hydrologic conditions along elevation gradients (Hodges 1997, Battaglia et al. 2002). In this study, while soil treatments had minimal effect on planted tree seedling growth or survival, it appears that topography treatments resulted in differential growth and survival of seedlings that resembled patterns in natural bottomlands. Survival of all species in the mature mix was highest on ridges, and survival of most of these species was lowest on mounds and pools. While many of these species are moderately flood tolerant, extended inundation in the pools through the summer months likely reduced growth and survival. In addition, the coarser substrate on ridges may have improved growth and survival as many of these species are better adapted to coarse alluvium deposited on ridges (Allen et al. 2001).

Species in the pioneer mix were less affected by topography. All of these species are at least moderately tolerant of flooding, develop adventitious roots, and often occur in sloughs, flats, and depressions (Hook 1984, Allen et al. 2001). As such, prolonged inundation on flats and mounds and pools did not affect these species as it did those in the mature mix. In fact, growth was higher for *C. occidentalis* and *S. nigra* on mounds and pools and ridges, and growth of *F. pennsylvanica* was highest on mounds and pools.

These results indicate the importance of including a variety of species with various flood tolerances and from different functional groups instead of limiting

plantings to a few highly desirable species such as *Quercus* spp. and *Carya* spp. as is done in most reforestation and bottomland restoration efforts. Species that were used in the pioneer mix are usually considered weedy species and are not often included as part of restoration strategies. However, in this case, these species generally had higher survival in mounds and pools and flats than any of the more desirable species in the mature mix. Introducing pioneer species that have higher survival and growth rates than later successional species can provide benefits such as establishing a quick canopy cover and amelioration of soil and hydrologic conditions (Duloher et al. 2000). While the modification of site conditions can aid in the growth and establishment of more desirable species, it may be necessary to thin pioneer species after a few years to reduce competition with later successional species (Duloher et al. 2000). In addition, introducing a mixture of pioneer species and later successional species with various flood tolerances across topographic positions maximizes chances of establishment and survival of a diverse tree canopy of species most adapted to site conditions.

#### *Colonizing species*

Topography treatments had significant effects on composition and distribution of colonizing species. In 2005, strong patterns arose in the distribution of colonizing species based on species' flooding tolerances. *M. officinalis* increased from 3% to 20% composition on ridges but remained a minor component in mounds and pools and flats. This species can expand rapidly, but because it is poorly adapted to prolonged inundation (Reed 1988), expansion was most likely restricted to drier ridges. Species adapted to flooding, such as *Echinochloa* spp., *A. coccinea*, and *Cyperus* spp., were more dominant on wetter mounds and pools than drier ridges or flats, while species less adapted to flooding, such as *C. fasciculata* and *I. annua*, were most dominant on ridges (Reed 1988, Ahn et al. 2004).

Colonizing woody species also showed a definite pattern in distribution based on topography. All species except *P. deltoides* showed a preference for ridges. Numbers of *P. deltoides*, a pioneer species that germinates best on wet soils, were highest on mounds

and pools (Allen et al. 2001). In 2005, 99 out of 100 individuals of *Ulmus* spp. were located on ridges. These species germinate best on moist, but not wet, soils (Allen et al. 2001).

While topography treatments resulted in differential colonization of various species, litter cover appeared to affect numbers of colonizing species. Litter cover has been shown to reduce species richness in many plant communities (Xiong and Nilsson 1999). It has been suggested that factors such as flooding that remove litter may increase species richness (Xiong et al. 2003). Results of this study support that hypothesis. In 2005, mounds and pools had significantly lower litter cover than ridges and flats and highest average species richness.

Topography also created a gradient in terms of species colonization on the ridges. Nine of the 35 species encountered on ridges in 2005 exhibited differential species composition based on elevation (Table 4.4). Obligate and facultative wetland species colonized the bottom of ridges but were largely absent at higher elevations. This pattern increased with time as *Cyperus* spp. increased in abundance on ridge bottoms from 0.42% in 2004 to 8.2% in 2005. Likewise, *A. coccinea*, *L. oryzoides*, *Ludwigia* spp., *Sesbania* spp., and *T. latifolia* all had higher composition at the bottoms of ridges than at higher elevations in 2005.

Species richness was also higher at the base of ridges, which may be explained by a number of factors. Higher soil moisture may have allowed for improved germination resulting in establishment of more species. Seeds dispersed by water may also have been deposited at the base of ridges during inundation, while this dispersal mechanism would not have operated at higher elevations. Litter cover was also significantly lower at the bottom of ridges, and may have reduced germination at higher elevations (Xiong and Nilsson 1999).

## **Conclusions**

The objective of this study was to assess biotic and abiotic responses to created topographic heterogeneity, planting mixtures, and soil treatments as part of bottomland



forest restoration. Restoration efforts resulted in differential nutrient dynamics, seedling growth and survival, and colonizing species composition and distribution. Because the top 5 m of soil had been removed from the site, it was thought that planted pioneer seedlings would have had higher survival and growth than later successional species. Results indicated that seedling survival was likely more dependant on hydrologic conditions than soil conditions. However, planting a mixture of pioneer species and mature species provided a diverse species mix and resulted in survival of many species regardless of hydrologic variations.

Erosion control mats were used in this study as a surrogate for mulch. The addition of mulch in harsh site conditions can increase soil organic matter and other soil properties such as cation exchange capacity (Athy et al. 2006). In this study, erosion control mats had relatively minor influence on survival and growth of planted seedlings. Erosion control mats also had little influence on colonizing species composition. Under the conditions at this site, erosion control mats probably would not be necessary to improve site conditions.

The creation of microtopography had the greatest influence on nutrient dynamics, seedling growth and survival, and colonizing species composition. Topographic heterogeneity resulted in diverse hydrologic conditions that affected all variables measured in this study. Creating microtopography similar in structure to that formed naturally has initiated patterns and processes that may resemble those in bottomland hardwood forests.

## CHAPTER V

### ASSESSING THE USE OF GIS ON STUDENT MOTIVATION AND CONCEPTUAL KNOWLEDGE IN AN UNDERGRADUATE ECOLOGY LAB

#### Introduction

Problem-based learning (PBL) is “an instructional (and curricular) learner-centered approach that empowers learners to conduct research, integrate theory and practice, and apply knowledge and skills to develop a viable solution to a defined problem” (Savery 2006). First developed in the 1950s and 1960s, PBL is becoming increasingly popular in science education as it engages students, develops higher-order thinking skills, improves knowledge retention, and enhances motivation (MacKinnon 1999, Dochy et al. 2003, Savery 2006). Techniques for creating problem-based learning environments vary, but field labs and the use of computers and other instructional technologies have been used in PBL to effectively increase motivation and conceptual knowledge (Hudak 2003, Kerfoot et al. 2005, Taradi et al. 2005).

Geographic information systems (GIS) have long been used by researchers as tools to store, manage, analyze, and display spatial data, but they are also increasingly being used by educators in K-16 classrooms as a means to support PBL (Summerby-Murray 2001, Drennon 2005). GIS has the potential to enhance learning by creating a student-centered inquiry environment, creating links between policy and science to help students solve real-world problems, enhancing interdisciplinary learning, enabling students to use the same tools as professionals, and being accessible to a wide range of learners (NRC 2006). These functions of GIS in education meet the National Research Council’s recommendations for effective learning, which include creating an environment that is learner, knowledge, assessment, and community centered (Bransford et al. 1999). In addition, GIS has been shown to improve student attitude by increasing the relevance of the subject to the student (West 2003).

Despite the increased interest in GIS in the classroom, there is much discussion as to the appropriate use of GIS in an education setting (Brown and Burley 1996, Chen 1998, Bednarz 2004). The lack of teacher training, the unavailability of computer resources, the inherent complexity of GIS, and the time spent teaching technology at the expense of science content are some barriers to bringing GIS into the classroom (Lloyd 2001, Baker 2005). Additionally, the effectiveness of GIS in terms of student impact is still under debate. This is due, in part, to scant empirical evidence as to its effectiveness as a teaching tool. Much of the literature that considers impacts of GIS in teaching is based on subjective, anecdotal case studies. Some research, however, has shown potential benefits for GIS to be used in secondary and undergraduate classrooms. For example, Patterson et al. (2003) found that AP high school students that used GIS outperformed college undergraduate students who did not in terms of geographic skills and concepts. Kerski (2003) reports that using GIS improved high school students' motivation towards geography, and GIS was a particularly helpful tool in raising final course grades for average and below-average students. In an undergraduate geography course, Lloyd (2001) found that students most valued their time spent on computer-based instruction as opposed to other learning styles, and in terms of content, Lloyd concluded that "students learn at least as well using computer-based instructional materials as they do with traditional approaches to learning." However, Pedersen et al. (2005) found that student performance in an undergraduate geography course was similar between those that used computer-based maps and those that used paper maps, but students preferred paper maps because of a general dislike of computers and the inability to view the entire map on the computer monitor. Likewise, Proctor and Richardson (1997) found little improvement in learning outcomes of students that used a multimedia GIS in an undergraduate human geography course.

The inconsistencies reported between studies show that much is to be learned about impacts of GIS on student attitude and conceptual understanding under different settings. Furthermore, most of the research on GIS education in undergraduate classrooms has been conducted in geosciences while assessing impacts on students'

spatial abilities. Few studies consider the use of GIS as a means to teach ecological concepts. The objective of this study was to determine if GIS could be used in an introductory ecology lab to enhance knowledge of ecological concepts without the need for students to have a fundamental understanding of GIS technology. More specifically, the research questions were: (1) will the use of GIS improve students' attitudes toward an undergraduate ecology lab, and (2) are there differences in conceptual knowledge of wildlife habitat features, spatial attributes, and population sampling techniques between students that use GIS and those that do not? Of additional interest were how impacts of GIS might differ based on student grade level, major, and achievement level.

## **Methods and Assessment**

### *Participants and experimental design*

This study was conducted in an undergraduate Fundamentals of Ecology Laboratory at Texas A&M University during Fall of 2004 (Year 1) and Fall of 2005 (Year 2). The lab served as an elective science course for non-science majors and a required course for natural-science majors. Ten lab sections were included each year with up to 140 students. Each lab section consisted of up to 14 students. Data analysis was based on 102 students in Year 1 and 100 students in Year 2 based on students' willingness to participate and numbers of completed surveys and tests.

This course was designed as a field-based inquiry lab addressing fundamental principles of ecology and methods of ecological inquiry. Teaching methods included a short lecture-style introduction to ecological concepts followed by field-based research and hands-on data collection. All ten sections were taught in this traditional way, but five sections (referred to as Field+GIS) were randomly selected to include the addition of GIS. In Year 1, few students indicated that they had ever used GIS (up to two students per section), and in Year 2, 24 students indicated that they had taken a course that included instruction on GIS. The same instructor taught all twenty sections over both years.

All students were instructed in the basic principles of habitat features and requirements (e.g., food, cover, disturbance) and spatial attributes (e.g., habitat size and adjacency, distribution, and density of patch types). They were then introduced to the Aggie Squirrel Project in which researchers at Texas A&M were tracking fox squirrels (*Sciurus niger*) on campus fitted with radio collars with the purpose of gaining an understanding of urban fox squirrel population ecology. Students were instructed to make a prediction describing what type of habitat squirrels would utilize on campus and why. They were divided into groups of 3 to 4 students and given frequencies of collars for specific squirrels and the radio telemetry equipment needed to locate them. The data students recorded included the location of each squirrel, a description of the general environment, and estimates of distances between squirrels and walkways, buildings, trees, and open grassy fields. In Year 2, most of the batteries in the squirrels' collars had expired making the tracking exercise impossible. In an attempt to maintain the consistency of the experience between years, the instructor hid active collars in trees, and students were instructed to use the telemetry equipment to find the collars. The students thought they were searching for squirrels, as it was not disclosed to them that the collars on the squirrels were no longer operational. A few students, however, located the collars in the trees. Following the field exercise, the students returned to the lab to discuss their findings and to compare them with initial hypotheses.

The five Field+GIS sections were then provided an additional 1300 squirrel locations that were obtained by the Texas A&M researchers and imported into a GIS (ArcView 3.2a©). Students were able to view squirrel locations represented by points on an aerial photo of campus (Digital Ortho Quarter Quadrangles). They were then given step-by-step instructions on how to generate grids of distances of squirrels to trees and squirrels to buildings. The databases created in ArcView were exported to Microsoft Excel© to create frequency distributions of distances of squirrels to trees and buildings. The results of these analyses were compared with original student-generated hypotheses, and additional question and hypothesis creation ensued. The entire lab procedure was conducted during one three-hour lab period.

Following the Year 1 study, it was recognized that the time of exposure to the concepts in the lab was different between the groups as the Field+GIS labs lasted 30 to 45 minutes longer than the Field-only labs. In order to equalize the time of exposure in Year 2, an additional discussion was given to the Field group following the field exercise. During the discussion, the instructor provided information on the habitat requirements and features of fox squirrels in rural environments, and the students discussed similarities and differences between these features with those they encountered on campus.

#### *Learning products and assessment*

A five-scaled pre/post survey, composed of 27 attitude questions adopted from West (2003), was used to assess students' attitudes toward the ecology course. The five scales included attitudes and perceptions of (1) students' efforts, (2) relevance of the subject, (3) satisfaction level, (4) performance, and (5) understanding. A five-point Likert scale was used to measure the level of agreement of the student with the statement, with a score of 5 = Strongly Agree, 4 = Agree, 3 = Neutral, 2 = Disagree, and 1 = Strongly Disagree. The same attitude surveys were used both years. A rubric was developed to assess students' responses to a pretest and posttest to determine gains in students' understanding of conceptual knowledge based on teaching methods. The rubric assessed conceptual understanding of habitat features, spatial attributes, and techniques of wildlife population sampling. The same pretest was given in both years; however, the posttest used in Year 1 to determine conceptual understanding was partially invalid as it was spatially oriented in nature. This test may have focused students' attention on spatial attributes at the expense of habitat features (see Results and Discussion) or given students less of an opportunity to comment on habitat features than on spatial attributes. The posttest used in Year 2 was adjusted to mirror the pretest to more accurately assess student understanding of all three concepts of interest. Cohen's kappa (Cohen 1960) was used to determine inter-rater reliability for the use of the rubric.

Initial kappa between two raters for 15 tests was 0.63. Following additional rater training, kappa improved to 0.82.

Effect size analyses using Cohen's  $d$  (Cohen 1988) were used to determine the magnitude of change from pre to post-survey and test scores within treatments. A value of 0.20 was considered a small effect, 0.50 was considered a medium effect, and 0.80 was a large effect (Cohen 1992). In addition, multivariate analysis of variance (MANOVA) procedure was used to determine differences between pre and post surveys and test scores, as well as time by treatment interactions (Field vs. Field+GIS). Analysis of covariance (ANCOVA) was used to determine differences between treatments with presurveys and pretests used as covariates. MANOVA and ANCOVA analyses were also performed to determine treatment effects by class (underclassmen and upperclassmen) and major (science and non-science majors). Analyses were also performed based on achievement level where pretest grades were used to separate students into higher (the top half of the scores) and lower (the bottom half of the scores) achievement groups. Statistical analyses were performed with SAS© software.

## **Results**

### *Attitudes*

In Year 1, attitudes for both Field and Field+GIS groups generally improved from pre to post surveys as indicated by effect sizes (Fig. 5.1a), although the magnitude of the changes were small ( $d = 0.3$  or less). Attitudes for upperclassmen in both groups improved in Effort, Relevance, and Understanding scales (Fig. 5.2). Non-science majors in the Field+GIS group showed a significant increase over the Field group in Effort (ANCOVA,  $F = 5.41$ ,  $P < 0.05$ ) and Relevance (ANCOVA,  $F = 3.46$ ,  $P < 0.10$ ), and science majors in both groups improved in Effort, Relevance, and Performance (MANOVA, Fig. 5.2). Higher achieving students in the Field group showed a greater increase in scores for the Understanding scale than higher achieving students in the Field+GIS group (ANCOVA,  $F = 2.94$ ,  $P < 0.10$ ). Scores for higher achieving students improved in both Field+GIS and Field groups in Relevance, Performance, and

Understanding as did scores in Effort for lower achieving students (MANOVA, Fig. 5.2).

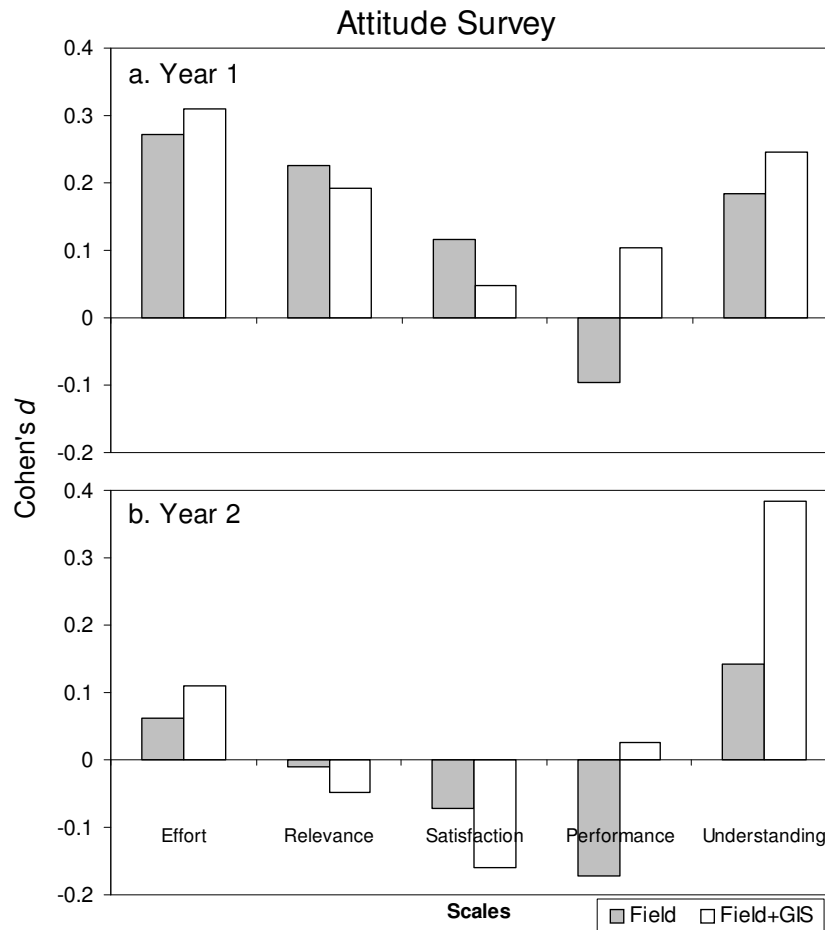


Figure 5.1. Results of effect size analysis on attitude surveys from the Field and Field+GIS groups in (a) Year 1 and (b) Year 2.

In Year 2, changes in attitude were negligible (Fig. 5.1b). Unlike Year 1 when changes were positive, attitudes declined in the Relevance, Satisfaction, and Performance scales, but the Field+GIS group indicated a small to medium increase in their perception of understanding ( $d = 0.38$ ). Scores for the Satisfaction scale declined for underclassmen in both groups (MANOVA,  $F = 4.28$ ,  $P = 0.05$ , Fig. 5.3), but



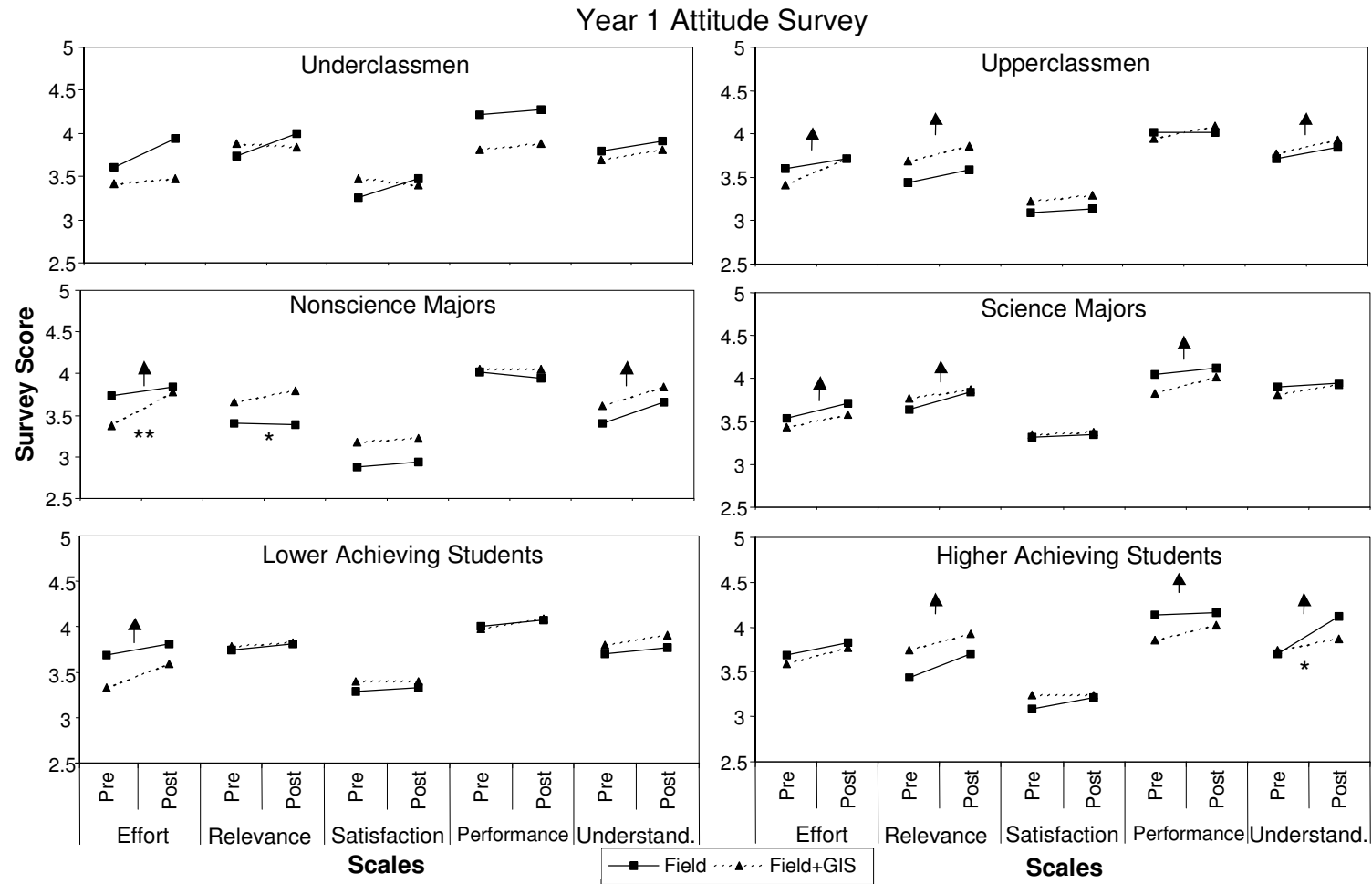


Figure 5.2. Pre/post survey scores from Year 1 measuring changes in student attitude. Asterisks (\*, and \*\*) indicate significant differences in ANCOVA between Field and Field+GIS groups at  $P < 0.10$  and  $0.05$ , respectively. The direction of arrows indicates significant time effects (MANOVA,  $P < 0.10$ ) with increasing or decreasing scores from pre to post tests.

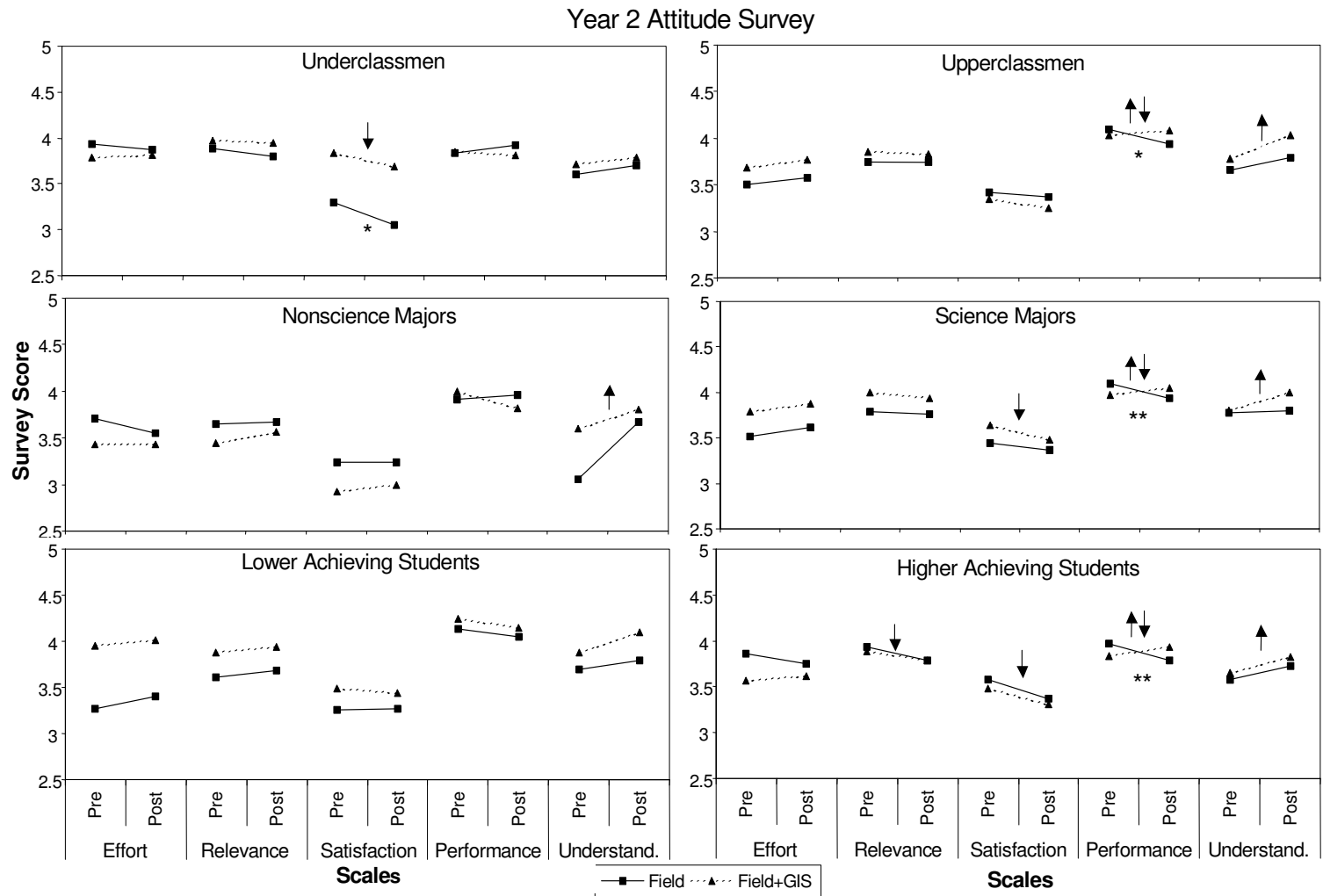


Figure 5.3. Pre/post survey scores from Year 2 measuring changes in student attitude. Asterisks (\*, and \*\*) indicate significant differences in ANCOVA between Field and Field+GIS groups at  $P < 0.10$  and  $0.05$ , respectively. The direction of arrows indicates significant time effects (MANOVA,  $P < 0.10$ ) with increasing or decreasing scores from pre to post tests.

ANCOVA results indicated the score for the Field+GIS group declined less than the Field group ( $F = 3.05$ ,  $P < 0.10$ ). Satisfaction scores also declined for science majors and higher achieving students (Fig. 5.3). Upperclassmen in the Field+GIS group also had significantly higher post-survey scores than the Field group on the Performance scale (ANCOVA,  $F = 3.41$ ,  $P < 0.10$ ). Post-survey scores were also higher for science majors and higher achieving students in the Field+GIS group on Performance (ANCOVA,  $F = 5.93$ ,  $P < 0.05$  and  $F = 5.67$ ,  $P < 0.05$ , respectively), while MANOVA indicated significant time by treatment interactions.

### *Conceptual knowledge*

Effect size analysis of pre and post tests in Year 1 revealed a small improvement in conceptual knowledge for the Field group as indicated by total score ( $d = 0.29$ ), but little change occurred in the Field+GIS group ( $d = 0.10$ , Fig. 5.4a). Scores for habitat features declined for both groups, but the Field+GIS group's decline was large ( $d = -0.94$ ). Alternatively, scores for spatial attributes increased by a magnitude of  $d = 0.99$  for the Field group and  $d = 1.41$  for the Field+GIS group (Fig. 5.4a). MANOVA results corroborate effect size results in that conceptual knowledge of spatial attributes, population sampling techniques, and total scores generally increased for both Field and Field+GIS groups, while scores on habitat features declined (Fig. 5.5). Underclassmen in the Field+GIS group performed worse than underclassmen in the Field group on habitat features (ANCOVA,  $F = 24.77$ ,  $P < 0.01$ ), and total scores were lower for the Field+GIS group as well (ANCOVA,  $F = 15.23$ ,  $P < 0.01$ ). Science majors in the Field+GIS group also had lower habitat feature scores and total scores than science majors in the Field group (ANCOVA,  $F = 6.24$ ,  $P < 0.05$ , and  $F = 4.46$ ,  $P < 0.05$ , respectively). Higher achieving students in the Field group had higher posttest scores on habitat features and total scores than those in the Field+GIS group (ANCOVA,  $F = 5.46$ ,  $P < 0.05$ ), but scores for both groups declined (MANOVA, Fig. 5.5). Lower achieving students in the Field group outscored lower achieving students in the Field+GIS group on habitat features and total scores (ANCOVA,  $F = 7.04$ ,  $P < 0.05$ ), but unlike scores for

higher achieving students, total scores improved from pre to post tests (MANOVA, Fig. 5.5).

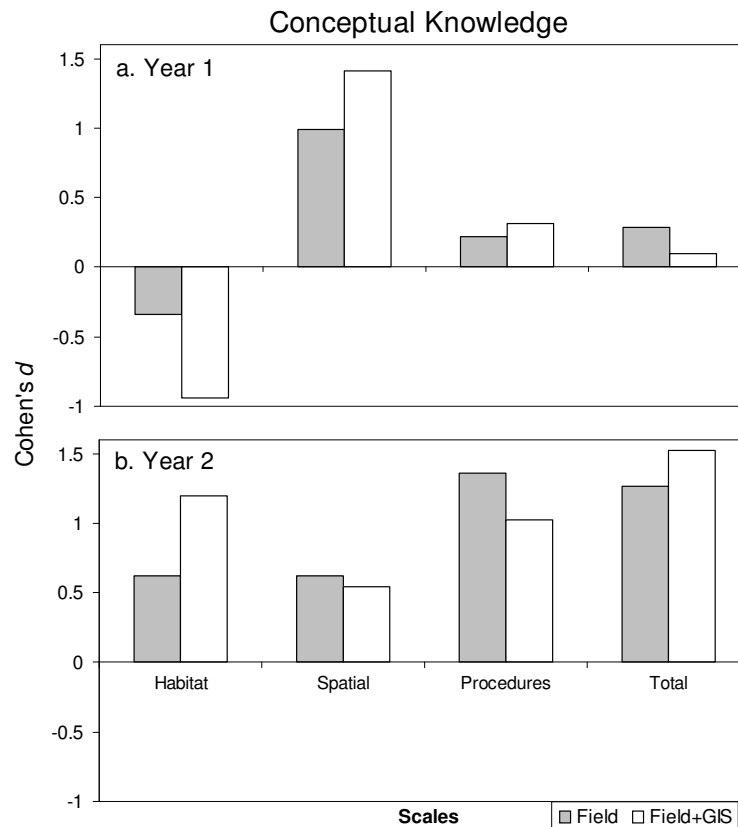


Figure 5.4. Results of effect size analysis on tests of conceptual knowledge from the Field and Field+GIS groups in (a) Year 1 and (b) Year 2.

In Year 2, the magnitude of change in conceptual knowledge scores varied from a low of 0.54 to a high of 1.52 (Fig. 5.4b). Total scores increased substantially for both groups, as did knowledge of population sampling. Knowledge of habitat features increased more for the Field+GIS group ( $d = 1.20$ ) than the Field group ( $d = 0.62$ ). ANCOVA also showed a significant difference in scores on habitat features between groups (ANCOVA,  $F = 3.06$ ,  $P < 0.10$ ), while MANOVA indicated increases for Field and Field+GIS groups in almost all categories (Fig. 5.6). Upperclassmen and science

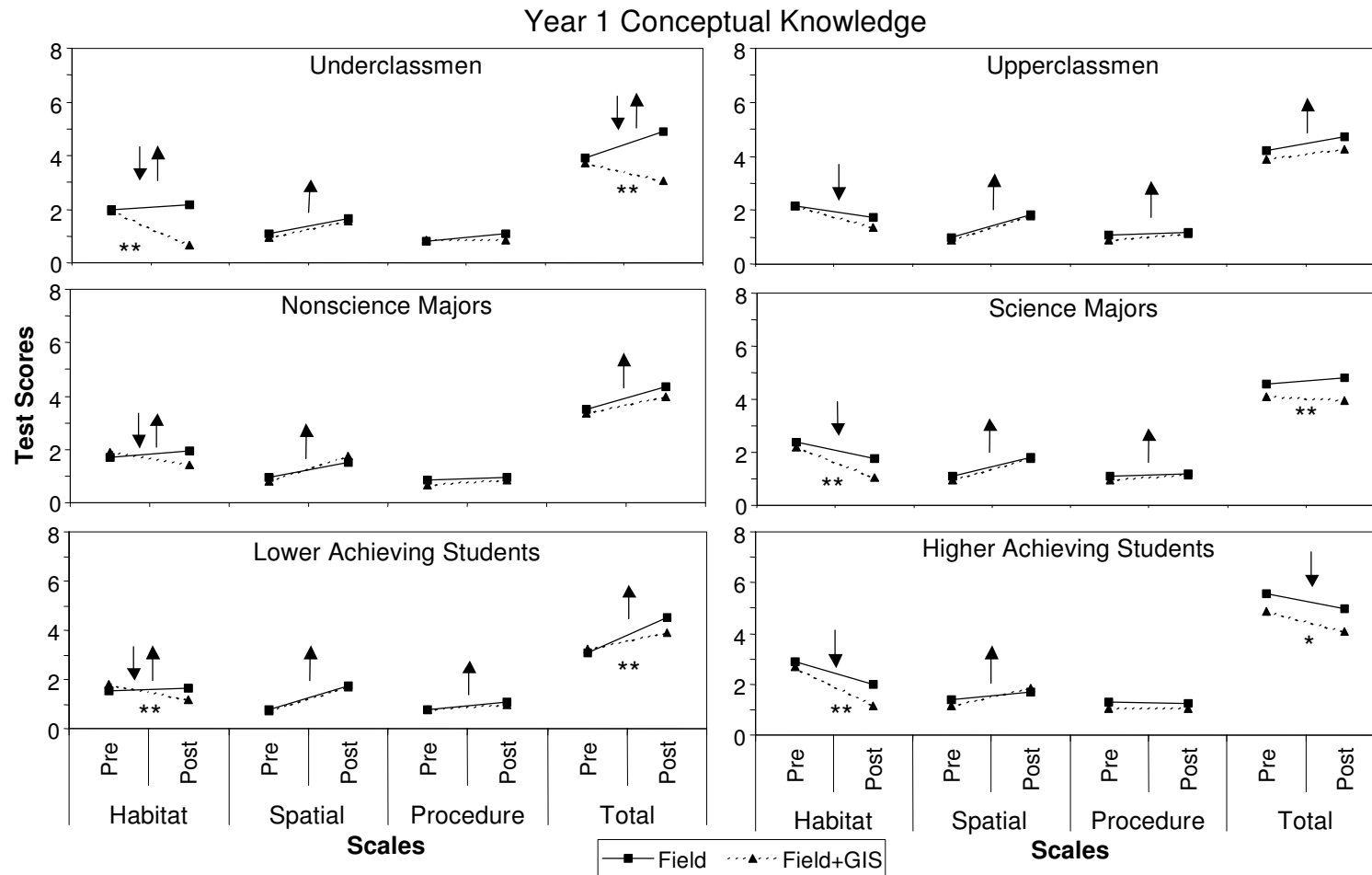


Figure 5.5. Pre/post test scores from Year 1 measuring conceptual understanding of habitat features, spatial attributes, population sampling techniques (labeled ‘procedures’), and total scores. Asterisks (\*, and \*\*) indicate significant differences in ANCOVA between Field and Field+GIS groups at  $P < 0.10$  and  $0.05$ , respectively. The direction of arrows indicates significant time effects (MANOVA,  $P < 0.10$ ) with increasing or decreasing scores from pre to post tests.

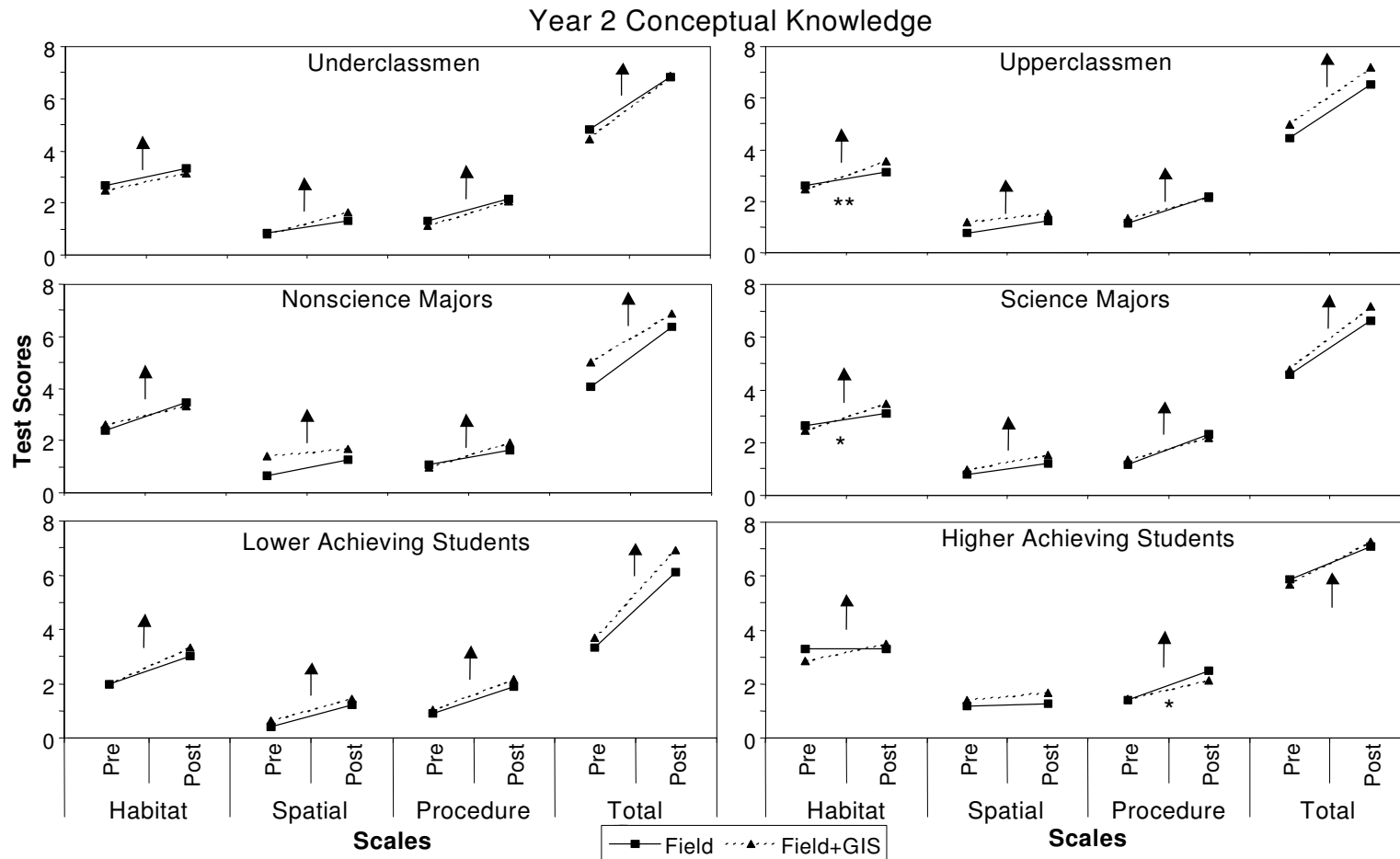


Figure 5.6. Pre/post test scores from Year 2 measuring conceptual understanding of habitat features, spatial attributes, population sampling techniques (labeled ‘procedures’), and total scores. Asterisks (\*, and \*\*) indicate significant differences in ANCOVA between Field and Field+GIS groups at  $P < 0.10$  and  $0.05$ , respectively. The direction of arrows indicates significant time effects (MANOVA,  $P < 0.10$ ) with increasing or decreasing scores from pre to post tests.

majors in the Field+GIS group scored higher on habitat features than their cohorts in the Field group (ANCOVA,  $F = 5.57$ ,  $P < 0.05$  and  $F = 4.62$ ,  $P < 0.05$ , respectively), but higher achieving students in the Field group scored higher on wildlife population sampling than those in the Field+GIS group (ANCOVA,  $F = 3.07$ ,  $P < 0.10$ ).

## **Discussion**

### *Attitudes*

In Year 1, student attitudes improved from pre to post surveys (Fig. 5.1a); however, the extent of the improvement varied between groups of students and for different scales. Nonscience majors in the Field+GIS group reported a greater increase in perceived effort and relevance of the subject than nonscience majors in the Field group, and higher achieving students' perception of understanding increased more for the Field group than the Field+GIS group (Fig. 5.2).

Although there were some differences between treatment groups, these differences may be outweighed by the fact that attitudes improved for both treatment groups in effort, relevance, understanding, and to a lesser degree, performance. The nearly equal improvement in attitude between the Field and Field+GIS groups indicates that something other than the GIS treatment improved students' attitudes. Because the field experience was similar for both groups of students, it is probable that improved attitudes resulted from the field-based nature of the lab. This finding is in agreement with Hudak's (2003) report that students enrolled in introductory geoscience courses preferred outdoor lab exercises to indoor lab exercises because they viewed outdoor lab exercises to be more interactive, interesting, and realistic. Additionally, Karabinos et al. (1992) found that outdoor field exercises makes students active participants thus creating enthusiasm for the subject. The field portion in this lab probably generated similar enthusiasm as attitudes improved for students in both Field and Field+GIS groups.

Unlike Year 1, student attitudes in Year 2 changed little or declined from pre to post surveys in four of the five scales (Fig. 5.1b). It is again thought that the field component of the lab had greater impact on attitude than the use of GIS. In Year 2,

students did not find radio-collared squirrels because the batteries in the collars had expired. Instead, students tracked collars that had been previously hidden in trees. Although the instructor gave the students the radio frequencies of the live collars and told the students to track the frequencies of the collars, the students assumed they were tracking squirrels. Throughout and following the field exercise, students expressed frustration and displeasure in not finding the squirrels they were supposed to be tracking, and the few students that located the hidden collars also expressed annoyance at being deceived. The result was decreasing attitudes for some of the students as satisfaction scores declined for underclassmen, science majors, and higher achieving students (Fig. 5.3). However, there were some significant differences between groups, and where differences existed, attitudes for the Field+GIS were higher than those for the Field group. Perhaps in these cases using a novel technology such as GIS that created a problem-based learning environment helped to compensate for the displeasure of the field experience resulting in students in Field+GIS group having a more favorable experience than those in the Field group. It has been reported that GIS coupled with field observation increases motivation (Bransford et al. 1999), and in an undergraduate geomorphology course, Wentz (1999) found that students enjoyed and valued the use of GIS when coupled with more traditional teaching methods. Likewise, Lo (2002) reported high satisfaction from students enrolled in an environmental literacy class that used a combined GIS-field approach. Although not definitive, the results of this study corroborate these previous findings as well as indicate that GIS helped dampen the effects of a disappointing field experience.

Tangential to the impacts of GIS, but nonetheless important, are the differences in attitude between years. These differences are likely due to the change in authenticity of the field experience. Authentic scientific inquiry is rarely taught in classrooms; much science taught in schools is done through simplified scientific tasks. These simple tasks, however, do not represent cognitive processes required for authentic scientific inquiry (Chinn and Malhotra 2002), nor do they maximize the potential for students to become engaged and motivated (Strong 2002). As student motivation is a vital component of



learning (NWREL 2001), authentic scientific inquiry should be used in undergraduate classrooms to foster critical thinking, enhance engagement, and improve motivation (Oliver 2006). The results of this study clearly indicate the difference in students' attitude when an authentic experience is conducted (in Year 1) and when they are exposed to a fabricated science experience (in Year 2).

### *Conceptual knowledge*

Because the instrument used in Year 1 to determine change in conceptual knowledge was ineffective, meaningful comparisons between years are difficult to make. However, comparisons can be made between treatment groups within the same year, and those comparisons yield interesting results. First of all, scores on habitat features declined for both groups in Year 1, but knowledge of spatial attributes improved (Fig. 5.4a). This is thought to be a result of the spatial nature of the posttest not lending itself well for students to be able to address questions related to habitat features. But while scores on habitat features declined for both groups, between-group comparisons show large and significant differences in the performance of students with the Field+GIS group scoring worse than the Field group (Figs. 5.4a and 5.5). Effect size analysis shows a three-fold difference in the magnitude of the decline between groups. Although there is little evidence in the literature to explain this finding, it follows that because GIS is a tool used to view, analyze, and manipulate spatial data, students who used GIS and then were given a test that was predominantly spatial in nature might have focused more heavily on the spatial aspect of the test at the expense of questions regarding habitat features. Students in the Field+GIS group may not have had a poorer understanding of habitat features than the other students, but their attention may have been drawn toward what they were more familiar with. Students in the Field group did not have GIS to reinforce spatial concepts, and therefore, may not have been influenced the same way as the Field+GIS group in completing the test. This interpretation is consistent with student responses to the spatial attributes section of the test where a larger magnitude of improvement occurred in the Field+GIS group than the Field group.

In Year 2, students' conceptual knowledge increased in all categories for both groups, but there were some differences between groups in habitat features and population sampling techniques. The Field+GIS group scored higher than the Field group on habitat features (Fig. 5.4b). This was particularly true for upperclassmen and science majors (Fig. 5.6). These two groups of students are typically more experienced and probably have greater knowledge of the subject matter due to previous course enrollment. Also, upperclassmen may be more skilled using computer technology than underclassmen (Lo et al. 2002). GIS could have assisted these students more than others in making connections to ecological concepts because of their advanced levels of knowledge and familiarity with computers.

Higher achieving students in the Field group scored marginally higher than higher achieving students in the Field+GIS group on population sampling techniques, but scores for all groups of students improved. It was expected that students who used GIS would have scored higher on this category as a result of exposure to additional techniques. However, this was not the case. This study also tested students' knowledge of spatial attributes such as adjacency of cover types, density and distribution of habitat types, and size requirements and limitations of suitable habitat. GIS is thought to improve spatial cognition by enabling students to visualize, analyze, and manipulate spatial data sets (NRC 2006); however, GIS did not improve students' conceptual knowledge of spatial attributes as assessed by the rubric. There may be a number of reasons for this, including: the assessment did not effectively measure gain in student knowledge pertinent to the use of GIS, navigation in the field improved students' spatial awareness for both groups of students so that differences were negligible (Thorndyke and Hayes-Roth 1982), or since the entire lab, including field and GIS portions, was conducted within one three-hour lab period, the time of student exposure to GIS was too short to induce a measurable change.

### *Implications*

This study shows that GIS can be used as an effective teaching tool in an undergraduate ecology lab without teaching the mechanics of GIS. Because of the complexity of the technology, GIS has been reserved for use by advanced students or those enrolled in an advanced college course (Carstensen et al. 1993). In the current study, however, lower achieving students in the Field+GIS group performed as well as those in the Field group. Likewise, there were no differences in conceptual knowledge between groups of higher achieving students. The approach taken in this undergraduate ecology lab was to teach with GIS as opposed to teaching about GIS, i.e., teaching applications of GIS with a focus on education instead of teaching GIS technology with a focus on training (Sui 1995). While teaching about GIS is important for students interested in using it extensively as a tool to analyze spatial datasets, teaching with GIS allows a focus on geographical and spatial knowledge acquisition for a particular application without the necessity of teaching the technicalities of the technology (Sui 1995). Reducing the complexity of GIS for students by creating automated GIS applications or using web-based GIS, can result in rich learning experiences while not limiting GIS use to advanced students (Brown and Burley 1996, Lloyd 2001, Baker 2005).

Besides the fact that GIS did not negatively impact student attitude or performance, of equal significance was that conceptual knowledge for both Field and Field+GIS groups increased substantially across all categories. This indicates that teaching with GIS compared with giving an additional follow-up lecture following the field experience yielded similar benefits. Why spend the time and resources involving students with GIS when the same result can be accomplished with a simple traditional lecture? Although the instruments used in this study found few differences between treatments, there may have been other benefits to the students that were not measured. For example, education technology has been found to meet the principles suggested by Bransford (1999) needed to create an effective learning environment (Boylan 2004). Additionally, GIS has a number of benefits that are not directly linked to attitude and

content knowledge. The National Research Council states that GIS supports the K-12 educational system by providing a challenging, real-world problem-solving context that embodies the principles of student-centered inquiry (NRC 2006). This certainly extends to undergraduate classrooms as well, and the use of GIS by undergraduates also prepares natural and social science majors to be able to use this tool as professionals. While teaching with GIS does not appear to be the cure-all for improving attitude and increasing ecological knowledge, GIS does provide other benefits important for the education of undergraduate students.

The effective use of GIS as a teaching tool may be dependent on the spatial and temporal scale of the investigation. In this study, students used GIS to examine phenomena operating at relatively small spatial scales—associations between squirrels, trees, and buildings. Students were able to observe these same phenomena during their field exercise. There may be little benefit to using GIS as a teaching tool when the spatial phenomena are small-scale and simple in nature. However, analyzing large-scale or spatially complex patterns and processes that are not easily observed during a classroom field exercise would likely bring substantial improvement in conceptual understanding. In addition, exploring temporal changes in spatial patterns is difficult without GIS. Using GIS to explore information across temporal scales may also bring appreciable benefits to student learning.

It is apparent that the field-based nature of the lab had a large impact on both attitude and conceptual knowledge and may have had a greater impact than students' use of GIS. Others have also indicated improved student attitudes and conceptual understanding following field labs as students become engaged in learning activities (Kern and Carpenter 1984, Karabinos et al. 1992). Kern and Carpenter (1986) found that while lower-order learning was similar between students in a traditional classroom setting and those in a field lab, students in the field lab exhibited increased levels of higher-order thinking over students in the classroom. Likewise, conceptual knowledge of students in an undergraduate earth science course improved more from their field and lab experience than from classroom lectures (Trop et al. 2000). The results of this and

other studies support recent calls in science-education reform that advocate incorporating hands-on, inquiry-based field activities into undergraduate courses to improve student learning (NSF 1996).

### **Conclusions**

While the evidence presented here does not overwhelmingly suggest that GIS directly improved student attitude and conceptual knowledge, GIS did not negatively impact student attitude or performance, and the trends indicate a slight improvement for those that used GIS. Both teaching methods resulted in improved student attitude and increased conceptual knowledge. However, the authenticity of the experience appeared to directly affect student attitude. These results indicate the importance of using authentic inquiry in undergraduate science classrooms. In addition, the effective use of GIS as a teaching tool may be scale-dependent. Small-scale phenomena that are simple in nature may be assessed as easily in a field exercise as with GIS. Using GIS to assess large-scale, complex patterns and processes, or changes over time may have a substantial impact on student understanding. Further studies are needed to determine direct benefits of teaching with GIS in undergraduate ecology classrooms.

## CHAPTER VI

### SUMMARY AND CONCLUSIONS

Results of these studies have a number of important implications for bottomland hardwood forest restoration and the use of GIS in an undergraduate ecology laboratory. A comparison of results between the study on mounds and pools and the study including ridges, mounds and pools, and flats reveals similar patterns that may indicate similar underlying processes. Within-plot variations in mounds and pools were similar to plot-level variations between ridge, flat, and mound-and-pool topographic treatments. For example, variation in soil properties generally followed the same patterns between ridges and mounds. Soil pH was higher on mounds compared with pools and highest on ridges compared with flats and mounds and pools. Nitrate was also higher in pools compared with mounds, and nitrate was highest in mounds and pools compared with ridges and flats. Tree seedling survival followed similar patterns as survival was generally higher on mounds compared with pools and ridges compared with mounds and pools. Finally, the composition of colonizing species was again similar between ridges and mounds, and mounds and pools and pools. The same species with highest percent composition on ridges also had highest composition on mounds.

Similar patterns of development in nutrient concentrations, seedling survival, and colonizing species composition between mound tops and ridges are likely a result of the hydrologic regime. Within mound-and-pool plots, mounds were inundated less than slopes and pools, and at the plot level, ridges were inundated the least, and mounds and pools were inundated the longest. Topographic positions higher in elevation had less inundation than those at lower elevations. Water inundation appears to be the mechanism controlling nutrient concentrations, seedling growth and survival, and colonizing species distribution and abundance at multiple spatial scales.

### **Species Responses to Flooding Regimes**

Hydrology is considered one of the most influential factor regulating the structure and function of wetlands. Because of the drastic changes to stream hydrology caused by urbanization, floodplain wetlands such as bottomland hardwood forests are particularly susceptible to altered urban flooding regimes. Due to the importance of hydrology in determining vegetative species survival, productivity, and composition, many studies have focused on determining the flood tolerance of a variety of bottomland hardwood species. However, these types of studies do not answer questions regarding species' flooding tolerance under various flooding regimes. The objectives of this study were to (1) determine responses of bottomland hardwood seedlings' productivity to experimental urban flooding regimes, specifically altered flooding frequency and duration, and (2) determine whether urban flooding regimes impact species from various wetland indicator groups differently. Results indicated that urban flooding regimes have direct effects on the productivity of bottomland species. Specifically, an urban flood with higher frequency and longer duration reduced growth in all three species tested. However, specific responses varied between species from different wetland indicator classes. The implications of altered productivity as a result of urbanization-induced changes in flooding regimes include possible shifts in species composition, the importance of proper species selection in wetland restoration based on hydrologic conditions of the site, and the significance of selecting appropriate reference sites and reference standards.

### **Created Mound-and-Pool Microtopography**

Spatial and temporal heterogeneity are thought to promote diverse communities, and microtopography has been shown to be an important component of bottomland hardwood forests as it influences hydrologic regimes, nutrient dynamics, and species distribution. Despite its importance, creation of microtopography is rarely included in bottomland restoration efforts. The objective of this study was to determine the responses of planted and colonizing species to soil treatments and created

microtopography and their influence on soil physiochemical development within a mound-and-pool complex. Microtopography created a spatially heterogeneous system in terms of hydrology, soil nutrient concentrations, planted seedling growth and survival, and colonizing species distribution, which was similar to that found in natural bottomlands. Erosion control mats, however, had no effect on colonizing species richness and little effect on percent composition of individual species. Erosion control mats also did not improve growth or survival of most planted species, and, therefore, does not appear to be a beneficial restoration strategy in the mound and pool complex of this study.

### **Large-scale Topographic Heterogeneity, Planting Mixtures, and Soil Amendments**

The natural development of topographic heterogeneity within a bottomland forest can be a complex process that develops over long time periods and differs at various spatial scales. Recent bottomland restoration efforts have largely focused on hydrologic restoration, but restoring topographic heterogeneity may help reestablish hydrologic conditions. The objective of this study was to assess biotic and abiotic responses to created topographic heterogeneity, planting mixtures, and soil treatments as part of bottomland forest restoration. Restoration efforts resulted in differential nutrient dynamics, seedling growth and survival, and colonizing species composition and distribution. Topographic heterogeneity resulted in diverse hydrologic conditions that affected all variables measured in this study. Results indicated that seedling survival was likely more dependant on hydrologic conditions than soil conditions. However, planting a mixture of pioneer species and mature species provided a diverse species mix and resulted in survival of many species regardless of hydrologic variations. Erosion control mats had relatively minor influence on survival and growth of planted seedlings and exerted little influence on colonizing species composition. In this study, erosion control mats probably would not be necessary to improve site conditions.



### **Use of GIS on Student Motivation and Understanding**

Researchers have long used GIS as a tool to store, manage, analyze, and display spatial data, but educators are using it increasingly as a means to support problem-based learning. Despite the increased interest in GIS in the classroom, there is little evidence to support assertions that the use of GIS can improve students' conceptual knowledge. The objective of this study was to determine if GIS could be used in an introductory ecology lab to enhance knowledge of ecological concepts and improve attitudes. While results of this study do not overwhelmingly suggest that GIS directly improved student attitude and conceptual knowledge, GIS did not negatively impact student attitude or performance. However, trends indicate a slight improvement for those that used GIS. Student attitude appeared to be directly affected by the authenticity of the science experience. In addition, the effective use of GIS as a teaching tool may be scale-dependent with more improvement in understanding following assessments of large-scale, complex patterns and processes as opposed to simple phenomena operating at small-scales directly observable in the field.

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