Evaluating the Success of Bottomland Forest Restoration In the Upper Mississippi Valley

by Lindley B. Ballen, Bachelor of Science

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ABSTRACT

EVALUATING THE SUCCESS OF BOTTOMLAND FOREST RESTORATION IN THE UPPER MISSISSIPPI VALLEY

by

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Since European settlement, clearing for agriculture, changes in hydrology, and urbanization have reduced the coverage of bottomland forest (BLF) in the Upper Mississippi Valley (UMV) by 46%. Recently, emphasis has been placed on restoring BLF, which provides vital ecosystem services (e.g., enhanced water quality, nutrient cycling, and wildlife habitat). Beginning in 1998, the US Army Corps of Engineers has restored BLF on many sites in the UMV, ranging in area from less than 1 ha to 120 ha. Root production method (RPM®) seedlings of three species of bottomland oaks and pecan have been planted to rapidly establish large-seeded species that are not regenerating under current conditions, with the expectation that light-seeded species (e.g., silver maple, green ash, eastern cottonwood, elm) will colonize passively. A chronosequence of nine restoration sites, ranging in age from 1 to 23 yr since planting, and two mature BLF reference sites was used to assess restoration success. Five 0.1-ha circular plots were randomly located at each site. Planted trees and natural recruits with a diameter at breast height (DBH) greater than or equal to 2.5 cm were identified, tagged, and their basal diameter, DBH, and height were measured. Density of shrubs was assessed in belt transects with a total area of 100 m^2 and cover of herbaceous species was estimated in twenty 0.5 m^2 quadrats. Tree variables by species (mean basal diameter, mean height, density, and dominance) and community variables (richness and Simpson diversity of each stratum, total tree dominance, total shrub density, total herbaceous cover and the percent exotic herbaceous species cover) were calculated at the plot scale. Trajectories of change in tree size and community structure were examined using generalized linear modeling, relative to their values in reference sites. Tree height and diameter increased with time since restoration for all species. *Quercus palustris*, *Q. macrocarpa*, and *Q. bicolor* are all on track to achieve dimensions typical of mature BLF within 27 to 37 yr since planting. However, the dominance models for these trees show general declines which may indicate decreasing survivorship among planted trees and no recruitment of new seedlings. Pecan has suffered high mortality and without replanting it will be underrepresented in the restored forest. Tree dominance, richness, and diversity peaked and then decreased. Both total shrub density and total herbaceous cover showed no trend with time, although diversity of both shrubs and herbaceous vegetation slightly increased. Comparison with reference sites suggest that the shrub density, although there is no trend with time, is still in line with reference plot values. Exotic cover peaked between 10 to 15 years and began to decline to levels similar to reference plots. Overall, the results indicate some restoration success (tree growth rates, shrub diversity, herbaceous richness and diversity, and declines in exotic species cover) but suggest that replanting will be necessary in most sites to overcome mortality due to prolonged flooding and other factors (e.g. white-tailed deer browsing, inhibition of tree recruits by dense grass cover). Accurate mortality data for planted tree species is necessary to evaluate and improve the success of future USACE restorations. Ideally, a subset of trees should be tagged immediately after planting and these trees should be monitored at regular intervals. Frequently updated records will allow the USACE to make site-to-site management adjustments.

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CHAPTER I

INTRODUCTION

Problem Statement

Bottomland forest (BLF) is found on river floodplains in the central and southeastern United States. BLF provides a variety of important ecosystem services including nutrient cycling, enhanced water quality, flood control, erosion control, and wildlife habitat (Kellison & Young, 1997; King & Keeland, 1999). Since Euro-American settlement, clear cutting (for timber, riverboat fuel, and farming), changes in hydrology (due to the construction of levees and the lock and dam system), and urbanization have reduced BLF area in the Upper Mississippi Valley (UMV) from 56% to less than 30% cover of the floodplain (Nelson *et al.*, 1994).

In the early 1990s the U.S. Army Corps of Engineers (USACE) began to restore BLF on a range of sites within the UMV in Missouri and Illinois. The objective was to quickly develop suitable wildlife habitat by planting hard-mast (acorn and other nut) producing trees, which provide an important food source for many animals native to Illinois and Missouri. Root Production Method (RPM®) seedlings (Forrest Keeling Nursery, 2013) of bottomland oaks (*Quercus* spp.), pecan (*Carya illinoinensis*), and several other species were planted, with the expectation that small-seeded tree species, such as silver maple (*Acer saccharinum*), green ash (*Fraxinus pennsylvanica*), Eastern cottonwood (*Populus deltoides*), and elms (*Ulmus* spp.) would colonize passively. Relative to light-seeded species, oaks and pecan are dispersal limited (King & Keeland, 1999; Battaglia *et al.*, 2008) and altered hydrology, has prevented their natural establishment on most sites. Although all BLF tree species have a degree of flood tolerance, changes in hydrology have increased the frequency of flooding and flood level depth above the tolerance limits of some species and have made it difficult for seedlings to establish (Simmons *et al*., 2007). Given the challenges newly planted trees face it has become essential to study their progress.

Purpose of the Study

While many studies have evaluated the success of BLF restoration in the Lower Mississippi Valley (e.g., Gardiner & Oliver, 2005; King & Keeland, 1999; Twedt, 2006), few studies have examined restoration success in the UMV (though see Grossman *et al.,* 2003; Dey *et al*., 2004). My research utilized a chronosequence of nine restoration sites, ranging in age from 1 to 23 yr, to study BLF succession and evaluate restoration success. Vegetation trajectories in these nine sites was compared to two mature BLF locations.

Significance

The USACE commits significant resources to restoration projects each year, so the results of this project may be of considerable economic value in informing future restoration planning and practices. Data collected on temporal changes in vegetation structure, and composition will help discern whether restoration techniques are effective and may increase the success of future restoration projects.

Additionally, data collected in this study were used by a fellow graduate student to model changes in avian communities during BLF succession (Le, 2014). BLF located along or near migration corridors provide habitat, cover, and food for many different species of birds during seasonal migration. Bird diversity may turn out to be an excellent indicator of success in restoring BLF with a complexity of composition and structure that is comparable to reference sites.

Hypothesis

I hypothesize that restored sites are on track to approach the structure and composition of mature BLF. If so, trajectories should be in the direction of values typical of mature bottomland forest.

Predictions

1. Planted trees are growing well and survival is high (i.e. density will not decrease). Planted trees increase in height and diameter and their dominance increases with time since restoration.
2. Tree richness and diversity should increase with respect to age, especially after 11 years of age when mowing is discontinued and natural recruits are able to establish.
3. The shrub layer will increase in density but may begin to decline in relationship to canopy closure.
4. Shrub richness and diversity will increase once mowing is discontinued and shrub species are better able to establish.
5. Total herbaceous cover will decline with stand age in relation to increasing canopy cover.
6. Herbaceous richness and diversity will increase with time since restoration, especially as restoration plots become established and attract a variety of bird species. Birds act as important seed dispersers and may bring additional seed varieties as they use BLF for coverage.
7. Exotic species cover may also decline with stand age in relation to increasing canopy cover.

CHAPTER II

LITERATURE REVIEW

BLF Importance

It is estimated that 13 million ha of BLF occur in the river floodplains of the United States (Wigley & Roberts, 1997). The importance of BLF cannot be underestimated as it provides many essential ecosystem services. It provides an important habitat for many species of native fauna such as beavers, turkeys, deer, bears, small mammals, amphibians, reptiles, and many more. It is estimated that the floodplain forests of the Upper Mississippi Alluvial Valley are home to about 290 species of birds, 57 species of mammals, and 45 species of amphibians and reptiles (Nelson & Wlosinski, 1999). During the spring and fall migratory seasons, waterfowl and other birds follow waterways and many use BLF for cover, food, and breeding grounds (Knutson *et al.*, 1996). Oak, pecan, and other hard-mast producing trees provide an important, high quality food source for many including turkeys, deer, and other mammals (Kellison & Young, 1997; Leach *et al.,* 2012). Other important functions include flood control, levee protection (Allen *et al.*, 2003), contaminant and nutrient filtering, and greenhouse gas reduction (Yu *et al.*, 2008). From an economic aspect, BLF also provides an important source of timber and can be an important place for recreational activities such as hunting. These and many more are all reasons why it is essential to restore and conserve BLF.

Historical Distribution and Composition

Pre-Euro-American settlement BLF contained a mosaic of old growth forest interspersed with younger, early successional forest (Wigley & Roberts, 1997). Frequent flooding, high winds, insects, and disease helped to create open patches of canopy within older stands containing *Quercus* *spp*. and *Carya spp*. that were readily colonized by early successional species such as *Fraxinus spp*., *Populus spp*., *Acer spp*., *Salix spp*., etc. Native Americans also influenced BLF structure and distribution by frequently clearing large tracts of forest for building, farming, fuel, and hunting purposes (Gardner & Lockhart, 2007; Hamel & Buckner, 1998; Wigley & Roberts, 1997). Historically, forest composition in the Lower Mississippi Valley was most likely dominated by sweetgum (*Liquidamber stryaciflua*), hackberry (*Celtis occidentalis*), and sugarberry (*Celtis laevigata*) (Ouchley *et al*., 2000), and oak species (*Quercus spp*.) typically represented less than 50% of the total forest composition (Hanberry *et al*., 2012). Northern and Midwestern BLF contained a mix of oak, pecan (*Carya illinoinensis*), and other hickories (*Carya spp*.) along with Eastern cottonwood (*Populus deltoids*), silver maple (*Acer saccharinum*), hackberry (*Celtis occidentalis*), sycamore (*Platanus occidentalis*), and black willow (*Salix nigra*) (Shaw *et* al., 2003).

Following Euro-American settlement, BLF habitat was quickly and significantly modified. Current BLF cover in the Lower Mississippi Valley is estimated to be less than 70% of what it originally was in the late 1700s (Stanturf *et* al., 2000) and only 2.8 million ha of the original 10 million ha remains (King & Keeland, 1999). Current BLF cover in the Upper Mississippi Valley has been reduced from 56% to less than 30% cover of the floodplain (Nelson *et al*., 1994). The most extensive changes began in the early 1800s as settlers began to convert productive alluvial soils to agriculture (Stanturf *et al*., 2000). Highly productive floodplain forests were converted to agricultural cropland. Lower lying floodplain forests that were more flood-prone were logged for timber and steamboat fuel (Nelson *et al*., 1994). Additional losses can be attributed to the Swamp Land Acts of 1849-1850 which allowed federally owned swamplands, including BLF, to be filled and drained (Stanturf *et al*., 2000). Additionally, the construction of locks, dams, and levees have permanently changed the distribution and composition of floodplain forests

Hydrology

Hydrology is considered by many to be one of the most dominant factors influencing the distribution, composition, and productivity of BLF tree species (Nelson *et al.*, 1994). Periodic, seasonal flooding of BLF brings additional nutrients and moisture (Anderson & Mitsch, 2008; Wigley & Roberts, 1997). Flooding can also decrease the oxygen content in soil and can contribute to reduced growth rates and tree mortality if flooding is too intense and too lengthy.

Changes in hydrology began in the early 1700s, soon after the founding of New Orleans. The French colonial government required landowners install levees within one year of settling. Levees, which are installed adjacent to or near a river, lake or sea, are earthen walls that regulate water levels and protect urban and agricultural areas from flooding. By 1735, levees stretched along both sides of the river within the New Orleans territory (Baker *et al.*, 1999). From 1929 to 1942 the lower Mississippi River was modified by the installation of 16 bendway cutoffs (to straighten the channel) and continuous levees (Schramm *et al.*, 2009). To date, the Mississippi levee system covers over 5630 km (USACE, 2014).

Increases in river traffic brought on the need for additional and lasting changes. The Mississippi, Illinois, and Missouri Rivers have long been a major route for commercial transportation and Americans found it necessary to better control navigation by modifying river flow and channel depth. As early as 1879, Congress established the Mississippi River Commission whose purpose was to create a plan for river modification that would deepen the navigation channel and reduce flooding. In 1928, the Mississippi River and Tributaries Project was established by congress following the most disastrous Mississippi River flood ever recorded in 1927 (Baker *et al.*, 1999). Its purpose was to outline a comprehensive plan for river modification and it is primarily responsible for present changes in hydrology (Baker *et al.,* 1999). In the 1930s a series of locks and dams were built between St. Louis, Missouri and St. Paul-Minneapolis, Minnesota and along the Illinois River all the way up to Chicago, Illinois (Nelson *et al.*, 1994). Then, in the early 1990s, another series of locks and dams were installed along the Illinois, Missouri, and Mississippi Rivers to control river depth and improve navigation (Dey *et al.*, 2000).

These permanent changes in hydrology had serious implications for BLF. Low-lying floodplain forests immediately adjacent to rivers with impoundments were permanently submerged (Nelson *et al.*, 1994). These forests were eventually replaced with aquatic habitats. BLF located on the outboard side of a levee experience higher levels of flooding and more frequent flooding events. BLF located on the inboard side of a levee would experience less frequent inundation. The construction of locks, dams, and levees has reduced the flood plain width by restricted flooding to 10% of its former floodplain (Schramm *et al.*, 2009). Additionally, these changes have shortened the spring flood pulse from 5 months to an average of two months (March to May) (Schramm, 2009). The result is increased frequency of flooding and flood level depth above the tolerance limits of some species and a permanently changed disturbance regime for BLF. This has led to shifts in forest composition to more flood tolerant species and increases in mortality of less flood tolerant species (Simmons *et al.*, 2007). Alternatively, Gee *et al.* (2014) found that species composition along the inboard side of the levee (less frequently flooded) shifted from flood-tolerant oak to flood-intolerant sugarberry (*Celtis laevigata*). Increased flood level depth also causes high seedling mortality and may prevent the replacement of fallen trees. In addition, altered flooding regimes may change dispersal patterns for species that rely on water transport as a means of dispersal. These trends may become amplified when coupled with climate change.

Total hydraulic restoration is not a practical or an economically realistic goal. Anthropogenic changes to the flooding regime have strong implications for land management and conservation. Land managers must take this into account during site selection and restoration planning, as some tree species are better suited to wetter habitats. Additionally, depending on the location, land managers may choose different planting methods and/or site preparation and management techniques. For example, direct seeding may be less effective than planting saplings since taller trees may have an initial advantage in areas prone to frequent flooding. Site preparation and management techniques that may lead to enhanced survival of planted trees include soil bedding or berm planting (Kabrick *et al.*, 2005), species selection based on site topography (Gardiner *et al.*, 2004), or interplanting with early successional trees like cottonwoods (Grossman *et al.*, 2003; Stanturf *et al.*, 2000; Twedt, 2006).

The present focus is placed on restoring function and structure of BLF. Land managers frequently opt to plant hard-mast producing species such as oaks, pecan, and hickory in greater proportions than they were historically. Oaks and other hard mast producing species are typically favored because they promote native biodiversity by providing food for many types of fauna, and they create a habitat that is favorable for waterfowl, deer, and turkey hunting. Additionally, hard-mast species are planted at higher densities in order to compensate for poor natural regeneration.

Improving Restoration Success

There are many studies that have evaluated restoration success. Many of these studies focus on restoration projects in the Lower Mississippi Alluvial Valley and few studies have gauged the success of BLF restoration in the Upper Mississippi Alluvial Valley. Some research topics include how to recruit target species, survival rates of planted trees, control of herbaceous and woody competition, the effects of deer browsing, the best site preparation methods, and interplanting.

Management entities are usually concerned with increasing recruitment of target plant and tree species which in turn increases the fauna community richness and diversity. Several studies suggest that restoration sites need to be within 100 m of an existing seed source (Allen, 1997; Twedt, 2006). The best way to accomplish this goal is to restore fields that are adjacent to quality forest stands when possible. If restoration near quality forest is not possible, then quick development of vertical structure can be used to attract bird visitation and increase seed deposition (Twedt *et al.*, 2002; Twedt *et al.*, 2010). Interplanting fast growing tree species, such as cottonwood and ash, can also be used as a method to quickly increase vertical structure (Twedt, 2006).

Survival of planted seeds and seedlings is probably of greatest concern to land managers. Low survival is not only economically costly but also counterproductive to restoration goals. Land managers typically overplant to account for seedling mortality. Survival rates that are less than 75% are economically undesirable (Ezell *et al.*, 2007).

Many studies have looked at ways to enhance seedling survival and there are many reasons why seedling survival can be reduced. Some studies have looked at differences in site preparation methods, reducing herbivory, and the control of herbaceous and woody competition.

Site preparation can enhance survival by creating suitable growing conditions. Disking is a common practice that improves soil structure, reduces compaction, and reduces existing plant cover (Allen *et al.*, 2004; *Shaw et al.*, 2003).

Seedling survival can be improved by controlling herbaceous and shrubby competition. Initially, weed control can be accomplished by disking a site. Additional follow-up weed control can be done by mowing or in some cases with herbicides (Ezell *et al.*, 2014; Grebner *et al.*, 2004). Mounding or berm planting is another method for enhancing seedling survival. Shaw *et al.* (2003) found that mounding helped to decrease soil water content and drained more quickly than non-mounded soil. Planting seedlings on slightly elevated topography may give young trees a slight advantage during moderate flooding. Many land managers also opt to install devices that reduce herbivory on planted seedlings. Deer browsing and rubbing can have a substantial effect on seedling mortality and deer guards do provide some protection (Ruzicka *et al.*, 2010). McGuire

(2014) found that trees that were protected with wire mesh deer guards had significantly higher basal diameter MRGR than trees without guards. However, the protection that guards provide may be limited and ecosystem recovery from browsing effects may be slow. Tanentzap *et al.* (2009) looked at a forest in New Zealand and found that even with population control and the addition of enclosures, vegetation recovery was slow.

Planting Techniques

Several planting techniques, with varying levels of effort and benefit, have been employed by land managers to convert unproductive farmland to BLF. The method used by restoration managers will depend on the outcomes desired and the available resources. Passive restoration, which requires the least amount of effort, allows an abandoned field to develop into a forest through dispersal from nearby trees. Although this method requires the least amount of attention there can be significant trade-offs. Tree recruits are limited to those that are within dispersal distance and dispersal is typically limited to distances within 100 m (Stanturf *et al.*, 2000). Additionally, tree diversity is often low since light-seeded species that are bird-dispersed or wind-dispersed tend to dominate over large-seeded species, which are dispersed by flooding or mammals (Battaglia *et al.*, 2008; Stanturf *et al.*, 2009).

An alternative method involves the direct seeding of oaks and other large seeded species at regular intervals. Seeding density generally ranges from 2,470 to 3,700 seeds per ha to account for high seed predation (Gardiner & Oliver, 2005). Oak species are typically favored for restoration planting because they provide food for wildlife and are dispersal limited (Battaglia *et al.*, 2008; King and Keeland, 1999; Ouchley *et al.*, 2000). Direct seeding requires a moderate amount of effort and is cost-effective. The technique has experienced differential rates of success and factors such as flooding, seed predation of planted caches, and herbivory of young shoots have reduced seedling growth and the success of direct seeded sites (Gardiner *et al*., 2004; Stanturf *et al*., 2009).

A third, more costly method involves planting bare root, ball and burlap, or RPM® seedlings. Bare root trees are grown in the ground and the soil is removed when the trees are dug out of the ground. Ball and burlap trees are also grown in the ground but their root mass with intact soil is wrapped in burlap before it is transported. RPM® seedlings are grown in mesh containers that encourage the growth of a fibrous root system. Seedlings are usually planted at a density of 746 seedlings per ha (Gardiner & Oliver, 2005). This technique requires the most effort and can be costly but it is typically the most successful of all the restoration techniques. Reforestation with RPM® seedlings may be the most effective, although the most costly, as these trees seem to exhibit high rates of survival and early acorn production (Dey *et al*., 2004; Grossman *et al*., 2003).

Regardless of the planting method, follow up maintenance and monitoring are necessary at most restored sites. Significant resources are committed to restoration projects each year and oftentimes funds only cover the cost of site preparation and planting. Limited or no monitoring is done on most restored sites, despite the fact that the trees on newly restored sites face a variety of challenges, such as flooding, deer browsing or rubbing, wind and storms, and competition from invasive species. It is essential to monitor these sites to ensure that a diverse mix of species is maintained and the majority of trees that are planted survive. Frequent monitoring is the best way to know if restoration techniques are effective.

Chronosequences

This study utilizes a chronosequence to evaluate restoration success. Chronosequences are used to study succession by using a series of similar plots with different ages, assuming that the variation observed among plots represents the sequence of changes that would occur on a single plot over time. Chronosequences provide a method for studying and modeling the progress of restorations and may be an effective research technique for evaluating restoration success. They have been used to evaluate changes in soil composition, the distribution of insects and birds in relation to stand age, and vegetation succession and health by using similarly managed sites that have been restored at different periods of time (Walker *et al.*, 2010; Pawson *et al.*, 2009; Vina, 2012). The use of chronosequences involves several assumptions that must be met in order for results to be valid. When selecting sites, care should be taken that they have been planted with a similar mix of species, have been managed in comparable ways and have experienced the same abiotic and biotic factors over time (Walker *et al.*, 2010; Uren & Parsons, 2013).

CHAPTER III

METHODS

Study Area

From a database of more than 60 USACE restoration sites in the UMV, nine restoration sites, ranging in age from 1 to 23 years, were selected that had been planted with a similar mix of tree species and managed in a similar manner (Figure 1, Table 1). Sites needed to be at least 3.4 ha in area, in order to avoid edge effects and fit five circular 0.1-ha plots that were used for vegetation surveys.

All sites were mowed and then disked prior to tree planting. Some sites had berms created (RL, CC, PS, CI, and BH). All sites were planted with saplings of *Quercus palustris* (pin oak), *Quercus macrocarpa* (bur oak), *Quercus bicolor* (swamp white oak), and *Carya illinoinensis* (pecan) but plantings may have also included saplings such as *Diospyros virginiana* (persimmon), *Celtis occidentalis* (hackberry), *Juglans nigra* (black walnut), and a variety of other oak species. *Q. texana* (Nutall's oak) was also planted at Polhman Slough. *Q. texana* is not native to Illinois and is indistinguishable from *Q. palustris* unless acorns are present. During data collection both were lumped under *Q. palustris*. Trees were planted in rows with a spacing of 6 x 6 m or 9 x 9 m. Cloth mats measuring 1x1 m were also placed at the base of trees to reduce competition until the trees became established. Plastic deer guards were installed around the base of each trunk to protect against damage from deer rubbing. On most sites, a ground cover of grass (usually *Elymus virginicus* and *Agrostis gigantea*) was planted to inhibit the establishment of small-seeded trees. Planting grass around tree seedlings reduces the probability that wind-dispersed trees will establish before planted trees have a chance to become well established. Any wind-dispersed seeds that land within the grass cover will be less likely to out-compete the tall grass. Additionally, all sites are mowed between the rows of planted trees either annually or biannually until they are 11 years old. Mowing further reduces competition from small-seeded tree recruits and shrubs. Lastly, except for Brickhouse and American Bottoms, which were planted with ball and burlap seedlings, the restorations used 3 gallon container Root Production Method (RPM®) seedlings from Forrest Keeling Nursery. RPM® seedlings are produced using a special air root pruning process that encourages a large, dense root mass, rapid seedling growth, and early mast production (Grossman *et al*., 2003; Lovelace 1998).

Two mature BLF reference sites, one at American Bottoms and one at Rip Rap Landing, were selected as long-term targets to evaluate restoration success. Aerial photographs, from the Illinois Geospatial Data Clearinghouse, taken in the 1940s were visually analyzed to determine if both mature sites had BLF coverage (Figure 2 and 3). Site reconnaissance data were collected at Rip Rap Landing by Meghan Romano through the National Great Rivers Research and Education Center (NGRREC) summer internship program. Both sites contain oak and pecan trees that are estimated to be at least 70 years old. Very few mature BLF sites exist today that contain a diverse mix of large and light-seeded species and are of suitable size.

Table 1: Site List. Restoration and reference site list with abbreviations that are used throughout this thesis. Site age referes to the number of years since planting at the time data were collected.

Number	**Site Name**	**Abbreviation**	**Age (years)**	**Hectares**	**State**
1	Crater's Landing	CL	1	3.61	IL
2	Epping	EP	3	34.13	IL
3	Red's Landing	RL	5	7.02	IL
4	Calumet Creek	CC	7	7.08	MO
5	Chain of Rocks 3B	COR	11	7.03	IL
6	Pohlman Slough	PS	11	30.77	IL
7	Cuivre Island	CI	18	10.58	MO
8	Brickhouse	BH	23	16.06	MO
9	American Bottoms	AB	23	3.42	IL
10	American Bottoms Reference	ABR	70+	36.22	IL
11	Rip Rap Landing Reference	RRR	70+	115	IL

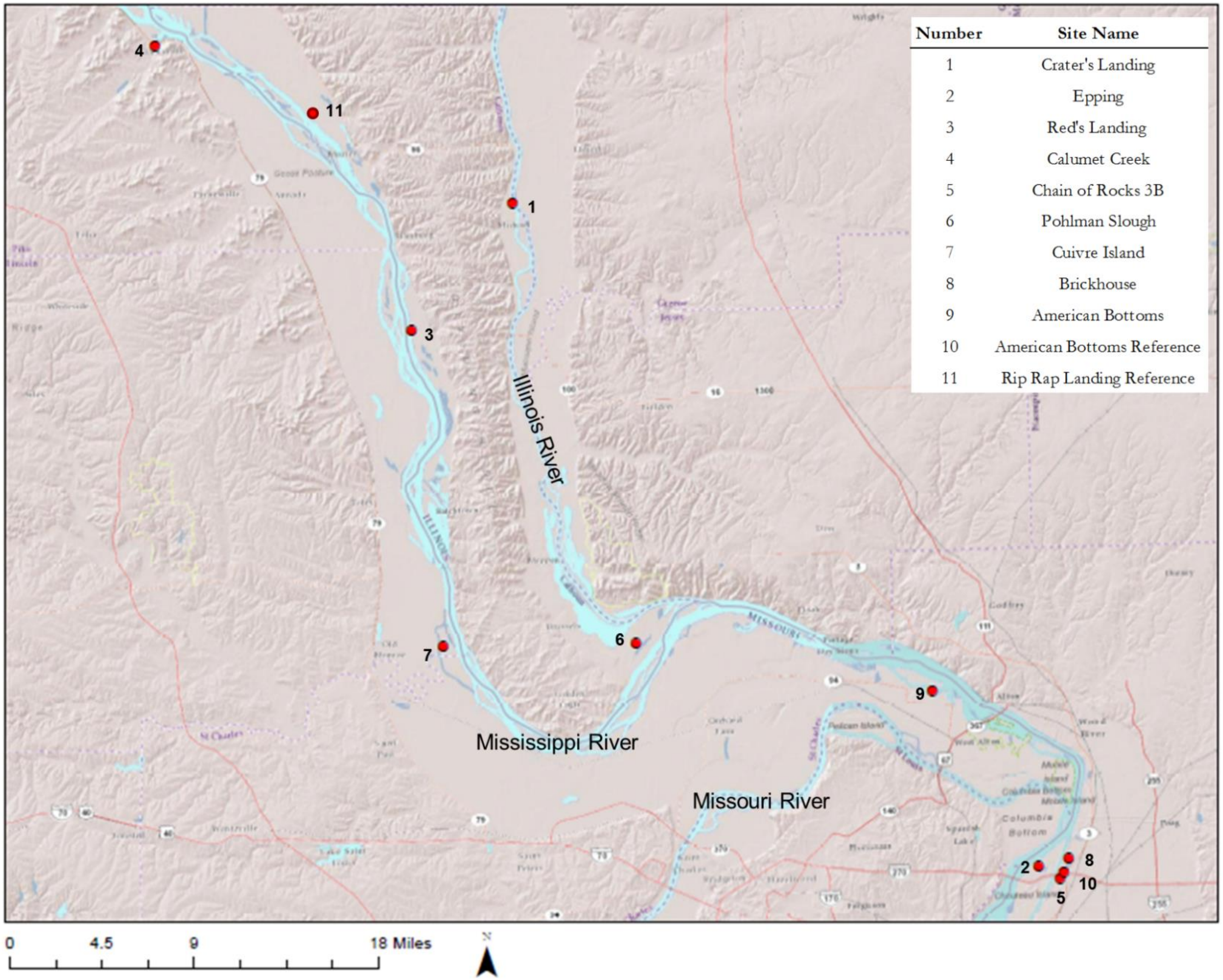

Figure 1: Map of study sites. Selected chronosequence of nine USACE restoration sites and two mature BLF reference sites in the floodplains of the Mississippi and Illinois Rivers. See legend for site data.

Figure 2: Aerial photograph of American Bottoms reference site. Photograph was taken August 31, 1941 of American Bottoms Reference (outlined in white).

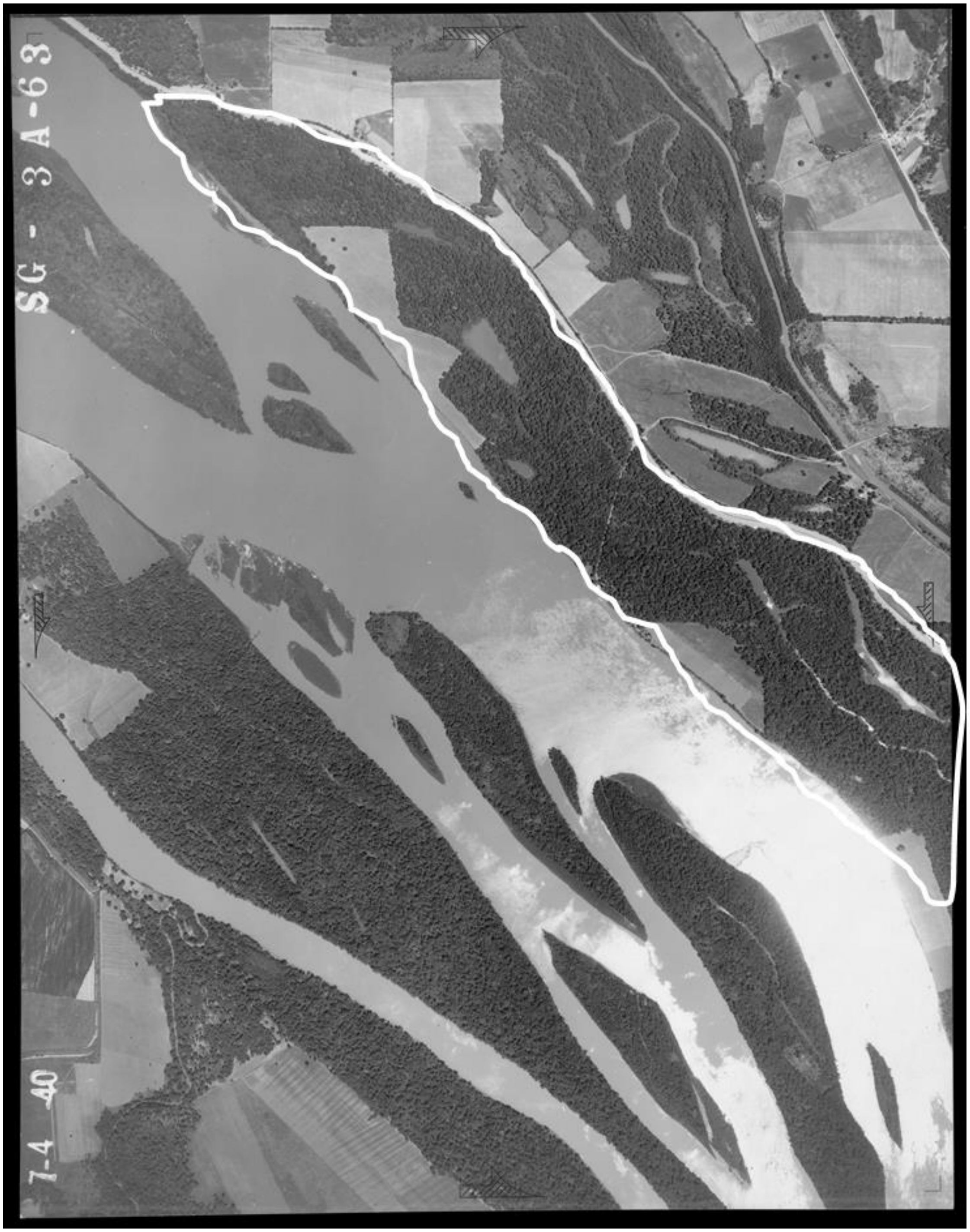

Figure 3: Aerial photograph of Rip Rap Landing reference site. Photograph was taken July 4, 1940 of Rip Rap Landing Reference (outlined in white).

All study sites were located adjacent to either the Mississippi or Illinois Rivers. The regional climate is continental, characterized by cold winters, a mild spring and fall, and hot summers. The mean winter temperature is 0.9°C, 13.1°C in spring, 25.2°C in summer, and 14.5°C in fall. Heavy rains and flooding typically occur during the spring and fall season. Mean annual precipitation is 96.1 cm and the total precipitation for 2013 was 108 cm. May and June were the wettest months for 2013.

Sampling Design

At each site, five 0.1-ha circular plots were established for vegetation surveys and for future monitoring (Figure 4). Except for Rip Rap Landing, all plot locations were randomly determined using ArcGIS Desktop 10 (ESRI 2010). The parameters for plot selection included placing a 30 m buffer zone around the edge of each site polygon to avoid edge-effects and a minimum of 50 m distance between plot centers. The plots at Rip Rap Landing were chosen based on previous reconnaissance data by Meghan Romano (2007). Four of the five plots had been previously marked with steel T-posts and the fifth plot was chosen by selecting a location from the reconnaissance dataset that contained mature oaks. This was done to ensure that the chosen sampling area only represented mature forest.

In the field, a GPS receiver (Garmin GPSMAP 62S) was used to locate the center of each plot. Each plot was permanently marked using a 1.8 m steel T-post and the center coordinate was accurately recorded. The top of the post was tagged with the plot number. Because some sites are mowed once a year, USACE requested that all posts be placed within a line of planted trees. When a coordinate fell between two tree lines a coin was tossed to decide which tree line the T-post should be placed in. UTM Coordinates for the center of each plot are listed in Appendix A.

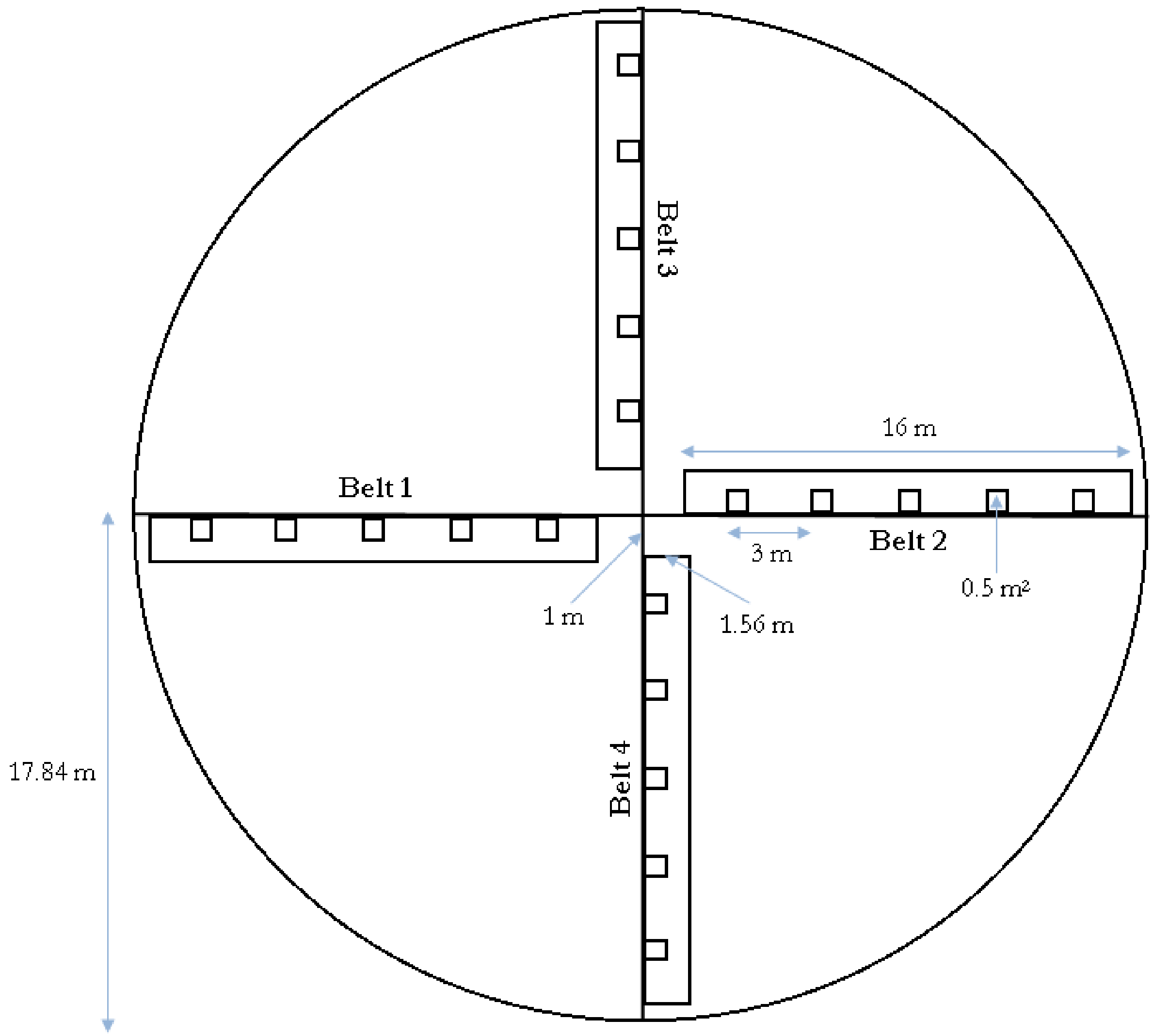

Figure 4: Sampling plot design. All trees within the 0.1 ha sampling plot were tagged. Four 17.84-m transects were set at right angles. Five 0.5-m^2 quadrats were placed along each transect for visual estimation of herbaceous species cover and measurement of litter depth and vegetation height. Shrubs ≥1 m high were tallied in each of the 16 x 1.56 m belts.

Procedures for Data Collection

Vegetation data were collected from all plots beginning May 2013 and ending September 2013. From the center of each plot a transect tape was used to measure a radius of 17.84 m (Figure 4). A total of four transects were set at right angles from each other. When possible, two of the four transects were placed along the tree line. Otherwise, transects were laid out in cardinal directions. Five quadrats of 0.5 m^2 were completed along the radius of each transect with the first quadrat placed 2 m from the center of the plot and subsequent quadrats placed at 3 m intervals. A PVC pipe square measuring 0.707 x 0.707 m was used to help delineate the quadrat. Within each quadrat, percent cover of plant species was visually estimated using modified Braun-Blanquet classes (Van der Maarel, 1975) (Table 2). Maximum litter depth and maximum herbaceous and shrub height were also recorded in centimeters within each quadrat. The total area sampled within each plot for herbaceous was 10 m^2.

Additionally, four belts that begin 1 m from the center of the plot were used to obtain stem counts for shrubs 1 m or greater in height (Figure 4). Each shrub stem that fell within the 1.56 x 16 m belt was tailed. Any tree recruits at least 1 m tall but with a DBH less than 2.5 cm diameter (measured 140 cm above ground surface) within this belt were included in the tally. These trees were not tagged. The total area sampled for shrubs within each plot was 100 m^2.

Tree tagging and identification began in August and ended early October 2013. Within each 0.1-ha plot, all planted trees and natural recruits taller than 1 m with a basal diameter of at least 2.5 cm were tagged with numbered aluminum tags to allow future observations of growth and mortality. In most cases, trees were identified to the species level, although some trees were only identified to the genus level. Tree diameter and height were measured at the end of the growing season starting in November 2013 and ending March 2014. Trees with a diameter at breast height (DBH; measured 1.4 m above ground level) of at least 5 cm had their tags nailed with aluminum nails 1.5 m above ground level. Trees with a DBH less than 5 cm were tagged using a loop of 20 gauge galvanized steel wire around the base of the tree. To avoid any risk of girdling the tree, the loop was 30 cm in diameter and the ends were hooked together but not twisted.

Diameter was measured using a diameter tape for all tagged trees within the plot. Basal diameter was recorded for all trees with a DBH less than 2.5 cm. Trees with a DBH greater than or equal to 2.5 cm had both their basal diameter and DBH measured and recorded. Basal diameter was measured 30 cm from the ground and DBH was measured 140 cm from the ground. Tree height was measured using a telescopic surveying target pole for trees that are 3 m tall or less. An OPTi-LOGIC Laser RangeFinder/Hypsometer 400LH was used to measure height for trees that were taller than 3 m.

Table 2: Modified Braun-Blanquet classes. These were used for visually estimating herbaceous cover.

Cover Class	Percent Cover Limits (%)	Midpoint Cover (%)
0	0 (absent)	0
1	< 1	0.5
2	1 – 4.99	3
3	5 – 24.99	15
4	25 – 49.99	37.5
5	50 – 74.99	62.5
6	75 – 94.99	85
7	95 – 98.99	97
8	≥ 99	99.5

Data and Statistical Analyses

All data were entered into Microsoft Excel 2010 and organized using pivot tables. Data were pooled at the plot level for each site and was either averaged or summed depending on the variable to be calculated. Tree variables by species and community variables by stratum were calculated from pivot table summaries. Mean basal diameter (cm), mean height (m), density (trees/ha), and dominance (m^2/ha) were calculated for *Q. palustris, Q. macrocarpa*, and *Q. bicolor*. Richness was calculated for each stratum (tree, shrub, and herbaceous) by counting the number of species present. Simpson's diversity was also calculated for each stratum using the equation:

$$Simpson's\ diversity = \frac{1}{\sum_{i=1}^{s} p_i^2}$$

Where s is the number of species and p_i is the proportional abundance of species i. Proportional abundance is the abundance expressed as a proportion of the total abundance.

Total tree dominance (m^2/ha) for all tagged trees and total shrub density (stems/ha) were calculated. Tree saplings (not tagged) that were greater than 1 m tall but with a DBH less than 2.5 cm were also included in shrub analyses. Total herbaceous cover (%) and proportion of exotic herbaceous species cover (%) were also calculated.

Time trajectories of tree and vegetation variables were analyzed using Generalized Linear Models (GLM) in SAS 9.3. GLMs are a type of regression that allows the response variables to have a distribution model other than normal. All GLMs used a log link function. A quasi-Poisson error distribution was used for all models except the density models for *Quercus palustris, Q. macrocarpa*, and *Q. bicolor* which used a negative binomial error distribution. The log link function allows the mean of the dependent variable to be related to the linear term ($a_1 Age$). GLMs were created using both age and age squared.

$$\ln(y) = a_0 + a_1 Age + a_2 Age^2 + \varepsilon$$

Where y is the variable being modeled, Age is the time since restoration, a_0, a_1, and a_2 are fitted parameters, and ε is a random error with a quasi-Poisson distribution (for which the variance is proportional to the mean) or a negative binomial distribution.

Both models were compared and the model with the lowest Akaike information criterion (AIC) and best fit statistics was selected (excluding the models for height and basal diameter for planted tree species). The density model for *Q. macrocarpa* was very close to significant, however, its use was justified on the basis that since basal diameter is increasing and dominance is decreasing, density must logically be decreasing.

The models for planted tree species height and basal diameter were selected differently. The parameters of age and age squared were both run and the fit statistics were analyzed, however, only the model using age was used since these models showed growth as increasing and not decreasing. Several of the models for height and basal diameter predicted that growth was negative and while the AIC value may have been better, the model was not biologically plausible.

Each response variable from the GLM was plotted against age using SigmaPlot 11.0. Lower and upper confidence intervals were plotted along with the predicted values. Mean reference site data were plotted with standard error bars.

The proportional change in height and basal diameter per yr was calculated. Since all the GLMs for planted tree height and basal diameter fit best using only the parameter of age, this was calculated by taking the exponent of a_1 from the GLM.

It was important to know when planted trees are expected to reach heights and basal diameters similar to those encountered at reference sites. To calculate this the equation from the GLM for height and basal diameter was rewritten to solve for age.

$$Age = \frac{\ln(d) - a_0}{a_1}$$

Where d is the mean height or mean basal diameter at the reference site, a_0 is the intercept from the GLM, and a_1 is the slope from the GLM.

CHAPTER IV

RESULTS

A summary of modeling results for each variable are presented in Appendix B. A total of 1160 trees were tagged and measured and twenty-four different tree species were found within the 55 sampling plots (Appendix C). Six different shrub species were found and tallied within the 55 sampling plots (Appendix D). A total of 91 different herbaceous plant species were found within the 55 sampling plots (Appendix E). Eighteen out of the 91 herbaceous species were exotic.

Planted Tree Basal Diameter

Q. palustris had a proportional change in basal diameter of 1.10/yr (Figure 5). *Q. macrocarpa* had a proportional change in basal diameter of 1.08/yr (Figure 6). *Q. bicolor* had a proportional change in basal diameter of 1.12/yr (Figure 7). Planted trees will attain similar diameters to reference site means in 27-38 years since planting (Table 3).

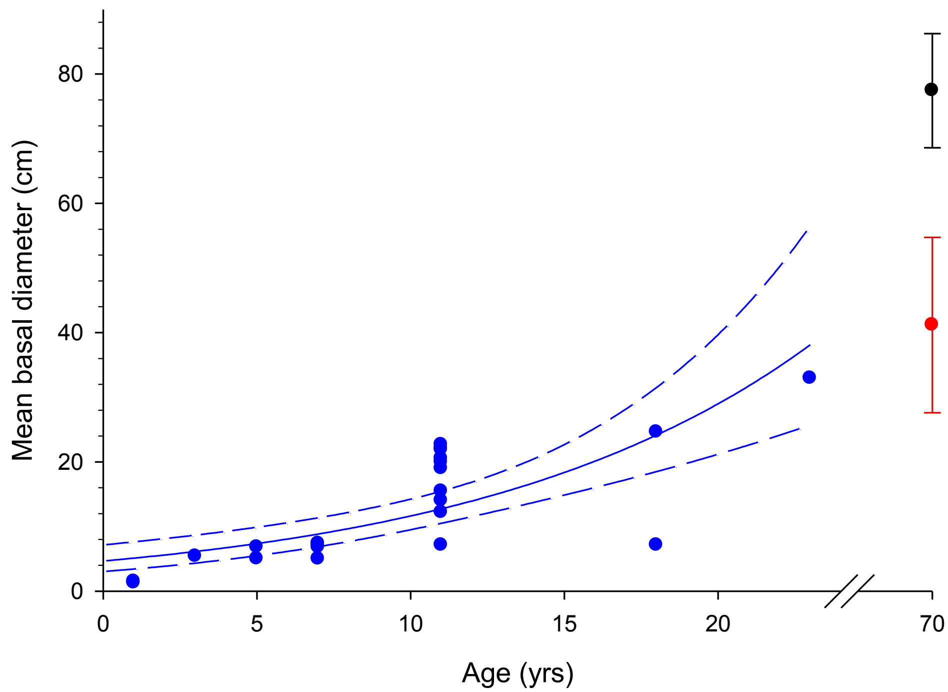

Figure 5: *Quercus palustris* basal diameter against age. GLMs used a log link function and a quasi-Poisson error distribution. Lower and upper confidence intervals were plotted along with the GLM predictor model (blue lines). The mean and standard error were plotted for reference data (ABR in red, RRR in black). Restoration plots are indicated by the blue points.

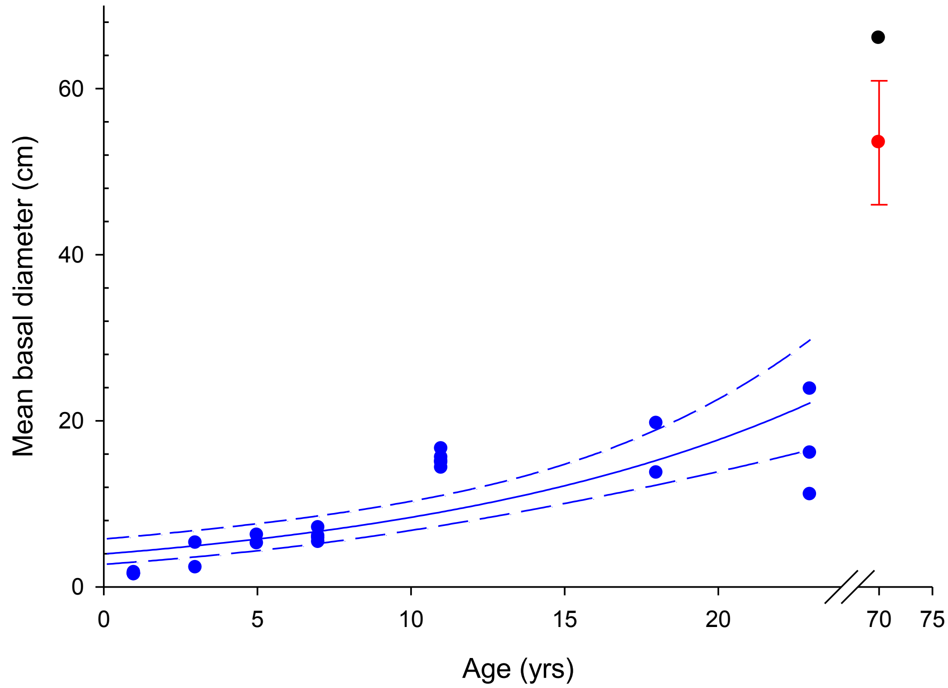

Figure 6: *Quercus macrocarpa* basal diameter against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5. Error bars and mean were not calculated for RRR since only one tree was found.

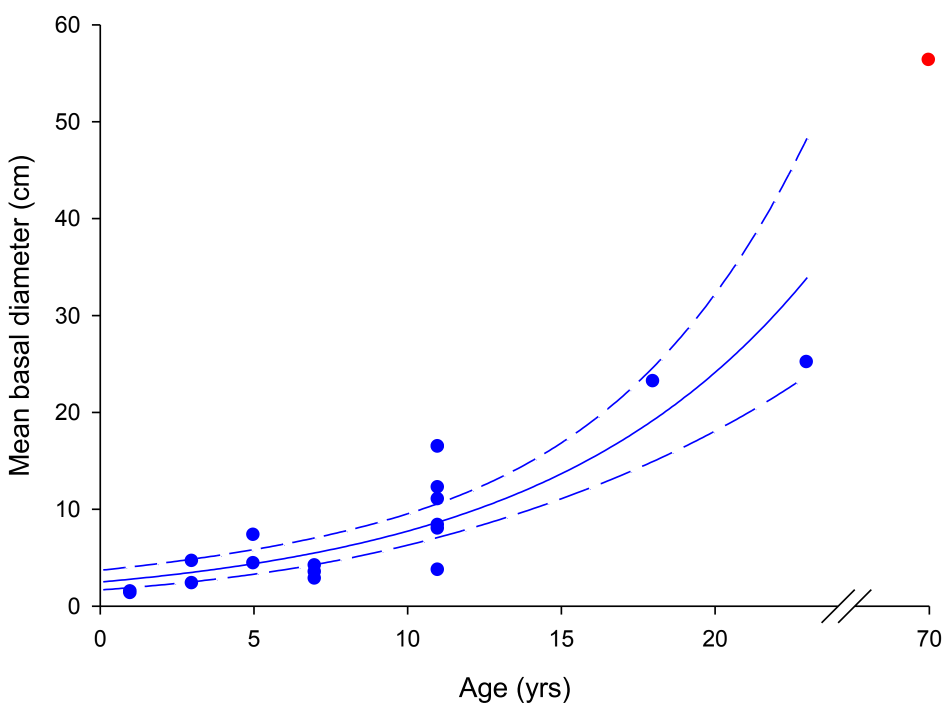

Figure 7: *Quercus bicolor* basal diameter against age. GLMs used a log link function and a quasi-Poisson error distribution. Confidence intervals are as in Figure 5. *Q. bicolor* was only found in one plot for ABR only.

Table 3: Proportional change in basal diameter per yr. This was calculated for each planted species. The number of years until planted trees reach basal diameters similar to ABR and RRR was also calculated.

Species	Proportional Change in Size per yr	Mean BD (cm) ABR	Mean BD (cm) RRR	Years until ABR	Years until RRR
Quercus palustris	1.10	63.09	77.43	29	31
Quercus macrocarpa	1.08	53.47	66.03	35	38
Quercus bicolor	1.12	56.30	-	27	-

Planted Tree Height

Quercus palustris has a proportional change in height of 1.08/yr for height (Figure 8). *Quercus macrocarpa* had a proportional change in height of 1.07/yr (Figure 9) and *Quercus bicolor* had a proportional change in height of 1.08/yr (Figure 10). Growth rates for *Carya illinoinensis* (one of the RPM® trees planted by USACE) could not be calculated because few planted trees of this species occurred in our plots. Planted trees will reach heights similar to reference site means in 31 to 34 years since planting (Table 4).

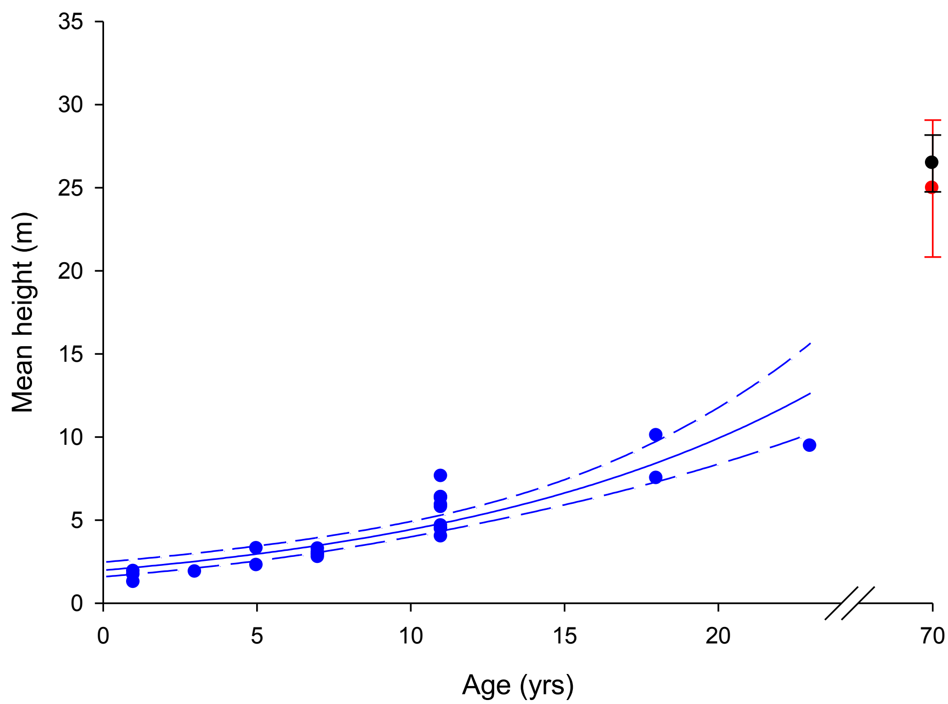

Figure 8: *Quercus palustris* height against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

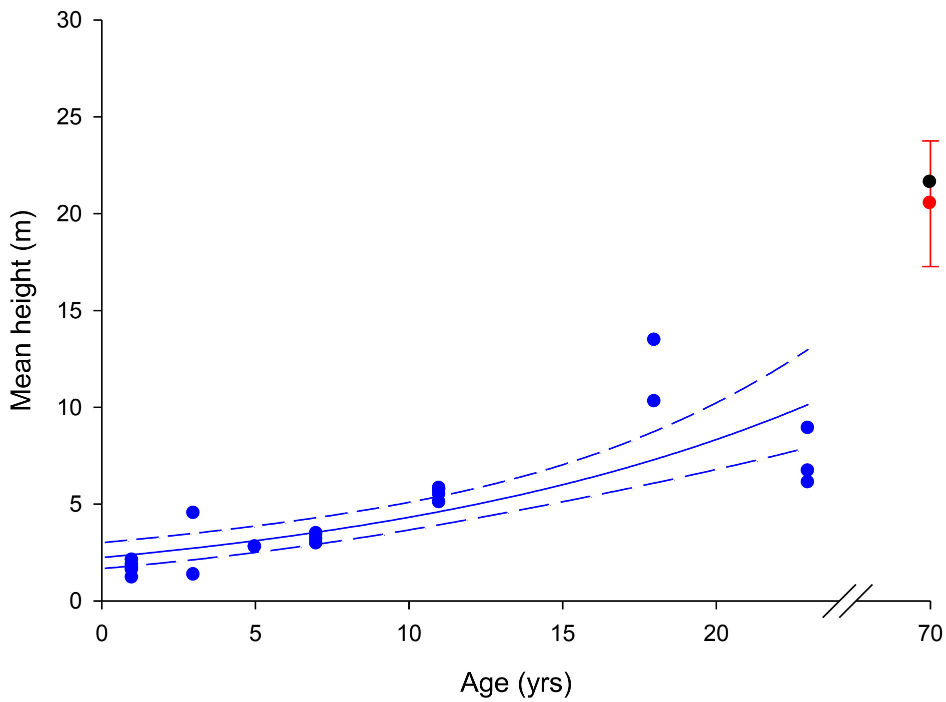

Figure 9: *Quercus macrocarpa* height against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

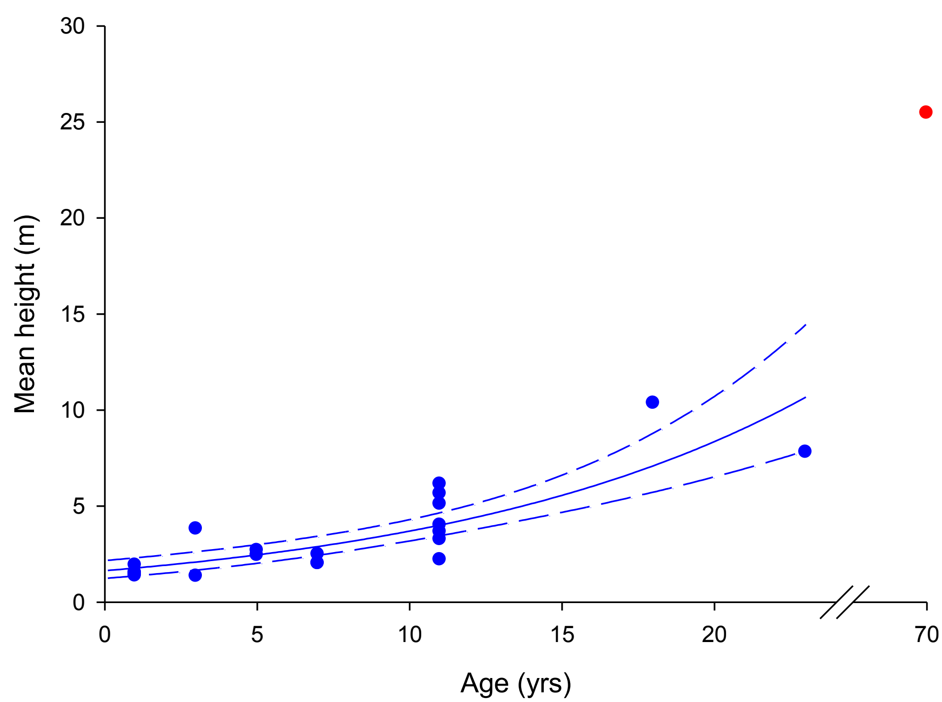

Figure 10: *Quercus bicolor* height against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Table 4: The proportional change in height per yr. This was calculated for each planted species. The number of years until planted trees reach heights similar to ABR and RRR was also calculated.

Species	**Proportional Change in Size per yr**	**Mean Height (m) ABR**	**Mean Height (m) RRR**	**Years until ABR**	**Years until RRR**
Quercus palustris	1.08	24.95	26.46	31	32
Quercus macrocarpa	1.07	20.51	21.60	34	34
Quercus bicolor	1.08	24.45	-	33	-

Planted Tree Density

Quercus palustris density increased, peaked between 10 to 15 years, and then began to decline (Figure 11). The density for *Q. macrocarpa* (Figure 12) and *Q. bicolor* (Figure 13) have descending models.

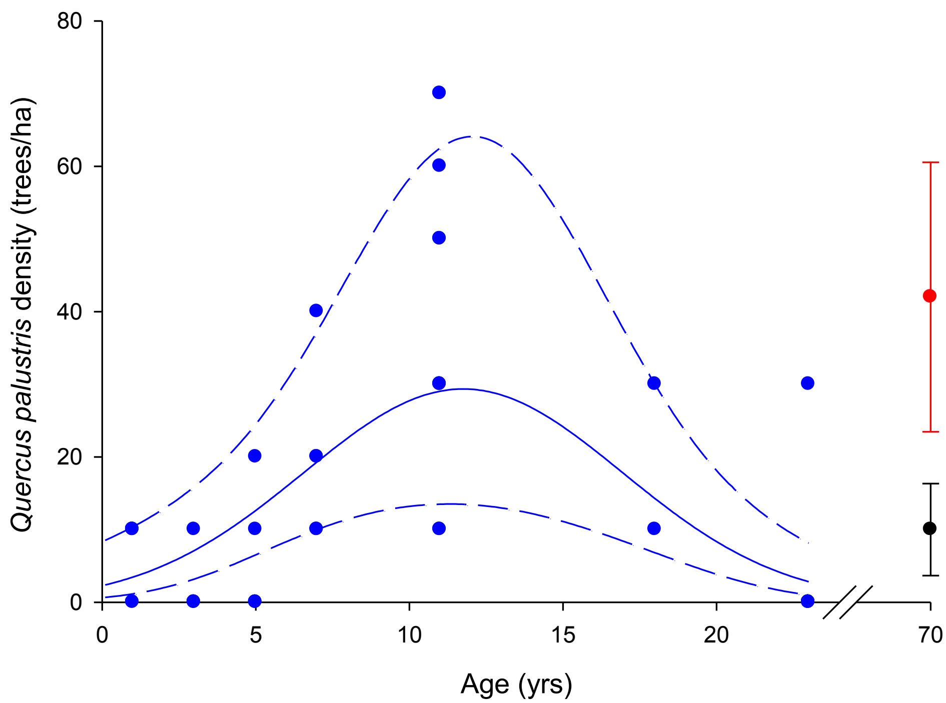

Figure 11: *Quercus palustris* density against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

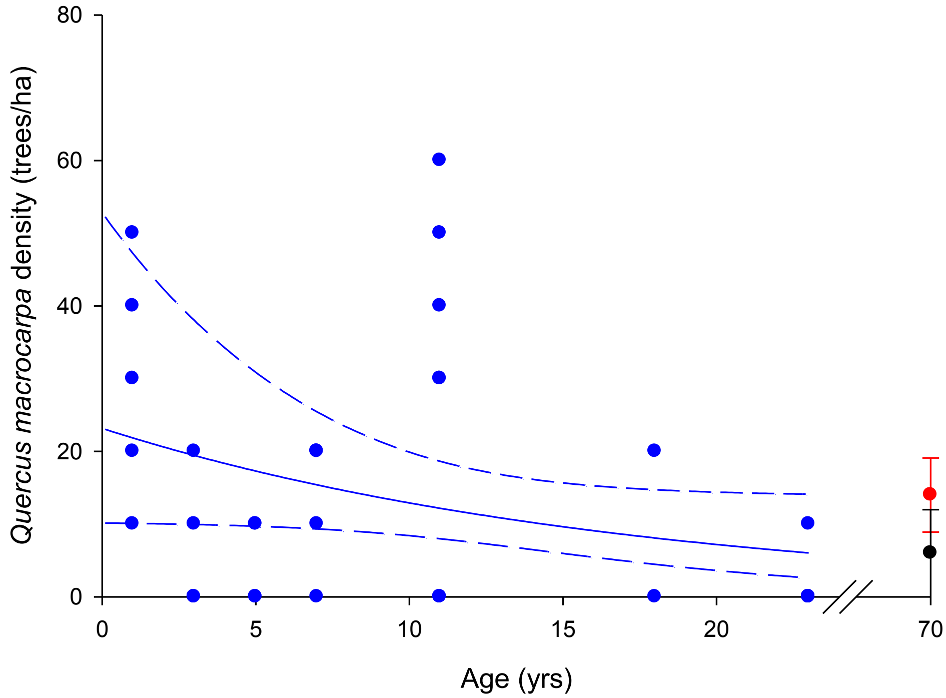

Figure 12: *Quercus macrocarpa* density against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

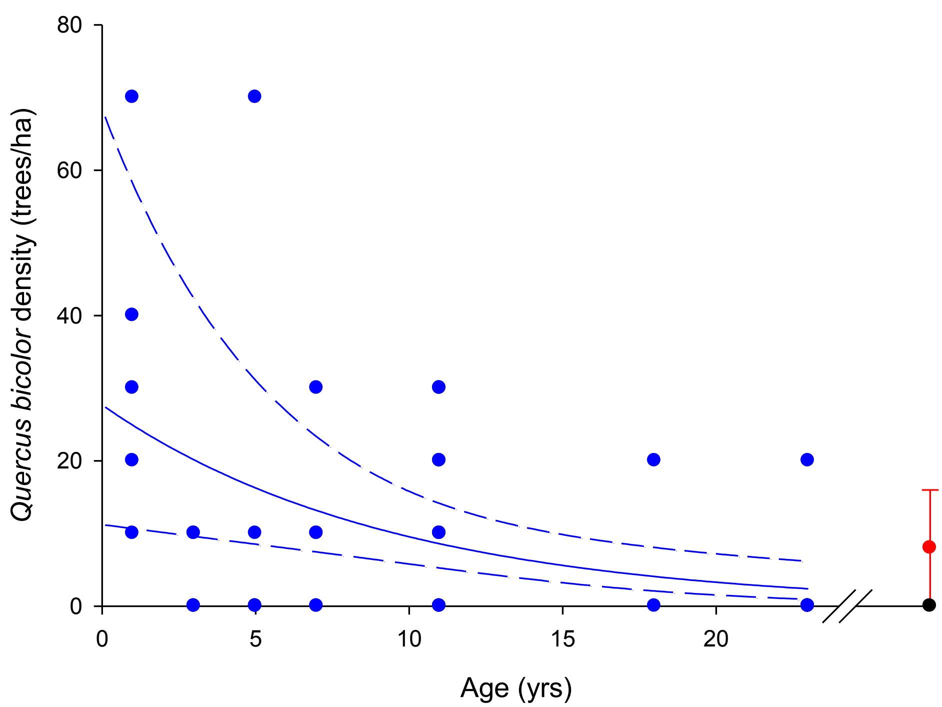

Figure 13: *Quercus bicolor* density against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Planted Tree Dominance

Q. palustris dominance peaked at 15 years and decreased (Figure 14). *Q. macrocarpa* peaked at 15 years and then began to decline (Figure 15). *Q. bicolor* peaked just after 15 years and then began to decline (Figure 16).

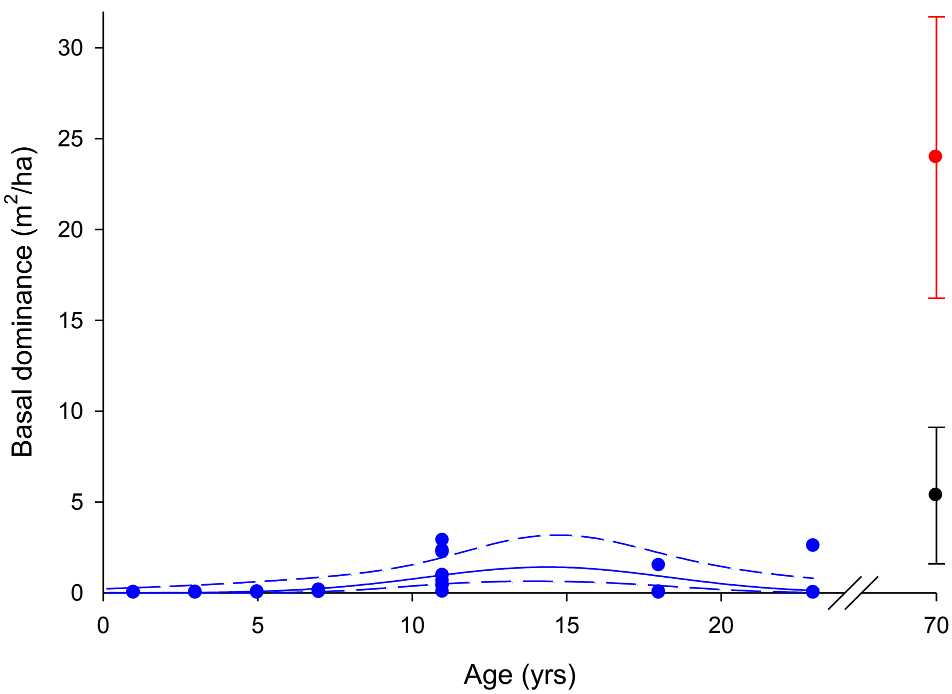

Figure 14: *Quercus palustris* basal dominance against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

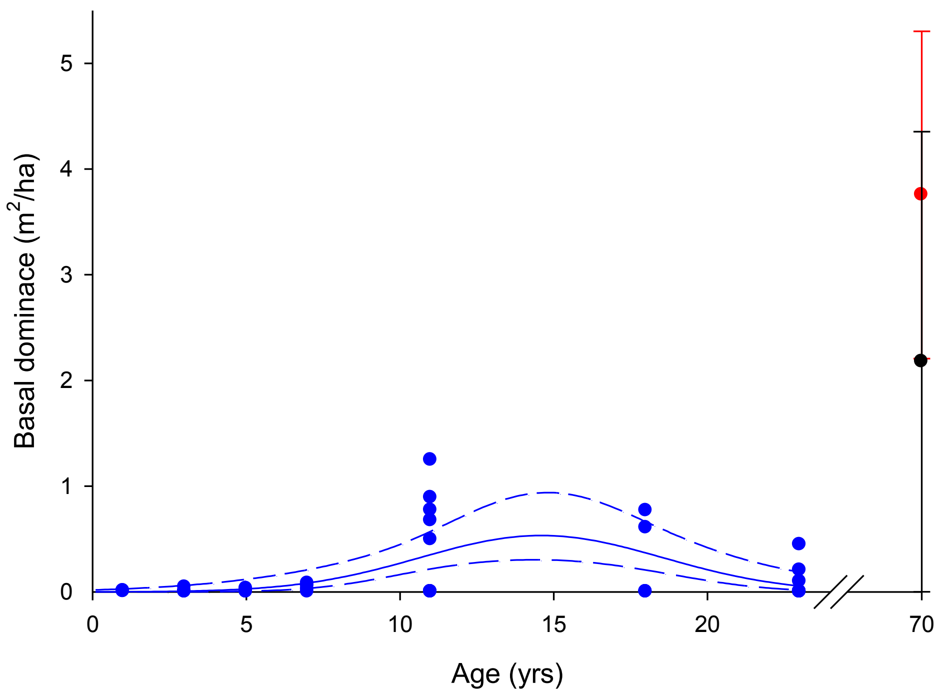

Figure 15: *Quercus macrocarpa* basal dominance against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

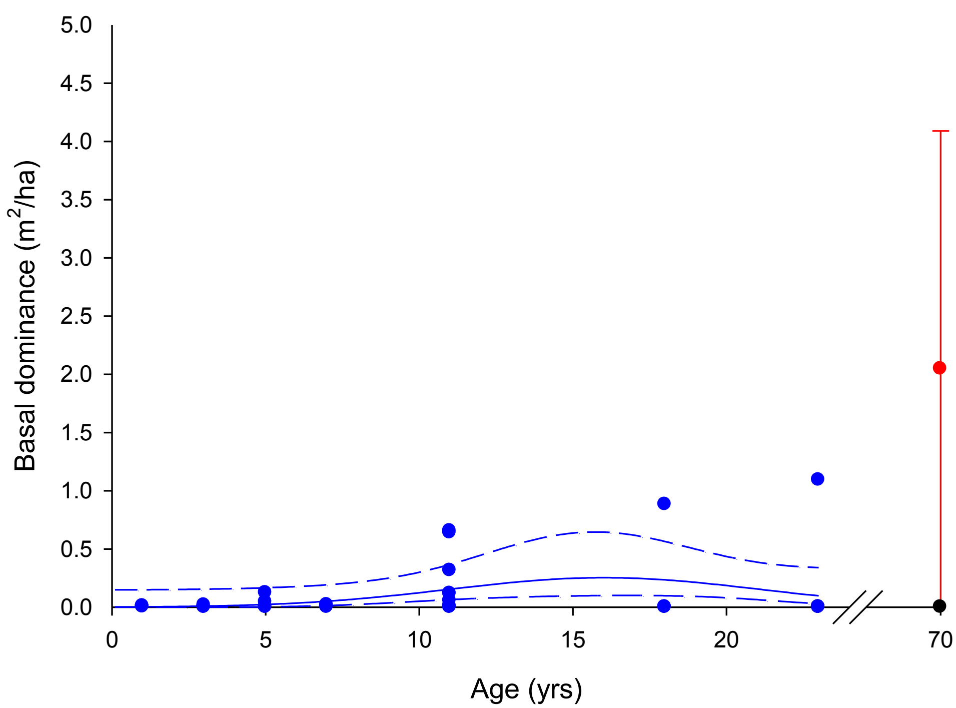

Figure 16: *Quercus bicolor* basal dominance against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Total Tree Dominance, Richness, and Diversity

Total basal dominance increased, peaked between 15 to 20 years and then decreased with age (Figure 17).

Richness peaked slightly between 10 to 20 years and then decreased (Figure 18). Simpson diversity had a slight peak between 10 and 15 years (Figure 19).

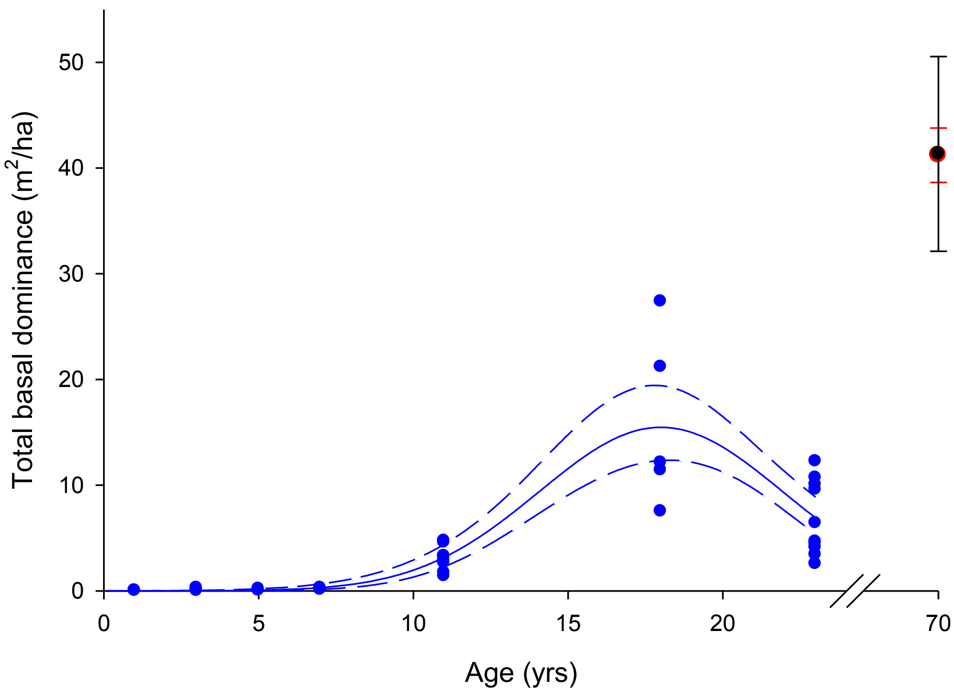

Figure 17: Tree total basal dominance against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

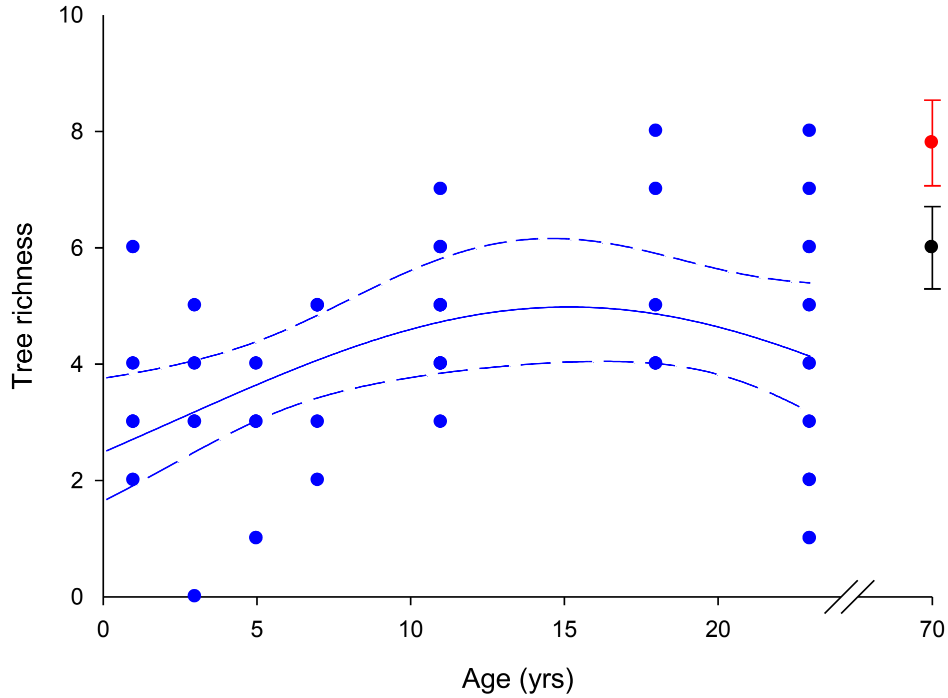

Figure 18: Tree richness against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

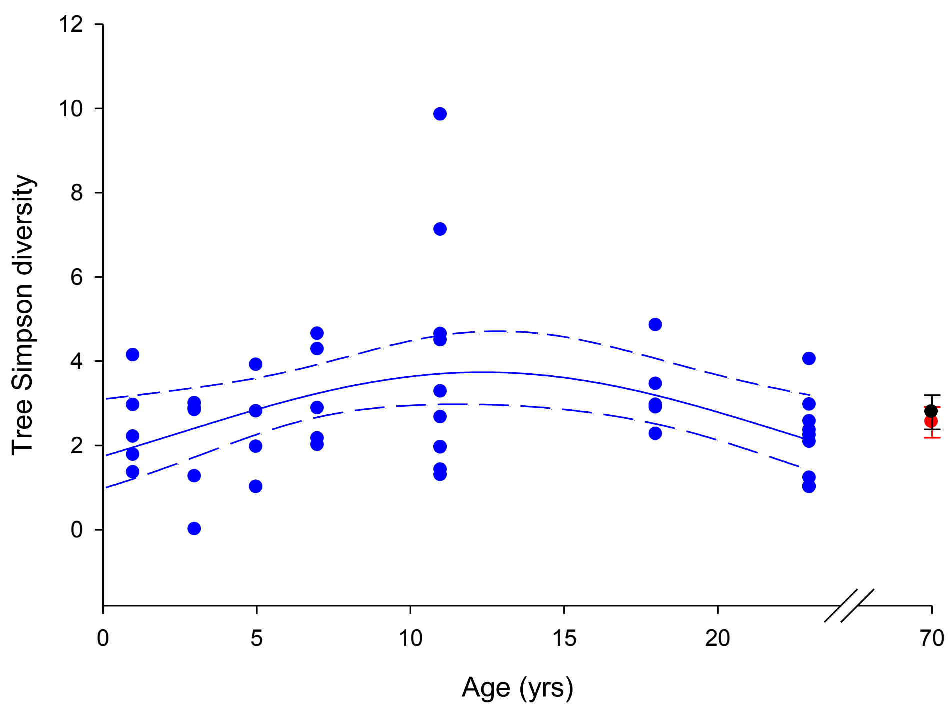

Figure 19: Tree Simpson diversity against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Shrub Density, Richness, and Diversity

Tree saplings that were greater than 1 m in height but had a DBH less than 2.5 cm were also included in shrub analyses. Total shrub density showed no trend with time (Figure 20). Shrub richness also showed no trend with age (Figure 21). Shrub Simpson diversity increased slightly with stand age (Figure 22).

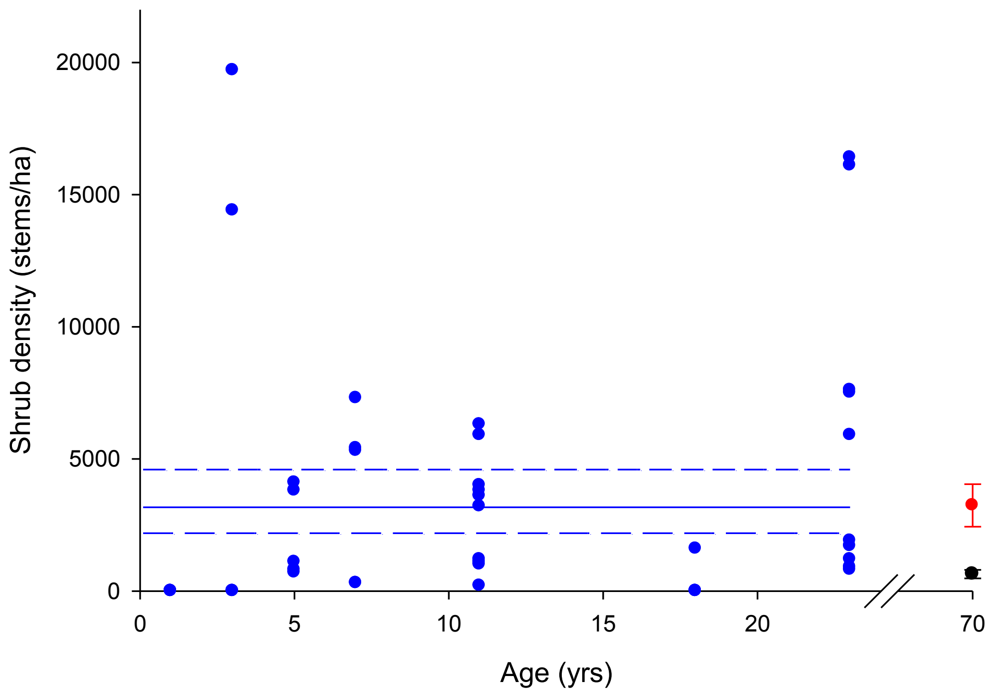

Figure 20: Shrub density against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

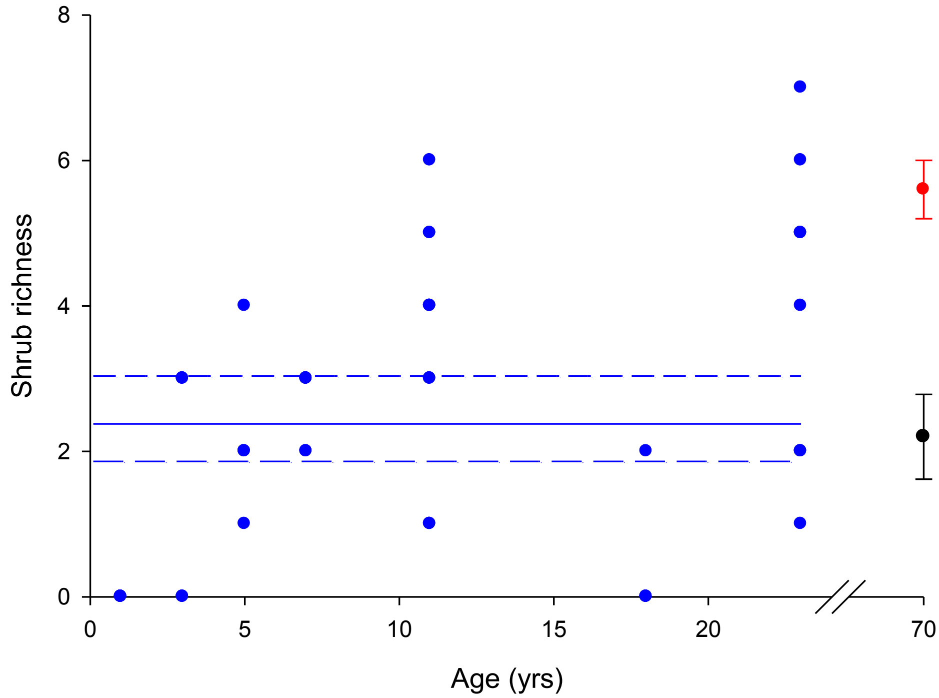

Figure 21: Shrub richness against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

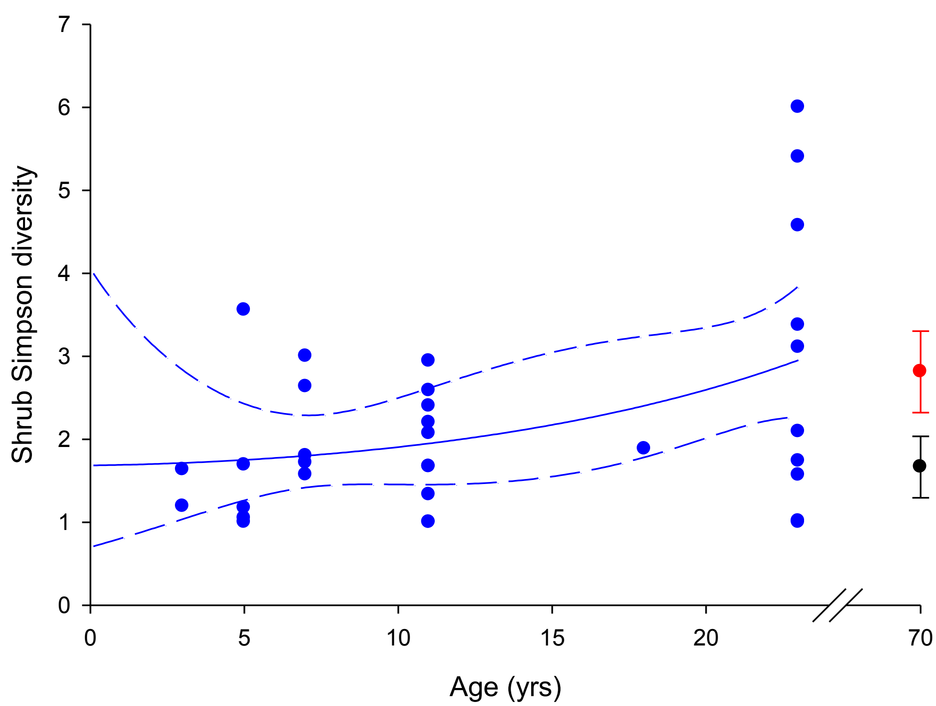

Figure 22: Shrub Simpson diversity against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Total Herbaceous Cover, Richness, and Diversity

Total herbaceous cover showed no trend with stand age (Figure 23). Herbaceous richness peaked between 10 to 20 years since restoration and then declined slightly (Figure 24). Herbaceous Simpson diversity moderately increased with time since restoration (Figure 25).

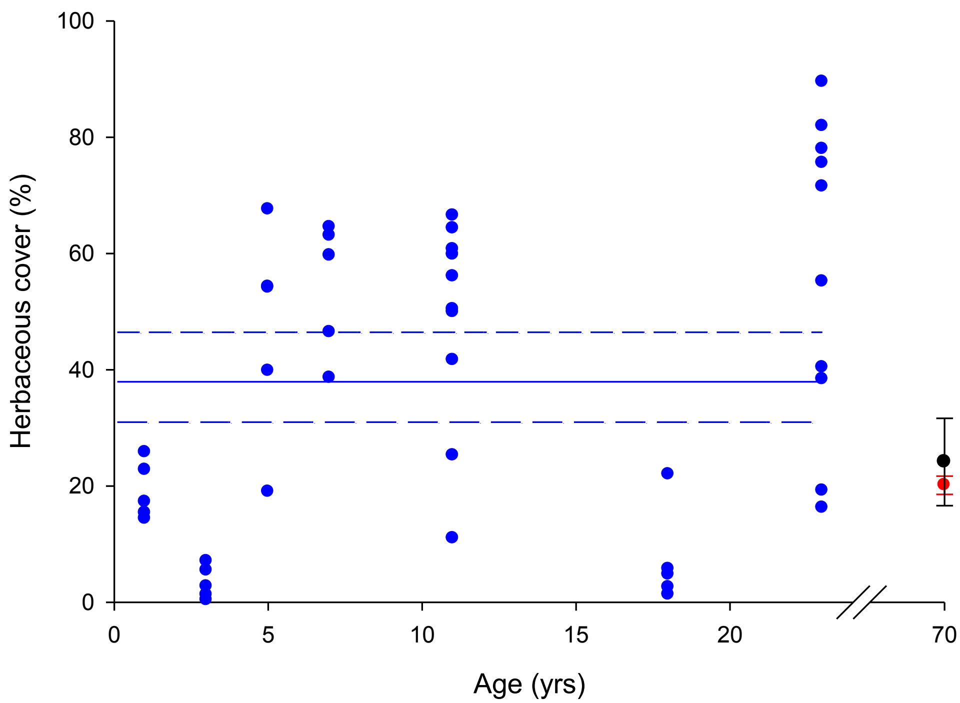

Figure 23: Herbaceous cover against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

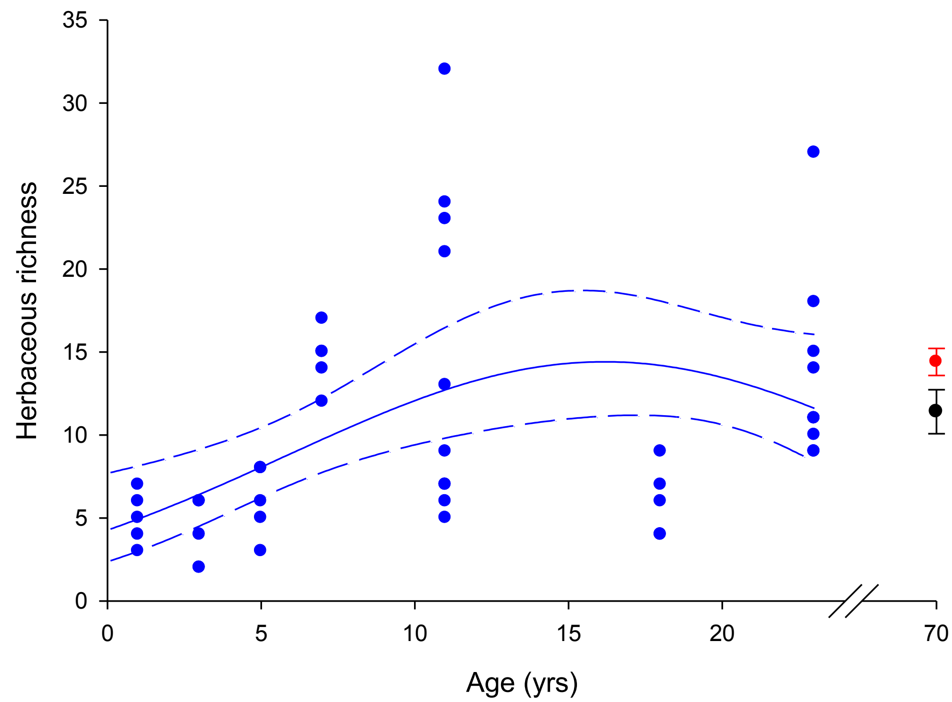

Figure 24: Herbaceous richness against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

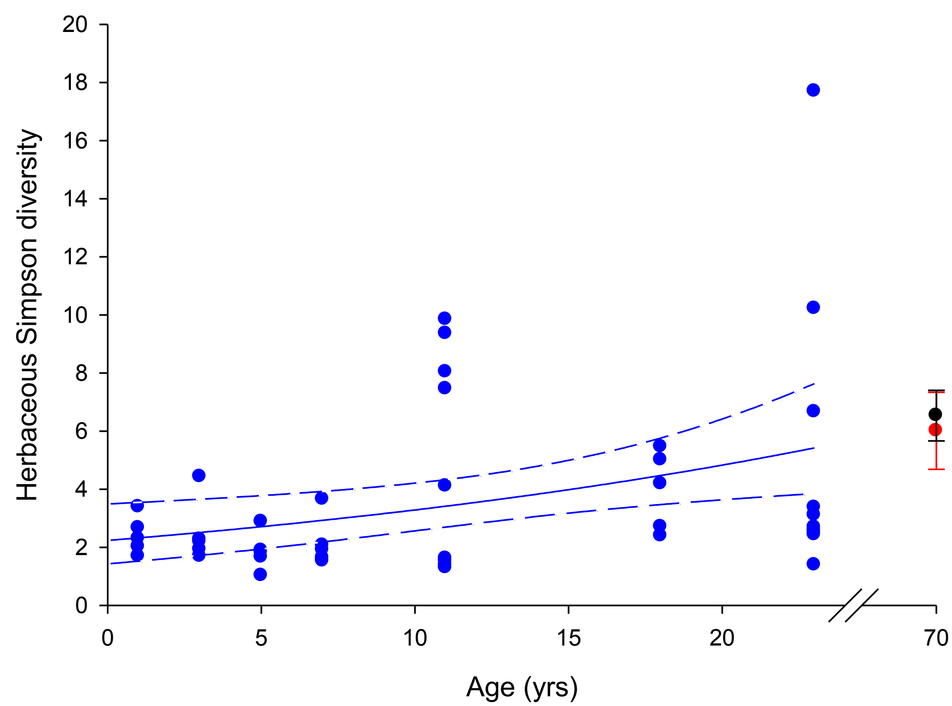

Figure 25: Herbaceous Simpson diversity against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure 5.

Percent Exotic Cover

Percent exotic herbaceous species cover peaked around 15 years and then began to decrease (Figure 26). See appendix E for a list of exotic species.

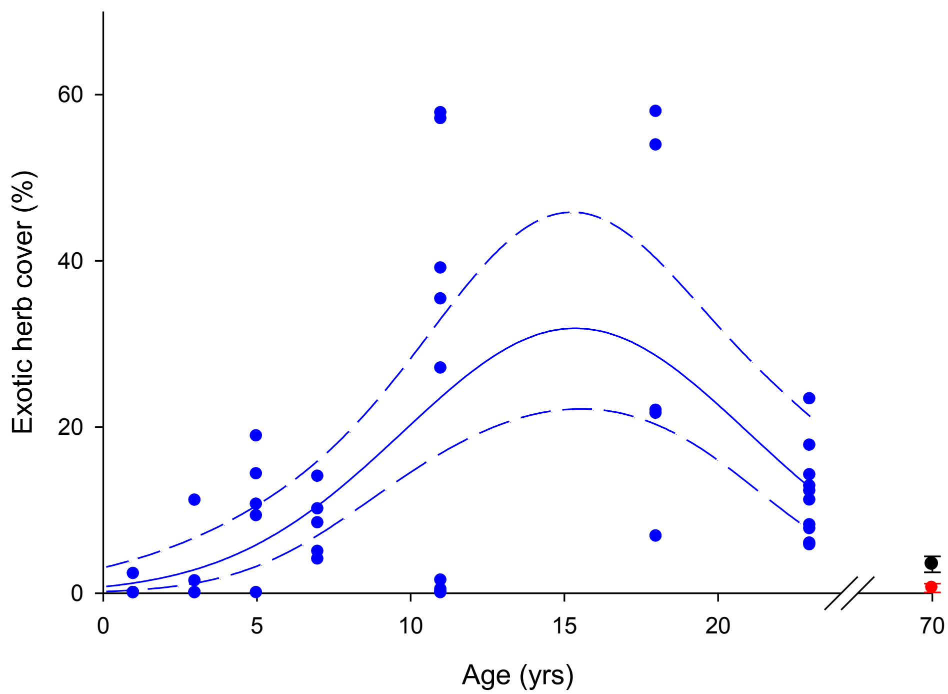

Figure 26: Percent exotic herbaceous cover against age. GLMs used a log link function and a quasi-Poisson error distribution. Reference plot data and confidence intervals are as in Figure

CHAPTER V

DISCUSSION

Tree Stratum

Of the planted tree species *Q. palustris* and *Q. bicolor* have the highest proportional change in height and basal diameter per year, followed by *Q. macrocarpa* (Table 3 and 4). These results are interesting since Allen *et al.* (2004) list *Q. palustris* and *Q. bicolor* as moderately flood tolerant and *Q. macrocarpa* as intolerant of flooding. It could be expected that, given the stress of flooding, *Q. macrocarpa* would exhibit the lowest mean relative growth rates.

Restoration sites will achieve trees of similar height to mature BLF in 31-34 years since planting and will have similar basal diameters in 27-37 years since planting (Table 3 and 4). Our oldest sites are 23 years old and if growth rates remain constant, the model suggests that these trees will attain similar heights and diameters in just a few years.

Despite the fact that planted trees will reach similar heights and basal diameters to mature BLF, dominance is still decreasing and lower than reference sites. Although trees are increasing in size the density of planted trees is decreasing. This is an indication of decreasing survivorship of planted trees.

Growth, dominance, and density could not be accurately modeled for *Carya illinoinensis*. This is because too few trees were encountered at restoration and reference plots. Pecan may be particularly sensitive to flooding. Additionally, *Carya illinoinensis* may be particularly susceptible to increased deer browsing. RPM® saplings of *Carya illinoinensis* are generally shorter than other saplings of oaks and therefore may be easier to reach. The foliage of *Carya illinoinensis* may also be considered more palatable than other species of trees. Twedt (2006) suggested that interplanting early successional trees within the hard mast mix will increase the probability that deer will overlook some of these trees. The USACE may be able to increase survival of *Carya illinoinensis* by allowing shrubs and light-seeded tree species to establish earlier (i.e. discontinue mowing sooner).

Total tree basal dominance is significantly lower and declining in restoration sites compared to reference sites. If the current trajectory continues, restoration sites will not reach values similar to mature forests. This is an indication of decreasing survivorship among planted trees and low recruitment of new seedlings. One of the oldest restoration sites, American Bottoms (23 years since planting), has a dense cover of grass that may be suppressing natural tree and shrub recruits and preventing canopy closure.

Tree richness and diversity were expected to increase after 11 years of age since USACE usually stops mowing restoration sites after 11 years of age. The model for tree richness also showed an initial increase and then a general decline after 15 years. Tree diversity began to decline after 11 years.

Richness values in the restoration sites ranged from about 1 to 8 species with a mean of 4 species, while values from reference sites ranged from about 5 to 8 species with a mean of 7 species. This difference is not large and subsequent research may show reference sites have increased richness values that are closer to reference sites. Additionally, Devall (1990) found similar richness values ranging from 5 to 10 in an old growth forest in Louisiana.

Tree Simpson diversity may approach values similar to reference sites but overall tree diversity is low. Diversity estimates for restoration sites and reference sites are generally low, between 2 to 5, except for COR restoration site which has a range of 3 to 10. Although low, Lockhart *et al.* (2010) looked at structure and composition in an old growth BLF in south-central

Arkansas found a diversity value of 4.7 within plots that were smaller (800 m^2) than those used within this study (1000 m^2).

Many of the restoration sites had substantial colonization by light-seeded species such as Eastern cottonwood (*Populus deltoides*), willow (*Salix spp.*), box elder (*Acer negundo*), green ash (*Fraxinus pennsylvanica*), and silver maple (*Acer saccharinum*). Colonization was also very homogenous, where only one or two different light-seeded tree species were able to colonize a plot. Low tree richness and diversity may become an issue at sites that are dominated by *Fraxinus* or *Ulmus* species. Should the emerald ash borer become introduced, a serious infestation would wipe out most ash trees. On that same note, a serious outbreak of Dutch elm disease could wipe out much of the *Ulmus* population. On sites where *Fraxinus* or *Ulmus* colonization is high losses could delay canopy closure. *Fraxinus* represented 72% of the total trees at American Bottoms, 41% at American Bottoms reference site, 23% at Brickhouse, and 16% at Cuivre Island. *Ulmus* represented 48% of the total trees at Cuivre Island and 19% at American Bottoms reference site.

Accurate mortality data for planted trees is necessary. Spring 2013 was marked by uncharacteristically high levels of rainfall and planted trees on the younger sites were overtopped by flood waters and may have experienced high rates of mortality. We could not obtain complete records of tree plantings for most of the restoration sites and so were unable to estimate mortality. Accurate mortality data is needed to improve the success of future plantings. Tree mortality is most likely higher initially and some tree species may do better if planted in areas of higher elevation (e.g. *Carya illinoinensis*). Ideally, a subset of trees should be tagged during the planting that would allow these trees to be monitored each year. Over planting is necessary to account for normal seedling loss due to abiotic and biotic factors. However, more research is needed to ensure that seedling mortality is within an acceptable range and that tree species are appropriately matched to site conditions (i.e. soil type, topography, flooding frequency).

Most of the restoration sites used RPM® trees (except Brickhouse and American Bottoms). RPM® oak trees produce acorns 10 to 15 years earlier compared to conventionally produced trees and some studies have observed seed production as early as 4 years old (Dey *et* al., 2004). It would be interesting to characterize the hard mast production and viability at these restored sites to calculate how much wildlife food is being produced and if these seeds have the potential to germinate. General and informal field observations during this study noted that some sites that were of suitable age were producing acorns while others were not. Most of the acorn production was noted to have taken place at the sites that were the least flooded for the shortest period of time. Future research may indicate that severe flooding may inhibit mast production.

Shrub Stratum

Total shrub density did not show a trend with time. This could be due to the mowing of sites. Although shrub richness initially increased, later trends show a gradual decrease or leveling off. Shrub richness approaches values similar to those of Rip Rap Landing Reference but does not seem to be approaching greater values like those of American Bottoms Reference. Shrub diversity shows a slight increase and is expected to reach values typical of mature BLF. Overall, richness and diversity are low for shrubs. Shrub richness for the restoration sites ranges between 1 to 6 species with a mean of 2 species, and 1 to 7 species with a mean of 4 species for reference sites. Other studies have reported richness values that were substantially higher. In southern Illinois on Horseshoe Lake Island, Robertson *et al.* (1978) reported shrub richness values as high as 59 species in secondary growth BLF, although their sampling plots were slightly smaller at 40 m^2. Huffman (1980) found that the number of woody species ranged from 15 to 20 in undisturbed BLF stands in southern Arkansas. Low shrub richness and diversity in the restorations sites could also be explained by the bi-annual mowing of restoration sites until 11 years since planting but it does not explain why shrub richness and diversity would be low in the mature reference sites.

Herbaceous Stratum

I did not see a general decline in total herbaceous cover that would be expected with the development of a forest canopy. Total herbaceous cover showed no trend with stand age. Irregular variation in herbaceous cover among sites (Figure 23) could be attributed to the extreme flooding of spring 2013. Spring 2013 experienced higher levels of precipitation which raised river levels and left some sites flooded for weeks with several meters of water. Some plots were sampled only a few weeks after the receding floods. Although every effort was made to allow the recovery of vegetation once flood waters receded, it is possible that some sites may not have had enough time to recover and therefore would have had a lower percentage of herbaceous cover. Differences in vegetation type and total cover by site may also be attributed to the season that data were collected in (i.e. spring vs. summer). Herbaceous vegetation and shrub data were collected starting in early June and ended late August. This slight, temporal difference could have had a large impact on the type of species present as the vegetation community continues to develop and change with the season. Differences could also be due to site proximity to a seed source. Most sites were located adjacent to older forests their seed source may be of better quality. Despite these differences, herbaceous richness and diversity do appear to be on track to approach values similar to mature BLF.

Exotic cover trends observed in our data are consistent with trajectories reported in the literature (McLane *et al.*, 2011). Exotic cover should continue to decline with stand age as canopy cover increases and should reach levels similar to reference sites. Additionally, exotic species could be sensitive to increased flooding and duration experienced at some of these sites during spring 2013. Predick and Turner (2008) found that exotic species occurred less frequently and at lower abundances in sites that were prone to flooding.

Chronosequences

Long term studies are the best way to evaluate forest growth. However, land managers need assessment information quickly in order to adjust restoration practices if necessary. Chronosequences can be a very useful technique for studying secondary succession and for evaluating restoration success more quickly (Allen, 1997). Nevertheless careful consideration and evaluation of sites selected for the chronosequence must be a priority.

It is important to consider and discuss the validity of the use of the chronosequence for this study. The review by Johnson & Miyanish (2008) discussed the use of chronsequences and cautioned against violating its key assumption- that all sites, regardless of age, all have the same biotic and abiotic pressures. Because all these sites were initially managed in a similar manner (e.g. same tree species were used, same site preparation methods) it gave us the unique opportunity to study the succession of these restored sites. However, although the sites were managed similarly, it is important to consider the fact that topographical, locational, and hydrological differences between the sites may violate some of the assumptions of the chronosequence. Proximity of a diverse seed source to the restoration site and differences in flooding based on site distance from the river may violate this assumption for some of the sites. For example, Calumet Creek restoration site was surrounded by older forest which may provide a significantly different seed bank than the COR restoration site, which is located near a busy highway. Some sites were also significantly flooded for several weeks (e.g. Crater's Landing, Brickhouse) while others only experienced moderate flooding (e.g. American Bottoms Restoration and Reference, COR). Further studies comparing site elevation to tree mortality could indicate additional differences between sites.

Although all the assumptions of the chronsequence have not been met, it is still a useful tool for gathering baseline information on restoration success and could be used as an indicator for failing sites that may require further research and attention. Regardless of site to site differences between each restoration, the chronosequence used within this study is a good method for evaluating site succession and restoration success. Tree growth can clearly be seen to be occurring and it is possible to track mortality rates if the trees within sample plots are tagged and tracked from the start of the restoration process.

It may not be possible to use the chronosequence method to continue research on the same sites that were evaluated in this study. The USACE has done supplemental plantings on several of the sites and has plans for future supplemental plantings. Polhman Slough had a supplemental planting done in 2004. Additionally, Red's Landing and Calumet Creek had supplemental plantings in 2011, although the number of trees is known. In October 2013, a large section of Epping was accidentally mowed through (no tagged plots were affected) by the Laclede Gas Company and that area has been replanted. Supplemental plantings make it difficult to appropriately evaluate the age of the site. Unfortunately, complete records on the number and species of trees are not always available. Furthermore, the USACE has changed the size of the RPM® trees they use for planting. Recent supplemental plantings done by USACE have used 15 gallon RPM® trees instead of the original 3 gallon RPM® trees. All of these changes make it impossible to continue using the chronosequence as a method for evaluating restoration at these specific sites. It is recommended that any continued research evaluate these sites independently.

Recomendations

The USACE commits significant resources to BLF restoration projects each year, so the results of this study may be of considerable value in determining future restoration practices and may increase the success of future restoration projects. However, more research is needed. Follow-up research has not been conducted on these sites after planting and this is the first study of its kind.

Based on these findings I would suggest an adaptive management approach where sites are frequently monitored and management adjustments (i.e. interplanting, discontinuing mowing earlier, tree placement, etc.) are made on case by case basis. Other studies have found this to be the best approach, especially given climate change and substantial human alteration of environments (Fabricus & Cundill, 2014; Failing *et al.*, 2013). Accurate records of tree plantings, including the number of each species planted, should be kept for each site. Additionally, permanent monitoring plots should be established at each new restoration and a subset of those trees should be tagged. Updated records should be maintained that include the tree size and the density of each planted species within the monitoring plots. This should make it easier for the USACE to ascertain whether or not each site has an acceptable number of trees per ha and the mortality trends for each planted species.

Additionally, restorations may be more successful by considering the local hydrology (i.e. flooding depth and frequency) along with site elevation and topography. Planted trees need to be better matched to site conditions to reduce seedling mortality. A typical planting involves selecting several types of trees that are assumed to grow well at a given site and then these trees are planted as they are unloaded from the truck. Tree placement could have the greatest impact on restoration success and needs to be given more consideration. Within natural, unaltered systems the distribution and sorting of BLF trees are strongly controlled by soil moisture, site topography, and flooding duration (Battaglia *et al.*, 2004; Gardiner *et al.*, 2004; Simmons *et al.*, 2009) (see also, Figure 27). Why, then, do land managers not plan restorations based on site topography and hydrology? The elevation and topography of each site should be mapped prior to planting and planting crews should be given detailed instructions on where to plant each tree species. Moreover, tolerance ratings are mostly based on field observations rather than experimental research. Better tolerance ratings for BLF tree species are needed and too little research is being conducted in this area.

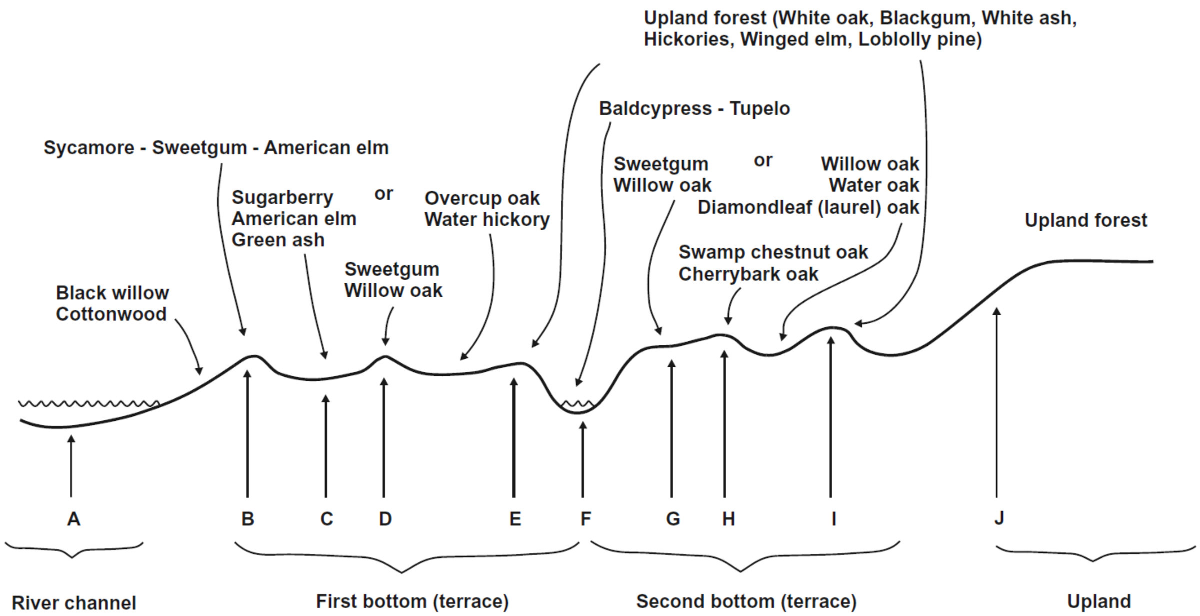

Figure 27: Typical Southern Mississippi Alluvial Valley BLF Distribution. Distribution is based on flooding and site topography (modified from Wharton *et al.*, 1982).

Land managers may also want to consider letting the most frequently and heavily flooded sites self-regenerate instead of diverting funds to try to restore these sites. Given that restoration can be expensive it may be more cost effective to allow the most altered sites to regenerate without assistance- even if that means that only light seeded species will grow and/or diversity is low. Regardless of which BLF tree species colonize these sites, some benefits will still be retained (i.e. evapotranspiration, erosion control, and some structural cover and habitat).

The USACE may also want to consider interplanting light-seeded species amongst the oaks and pecan. This can be done during the initial planting or during subsequent years after oaks and pecan species are of suitable size. Several studies have had positive results with interplanting (Twedt, 2006; Stanturf *et al.*, 2009). Fast growing, early successional trees increase evapotranspiration, provide habitat structure, and aid in canopy closure (Dey *et al.*, 2010; Twedt & Wilson, 2002). Twedt (2006) found that by planting clusters of light seeded species within a BLF restoration site species diversity, stem density, and vertical structure increased. Interplanting may be a useful way to increase the diversity of light-seeded species in areas that are not directly adjacent to a suitable seed source (i.e. greater than 100 m). Faster growing trees may provide enough structure to increase bird visitation and seed deposition which would lead to increases in herbaceous, shrub and tree diversity (Twedt *et al.*, 2002). Another way to increase canopy cover and tree diversity is by discontinuing mowing sooner than 11 years since planting to allow natural tree recruits to become established.

CHAPTER VI

CONCLUSION

Results indicate that restoration is successful in some respects but not in others. Tree growth rates are on track to reach sizes typical of mature BLF. Herbaceous richness and diversity approach mature BLF values and exotic herbaceous species begin to decline after 15 years. On the other hand, trajectories of tree dominance, tree and shrub richness, and herbaceous cover do not approach mature BLF values. Additional management, including replanting and mowing of the herbaceous vegetation, will be necessary at most sites to steer the restoration sites towards the structure and diversity typical of reference sites.

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APPENDIX A

Table 5: Coordinates for the Plot Centers. Coordinates are listed in Universal Transverse Mercator (UTM) form. All plots are located in UTM zone 15 S.

Site	Plot	Easting	Northing
CL	1	706225.01	4347727.44
CL	2	706201.97	4347782.37
CL	3	706241.95	4347674.57
CL	4	706164.84	4347880.24
CL	5	706182.47	4347834.06
EP	31	747139.84	4295325.00
EP	43	747240.16	4295371.45
EP	60	747389.93	4295421.67
EP	99	747689.65	4295572.11
EP	112	747739.79	4295721.45
RL	4	698218.40	4337535.40
RL	5	698178.69	4337806.50
RL	9	698205.83	4337861.61
RL	10	698184.24	4337928.81
RL	11	698218.05	4337754.18
CC	196	678123.85	4359960.93
CC	197	678219.80	4359948.69
CC	200	678151.53	4360069.28
CC	201	678191.34	4360024.66
CC	204	678218.40	4360085.25
COR	2	749110.92	4294376.63
COR	5	749178.60	4294382.08
COR	7	749227.23	4294440.27
COR	9	749256.91	4294492.31
COR	10	749289.34	4294539.99
PS	1	715955.62	4312954.84
PS	2	715918.46	4312794.10
PS	4	716092.56	4313193.85
PS	5	715996.24	4313103.13
PS	6	715846.06	4312488.00
CI	164	700704.52	4312543.93
CI	165	700681.95	4312681.08
CI	166	700689.54	4312757.90
CI	167	700698.69	4312875.86
CI	173	700739.20	4313022.38
AB	118	749822.81	4296025.59

AB	119	749873.18	4296000.51
AB	120	749815.96	4296077.59
AB	121	749874.04	4296056.09
AB	122	749858.60	4296104.49
BH	133	739089.70	4309322.28
BH	140	738889.40	4309371.78
BH	143	739039.59	4309371.87
BH	148	738789.79	4309422.10
BH	152	738989.44	4309422.57
ABR	3	750421.92	4296443.34
ABR	5	750374.24	4296300.72
ABR	6	750631.03	4296321.05
ABR	7	750585.54	4296164.06
ABR	8	750602.45	4296510.14
RRR	1	690647.74	4354884.44
RRR	2	691427.86	4353841.00
RRR	3	691687.75	4353573.95
RRR	4	691490.74	4353642.46
RRR	5	691229.24	4354410.74

APPENDIX B

Table 6: Summary Model Data. Each variable and the fitted equation are listed below. The density model for *Q. macrocarpa* is very close to significant, however, its use is justified on the basis that since basal diameter is increasing and dominance is decreasing, density must logically be decreasing.

Variable	**Fitted Equation**	**Pearson X^2**	**P**	**AIC**
Q. palustris Basal Diameter	ln(y)=1.5406+0.0913*Age	22.0	<0.0001	165.8
Q. macrocarpa Basal Diameter	ln(y)=1.3764+0.0749*Age	41.5	<0.0001	132.3
Q. bicolor Basal Diamter	ln(y)=0.9109+0.1136*Age	30.5	<0.0001	106.5
Q. palustris Height	ln(y)=0.6821+0.0806*Age	6.2	<0.0001	88.3
Q. macrocarpa Height	ln(y)=0.8071+0.0657*Age	13.7	<0.0001	91.0
Q. bicolor Height	ln(y)=0.4931+0.0815*Age	7.9	<0.0001	73.7
Q. palustris Density	ln(y)=0.8255+0.4346*Age-0.0185*Age^2	36.8	0.0002	283.9
**Q. macrocarpa* Density	ln(y)=3.1425-0.0584*Age	19.3	0.0680	279.6
Q. bicolor Density	ln(y)=3.3213-0.0346*Age	25.1	0.0021	253.9
Q. palustris Basal Dominance	ln(y)=-6.3231+0.9289*Age-0.0323*Age^2	63.1	0.0088	64.6
Q. macrocarpa Basal Dominance	ln(y)=-7.4333+0.9323*Age-0.0319*Age^2	11.4	0.0002	35.2
Q. bicolor Basal Dominance	ln(y)=-6.4157+0.6291*Age-0.0196*Age^2	18.4	0.0686	29.7
Tree total Basal Dominance	ln(y)=-7.7143+1.1614*Age-0.0323*Age^2	47.5	<0.0001	144.1
Tree Richness	ln(y)=0.9101+0.0918*Age-0.0030*Age^2	35.3	0.0426	186.0
Tree Simpson Diversity	ln(y)=0.5559+0.1235*Age-0.0050*Age^2	105.1	0.0578	171.2
Shrub Density	ln(y)=8.0623	338771.3	<0.0001	267026.5
Shrub Richness	ln(y)=0.8662	73.4	<0.0001	191.2
Shrub Simpson Diversity	ln(y)=0.3757+0.0302*Age	16.5	0.0082	105.5
Herbaceous Cover	ln(y)=3.6357	801.7	<0.0001	1176.3

Herbaceous Richness	$\ln(y)=1.4512+0.1504*Age-0.0047*Age^2$	146.7	0.0182	325.6
Herbaceous Simpson Diversity	$\ln(y)=0.8040+0.0385*Age$	91.3	0.0072	211.7
Exotic Herbaceous Cover	$\ln(y)=-0.2594+0.4847*Age-0.0158*Age^2$	421.7	<0.0001	641.3

APPENDIX C

Table 7: List of Tree Species from all Sites. Twenty-four tree species were found within the 55 sampling plots. Presence at a particular site is recorded as "+". For site abbreviations see Table 1. At Pohlman Slough, *Q. texana* was indistinguishable from *Q. palustris*. Both were categorized as "*Q. palustris*".

Species	**Common Name**	**Total Number Tagged**	**CL**	**EP**	**RL**	**CC**	**COR**	**PS**	**CI**	**AB**	**BH**	**ABR**	**RRR**
Acer negundo	Box Elder	153							+		+	+	+
Acer saccharinum	Silver Maple	104									+	+	+
Asimina triloba	Paw Paw	1				+							
Betula nigra	River Birch	1							+				
Carya illinoinensis	Pecan	27		+	+		+	+					+
Celtis occidentalis	Hackberry	39							+			+	+
Cornus drummondii	Dogwood	3										+	
Crataegus spp.	Hawthorn	2											+
Diospyros virginiana	Persimmon	36	+	+	+	+	+	+		+	+	+	
Fraxinus pennsylvanica	Green Ash	237				+	+		+	+	+	+	+
Gleditsia aquatica	Water Locust	3							+		+		
Gleditsia triacanthos	Black Locust	3										+	
Gymnocladus dioicus	Kentucky Coffeetree	1							+				
Ilex amelanchier	Swamp Holly	1										+	
Juglans nigra	Black Walnut	6	+	+		+							
Morus sp.	Mulberry	22			+						+		+
Platanus occidentalis	Sycamore	10							+	+		+	+
Populus deltoides	Cottonwood	45		+		+		+			+		+

Quercus bicolor	Swamp White Oak	54	+	+	+	+	+	+	+		+	+	
Quercus lyrata	Overcup Oak	5		+	+		+						
Quercus macrocarpa	Bur Oak	74	+	+	+	+	+		+		+	+	+
Quercus palustris	Pin Oak	94	+	+	+	+	+	+	+	+		+	+
Salix nigra	Black Willow	3		+									
Ulmus americana	American Elm	181							+		+	+	+

APPENDIX D

Table 8: List of Shrub Species from all Sites. A total of six different shrub species were found within the sampling plots. Tree saplings that were included in the shrub count (height ≥ 1 m but DBH < 2.5 cm) are not represented in this table. Presence at a particular site is recorded as "+". For site abbreviations see Table 1.

Species	Common Name	CL	EP	RL	CC	COR	PS	CI	AB	BH	ABR	RRR
Apocynum cannabinum	Dogbane	+	+		+	+	+		+	+		
Campsis radicans	Trumpet Vine	+		+	+	+	+	+	+	+	+	+
Cephalanthus occidentalis	Buttonbush											+
Ilex decidua	Possumhaw										+	+
Lonicera mackii	Bush Honeysuckle					+			+			
Morus sp.	Mulberry			+		+	+			+		+

APPENDIX E

Table 9: List of Herbaceous Plant Species from all Sites. Ninety-one herbaceous plant species were found within the 55 sampling plots. Eighteen of those species were exotic. Presence at a particular site is recorded as "+". For site abbreviations see Table 1.

Species	**Common Name**	**Status**	**CL**	**EP**	**RL**	**CC**	**COR**	**PS**	**CI**	**AB**	**BH**	**ABR**	**RRR**
Abutilon theophrasti	Indian Mallow	Exotic									+		
Agastache nepetoides	Yellow Giant Hyssop	Native			+				+			+	+
Alisma subcordatum	American Water Plantain	Native								+			
Allium vineale	Wild Garlic	Native								+			
Ambrosia trifida	Giant Ragweed	Native	+		+	+		+			+		
Ampelopsis cordata	Heartleaf Peppervine	Native									+		+
Apocynum cannabinum	Dogbane	Native	+	+		+	+	+		+	+		
Arisaema dracontium	Green Dragon	Native										+	
Asclepias syriaca	Common Milkweed	Native		+			+						
Bidens frondosa	Devil's Beggartick	Native									+		
Boehmeria cylindrica	False Nettle	Native			+		+						+
Bromus inermis	Smooth Brome	Exotic								+			
Bromus racemosus	Bald Brome	Native		+			+	+					
Campsis radicans	Trumpet Vine	Native	+	+	+	+	+	+	+	+	+	+	+
Cardiospermum halicacabum	Balloon Vine	Exotic									+		
Carduus nutans	Nodding Plumeless Thistle	Native					+						
Carex crus-corvi	Ravenfoot Sedge	Native								+		+	
Carex eburnea	Bristle-leaved Sedge	Native		+		+	+			+			
Carex festucacea	Fescue Sedge	Native				+						+	
Carex frankii	Frank's Sedge	Native				+							
Carex grisea	Wood Gray Sedge	Native					+						
Carex hyalinolepis	Shoreline Sedge	Native									+	+	

Cephalanthus occidentalis	Buttonbush	Native											+
Cerastium brachypetalum	Gray Chickweed	Exotic					+						
Chamaecrista fasciculata	Partridge Pea	Native									+		
Cichorium intybus	Common Chicory	Native				+	+	+					
Cirsium discolor	Field Thistle	Native					+						
Commelina communis	Asiatic Dayflower	Exotic					+		+				
Convolvulus arvensis	Field Bindweed	Exotic								+		+	
Cubelium concolor	Green Violet	Native					+						
Cynanchum laeve	Honeyvine	Native	+	+	+	+		+	+	+	+		+
Cyperus esculentus	Yellow Nutsedge	Native		+	+			+			+	+	
Daucus carota	Queen Anne's Lace	Exotic					+			+		+	
Desmanthus illinoensis	Illinois Bundleflower	Native		+			+				+		
Desmodium paniculatum	Panicledleaf Ticktrefoil	Native				+	+			+			
Echinochloa crus-galli	Barnyard Grass	Native					+						
Elymus virginicus	Virginia wildrye	Native				+	+					+	
Euonymus fortunei	Winter Creeper	Exotic					+						
Euphorbia nutans	Eyebane	Native									+		
Galium aparine	Stickyweed	Native					+			+		+	
Geranium carolinianum	Carolina geranium	Native					+						
Glyceria spp.	Mannagrass	Native										+	
Glyceria striata	Fowl Mannagrass	Native									+	+	
Hibiscus lasiocarpos	Rosemallow	Native									+		
Hordeum pusillum	Little Barley	Native					+						
Impatiens capensis	Jewelweed	Native							+			+	+
Ipomoea hederacea	Ivyleaf Morning-glory	Exotic				+		+			+		
Ipomoea lacunosa	Whitestar	Native	+										
Iris virginica	Virginia Iris	Native										+	
Laportea canadensis	Woodnettle	Native											+

Lippia lanceolata	Lanceleaf Fogfruit	Native			+						+		
Lolium arundinaceum	Tall Fescue	Exotic					+			+			
Lolium pratense	Meadow Fescue	Exotic					+			+			
Lonicera japonica	Japanese Honeysuckle	Exotic					+						
Lonicera maackii	Bush Honeysuckle	Exotic					+						
Lysimachia ciliata	Fringed Loosestrife	Native										+	
Medicago lupulina	Black Medick	Native					+			+			
Menispermum canadense	Canadian Moonseed	Native										+	+
Panicum philadelphicum	Philadelphia Panicgrass	Native		+		+							+
Pastinaca sp.	-	Exotic					+						
Persicaria pensylvanica	Smartweed	Native	+	+	+				+		+	+	+
Phalaris arundinacea	Reed Canarygrass	Native				+	+						
Physalis longifolia	Wild Tomatillo	Native				+					+		
Physostegia virginiana	Obedient Plant	Native									+		
Phytolacca americana	American Pokeweed	Native				+							
Plantago lanceolata	Narrowleaf Plantain	Native					+						
Ranunculus hispidus	Bristly Buttercup	Native										+	
Rorippa sylvestris	Creeping Yellowcrest	Exotic			+						+		+
Rubus argutus	Sawtooth Blackberry	Native					+			+		+	
Rumex altissimus	Pale Dock	Native				+							
Rumex crispus	Curly Dock	Native	+	+	+	+	+	+					
Sagittaria latifolia	Broadleaf Arrowhead	Native			+								
Saururus cernuus	Lizard's Tail	Native							+			+	
Setaria viridis	Green Bristle Grass	Native				+					+		
Sicyos angulatus	Bur Cucumber	Native							+				+
Sida spinosa	Prickly Fan Petals	Native									+		
Smilax hispida	Bristly Greenbriar	Native							+			+	+
Solanum carolinense	Carolina Horsenettle	Native				+	+	+		+	+		

Solidago altissima	Goldenrod	Native		+		+	+	+		+	+		
Spermacoce glabra	Smooth False Buttonweed	Native						+			+		
Spiranthes ovalis	October Lady's Tressess	Native									+		
Stachys tenuifolia	Smooth Hedgenettle	Native									+		
Stellaria media	Common Chickweed	Native					+				+		
Symphyotrichum pilosum	hairy White Oldfield Aster	Native	+										
Toxicodendron radicans	Poison Ivy	Native				+	+		+	+	+	+	+
Trifolium hybridum	Alsike Clover	Exotic					+			+			
Triodanis perfoliata	Clasping Venus' looking-glass	Exotic				+	+						
Veronica arvensis	Corn Speedwell	Exotic					+						
Viola sororia	Common Blue Violet	Native				+					+		
Vitis riparia	Riverbank Grape	Native		+		+	+	+	+	+	+	+	+
Xanthium pennsylvanicum	Common Cocklebur	Native						+					+