Early Successional Processes of Experimentally-Reclaimed Mine Sites in Eastern Ohio
and the Restoration of American Chestnut

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This dissertation titled
Early Successional Processes of Experimentally-Reclaimed Mine Sites in Eastern Ohio
and the Restoration of American Chestnut

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ABSTRACT

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Early Successional Processes of Experimentally-Reclaimed Mine Sites in Eastern Ohio and the Restoration of American Chestnut

Director of Dissertation: Brian C. McCarthy

Surface mining for coal represents one of the most severe and extensive forms of disturbance throughout the Appalachian region of the Eastern Deciduous Forest. Federal law has required reclamation of mined sites to mitigate environmental hazards since the late 1970s. However, the methods typically utilized do not allow for a return to the forest cover found on these sites prior to mining. New reclamation methods have been proposed that aim to expedite a return to native forest cover through the use of compaction-reducing soil-preparation methods. One of the major components of the Eastern Deciduous Forest, prior to its demise due to an introduced pathogen, was the American chestnut (*Castanea dentata*). The original extent of the American chestnut and the Appalachian coal fields are nearly concurrent, possibly making these experimentally-reclaimed sites ideal for restoration plantings of disease-resistant chestnut hybrids. The work here has been conducted on an experimentally-reclaimed mine site in Belmont County Ohio, and attempts to shed light on two broad questions: 1) What can be expected during restoration plantings of American chestnuts on experimentally-reclaimed mine lands, and 2) What types of plant communities are becoming established on these sites during the initial natural colonization phase of succession that is occurring there? I found
effects of distance from the remnant site forest edge, microtopographic position, the amount of existing vegetation cover at the site, and a small set of soil chemical variables to influence the success of American chestnut establishment. In addition to the physical environment, the hybrid nature of restoration chestnuts and how traits characteristic of each parental lineage in the hybrid offspring may be related to success on this site. The plant communities naturally developing during early succession at the site differ significantly among a set of microtopographic positions specific to this type of reclamation, possibly due to the soil physical properties unique to each position. The spatial distribution of naturally-colonizing woody species also differs in relation to the same set of microtopographic positions as well as species identity, edge distance, and possibly competition. The results of this work make an important contribution to the current understanding of the use of the experimental Forestry Reclamation Approach method in mine land reclamation, effects of anthropogenic disturbance and natural recovery processes on the landscape, as well as how well chestnut restoration will contribute to reclamation.
DEDICATION

For Julia
ACKNOWLEDGMENTS

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My roommate, landlord, and good friend Ryan Homsher deserves special recognition for the countless hours of field work and support during this entire time of my life.
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CHAPTER 1 : INTRODUCTION TO THE DISSERTATION

Disturbance and succession are key processes inherent to plant community dynamics. Disturbances occur over a variety of scales due to a variety of reasons but all ultimately result in a change to the vegetative composition or physical environment of a local area (Pickett et al. 1987). “Disturbance” has been defined by Pickett and White (1985) as, “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment.” In natural systems, disturbances can have an effect on species diversity similar to predation by removing dominant competitors preventing competitive exclusion (Paine and Levin 1981). Disturbance can also prevent competitive exclusion of two or more species by creating a heterogeneous environment with differing resource levels accommodating different species (Levin and Paine 1974, Denslow 1980). Variations in frequency and intensity of disturbance have effects on species diversity ranging from being critical to the maintenance of that diversity to causing long-lasting irreversible changes in community structure (Connell 1978).

Following disturbance, forest ecosystems begin a series of short- and long-term processes related to vegetation change. Various explanations have been offered to describe the successional process beginning with Clements (Clements 1916). Models have included “relay floristics” where groups of plant species replace one another until stability is achieved to “initial floristics” with emphasis placed on the propagule pool
(Egler 1954). Connell and Slatyer (1977) proposed a set of three hypotheses to explain successional processes under different conditions. The Connell and Slatyer models propose three different scenarios: the first (facilitation) model involves early successional species arriving at a site, colonizing and modifying the site in some way to make the site more amenable to later successional species. The second (tolerance) model involves any species arriving and colonizing the site and making that site non-amenable to early successional species but not inhospitable to other late successional species. The third proposed model (inhibition) involves arrival and colonization of any species (early or late successional), which then makes the site less amenable for both early and late successional species. All of these processes ultimately arrive at a state where subsequent changes in composition are the result of a new type of disturbance that removes living individuals from the system and releases growing space. Pickett et al. (1987) described a set of successional causes, contributing processes and modifying factors to those processes. These ideas are based on the species’ performance related to various site conditions and the effect that performance has on the successional trajectory.

Ecological restoration is the process of “assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” (SER 2004). It could also be said it is essentially the process of directing succession along some desired trajectory. Under ideal conditions, once this trajectory is realized, the system may no longer require continued input. Selection of proper goals for the final state of the restored system is therefore of great importance (McCoy and Mushinsky 2002, Ruiz-Jaen and Aide 2005).
Restoration ecology has been described as a kind of “acid test” of the understanding of ecological processes (Bradshaw 1987). The scientific discipline of restoration ecology underpins the practical nature of ecological restoration. Much of Egler’s Initial Floristics Hypothesis is reflected in current-day restoration in the idea that if the correct propagule pool is initially established then a desired successional trajectory will follow. The workings of an ecosystem can be elucidated by studying its constituent parts.

Mine land reclamation represents an important opportunity to study the mechanisms of succession across varying conditions. In Ohio, at least 180,000 hectares of land have been affected by surface mining (http://ohiodnr.com/mineral/federal/tabid/17851/default.aspx). Prior to WWII, most coal was mined subterraneously, but increased demand and the development of technology made large scale surface mining both necessary and possible. Most surface mined land in eastern Ohio and other Appalachian states is located within the Eastern Deciduous Forest. Understandably, much of this mined land bears little resemblance to its former forested state, even after many decades of succession.

Years of relatively unregulated mining coupled with a lack of reclamation activity led the federal government to enact the Surface Mine Control and Reclamation Act (SMCRA) in 1977. The law requires that mining companies secure necessary permits before mining operations begin, prohibited mining on certain types of lands, and gave the government oversight of these processes. The law also established a bonding process to
ensure reclamation following mining (Surface Mining Control and Reclamation Act 1977).

Current laws emphasize a quick return to vegetative cover (usually 5 years) for bond release and this has led most coal operators to seek a quick reclamation strategy. Instead of forest, these sites are usually returned to grasslands dominated by fescue species, nitrogen-fixing clovers and lespedezas, and sometimes by low-quality nitrogen-fixing shrubs and tree species. Succession often becomes stalled on these sites where native species are unable to compete with introduced species. Research has shown that post-law planting of sites with aggressive ground-cover species to minimize erosion actually has the effect of slowing long-term recovery back to forest (Holl 2002).

One of the primary goals of the restoration ecologist is to direct succession along a modified trajectory. In the case of formerly forested mined land, the goal is to return to an eastern deciduous forest ecosystem. Heretofore, this objective has rarely been met. A variety of hurdles need to be overcome along the path of succession for this to occur. Post-mining practices have traditionally yielded a substrate very dissimilar to that found in the original or surrounding forest lands. Attention was spent ensuring the stability of land to prevent severe erosion and mudslides that had previously plagued post-mining landforms (Wooley 1978). The substrate deposited after a mining operation is usually a highly compacted material lacking in nutrient content.

New techniques in soil preparation are being advanced by the Appalachian Regional Reforestation Initiative (ARRI) to improve growing conditions on previously
mined land. The Forestry Reclamation Approach (FRA) consists of five steps to return
mined land to forest cover. First, a suitable medium for tree growth needs to be
established that is no less than 1.3 m deep and comprised of the best material available.
Second, the topsoil (or topsoil substitute) must be loosely graded with no more than one
bulldozer pass (or not at all) to minimize compaction. Third, less aggressive ground
covers that are compatible with trees need to be planted. These ground covers also need
to be sown at a rate which does not cause them to out-compete planted tree seedlings.
Fourth, early successional trees need to be planted along with more commercially
valuable late successional hardwoods to ensure soil stability and wildlife cover. Fifth,
restoration work must be performed by experienced and reputable tree-planting
contractors (Zipper et al. 2011a). Reclamation to forest rather than grassland has both
economic advantages as well as ecological benefits (Zipper et al. 2011b). Demand for
high quality hardwood lumber has not diminished and reforestation provides better
wildlife habitat compared to grassland. This method of reclamation represents an attempt
to apply an ecologically-minded approach to mine reclamation versus the agronomic
approach of traditional SMCRA-approved methods (Pfannstiel and Wendt 2002).

American chestnut (*Castanea dentata* (Marsh.) Borkh.), which once made up a
large portion of the eastern deciduous forest (Fig 1.1), exhibits good tolerance to the high
light conditions typically found on reclaimed mine sites (Joesting et al. 2007). The
species is also noted for its ability to tolerate poor site quality during its early
developmental stages (Jacobs 2007). The ability to thrive in high light conditions on poor
soils where other hardwood species tend to perform poorly (Groninger et al. 2006) makes American chestnut an ideal candidate for mined-land reclamation projects.

American chestnut was once a dominant canopy tree in the Eastern Deciduous Forest with a range extending from southern Maine to northern Georgia. Pollen records indicate a dominance of chestnut and oak species in the entire Appalachian range for roughly 2500 years prior to decline (Paillet 2002). The species has been all but eliminated from the forest canopy as the result of an introduced fungal pathogen. Chestnut Blight (*Cryphonectria parasitica* (Murrill) Barr) is believed to have been introduced to the United States via a specimen tree at the Brooklyn Botanical Garden in 1904. Chestnut blight is a diffuse canker that kills trees by inserting its hyphae through the bark into the tree’s vasculature, causing a disruption of the xylem and phloem eventually girdling the tree and killing it (Liebhold et al. 1995).

Species that replaced chestnut were typically oak (*Quercus* spp.) or hickory (*Carya* spp.), but red maple (*Acer rubrum*) and ash (*Fraxinus* spp.) were also noted to be replacing Chestnut in canopy gaps in the 1950’s (Woods and Shanks 1959). While oak and hickory serve to support wildlife via the production of hard mast, none of the replacement species equaled the fecundity and food quality of American chestnut. Studies show that mast output and correlated vertebrate carrying capacity dropped markedly with the decline of American chestnut (Diamond et al. 2000, Gilland et al. 2012). The decline of American chestnut is also believed to have markedly altered various ecosystem
processes (decomposition and nutrient cycling) in terrestrial environments (Ellison et al. 2005).

The American Chestnut Foundation has been working toward developing a blight-resistant hybrid chestnut tree for the last 25 years. The hybrids have been developed using a backcross breeding program with Chinese chestnuts (*Castanea mollissima* Blume.) to produce a hybrid identical in characteristics to the American chestnut while retaining the blight resistance of the Chinese chestnut (Jacobs 2007). The latest generation of hybrids (BC$_3$F$_3$) is 15/16ths American chestnut character while still displaying resistance to chestnut blight. The latest hybrid generation is now becoming available in sufficient quantities for experimental purposes (Diskin et al. 2006).

Preliminary studies of these hybrids have shown that American chestnut is an ideal candidate for mine-land reclamation projects. American chestnut seems to tolerate the exposed, rocky, and acidic characteristic of mine soils. Results have been promising on sites with varying soil treatments to relieve soil compaction characteristic of reclaimed mines (Jacobs 2007). Data from a long-term study in Wisconsin has shown that American chestnut has the ability to reproduce prolifically when introduced in relatively small numbers. At that study site, chestnut has colonized an oak-hickory stand over a period of roughly 80 years following the introduction of approximately ten trees around the turn of the century (McEwan et al. 2006).

The FRA approach is currently being implemented at study sites in Ohio with the hopes of furthering the reclamation of former mine land to forest, as opposed to
grassland. American chestnut stands to make reclamation to forest an attainable goal while at the same time granting an opportunity to reintroduce the species to the forest landscape.

With the work presented in this dissertation, I have attempted to shed light on some of the factors controlling the establishment success of American chestnut on sites reclaimed using the FRA end-dump method. I have also examined the role the hybrid lineage between Chinese and American parents may play in the success of using blight-resistant hybrid chestnuts for mine-land reclamation projects. Finally, I have observed the effects of the physical micro-environment on the plant communities that are naturally developing at a site reclaimed using the FRA.

**Literature Cited**


Figures and Legends

Figure 1.1: Original, pre-blight range of the American chestnut. Digital representation from Little’s “Atlas of United States Trees.” Available at <http://gec.cr.usgs.gov/data/atlas/little/castdent.pdf>
CHAPTER 2: REINTRODUCTION OF AMERICAN CHESTNUT (*CASTANEA DENTATA*) ON RECLAIMED MINE SITES IN OHIO: MICROSITE FACTORS CONTROLLING ESTABLISHMENT SUCCESS

**Abstract**

Microsite availability is crucial for recruitment success in natural populations as well as populations being established for restoration projects. Understanding the specific microsite requirements of a particular species targeted for restoration will increase the probability of success of any restoration project. Surface mining for coal represents one of the largest anthropogenic disturbances to the forests of the eastern United States. The original natural range of the American chestnut (*Castanea dentata* (Marsh.) Borkh.) overlaps the extent of the Appalachian Coal Basin. With American chestnut being readied for reintroduction trials, I sought here to determine some of the effects of microsite conditions on the establishment success of American chestnut on mine sites reclaimed using new, compaction-reducing techniques (i.e., “end-dump” reclamation) that create a series of loosely dumped mounds roughly 8 m diameter and 3 m tall to serve as a planting substrate. Specifically, I examined the effects of distance from existing forest edge, amount of existing cover of vegetation, small-scale topographic position, and a small set of soil variables on the growth and survival over three seasons of American chestnut seedlings planted on a reclaimed mine site in east-central Ohio. I found decreased tree survival adjacent to existing forest edges and greater annual growth rates at distances of 20 and 50 m from the existing forest edge. Microtopographic position had
a significant effect on seedling growth and survival—seedlings planted higher on mounds had increased mortality and lower growth than those on the side slopes of those mounds. The amount of existing vegetative cover also affected survival and growth; trees growing in plots with higher vegetative cover values showed increased growth and survival. The compaction-reducing reclamation approach used here is a relatively new and novel. Promising results that have been observed to date using American chestnut and this method seem to indicate that the combination may be very effective at restoring functional forests on lands degraded by surface mining.

**Introduction**

Availability of a suitable microsite, a small-scale site suitable for the establishment of an individual, is crucial for recruitment in natural populations (Eriksson and Ehrlén 1992, Elmarsdottir et al. 2003); and restoration projects of anthropogenically-disturbed sites (Knapp et al. 2008, Kelly and Wentworth 2009). Understanding the specific microsite requirements of a particular species targeted for a specific type of restoration will increase the probability of success of any restoration project (Young et al. 2005).

Edge influences that affect recruitment success exist at varying distances from anthropogenically-created forest edges (Harper et al. 2005). Existing herbaceous cover may have effects on seedling establishment by providing seed-predator cover (Gill and Marks 1991, Pusenius and Ostfeld 2002). Likewise, intact vegetative cover may inhibit natural colonization of woody seedlings via competition for water, light and mineral
resources (Davis et al. 1999). However, vegetative cover has also been observed to facilitate the emergence of certain large-seeded species via retention of soil moisture and shading effects (DeSteven 1991).

Surface mining for coal represents one of the largest anthropogenic disturbances to the forests of the eastern United States. The 1977 Surface Mining Control and Reclamation Act (SMCRA) sought to remedy many of the land-degradation problems previously associated with the closing and/or abandonment of surface-mining projects (Wooley 1978). Establishment of pasture has been common on mined lands, offering an easily achievable means of satisfying the requirements of the law, but resulting in plant communities that remain in a state of arrested succession for decades—the previous forest condition is rarely if ever achieved (Ashby 1991, Figure 2.1). The observed lack of recovery is likely due to a suite of factors including soil compaction and poor drainage caused by heavy equipment employed during the final stages of reclamation and aggressive competition from non-native ground cover species typically utilized to satisfy bond requirements (Holl and Cairns 1994, Zipper et al. 2011b).

New reclamation methods (Forestry Reclamation Approach; FRA) have been proposed to promote an expedient return to forest cover, while still meeting federal guidelines for bond release. These methods emphasize the use of non-compacting ground-preparation methods and less aggressive vegetative cover (Zipper et al. 2011a). Increased growth rates of desirable woody species have been noted where soil-ripping has been implemented to relieve compaction (Ashby 1997, McCarthy et al. 2010). Using
a process known as end-dump reclamation, the original approximate contour of the land is first restored as per SMCRA regulations, but contrary to past practices, the final layer of material is dumped into large piles (~5 × 5 m in diameter, 2–3 m tall) that provide a loose rooting medium into which trees are then planted (Figure 2.2). The success of these projects may ultimately depend on the selection of proper tree species suitable for the site with a set of microhabitat conditions best suited for those tree species (Young et al. 2005).

Of the nearly one million hectares affected by surface mining in the United States, nearly 40% are located in the Appalachian Coal Basin ranging from Northern Pennsylvania to Alabama (USGS 1996). The natural range of the American chestnut (Castanea dentata (Marsh.) Borkh.) is nearly concurrent with the extent of the Appalachian Coal Basin. American chestnut was historically a dominant species in the eastern forest comprising up to 50% of basal area in some stands (Russell 1987) and was important both economically as a timber source and as a wildlife food source. American chestnut became functionally extinct as a canopy component after the introduction of the chestnut blight in the early 20th century. The species remains predominantly as a shrub, resprouting in the understory but rarely reaches sexual maturity before succumbing to repeat infection of the blight (Paillet 1993).

The American Chestnut Foundation (TACF) has conducted a 25-year program to breed a blight-resistant chestnut tree (Burnham et al. 1986). The backcross-breeding program seeks to incorporate the blight-resistance of the Chinese chestnut (Castanea
mollissima Blume) into a tree that will be functionally equivalent to the American parent (i.e., a tall-growing, straight-trunked timber tree). TACF is currently beginning field trials to gain insight into the field performance of the trees prior to large-scale deployment (Jacobs 2007). A successful reintroduction program for the species will yield multiple benefits including increased forest cover, enhanced mast for wildlife, potential timber value, and benefits for carbon sequestration.

Due to the decline of chestnut prior to the era of modern scientific investigation, there remains a paucity of data regarding its establishment requirements. However, some recent work has attempted to shed light on the site requirements and best establishment practices for American chestnut in different ecosystems (McCament and McCarthy 2005, Rhoades et al. 2009, Robertson and Davis 2011). Research conducted on reclaimed strip mines in Ohio has shown American chestnut to be a well-performing candidate species when treatments to relieve soil compaction are applied (McCarthy et al. 2008) (Fig. 2.3). McCarthy et al. (2010) reported survival rates of ~70% in chestnuts introduced as bare-root seedlings on mine sites in eastern Ohio. Merging the Forestry Reclamation Approach and the reintroduction of American chestnut provides a unique opportunity to restore large tracts of the eastern deciduous forest damaged by surface mining. Ultimately, an understanding of what factors determine the success of chestnut restoration plantings will increase the chances of achieving the goal of returning the species to a functional presence in the Eastern Forest.
I sought here to determine what effects some elements of the microenvironment have on the establishment and performance of the target species, American chestnut, on reclaimed mine sites. The factors considered included: 1) distance from remnant habitat, 2) amount of existing vegetative cover, 3) microsite slope position and aspect, and 4) soil nutrient availability. To this end, I examined these questions on a reclaimed mine site in Ohio employing the relatively new, compaction-reducing reclamation method.

**Methods**

**Study Site**

The study was conducted at the Jockey Hollow Wildlife Management Area (JHWMa) in Belmont County, Ohio (40° 11’ 42.25” N, 81° 06’ 10.23” W, elevation 365 m.). The climate of eastern Ohio is humid-continental (Peel et al. 2007). Temperatures range from -7 °C (mean January low) to 27.8 °C (mean July High), mean annual precipitation is 103.3 cm (NCDC, Cadiz, OH). The site was procured by the Ohio Department of Natural Resources in 2004 and is now managed primarily as a public hunting facility. The site is composed of some second-growth hardwood forest, but mostly rolling grasslands that are the result of previous mining activities and reclamation via SMCRA-approved methods. Portions of the site were re-mined for coal from 2007–2009 and reclaimed using “end-dump” reclamation.

**Seedling Source and Planting Methodology**

All seedlings utilized in this study were nursery-grown and lifted as 1-0 bare-root pure American chestnut seedlings from the Ohio Division of Natural Resources, Marietta
State Tree Nursery (Ohio Dept. of Natural Resources, Marietta OH). All seedlings were planted by excavating a hole large enough to accommodate the root mass (as opposed to dibble-bar planting). Previous plantings of American chestnut in eastern Ohio have shown extensive levels of browsing. To protect from deer and small mammals, each seedling received a tree planting tube (120 cm tall; Plantra Inc. Mendota Heights, MN; www.plantra.com). All seedling planting was conducted in March 2009.

**Experiment 1: Distance to Forest Edge**

In order to examine the effect of distance from remnant habitats, 240 chestnut seedlings were planted along 40 transects at distance intervals of 5, 20 and 50 m extending from the edge of adjacent remnant forest into a previously mined area reclaimed using the FRA end-dump method. There was no appreciable topographic gradient along the transects. Seedlings were monitored monthly for survival during the 2009–2011 growing seasons. Growing season was defined by the dates where temperatures had less than a 50% chance of falling below 0° F (-18° C, Bauerlein 2005). Growth was measured as the increase in height at the terminal bud from before bud break (measurement taken in March) through late fall (measurement taken in early November) to account for any browse or dieback that may have occurred during the winter between growing seasons.

**Experiment 2: Mound Position Effects**

Extremes in climate in the central Appalachians (the location of this study site) are found on the southwest- and northeast-facing slopes with the southwest-facing slopes
exhibiting the driest conditions (Tajchman et al. 1988, Boerner 2006). This experiment examined whether differences in microclimate could be observed at the scale of individual mounds, created by the FRA reclamation approach. To evaluate this effect, 240 seedlings were planted on 80 mounds in an area reclaimed using the FRA approach such that on each mound I planted one seedling each on the northeast side, southwest side, and the top of each mound (i.e., three seedlings per mound). Seedlings were monitored for survival during the 2009–2011 growing seasons. Growth was measured as the increase in height at the terminal bud from before bud break (measurement taken in March) through late fall (measurement taken in early November) to account for any browse or dieback that occurred during the winter between growing seasons.

Experiment 3: Effects of Competing Vegetation

The portion of the study site used for these experiments had been mined and reclaimed in the fall and winter of 2008–2009. No herbaceous ground cover was purposefully seeded at these sites during the reclamation process. Thus, all vegetative cover for this experiment originated from the seedbank or dispersal of propagules from sources nearby. After vegetative emergence, the vegetative cover on each mound utilized for Experiment 1 (Distance to Forest Edge) and Experiment 2 (Mound Position Effects) was visually estimated (200 mounds total; 120 from Experiment 1, 80 from Experiment 2). Vegetation estimates were collected in July of each year (2009, 2010, and 2011). Vegetative cover was classified as being “light” (0 < 30% cover), “intermediate” (30 < 70% cover), or “heavy” (> 70% cover).
Soil Quality

To examine the effects of soil characteristics, soil samples were collected from 140 mounds at the site, across the site-specific experimental treatments. Soil samples were taken from plots in half the transects in Experiment 1 (Distance to Forest Edge; 60 samples) and the 80 mounds utilized in Experiment 2 (Mound Position Effects) in June 2010 (during the second growing season). The transects used for soil sampling were randomly chosen using their transect ID number. Ten, 5-10 cm deep soil cores were collected from each mound and homogenized. Soil samples were then transported to the lab and passed through a 2 mm sieve to remove large particles prior to analysis. Soil moisture was determined gravimetrically by drying at 85°C for 24 h. Soil pH was determined with a Fisher Scientific XL 15 pH meter (Fisher Scientific, Suwanee GA) using a 1:1 solution of soil and deionized water (Page et al. 1982). Soil total C and N were estimated via combustion using an Elementar C:N analyzer (Elementar, Hanau, DE).

Data Analysis

The effects of planting distance and mound position on survival were analyzed independently using the Cox proportional hazard likelihood model (Cox 1972) in the package “survival” in the R language (R Development Core Team 2010). Effects of vegetative cover were analyzed independently and ultimately included as a covariate for survival analyses for planting distance and mound position. Differences in annual growth rate between treatments in each experiment were analyzed using one-way analysis of
variance (ANOVA). If an $F$-test yielded a significant ($P < 0.05$) result, individual treatment means were subsequently compared using Tukey’s Honestly Significant Difference (HSD). Dependent variables were tested for assumptions of normality using the D’Agostino omnibus test for normality (D'Agostino et al. 1990) in the “fBasics” package in R and a Fisher’s $F_{\text{max}}$-test (Sokal and Rohlf 1995, Wuertz 2010) for homogeneity of variance in the base R stats package. Due to limitations of the experimental design (i.e., “distance from forest edge” and “mound position” were not measured in the same experiment, but vegetation cover was estimated for mounds in both experiments), it was only possible to test for interactive effects between distance to forest edge and vegetation cover in Experiment 1 and between mound planting position and vegetation cover in Experiment 2. Thus, the effect of vegetation cover was analyzed as an independent effect in experiment-3 and as a covariate in experiment-1 & -2. Correlations between measured soil variables and annual growth rate (AGR) were analyzed via Pearson’s product-moment correlation coefficient (Zar 1999). AGR was determined as the difference in height at the terminal bud of the seedling between the end of the growing season and just prior to leaf flush in the preceding spring.

**Results**

*Experiment 1: Distance to Forest Edge*

Distance from remnant habitat showed a significant effect on seedling survival rates (Cox proportional hazard model; Likelihood = 33.76, df = 2, $P < 0.001$). Seedlings planted on transects 5 m from the remnant forest exhibited significantly lower survival
rates, whilst seedlings at the 20 and 50 m distances did not show significantly different survival rates between positions (Fig. 2.4). Distance from remnant habitat did not have an effect on AGR in the first year but these values were significantly lower at the adjacent planting position the second year ($F = 15.22$, df = 1, $P < 0.01$). Third year AGR was significantly lower in the adjacent and 20 m positions compared with the 50 m positions ($F = 17.49$, df = 1, $P < 0.001$; Fig. 2.5).

**Experiment 2: Mound Position Effects**

Planting position relative to the “mound” had a significant effect on seedling survival (Likelihood ratio test = 4.07, df = 2, $P < 0.05$), seedlings planted at the top mound position had greater mortality than seedlings at either aspect on the shoulder of the mound (Fig. 2.6). Relative planting position also showed a significant effect on annual growth rate in the first year ($F = 9.72$, df = 1, $P < 0.01$) where seedlings planted at the top mound position showed a significantly lower annual growth rate compared to seedlings planted at either aspect position on the shoulder of the mound. Second year growth results were not significantly different among planting positions. Growth rates were significantly lower in the third year at the top position compared to either shoulder aspect ($F = 27.89$, df = 1, $P < 0.01$; Fig. 2.7).

**Experiment 3: Effects of Competing Vegetation**

I found distinct differences in survival rates as a function of vegetative cover classes (Cox proportional hazard model Likelihood = 6.6, df = 2, $P < 0.01$). Heavy vegetative cover plots showed significantly higher survival rates compared to
intermediate and low vegetative cover plots (Fig. 2.8). Annual growth rate was significantly different among all vegetative cover classes in the first year \((F = 38.15, \text{df} = 1, P < 0.001)\). Second year AGRs were significantly lower in the light vegetative cover plots \((F = 13.67, \text{df} = 1, P < 0.001)\). In the third year, AGR was significantly higher in the intermediate vegetative cover plots \((F = 8.03, \text{df} = 1, P < 0.01)\) (Fig. 2.9). The dominant herbaceous species on the mounds were *Phytolacca americana* L. (pokeweed), *Polygonum sagittatum* L. (tear-thumb), *Polygonum pensylvanicum* L. (smartweed), and *Tussilago farfara* L. (colt’s foot). There was no significant interaction between distance from the forest edge and vegetation cover on growth rate (Distance*Cover for years 1–3, \(F = 0.005, 0.048, 0.440\) and \(P = 0.94, 0.83, 0.51\), respectively). Likewise, there was also no significant interactive effect between vegetation cover and mound planting position (Yr-1, \(F = 2.75, P = 0.09\); Yr-2, \(F = 0.404, P = 0.53\); Yr-3, \(F = 0.743, P = 0.389\)).

**Soil Quality**

Soil characteristics varied widely across the site. As expected, soil moisture and AGR were significantly correlated \((r = 0.0094, P = 0.0143)\). Mean soil pH was 6.48 with a range of 5.65–7.85. Soil metrics differed between some of the vegetative cover classes. Plots classified as having heavy vegetative covered showed significantly higher soil N levels \((F = 107.47, \text{df} = 1, P < 0.001)\).

**Discussion**

The performance of desirable species in any reclamation project can be influenced by the creation of optimal regeneration sites for that species. In many cases, creation of
species-targeted sites may not be possible at the time of reclamation work or physical site preparation. Where this occurs, an understanding of the effects of the microsite conditions that will be created can increase the chances for successful reclamation and/or restoration. Species-specific sensitivity to environmental conditions has been observed in various studies on unreclaimed mine spoils (e.g., Brown 1962) but this study is one of the first to report on conditions found on sites reclaimed using end-dumping methods. Applying the Forestry Reclamation Approach may yield a successful return to forest cover but considerations need to be taken at the time of planting in order to maximize performance of desirable species. Mine reclamation sites may provide an interesting venue for the restoration of the American chestnut (McCarthy et al. 2008, McCarthy et al. 2010, Fields-Johnson et al. 2012), which I have focused on here.

A significant, negative edge effect on establishment success was observed at the study site. Most studies of establishment in relation to distance from forest edge have reported greater establishment success closer to the forest edge, attributed to lessened dispersal distance from parent trees. This is especially true in nut-producing trees where effective dispersal distances are shortened by the weight of the seed and reliance on dispersers with different ranges. However, with planted seedlings, dispersal distance would not be a factor in determining establishment with relation to an existing forest edge. Thus, factors other than dispersal distance are likely responsible for the pattern of survival seen in this study. The shade-tolerance status of American chestnut has been examined in several studies (Wang et al. 2006, Joesting et al. 2007) and demonstrated to
be mid-tolerant. Thus, it is possible that seedlings at the 5 m position from the forest edge might have suffered from shading effects associated with the existing forest edge throughout parts of the day. Significant shading effects on establishment success adjacent to remaining forest edges have been observed in plantation settings before (Hansen et al. 1993).

In orchard settings, extensive vole damage has been observed on American chestnut seedlings planted with protective tubes, which may be providing refugia for girdling rodents (pers. obs., B.C. McCarthy). Most mounds at the site remained relatively non-vegetated for the first growing season and therefore provided little to no cover for rodent species. Seedling mortality increased dramatically in the second year of study, suggesting rodent cover provided by increasing vegetation may be influencing mortality via rodent gnawing. A similar observation has been made by Ida and Nakagoshi (1996) with Quercus and Fagus seedlings in temperate woodlands. Rodent predation pressure has also been found to differentially operate at varying distances from remnant habitat (Ostfeld et al. 1997). Browsing damage from large ungulates did not seem to be great enough to warrant further protection than provided by the planting tubes used in the studies. Browse damage in the form of missing terminal buds was not apparent on trees that had grown out of their protective tubes at the site during height measurements. Although not screened for in this study, phytophthora root rot (Phytophthora cinnamomi Rands) could have contributed to chestnut mortality (Anagnostakis 2001). Although the disease is considered to be a possible hindrance to restoration projects (Jacobs 2007), it is
typically a much greater concern in more heavily compacted, wetter soils (Rhoades et al. 2003). In a similar study conducted on uncompacted mine spoils in Kentucky, phytophthora was not observed (Adank et al. 2008). Additionally, greater mortality was seen in this study at the tops of mounds where conditions are likely to be drier and thus phytophthora incidence would be decreased, as well as a positive correlation between soil moisture and AGR. However, it is possible that seedlings planted closer to the forest edge would be more likely to be first infected by *P. cinnamomi* if the oomycete pathogen is present in the soil surrounding the reclaimed site. Future studies on FRA-reclaimed sites will need to determine the colonization abilities of phytophthora across a broad range of site locations to elucidate the interaction between climate, soil conditions and phytophthora-mediated chestnut mortality.

Heavier vegetative cover may not be competing with desirable chestnut species for light and soil moisture as was originally expected. Vegetation cover at the site did not overtop the planted seedlings during the observed years (the dominant species tend to form sprawling “mats”). In fact, heavier vegetative cover may be providing ecological benefits such as soil shading which lowers soil temperature and allows more soil moisture to be retained throughout the summer months. This facilitation may be of particular import in an exposed environment such as this (Bell and Ungar 1981). Additionally, higher soil nitrogen levels were evident on mounds with greater vegetative cover. The mechanism for increased nitrogen could be either through microbial action, decay of plant material, or presence of nitrogen-fixing plant species (which were present
but not the dominant vegetation at the site). However, correlation analysis between soil variables and seedling AGR showed no relationship between increased soil N and seedling performance. The lack of a relationship between soil N and chestnut growth suggests that previous levels of N on the site were probably sufficient for vegetation growth, and that the vegetation itself facilitated chestnut growth, probably through decreased solar exposure and evaporation. Additionally, root collar diameter was not measured, which may have given a better picture of nutrient effects on growth than only measuring height (Cannell 1989). Only total N was measured; measurement of biologically available nutrients would yield a clearer picture of the soil nutrient dynamics at work at sites reclaimed using the FRA.

Over the years, various forms of mounding have been used in restoration projects in an attempt to enhance establishment success of desired species (Sutton 1993, Löf et al. 2006). However, most of these studies have utilized mounds on the scale of centimeters whereas the mounds utilized in the FRA strategy are created on a meter scale. A typical mound is 2-3 m tall and 5-8 m across. It is likely that the mounds used here experience differences in the physical environment closer to those seen in tip-up mounds (e.g., Peterson et al. 1993) rather than mounds created for very fine-scale topographic relief in bottomland restoration (Anderson et al. 2007). The planting medium found at the top of the mound is often relatively unstable and prone to subsidence. Rhoades et al. (2009) found chestnut survival to be limited on extremely xeric sites where sand and coarse fragments (> 2mm) comprised more than 50% of the bulk soil volume. On the mounds
created by the FRA strategy, finer particles typically accumulate in the lower swale portions between the mounds leaving the upper portions typically coarser in texture. Coarse fragments here represented 30–40% at the side and top portions of the mounds respectively (Gilland, unpublished data). Seedlings planted at the top of the mound may then be more prone to drought in the extremes of the season or during periodic drought events. A minor concern with the FRA end-dump method is that in providing a looser rooting medium than previously found on reclaimed mine sites, excessive drainage may become a problem. This problem can be alleviated by not planting seedlings at the uppermost positions on the mounds where soil is less stable and prone to drying. Survival may also be practically increased through the inclusion of a moisture-retaining agent at the time of planting (Alm and Stanton 1993). American chestnut survival has been observed to increase after the application of a moisture-retaining polymer on reclaimed mine sites but these results were found only under direct-seeded conditions (Miller 2010), not with bare-root seedlings as used here.

The range of pH on the site was similar to what has been found at other reclaimed mine sites in Ohio (Zeleznik and Skousen 1996, Shukla et al. 2004). The reintroduction of American chestnut via mine land reclamation has been thought to be limited at circumneutral pHs. The pH values observed (slightly acidic to circumneutral) do not seem to have affected the performance of American chestnut on these sites as no correlation was found between growth and soil pH. Naturally-occurring trees are observed in greatest abundance at pH 4–6 and in extremely low abundance at soil pH.
levels above 6 or on otherwise calcareous soils (Russell 1987, Tindall et al. 2004). However, these studies are based on historic observations rather than experimental studies. Rhoades et al. (2009) reported growth rates roughly equal or slightly greater than the results seen here on “xeric” sites (i.e., ~10 cm yr$^{-1}$) and much greater growth rates on more mesic sites at pH of 4.3–4.6.

The relative scarcity of American chestnut in the landscape has likely held back much ecological research on the species. The promise of a blight resistant chestnut in the near future has promoted many researchers to begin looking at the ecology of this species in greater detail. The state of American chestnut restoration research now stands at a transition from generally greenhouse-based studies to field-based reintroduction trials. As blight-resistant hybrid trees become more freely available, the volume of research into the ecology of this species will likely become equivalent to that of other similar hardwood species. The Forestry Reclamation Approach is a new and novel reclamation method and the promising results that have been observed to date using the target species of this research (American chestnut) seem to indicate that this method may be very effective at establishing chestnut trees on lands degraded by surface mining.

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Figures and Legends

Figure 2.1: Arrested succession common on sites reclaimed to pasture under the Surface Mine Control and Reclamation Act (SMCRA). This site, photographed in 2010, was reclaimed in 1978.
Figure 2.2: Application of the end-dump reclamation performed at the Jockey Hollow Wildlife Management Area in eastern Ohio. The end-dump method results in a series of mounds of loose material that serve as a non-compacted rooting medium to promote tree growth.
Figure 2.3: Three-year-old American chestnut growing at the Jockey Hollow Wildlife Management Area in eastern Ohio where end-dump reclamation has been performed. The end-dump method results in a series of mounds of loose material that serve as a non-compacted root.
Figure 2.4: Experiment-1. Survival results for American chestnut at JHWMA by distance from remnant forest habitat. Lines indicate total proportion of surviving trees by planting position over time.
Figure 2.5: Experiment-1. AGR results for American chestnut at JHWMA by distance from remnant forest habitat. Significant differences in annual growth rates were observed among planting distances in years two and three. Boxplots follow standard construction: box represents interquartile range (25–75\textsuperscript{th} percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at $P < 0.05$ determined by Tukey HSD.
Figure 2.6: Experiment-2. Survival results for American chestnut at JHWMA by relative position on the “mound” created by the FRA strategy. Lines indicate total proportion of surviving trees for each planting position at each sampling time. Trees planted at the top of the mound showed significantly lower survival rates ($P < 0.05$, Cox proportional hazard) compared with seedlings planted on the shoulder of the mound at either aspect.
Figure 2.7: Experiment-2. AGR results for American chestnut at JHWMA by relative position on the “mound” created by the FRA strategy. Significant differences in annual growth rates were observed in years one and three. Boxplots follow standard construction: box represents interquartile range (25–75\textsuperscript{th} percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at $P < 0.05$ determined by Tukey HSD.
Figure 2.8: Experiment-3. Survival results for American chestnut at JHWMA by vegetative cover class. Mounds with heavy vegetative cover (> 70%) showed a significantly (P < 0.05) higher survival rate compared with intermediate (70% > 30%) and light (< 30%) vegetative cover (which did not differ from each other).
Figure 2.9: Experiment-3. AGR results for American chestnut at JHWMA by vegetative cover class where “light” (0 < 30% cover), “intermediate” (30 < 70% cover), or “heavy” (> 70% cover). Significant differences in AGR were noted among vegetative cover classes in all three years. Boxplots follow standard construction: box represents interquartile range (25–75th percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at P < 0.05 determined by Tukey HSD.
CHAPTER 3: VARIATION OF AMERICAN CHESTNUT (*CASTANEA DENTATA*) HYBRIDS ON NEWLY RECLAIMED MINE SITES IN EASTERN OHIO, USA

Abstract

Disturbance and succession are key factors in plant community dynamics. Surface mining for coal represents a significant form of anthropogenic disturbance on the landscape. Currently there is more than one million hectares of former mined land in the United States. Much of this land exists in a state of arrested succession as a result of previously used reclamation methods. New reclamation procedures are being examined to determine if they can aid in a return to original forest cover on former mine sites. The study was conducted on public lands that had been previously surface-mined for coal, reclaimed in 1978, and re-mined and reclaimed using new methods in 2007. In March 2008, 535 American chestnut seeds were planted at the study site in 3 experimental blocks of 38 to 50 replicates. Each replicate contained 5 seeds from 5 different genetic lines including 3 hybrid Chinese-American lines. Results of the planting experiment showed Chinese chestnut and early-generation hybrids to have significantly better performance based on growth and survival measurements. Morphological differences between the hybrid lines may be responsible for these differences as trees with a greater percentage of Chinese parental material possess a suite of leaf characters that may make those hybrids better suited for the arid, high-light conditions found on reclaimed mine sites. These results highlight the necessity for further exploration of the interactions between site and species characters in determining the outcomes of restoration projects.
Introduction

Disturbances ultimately result in a change to the vegetative composition or physical environment of a local area (Pickett et al. 1987). The communities that develop on disturbed sites are largely a function of propagule and microsite availability (Eriksson and Ehrlén 1992). When propagule availability is not limiting in a restoration project (i.e., where large numbers of seeds or individuals are anthropogenically introduced at a site), the adaptative ability provided by characteristics of the target-species to the disturbed area may influence establishment (Noble and Slatyer 1980). Understanding trait-environment interactions has the potential to aid greatly in the design and implementation of successful restoration projects (Funk et al. 2008).

Surface mining for coal represents one of the largest anthropogenic disturbance regimes in the United States. Since the passage of federal reclamation statutes in the late 1970s, mine sites have been mostly reclaimed as pasture or hay-land (Barnhisel and Hower 1997). Sites reclaimed to grass-land per federal mandate (Gerard 2000) frequently exhibit “arrested succession” and rarely return the previously existing state (i.e., forest) found on the site (Holl 2002). In addition to a lack of structural and species similarity, these systems also do not usually attain the same functions as the forest ecosystems existing on most of these sites in eastern North America (Simmons et al. 2008).

Nearly one million hectares have been affected by surface mining in the United States and roughly 40% of these lands are located in the Appalachian Coal Basin (USGS 2011). Concurrent with the extent of the Appalachian coal basin is the original region of
dominance of the American chestnut *Castanea dentata* Marsh. (Borkh.). American chestnut was once an important component of the Eastern Deciduous Forest (Braun 1950), making up nearly 25% of the basal area in certain locales (Paillet 2002). The species was an important wildlife food source and timber producer (Jacobs 2007). The introduction of the non-native pathogen *Cryphonectria parasitica* (Murrill) Barr, which causes the chestnut blight, effectively removed American chestnut from the forest canopy by the early 1950s (Loo 2009). In order to return the American chestnut to the eastern forest landscape, the American Chestnut Foundation instituted a backcross breeding program with blight-resistant Chinese stock starting in 1983 (Griffin 2008). Large enough quantities of hybrid trees are now becoming available for reintroduction testing purposes (Jacobs 2007). The hybrid chestnut trees currently in preparation for release from the American chestnut foundation are reported to be morphologically identical to the American chestnut (Diskin et al. 2006). Former coal mine sites that are currently being reclaimed using new, experimental compaction-lessening methods may be able to serve as ideal locations for the reintroduction of American chestnut. Early studies have shown impressive growth and survival by American chestnut on relatively inhospitable reclaimed mine sites (McCarthy et al. 2008, McCarthy et al. 2010, Gilland and McCarthy 2012).

Phenotypic plasticity can allow genetically similar organisms to respond to differences in habitat types or potential environmental stressors. High levels of phenotypic variation and adaptation between xeric and mesic sites have been observed
among many eastern hardwood species. For example, leaves of trees living in xerophytic environments tend to be smaller and thicker (Abrams et al. 1990) than those living in mesophytic environments. In some cases, phenotypic stress responses in identical genotypes may outweigh differences among regionally adapted genotypes (Abrams 1990, Abrams 1994). Under the extreme conditions found on reclaimed mine sites (e.g., high sunlight, dry soil), leaf characteristics of the Chinese parent of the blight-resistant hybrid chestnut may provide an adaptive advantage and may occur despite selection against that phenotype in the breeding program. Specifically, increased Leaf Mass per Area (LMA; an indicator of relative thickness and characteristic of the Chinese parent) may allow for better heat dissipation (Smith et al. 1997), while increases in leaf pubescence (another characteristic of the Chinese parentage) may also provide adaptive benefits under droughty conditions by increasing boundary layer resistance, reducing transpiration, and increasing albedo (Johnson 1975). Despite selecting specifically for “American” leaf characteristics, it is possible that more characteristic “Chinese” characters will be expressed in an adaptive manner under harsh field conditions among the hybrids.

The goal of this research was to determine if differences in overall performance (3-year survival and yearly height growth) as well as leaf and stem morphological or chemical metrics exist between parent and hybrid lines of American chestnut under field conditions. These results will provide insight to the adaptive behavior of the species when introduced on reclaimed mine sites and also provide insight in to the degree to which
desired morphological characteristics are expressed under adverse field conditions in a restoration setting after an artificial selection process.

Methods

Site Description

The study was conducted at the Jockey Hollow Wildlife Management Area (JHWMA) in Belmont County, Ohio, USA (40° 11’ 42.25” N, 81° 06’ 10.23” W, elevation 365 m). Temperatures range from -7 °C (mean January low) to 27.8 °C (mean July high), mean annual precipitation is 103.3 cm (NCDC, Cadiz, OH). The site was reclaimed using Forestry Reclamation Approach (FRA) methods to prepare a portion of the site that had been surface-mined for coal. The site was reclaimed in the winter of 2007 and at the time planted predominantly with a mixture of native, nut-producing hardwood seedlings. The FRA method used to reclaim the site is as follows: prior to seed planting, contour was restored with the final layer loosely dumped in piles to yield a less compacted surface in a process known as “end-dump” reclamation. This yields a surface consisting of a series of large mounds of mining overburden approximately 8–10 m in diameter and 3–5 m high (Fig. 3.1).

Hybrid Genetic Line Growth and Survival Comparisons

American chestnut seeds were planted at the Jockey Hollow Site in March 2008 in 3 experimental blocks of 38 to 50 replicates each. Two blocks were placed on opposing east and west aspects of roughly equal slope (25%), and the third was located at the top of the east-facing slope on a relatively flat terrain. The individual mounds making
up the 3 blocks were treated as replicates and were planted with 1 individual of each of 5
different genetic lines. These lines included pure American, pure Chinese, and 3
individual hybrid lines from the American Chestnut Foundation’s breeding program
(B$_1$F$_3$; B$_2$F$_3$; B$_3$F$_1$). These hybrids represent a successive order of “advancement” in the
American Chestnut Foundation’s breeding scheme (Burnham et al. 1986) toward the
ultimate goal of a tree that is morphologically indistinguishable from the American parent
but still retains the Chinese parent’s blight resistance. The 5 seeds were planted in
random order around the “shoulder” of each mound. Each seed received a 30 cm tall
protective tree-tube (i.e., plastic sleeve) buried halfway in the ground to prevent rodent
predation prior to germination as well as mechanical damage from rodents after
germination. Survival was assessed monthly for the first growing season and then at the
beginning and end of each subsequent growing season for the following 2 years. Height
to the terminal bud was measured prior to leaf-out in the spring and again in late October
of the same growing season to determine Annual Growth Rate (AGR, cm yr$^{-1}$).

Leaf Characteristics

A sample of ten branches from each phenotype was collected in September 2011
for measurement. After returning from the field, all leaves were removed from the twigs,
pressed and dried. After pressing and drying, leaf area was determined for all sampled
leaves using digital scans made of the leaves on a 1200 DPI flatbed scanner and the
software package ImageJ (http://rsbweb.nih.gov/ij/). After determining area, dry mass for
each leaf was determined in order to calculate leaf mass per area (g cm$^{-2}$). ImageJ was
also used to measure leaf:width ratio (cm). Stomatal density was measured using leaf peels and an Olympus CX41 light microscope at 40× magnification (Olympus America Inc., Melville NY). Pubescence on the underside of the leaf was determined visually using an Olympus SZ61 stereoscope using a visual scale from 1 (least pubescent) to 5 (most pubescent).

**Data Analysis**

All statistical analyses were performed using the R programming environment (R Development Core Team 2010). Differences in survival among lines were analyzed using Cox proportional hazard analysis in the R package “survival” (Therneau and Lumley 2010). Differences in AGR were analyzed with a Kruskal-Wallis test when assumptions of normality could not be met despite transformation attempts. LMA, leaf length:width ratio, and stomatal density were measured using one-way analysis of variance (ANOVA) after testing for assumptions of normality using the D’Agostino omnibus test for normality (D’Agostino et al. 1990) in the “fBasics” package in R and a Fisher’s $F_{max}$-test (Sokal and Rohlf 1995, Wuertz 2010) for homogeneity of variance in the base R stats package. Means were compared using Tukey’s Honestly Significant Difference (HSD) when an $F$-test yielded a significant ($P < 0.05$) result (Tukey 1977). Pubescence scores were analyzed using a chi-square goodness of fit test for categorical data (Zar 1999).
Results

Hybrid genetic line growth and survival comparisons

Seeds that had germinated by July 1, 2008 were considered to have successfully germinated for the year. Germination percentages ranged from 81% (pure American) to 90.2% (pure Chinese). Hybrid germination rates were intermediate between both parental lineages. Significant differences in survivorship over the 3 years were observed among hybrid lines (Cox proportional Hazard Likelihood = 16.98, df = 4, \( P = 0.002 \)). Individuals from the pure American genetic line fared worst over the study period with final survival rates near 35%, while the B1F3 hybrid lines had the greatest survivorship at nearly 60% (Figure 3.2).

A significant difference in height (\( F = 39.3, P < 0.01, \text{df} = 4 \)) was also observed among genetic lines in the first year with the pure American line showing significantly lower growth rates than the hybrid and pure Chinese lines. This pattern continued for the next 2 years where pure Chinese lines and earlier hybrid lines had greater Annual Growth Rates than pure American and later backcross lines (Fig. 3.3).

Leaf Characteristics

Leaf characters exhibited a somewhat linear gradation across the hybrid lines. Trees of pure Chinese origin, as well as those from the first and second backcross, had significantly thicker leaves than individuals from the third backcross line and pure American trees (\( F = 11.860, \text{df} = 4, P < 0.01, \text{Fig. 3.4} \)). Leaf length to width ratio was similar but not equivalent to the pattern of thickness. Pure Chinese and first backcross
trees had significantly lower L:W ratios than Pure American, BC₃, and BC₂ trees ($F = 14.780, \text{df} = 4, P < 0.001$). Stomatal density was not significantly different among any of the parents or hybrid lines ($F = 1.204, \text{df} = 4, P = 0.31$). Pubescence was not randomly distributed throughout generations ($\chi^2 = 242.078, \text{df} = 16, P < 0.01$). Instead, pubescence scores were distributed in a clumped fashion across generations where each successive backcross generation was less pubescent than the previous (Fig. 3.6).

**Discussion**

Linking broader ecological theory with the practice of ecological restoration will likely yield more successful restoration projects (Palmer et al. 1997). The importance of dispersal limitation versus niche limitation in community assembly has been widely debated (Stevens et al. 2004). However, in habitats undergoing restoration efforts where large amounts of propagules are being anthropogenically introduced, the interaction of environmental suitability and species-specific characteristics may be the determining factor in establishment success (Pywell et al. 2003). Here, I have observed some level of habitat preference evident among hybrid lines of chestnut that appear to possibly be underlain by difference in these lines’ adaptive morphology. Walker et al.’s (2006) study found plant characteristics to not be indicative of early colonization success. However, their work considered traits in the form of broader functional groups (e.g., grasses, forbs, N-fixers). Here, I attempted to use much narrower, species-specific traits which were likely to vary among hybrid lines that I also felt could be ecologically meaningful in the context of mine land restoration.
Leaf Mass per Area (LMA) can be effectively used as a proxy for leaf thickness or robustness. It has been proposed and generally accepted that thicker leaves or those with higher LMA are characteristic of arid or xeric environments (Givnish 1987). Increased LMA has been correlated with both decreasing precipitation as well as increasing site irradiance (Jurik 1986, Wright et al. 2004). Here, seedlings of pure Chinese ancestry had significantly thicker leaves compared with their pure American counterparts. This increased thickness may make pure Chinese individuals and those with a greater percentage of Chinese characteristics better suited for the high-light and relatively xeric conditions typically found at exposed mine sites reclaimed using the FRA end-dump method. The leaf-width ratio results were contrary to expected under the idea that pure Chinese and early hybrid lines might be better adapted to high-light conditions. Narrower leaves are more common in arid environments due to greater heat loss ability (Fonseca et al. 2000) and increased water use efficiency (Parkhurst and Loucks 1972). Here, I observed the opposite where Chinese trees and earlier parents had wider leaves (Fig. 3.5) than the American parents and later hybrids. The lack of significant differences in stomatal density was not unexpected as most studies examining stomatal density have found it to be under greater environmental control than inherent genetic control (Woodward 1987, Woodward and Bazzaz 1988). The increasing levels of leaf pubescence observed here may have a similar effect in mitigating water or light stress incurred on reclaimed mine sites (Gutschick 1999). Increased abaxial leaf pubescence in
pure Chinese and earlier hybrids may aid in water retention on mine sites by increasing boundary layer resistance and reducing water loss via transpiration (Press 1999).

These results indicate that many of the desired characteristics in the hybrid offspring can be recovered sufficiently via the American Chestnut Foundation’s breeding program in the allotted number of generations. Diskin et al. (2005) found most of the desired characteristics to be recovered in fewer generations than were needed here but the trees utilized in that study were grown in nursery conditions at the ACF’s Virginia breeding facility. This study also highlights the need for more comprehensive examinations of the biotic conditions found at mine sites reclaimed using the FRA end-dump method. The studies published thus far mostly utilize abiotic data collected at one point in time (e.g., gravimetric soil moisture) that may not adequately reflect the conditions realized by the trees being established on the site.

The selection process utilized in American chestnut could effectively utilize the data generated from studies like this to better refine the restoration breeding program. A focus on morphological fidelity to the American parent may ultimately not be the best approach for restoration depending on the reintroduction context. This is an issue that will likely not be easily solved as it gets at the heart of the theory of restoration ecology; namely the arrival at a suitable target for restoration. If simple mine land reclamation to wildlife habitat is the stated goal, then hybrid chestnuts displaying a more drought-tolerant, but “Chinese” morphology might yield better results at least initially. However, this is likely not long-term solutions as pure Castanea mollissimma trees are not likely to
achieve canopy status due to their inherent “low, spreading” architecture. Trees that are shade intolerant and unable to reach canopy stature (like the Chinese chestnut) would be unable to play a role in long-term forest establishment, hence the goal of an upright, timber-type “American-form” tree. Development of goal-specific cultivars by the American chestnut foundation might aid in more successful restoration attempts (e.g., a “mineland” cultivar, a “shade tolerant” cultivar, etc.) but also may limit genetic diversity in those lines if intensive inbreeding or cultural cloning is utilized. Additionally, restoration at the FRA end-dump sites may need to pursue a phased system of introduction suggested by Pywell et al. (2003) after initial extreme environmental conditions have been ameliorated by the use of early-successional species in order to ensure the success of desirable species.

Conclusions

The combination of the Forestry Reclamation Approach and the American chestnut may turn out to be an effective tool for the restoration of large portions of the Eastern Deciduous Forest affected by surface mining. Additional investigation into the identification of and interaction of traits that govern plant community assembly on reclaimed mine lands will likely aid in determining more successful restoration strategies. Whilst the overarching goal of the American Chestnut Foundation is to retrieve a blight resistant tree with the morphological similarity of the American parent through its breeding program, there may remain a significant use for intermediate hybrids that exhibit slightly different traits.
Acknowledgements

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leads to altered ecosystem structure and function. Ecological Applications 18: 104–118.


Figures and Legends

Figure 3.1: Application of the end-dump reclamation method performed at the Jockey Hollow Wildlife Management Area in Belmont County, Ohio, USA. The end-dump method entails placing stockpiled topsoil and overburden loosely in large mounds designed to be a non-compacted rooting medium to promote tree growth as opposed to older, compaction-inducing methods.
Figure 3.2: Survival results for American chestnut at JHWMA by hybrid genetic line. Pure American trees showed significantly greater mortality ($P < 0.01$) than pure Chinese trees and trees closer to the Chinese parentage (i.e., $B_1F_3$ hybrids).
Figure 3.3: AGR results for American chestnut at JHWMA by genetic line. Significant differences in annual growth rates were observed between lines in all 3 years. Boxplots follow standard construction: box represents interquartile range (25–75th percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at $P < 0.05$ determined by Tukey HSD.
Figure 3.4: Leaf mass to area ratio for American chestnut at JHWMA by genetic line. Boxplots follow standard construction: box represents interquartile range (25–75\textsuperscript{th} percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at $P < 0.05$ determined by Tukey HSD.
Figure 3.5: Histogram showing the distribution of underside leaf pubescence scores in hybrid and parent lines of American chestnut. Scores ranged from 1 (least pubescent) to 5 (most pubescent).
CHAPTER 4 : MICROTOPOGRAPHY INFLUENCES EARLY SUCCESSIONAL PLANT COMMUNITIES ON EXPERIMENTAL MINE LAND RECLAMATIONS

Abstract

Surface mining for coal represents one of the most severe forms of anthropogenic disturbance to the forests of the eastern United States. Reclamation methods adopted under federal law after the late 1970’s have led to a state of arrested succession, with a general failure to achieve pre-disturbance conditions. New methods of reclamation have been proposed with the goal of returning mined land to its former forested state through the use of compaction reducing techniques that create large amounts of microsite heterogeneity. I examined the effect of fine-scale topographic relief, soil physical properties, and reclamation style on early plant community development on an experimentally reclaimed mine site in eastern Ohio. I sampled plots at four microsite types and at different distances from the remaining forest edge in both experimentally and traditionally reclaimed areas of surface mine. Non-parametric multivariate ANOVA on distance matrices indicated significant differences in plant community composition among microtopographic positions and reclamation types. These microhabitat positions also exhibited significant differences in measured soil properties (e.g., soil moisture, texture, bulk density), which significantly affected community composition. Plots in the traditionally-reclaimed areas had no woody plant colonization, indicating a state of “arrested succession” common to older sites reclaimed using traditional methods. These results suggest that the creation of heterogeneous microsites at the time of reclamation markedly accelerates succession and promotes enhanced plant community diversity.
Expanded application of the methods used here could allow for a faster return to the former forested state of highly disturbed lands than previously used reclamation methods.

**Introduction**

Disturbance and succession form the underpinnings of ecological restoration theory. A stronger integration of the known body of work concerning the succession dynamics of the system targeted for restoration will likely increase the probability of reaching the desired restored state (Walker et al. 2007). Additionally, restored systems can be used to study and test successional mechanisms (“acid test” of Bradshaw 1987).

Surface mining for coal represents one of the most severe disturbances to the forest of the eastern United States, resulting in a complete disruption of both the biotic and abiotic components of the ecosystem. Of the nearly one million hectares of land affected by surface mining in the United States, nearly 40% is located in the Appalachian Coal Basin (USGS 2011)—an area contained entirely within the range of the Eastern Deciduous Forest biome of North America (Braun 1950).

The 1977 Surface Mining Control and Reclamation Act (SMCRA) sought to remedy many of the land-degradation problems previously associated with the closing (and abandonment) of surface-mining projects (Wooley 1978). Reclamation to grassland or pasture has been the most widely adopted method of satisfying the law’s short-term vegetation requirements for bond release (Gerard 2000). However, this reclamation method often results in a decades-long state of arrested succession where little to no natural tree recruitment occurs, and grassland persists in what was once a forested
landscape (Ashby 1991, Chaney et al. 1995; Fig. 4.1). The lack of succession and recovery to the natural forested condition is thought to be the result of soil compaction associated with heavy equipment usage and erosion prevention, and aggressive, non-native ground cover species used to satisfy bond requirements (Holl and Cairns 1994, Holl 2002).

Observations of sites reclaimed prior to the passage of SMCRA show that in the absence of heavy compaction and aggressive ground cover species, the forest that develops on those sites resembles intact forests on nearby undisturbed sites (Skousen et al. 2006). On reclaimed mine sites where soil compaction has been relieved, increased growth among native woody species targeted in forest restoration projects has been observed (Ashby 1997, McCarthy et al. 2010). In light of these observations, new methods of reclamation have been proposed that encourage the use of compaction reducing techniques and less aggressive ground covers (i.e., the “Forestry Reclamation Approach (FRA)”; Zipper et al. 2011). The goal of restoration projects utilizing the FRA approach is a return to native forest cover through the implementation of new techniques that alter the physical environment while the site is undergoing post-mining reclamation activities. The FRA approach creates a high level of fine-scale microtopographic relief as part of the “end-dump” method of compaction reduction (Fig. 4.1 inset). Small scale topographic relief has been observed to affect plant community development in natural forest communities (Beatty 1984, Peterson et al. 1990) managed *Quercus* plantations in
Europe (Löf et al. 2006), forested wetlands (Anderson et al. 2007), and prairie restoration projects (Biederman and Whisenant 2011).

Much of restoration is focused on essentially “speeding up” succession to arrive at some desired endpoint. Most mine land reclamation studies thus far have focused on planting desirable native species during reclamation with a specific final forest overstory community in mind, while overlooking the species that naturally colonize the site during succession. Research indicates that while active restoration is being pursued at most sites, a natural process of colonization will exist concurrently that may in fact affect the final outcome of the restoration project (Hobbs et al. 2007). My goal was to characterize these natural colonization patterns as they occur in non-natural environments created using “traditional” and the new FRA “end-dump” methods.

Specifically, I asked:

1. Do differences exist among plant communities at different microtopographic positions, distances from the remaining forest edge, and sites reclaimed using “traditional” methods?

2. Do differences in soil physical properties exist among microtopographic positions and are these differences reflected in plant community composition?

3. Do differences in the density and composition of woody plant colonization exist with respect to forest edge and between end-dump reclaimed sites and sites reclaimed using “traditional” methods?
By observing the patterns of natural colonization at the site I hope to provide some of the first observational data on the nature of succession on sites reclaimed using the FRA end-dump method compared with traditional reclamation.

Methods

Study Site

The study was conducted at the Jockey Hollow Wildlife Management Area (JHWMA) in Belmont County, Ohio (40° 11’ 42.25” N, 81° 06’ 10.23” W, elevation 365 m). The climate at the site is classified as humid-continental (Peel et al. 2007), with temperatures ranging from -7° C (mean January low) to 27.8° C (mean July High) and a mean annual precipitation of 103.3 cm (NCDC, Cadiz, OH). The site was procured by the Ohio Department of Natural Resources in 2004 and is now managed primarily as a public hunting facility. The site is 1,400 ha composed of some un-mined second-growth hardwood forest, but is mostly dominated by rolling grasslands, which are the result of previous mining activities and reclamation via SMCRA-approved methods.

Portions of the site were re-mined for coal from 2007–2009 and reclaimed using a new experimental method of mine reclamation known as the “end-dump” method (~100 ha). The end-dump method of reclamation restores the approximate original contour of the land per SMCRA regulation while the uppermost layer of material is dumped into large piles to serve as a loose rooting medium (Fig. 4.1 inset), with no subsequent strike-off or tracking-in via heavy equipment.
**Effects of microtopography on plant community development**

Plant inventories were conducted at JHWMA in July–August 2010 using a stratified sampling scheme. Within the portion of the Jockey Hollow site reclaimed with the end-dump method, six 50 m transects were placed perpendicular to the forest edge into the reclaimed mine area. Along each of these transects, three 100 m² plots (10 × 10 m) were established at 5, 20, and 50 m from the forest edge for a total of 18 plots. Eight 1 m² subplots (1 × 1 m) were established within each of the larger 100 m² plots at the “swale”, NE, SW, and “top” position of the mounds created by the FRA end-dump method (2 subplots per mound position). Aspect differences were examined as extremes in climate in the central Appalachians (the location of this study site) are found on the southwest- and northeast-facing slopes with the southwest-facing slopes exhibiting the driest conditions (Tajchman et al. 1988). Complex interactions exist among topography, soils, and plant communities at a variety of scales (Boerner 2006) and I sought here to examine if these effects occur in my system of interest.

An additional three transects were sampled in a portion of the study area that was reclaimed at the same time as the FRA reclamation but using the traditional “pasture” or “grass land” SMCRA-approved reclamation method. Along these transects, three 10 × 10 m plots were established at 5, 20, and 50 m from the forest edge and four 1 m² subplots were established within the larger 100 m² plots.

All herbaceous individuals in the 1 m² subplots in all transects were identified to species and percent cover was recorded. Woody seedlings by species were counted in the
10 × 10 m plots along the transects. Seedlings were defined as woody individuals < 1 m in height and < 2.5 cm diameter at breast height (dbh). There were no naturally occurring individuals with dbh > 2.5 cm at the site.

Soil properties

Within each vegetation sampling plot (1 m²), three soil cores were taken at each topographic position (Top of the mound, side slope, and swale between adjacent mounds) and homogenized. These samples were measured for gravimetric soil moisture, percent coarse fragments (i.e., > 2 mm), and texture (% sand, silt, clay) using the hydrometer method (Bouyoucos 1962). Two soil bulk density cores were taken from each topographic position at the same time for analysis (g cm⁻³) in areas reclaimed using the FRA techniques. In the grassland-reclaimed plots, two bulk density cores were taken within each larger 10 × 10 m plot (18 total cores).

Data Analysis

Mean values of plant species richness, mean percent cover and evenness (Pielou’s J) for each microhabitat type (top and sides of mounds, swales, and grassland) were calculated along with Simpson and the Shannon-Weiner indices of diversity (Peet 1974; Magurran 2004). A relative importance value (RIV) was calculated for each species where RIV = relative cover + relative density + relative frequency (Skeen 1973). Dependent variables were tested for assumptions of normality using the D’Agostino omnibus test for normality (D’Agostino et al. 1990) in the FBASICS package in R and a Fisher’s \( F_{\text{max}} \)-test (Sokal and Rohlf 1995, Wuertz 2010) for homogeneity of variance in
the base R stats package. Differences in values among habitat types were compared using
one-way Analysis of Variance (ANOVA) followed by Tukey’s Honestly Significant
Difference (HSD) when the $F$-statistic indicated a significant difference.

Nonmetric multidimensional scaling analysis (NMDS) was used to examine plant
community composition, and to assess relationships between microhabitats and
reclamation types with abiotic features (McCune and Grace 2002). The NMDS ordination
was based on Bray-Curtis dissimilarity distances ranging from a value of 0 indicating
sites are identical, to a value of 1 indicating complete dissimilarity (Wolda 1981). NMDS
uses an iterative search to converge on a solution that minimizes “stress” in the final
ordination configuration. “Stress” indicates the goodness-of-fit with values corresponding
to an acceptable indication of the similarities between samples (Clarke 1993). The
NMDS procedure was performed using the function metaMDS “Vegan” package
(Oksanen et al. 2010) in the R statistical computing program (R Development Core Team
2010). I examined how the environmental factors I measured could be influencing
vegetative composition using the envfit function over the NMDS by fitting vectors
representing environmental variables onto the ordination (Oksanen et al. 2010). I used
ADONIS (a multivariate ANOVA procedure for distance matrices) to compare diversity
among microhabitat positions and reclamation strategies. In R, the function Adonis was
utilized as it is a robust alternative to parametric MANOVA for describing how variation
is attributed to different experimental treatments (Anderson 2001).
Differences in soil characteristics among microhabitat types were examined using one-way ANOVA followed by Tukey’s HSD when the ANOVA indicated a significant F-value. Dependent variables were tested for assumptions of normality using the D’Agostino omnibus test for normality (D’Agostino et al. 1990) in the FBASICS package in R and a Fisher’s $F_{max}$-test (Sokal and Rohlf 1995; Wuertz 2010) for homogeneity of variance in the base R stats package.

Woody plant communities were analyzed based on stem density (stems per 100 m²) at the varying distances from the remaining forest edge in both the mound-style and grassland reclamation. One-way ANOVA was used to examine differences in seedling density among distances from the forest edge followed by Tukey’s HSD when the ANOVA returned a significant $F$-value. Woody seedling composition was examined as described above using NMDS followed by ADONIS to check for significant differences among distances.

Results

Plant communities across microtopographic positions and reclamation types

A total of 29 herbaceous species were found at the Jockey Hollow site across all microtopographic positions in the mound-reclamation area and the grassland reclamation area. Grassland-reclaimed areas were dominated by tall fescue (*Festuca arundinacea* Shreb. [RIV = 0.596]) that was seeded during reclamation. Areas reclaimed with the FRA
experimental methods were dominated by the N-fixing herb birdsfoot trefoil (Lotus corniculatus L. [RIV = 0.248]).

Species richness ($F = 3.58, P = 0.007$) and total cover ($F = 5.574, P < 0.001$) differed significantly among microhabitat positions. Plots in the swale position had both lower cover as well as richness. Evenness (Pielou’s J) also differed significantly among the microhabitat positions ($F = 2.49, P = 0.04$); Shannon and Simpson’s diversity did not differ significantly (Table 1). Sides of the mounds (NE and SW aspect) and top mound positions were dominated by nitrogen-fixing species in terms of relative cover, where the swale position was dominated by hydrophilous forbs, (Typha sp. in particular). Graminoids were found in relatively low abundance in all of the FRA-reclaimed plots, but dominated the traditional-type reclamation plots (Fig. 4.2).

Nonmetric multidimensional scaling showed clustering of plant communities in plots sampled at the different microhabitat types and reclamation strategies (i.e., traditional SMCRA reclamation vs. FRA [Fig. 4.3]). Multivariate ANOVA on the underlying distance matrix indicated that this clustering was significant, although microhabitat type only explained about 1/3 of the variation in species composition ($F = 19.8, R^2 = 0.321, P < 0.01$). Reclamation type significantly affected plant community composition ($F = 33.6, R^2 = 0.165, P = 0.0049$), but the percent of variation explained by the model did not increase appreciably when the parameters were combined. Examination of the NMDS ordinations did not indicate clustering of vegetation groups due to distance from the remaining forest edge. Multivariate ANOVA (Adonis) indicated
significant differences among distances from the forest edge but the percent variation explained being extremely low ($R^2 = 0.039$).

*Soil physical attributes among microhabitat types*

ANOVA indicated that soil moisture differed significantly among the microhabitat types ($F = 80.9$, df = 4, $P < 0.001$), but this difference was almost entirely attributable to differences between moisture in the swales and all of the other microhabitat types (Table 2). The majority of soil variables measured in this study varied significantly among microhabitat positions including coarse fragments ($\% > 2\text{mm}$ [$F = 51.3$, $P < 0.001$]), bulk density ($F = 34.1$, $P < 0.001$), % sand ($F = 9.79$, $P < 0.01$), and % clay ($F = 19.8$, $P < 0.001$). Percent silt did not vary significantly among microhabitat positions (Table 2). Percent moisture, bulk density, percent coarse fragments, percent sand, and percent clay were all significantly related to plant composition at $\alpha = 0.05$ (Fig. 4.3), but like distance to forest edge, these soil variables did not explain a large fraction of site variation. Vectors for bulk density and percent coarse fragments explained the most variation ($R^2 = 0.19$, $R^2 = 0.17$, respectively). All other soil variables except % silt were significantly ($P < 0.05$) related to community composition but the explanatory power was reduced; percent moisture ($R^2 = 0.12$), percent clay ($R^2 = 0.09$), and percent sand ($R^2 = 0.04$).

*Woody seedling colonization*

Woody seedling density increased with increased distance from the remaining forest edge ($F = 77.823$, df = 3, $P < 0.01$). No woody seedlings were found to be
colonizing in the grassland reclamation transects (Fig. 4.4). The NMDS ordination did not indicate any patterns of tree species colonization related to distance from the remaining forest edge. Multivariate ANOVA based on the distance matrices (using seedling density values to create a Bray-Curtis distance matrix) indicated that there was no significant grouping of specific seedling densities by distance ($F = 1.26$, $R^2 = 0.023$, $P = 0.135$).

Discussion

One of the central goals of plant community ecology is to be able to predict community composition from environmental data (Weiher and Keddy 1995). In this study, I saw evidence of different plant community assemblies developing on an experimentally reclaimed mine site, resulting at least in part from differences in the soil physical habitat at a fairly fine scale. The results of this study indicate that reclamation practices have been influencing the trajectory of plant community development on former mined lands in the eastern U.S. since the implementation of the SMCRA. From a forest regeneration standpoint, the implementation of methods approved by the Surface Mine Control and Reclamation Act has effectively halted the return of the formerly forested state of the bulk of these lands.

Microtopography has been shown to affect plant establishment success in wetland communities (Vivian-Smith 1997) and on colonization patterns in old-growth forests as a result of tree uprooting (Beatty 1984). Peterson et al. (1990) proposed the “microsite specificity hypothesis” pertaining to the surface structure found in old growth forests as a
result of tree uprooting. The authors proposed that differences exist in soil variables at the scale of pit and mound location result in differential colonization preferences. The FRA approach creates a highly differentiated set of microhabitats relative to fine-scale topography in a very similar fashion. I found significant differences in the soil physical environment among the different parts of the “end-dump” topography sampled here (Table 2) that were reflected in significant differences in the plant community at those positions (Fig. 4.3).

General patterns of vegetation cover at the site were consistent with observations at similar scales (i.e., tip up mounds; Peterson et al. 1990, Peterson and Campbell 1993) in that lowest total cover was found in the bottom position. Initial observations at the site indicated that there was likely to be a difference in soil texture between the “swale” portion of the mound and positions higher on the mound. Differences in texture were likely the result of finer material washing from positions higher on the mound into the swale and creating a poorly draining “mat” in the bottom of the swale position. These plots are subject to extended periods of inundation throughout the season due to impeded drainage that results in a set of water-holding ponds that remain at the site through roughly mid-July and into August in wetter years. The dominance of the exotic *Typha latifolia* here may be of interest in the future as its possible allelopathy (Jarchow and Cook 2009) may affect long-term successional dynamics even as the pools disappear due to subsequent subsidence by the mounds.
Differences in cover by functional groups were especially pronounced between the traditionally reclaimed plots and the FRA reclaimed plots (Fig. 4.2). The dominance of graminoids and N-fixers on traditionally reclaimed plots led to highly homogenous communities (Fig. 4.3) and was not unexpected given the reclamation method used. The dominance by N-fixing species at the top mound position is interesting in that it indicates the need to further examine the role of functional diversity in community assembly in restoration systems (Aerts and Honnay 2011).

 Bulk density has been observed as having an effect on early plant colonization in disturbed areas (Karim and Mallik 2008) and seedling establishment (Kozlowski 1999). Here however, I was somewhat more concerned with the effects of bulk density on recruitment of woody individuals than the herbaceous community. Bulk density was significantly different among the reclamation types (Table 2) supporting the notion by advocates of the FRA that excessive compaction exists on sites reclaimed using the traditional SMCRA-approved methods. Soil bulk density is likely one of the driving factors for the lack of woody seedling colonization on areas reclaimed using grassland methods as the results here for bulk density are high enough to impede root penetration (Zisa et al. 1979, Heilman 1981). The lack of tree colonization on grassland sites could also be due to increased competition. Total vegetation cover was higher in the grassland plots compared with most of the other microhabitat positions (Table 1). It is likely that heavily seeded ground cover in the grassland-reclaimed plots may also be limiting woody plant colonization. Studies in the North Carolina Piedmont region found increased
emergence of common woody species of the under vegetation cover followed by reduced
growth and survival in non-weeded plots in abandoned fields (DeSteven 1991a, 1991b).
The lack of woody plant colonization seen in the grassland plots may thus be the result of
interacting biotic and abiotic processes.

In addition to hypothetically increasing success of desirable native woody species,
these results indicate that the new type of FRA reclamation implemented at this site is
effective at initiating the early stages of succession for natural forest communities. The
results of this study lend support to the long-held belief by many land managers and
practitioners in mine-land restoration that the factors resulting in the state of “arrested
succession” observed on most reclaimed mine sites are both biotic and abiotic in nature.
The diversity related to differences in microhabitats created at the time of initial
reclamation will ultimately result in dramatically increased benefits in the possibilities for
conservation on formerly degraded lands.

**Implications for Practice**

- Traditional methods of mine-land reclamation are likely inhibiting a return
to forest cover on many sites due to the adverse soil conditions created by
those reclamation processes.

- The creation of a set of microsites with varying soil physical properties
may help to increase diversity on reclaimed sites by creating a diverse set
of plant assemblages at each microsite type.
• Despite the occurrence of natural recolonization by woody species on experimentally-reclaimed sites, active restoration in the form of seed or seedling addition may be necessary depending upon the desired restoration outcomes.

Acknowledgements

I would like to thank the Ohio Dept. of Natural Resources Mineral Division, the U.S. Dept. of the Interior Office of Surface Mining, and the Appalachian Regional Reforestation Initiative for facilitating this project as well as the Oxford Mining company for site access.

Literature Cited


### Tables

Table 4.1: Table 1. Species richness, total cover, and diversity values for “Traditional” grassland-style reclamation and “Experimental” microtopographic positions at Jockey Hollow Wildlife Management area. All values are means ± SE. Letters indicate significant pairwise differences between means within a row (Tukey HSD) at α = 0.05.

<table>
<thead>
<tr>
<th>Traditional reclamation</th>
<th>Experimental</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Grassland</td>
</tr>
<tr>
<td>S</td>
<td>3.26 (.19)</td>
</tr>
<tr>
<td>Total cover (%)</td>
<td>110.07 (3.01)</td>
</tr>
<tr>
<td>H'</td>
<td>0.861 (.058)</td>
</tr>
<tr>
<td>D</td>
<td>0.492 (.027)</td>
</tr>
<tr>
<td>J</td>
<td>0.746 (.016)</td>
</tr>
</tbody>
</table>
Table 4.2: Soil physical characteristics at Jockey Hollow Management Area by microhabitat position in both experimentally and traditionally reclaimed areas. All values are mean (±SE). Letters indicate significant pairwise differences within a row (Tukey HSD) at α = 0.05.

<table>
<thead>
<tr>
<th></th>
<th>Traditional reclamation</th>
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<th>Top</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Grassland</td>
<td>Swale</td>
<td>NW</td>
</tr>
<tr>
<td>Soil moisture (%)</td>
<td>12.4 (0.88)^a</td>
<td>29.1 (1.06)^b</td>
<td>14.91 (0.77)^a</td>
</tr>
<tr>
<td>Coarse fragments (%)</td>
<td>41.1 (1.18)^a</td>
<td>12.5 (1.04)^b</td>
<td>26.3 (1.99)^c</td>
</tr>
<tr>
<td>Bulk density (g cm(^{-3}))</td>
<td>1.84 (0.031)^a</td>
<td>1.41 (0.037)^b</td>
<td>1.24 (0.020)^cd</td>
</tr>
<tr>
<td>Sand (%)</td>
<td>22.4 (1.45)^a</td>
<td>12.4 (2.09)^b</td>
<td>22.4 (1.56)^a</td>
</tr>
<tr>
<td>Silt (%)</td>
<td>30.0 (0.98)</td>
<td>31.4 (1.31)</td>
<td>32.6 (1.08)</td>
</tr>
<tr>
<td>Clay (%)</td>
<td>47.6 (1.211)^a</td>
<td>56.1 (1.76)^b</td>
<td>45.01 (1.27)^ac</td>
</tr>
</tbody>
</table>
Figure 4.1: Traditional-style mine site reclamation to pasture at the Jockey Hollow Wildlife Management Area resulting in a state of arrested succession. The pictured reclamation work is ~30 years old. Inset: Experimental reclamation methods utilized at re-mined sections of JHWMA. The Forestry Reclamation Approach “end-dump” method attempts to promote successful tree recruitment via reduced soil compaction.
Figure 4.2: Relative cover of functional groups at Jockey Hollow Wildlife Management Area by microhabitat position: Grassland (GL), northwest (NW), southeast (SE), Swale, and Top. “forbs” indicates non- n-fixing herbaceous plants. Bar heights indicate mean relative cover, arrows indicate SE. Graminoid cover was greatest in plots reclaimed using tradition “grassland-style” reclamation. Letters indicate significant differences at $\alpha = 0.05$ among functional groups within each microhabitat position.
Figure 4.3: Nonmetric multidimensional scaling (NMDS) ordination of microtopographic positions: Grassland (GL), northwest (NW), southeast (SE), Swale, and Top. Environmental attributes represented by the vectors include: Bulk density (BD; g cm\(^{-3}\)), distance from the remaining forest edge (Distance), Percent soil moisture (Percent.Moisture), and percent coarse fragments (coarse.frag; % > 2 mm), percent sand, silt and clay. ADONIS indicated a significant difference between communities at different microtopographic positions \((F = 19.8, R^2 = 0.321, P < 0.01)\).
Figure 4.4: Density of woody seedlings by distance from the forest edge that remained after mining at the Jockey Hollow wildlife management area. Density of seedlings varied significantly among distances ($F = 77.8$, df = 3, $P < 0.001$). Boxplots follow standard construction: box represents interquartile range (25–75\textsuperscript{th} percentile), horizontal line displays the median, whiskers and outliers defined as per Tukey (1977). Letters indicate significance at $P < 0.05$ determined by Tukey’s HSD.
CHAPTER 5: SPATIAL PATTERNING OF EARLY WOODY COLONIZATION ON A RECLAIMED MINE SITE IN UNGlaciated OHIO

Abstract

Analyzing the spatial pattern of individuals within a plant community is useful for inferring the underlying mechanisms and processes governing community organization. Aggregated patterns can be caused by habitat-specificity or dispersal limitation, while overdispersed (regular) patterns are often caused by competition or predatory host-specificity or at long-term time scales geomorphic patterns. However, few studies have looked at how abiotic influences might drive spatial patterning. Here, I examined how small-scale topography, vegetation cover, and species identity might lead to differences in woody seedling spatial patterning on an experimentally-reclaimed surface mine in eastern Ohio, USA. Overall, I observed an aggregated (clumped) pattern of species distribution when all species were included, using Ripley’s K-analysis. Results of quadrat-count analysis, using fine-scale topography created by the reclamation approach, indicated preferential colonization of different microsite types by different species. The same method showed competitive effects from the dominant groundcover vegetation having an exclusionary effect on woody seedling emergence and establishment. These results indicate that fine-scale topography along with the ground-covering vegetation can influence spatial patterning of woody individuals during the earliest stages of succession following mine land reclamation.
Introduction

Point Pattern Analysis (PPA) can be used to understand ecological patterns observed at multiple scales (Ripley 1981, Moeur 1993, Perry et al. 2006) and infer process from those patterns (McIntire and Fajardo 2009). Understanding the spatial pattern of vegetation through the use of PPA is useful for elucidating which biological processes (e.g., competition, facilitation, and seed dispersal) might be at work to structure plant communities. PPA seeks to quantify patterns in vegetation structure by testing for aggregation or overdispersion of individual plants against a random distribution (Ripley 1981).

In successional time, there is typically a movement from an initial aggregated spatial pattern during colonization, which typically leads to a random distribution during mid-succession, and ultimately a regular pattern of dispersion as the community reaches maturity (Metsaranta and Lieffers 2008). Aggregated patterns can be indicative of dispersal limitations (e.g., limited “seed rain;” Forman and Hahn 1980) or positive, facilitative effects (e.g., amelioration of soil salinity from shading in salt marsh communities; Bertness and Callaway 1994, Eccles et al. 1999). Clumps in stem-mapped data can also be indicative of preferential establishment in suitable microsites (Schupp 1995). Overdispersed patterns of distribution can indicate competitive interactions (Pielou 1962, Kenkel et al. 1997), pathogen escape (Augspurger 1983), and host-specific herbivore interactions (Janzen 1970).
Surface mining for coal represents one of the largest anthropogenic disturbances to the hardwood forests of the eastern United States. Following this form of disturbance, establishment of pasture-land has been common on mined lands, with various studies revealing a state of arrested succession that may last for many decades, and the previous forest condition is rarely if ever achieved. The observed lack of recovery is likely due to a suite of factors at the end of the disturbance event including: excessive soil compaction, soil infertility, and/or poor drainage, coupled with the aggressive, non-native ground cover species typically utilized to satisfy bond requirements (Zipper et al. 2011). New reclamation methods (Forest Reclamation Approach; FRA) have been proposed to promote an expedient return to forest cover by manipulating the early successional environment, making it more suitable for tree colonization. These methods emphasize the use of non-compacting ground-preparation methods and less aggressive vegetative cover. After completion of mining operations, the original approximate contour of the land is first restored as per federal regulations, but contrary to past practices, the final layer of material is dumped into large piles that provide a loose rooting medium into which trees are more likely to establish successfully.

Re-colonization patterns of reclaimed sites after mining are governed by both biotic and abiotic factors that can result from direct intervention by land managers during the reclamation process (Bestelmeyer et al. 2003). The pattern of colonization may also be affected by the topography created by the sets of mounds that characterize the FRA
end-dump method analogous to variation observed in microtopography (“pit-and-mound”) influencing tree distribution in forests (Collins and Pickett 1982, Beatty 1984).

I examined the spatial distribution of woody species as they colonize a site reclaimed using the FRA end-dump method. Specifically, I asked: 1) Does the distribution of woody species during initial colonization vary from a random spatial pattern on these sites, and if so, does another dominant pattern exist at the site? 2) Does species identity of the dominant species at the site have an important effect on the observed spatial patterning of those species during early colonization? and 3) Does the microsite position created by the FRA end-dump method or presence or absence of dominant, mat-forming vegetation affect the spatial distribution of woody individuals during initial colonization?

**Materials and Methods**

**Study Area**

The study was conducted at the Jockey Hollow Wildlife Management Area (JHWMA) in Belmont County, Ohio (40°1’42.25” N, 8°06’10.23” W, elevation 365 m). Temperatures range from -7 °C (mean January low) to 27.8 °C (mean July high) mean annual precipitation is 103.3 cm (NCDC, Cadiz, OH). The site was procured by the Ohio Department of Natural Resources in 2004 and is now managed primarily as a public hunting facility. The site is composed of some pockets of second-growth hardwood forest, but mostly rolling grasslands that are the result of previous mining activities and reclamation via methods approved under the Surface Mining Control and Reclamation
Act (SMCRA) of 1977. Portions of the site were re-mined for coal from 2007–2009 and reclaimed using “end-dump” reclamation in accordance with FRA guidelines for newly reclaimed mines (Fig 5.1).

Field Methods.

In 2011, ten 20 × 10 m sampling plots were established at JHWMA in an area reclaimed using the FRA end-dump approach. The location and species identity of each woody individual in the plot was recorded as an (x,y) coordinate within the plot. Locations of the mound and swale topography within the sampling plots as well as the extent of dominant herbaceous vegetation were also included. Five of these plots were located adjacent to a forest edge that remained intact during and after mining operations, while the other five were located away from the forest edge and traversing the center of the reclaimed area (Fig 5.1). Digital versions of the maps were created using the spatstat package (Baddeley and Turner 2005) in the R programming environment (R Development Core Team 2011). The maps created were a “marked point pattern” where the mark for each point was the individual’s species identity.

Data Analysis

I used Ripley’s K function to examine the spatial patterning of the seedlings colonizing JHWMA. Ripley’s K is a distance method which measures the number of events (here, tree seedling colonizations) found up to a given distance of any particular event (Ripley 1981). The function as implemented using the “Kest” function in R is defined as:
The term $e_{ij}$ is an edge correction weight. I used Ripley’s isotropic method for edge correction for points close to the plot edge as the sampling plot (and as such, the window used for analysis) is rectangular (Ripley 1988, Haase 1995). In order to simplify visual analysis of the output, I transformed the $K$ function to the $L$ function using the following equation:

$$L(r) = \sqrt{\frac{K(r)}{\pi}}$$

the straight line $L_{\text{pois}}(r) = r$ which also adds the benefit of stabilizing the variance around the estimation term. It is implemented in R using the Lest function.

I next compared the output of $L(r)$ of the data to random bootstrapped processes generated under Complete Spatial Randomness (CSR) to determine whether spatial patterns were random, clustered, or uniform. I used Monte Carlo simulations ($n = 99$) to construct confidence envelopes around the theoretical $L_{\text{pois}}(r) = 0$ using the envelope function in R. When comparing plotted $L(r)$ using envelopes, values above the upper bound of the confidence envelope indicate an aggregated pattern, and values below the lower bound of the envelope indicate overdispersion, and values that fall within the envelope indicate that the data resemble a typical Poisson distribution (CSR).
I performed Ripley’s analysis for all woody stems in all plots. I also analyzed spatial patterning on a per-species basis, and for overstory and understory individuals separately. I conducted a monospecific analysis with *Platanus occidentalis* (American sycamore) and *Populus deltoides* (Eastern cottonwood), as they are the dominant early-colonizing species at the site (representing ~75% of all stems combined) and were present in all sampling plots.

The effects of microsite position and herbaceous vegetation cover were examined by dividing the plots into irregular quadrats according to the microsite positions on mounds (i.e., “top,” “side,” and “swale”), or the extent of groundcover vegetation and applying the $\chi^2$ test based on quadrat counts. I used the command `quadrat.test` which accepts tessellations and uses the tiles of the tessellation as the quadrats for the test of CSR.

I also compared total stem densities (stems m$^{-2}$) between plots sampled adjacent to the remaining forest edge using *t*-tests. All statistical analyses were performed using the `spatstat` package in R (Baddely and Turner 2005).

**Results**

Seven of the ten plots exhibited an aggregated spatial pattern during early colonization (Fig. 5.2). Different spatial patterns were observed between plots sampled adjacent to the forest edge versus those sampled away from the edge (Fig. 5.2). Plots 1–5 were located adjacent to a remaining forest edge while plots 6–10 were located away from the forest edge running through the center of the reclaimed area. Plots located
adjacent to the remaining forest edge showed a mixture of random (plots 1–3) and aggregated (plots 4 and 5) while all of the plots located farther into the center of the site showed aggregated arrangements (plots 6–10). Intensity of points (i.e., stem density) was significantly greater ($t = 7.86$, $P < 0.001$) in plots adjacent to the forest edge (mean = 3.334, SE = 0.294) than plots adjacent to the center mining road (mean = 2.171, SE = 0.358).

Different patterns of spatial arrangement were also observed when *Platanus occidentalis* and *Populus deltoides* were analyzed separately by species (Fig. 3, 4). *Platanus occidentalis* mostly adhered to a completely random spatial distribution in both plot classes (forest adjacent and distant; Fig. 3). An aggregated distribution was observed in some plots (i.e., plots 4, 7, 8, and 9). *Populus deltoides* exhibited an aggregated spatial patterning in almost all plots (Fig. 4), with all center plots highly aggregated but only two of the five forest-adjacent plots having an aggregated distribution (Fig. 4).

Mound position had a significant effect on the spatial patterning of all plots. Chi-square tests for distribution showed that when sampling plots were divided into subsections based on the relative position (i.e., “top” or “side” of the mound or “swale”), seedlings were not randomly distributed (Table 1). A similar result was observed when the same operation was performed and plots were divided into subsections (“mat of vegetation present” or “mat of vegetation absent”) based on the extent of dominant groundcover vegetation (Table 2).
Discussion

These results provide evidence that the spatial distribution of colonizing woody species during early succession on FRA-reclaimed mine sites is likely controlled by a combination of factors including: proximity to the remaining forest edge, the fine-scale micro-topography created by the FRA approach, the identities of the colonizing species, and competition with groundcover vegetation. Generalized conclusions about patterns of tree recruitment are not possible or easily modeled. As has been described previously, early succession was characterized by an aggregated pattern of species distribution when all species were considered, the same type of pattern observed in naturally disturbed systems (Martens et al. 1997, Metsaranta and Lieffers 2008; Fig. 3). The regular pattern observed in plots 1, 2, and 3 was unexpected at this very early successional time point (Ward et al. 1996), as this type of pattern usually does not emerge until competition between larger individuals begins to structure the plant community (Peet and Christensen 1987, Kenkel 1988).

The clumped pattern I observed during this early successional phase is most likely the result of increased establishment success within suitable microhabitats. Studies of desirable, introduced species have shown the importance of microsite conditions on the establishment of planted seedlings on sites reclaimed using the Forestry Reclamation Approach (Gilland and McCarthy 2012). The dominant species observed here are not likely constrained by an important determinant of spatial patterning—seed dispersal (Nathan and Muller-Landau 2000). Both species (*Populus deltoides* and *Platanus*...
*Platanus occidentalis* produce prolific amounts of wind-dispersed seeds (Burns and Honkala 1990). Unlike biotic vectors, wind dispersal is generally non-directional, so a random distribution might be expected in these species. Colonization and establishment will be determined by suitable regeneration microsites, and might include the size of the microsite (Rice 1987), the presence of a proper seedbed for germination (Thomas and Wein 1990), and adequate light and moisture levels (Beckage and Clark 2003).

The observed aggregated pattern was mostly unsurprising given the successional stage of the site; previous large-scale studies that have found aggregated spatial patterning to be the most common spatial arrangement, especially in woody species (Barot et al. 1999, Condit et al. 2000). However, I did notice some departures from this aggregated distribution, especially among plots located near the forest edge. Three of the forest-edge plots (plots 1, 2, and 3) exhibited random distributions when considering all species (Fig. 3) and when considering only *Populus deltoides* (Fig. 5). *Platanus occidentalis* was also randomly distributed on these three plots, as well as in another forest-edge plot and one center plot (Fig. 4).

I observed a significant difference in colonization with respect to the microtopographic position created by the FRA “end-dump” method of reclamation. Differences in the herbaceous community with respect to fine-scale topography have previously been seen on these sites (Gilland, personal observation), and are reminiscent of patterns of topographic preference described in the tip-up mounds created by treefalls in intact forests (Beatty 1984; Peterson et al. 1990).
The presence or absence of dense groundcover vegetation likely had an effect on the distribution of early colonizing individuals. The types of groundcover used during traditional mine-land reclamation have been observed (along with excessive soil compaction) to inhibit natural colonization by woody species (Holl and Cairns 1994). One of the most common herbaceous reclamation species (*Lespedeza cuneata*) was in fact common on the study site, likely remaining either in the soil seed bank from previous reclamation efforts, or transported by wind from older reclaimed sections of the site where it formed dense mats of vegetation. My analyses showed a departure from CSR when vegetation cover was considered to create a set of vegetation present/absent polygons which were then compared for colonization density (Table 2). Previous studies have posited that existing groundcover vegetation impacts woody seedling establishment by acting as a “filter” against the emergence of woody stems (George and Bazzaz 1999), preventing dispersed seeds from making contact with the soil and preventing germination (DeSteven 1991), or by providing protective cover for seed predators (Ostfeld et al. 1997; Pusenius et al. 2007). At the study site, mats of vegetation that exclude woody plant emergence may be resulting in the appearance of a pattern of overdispersion of individuals in some cases (e.g., plot 3) or may be driving some plots from a more biotically accurate aggregated spatial distribution to a mathematically random distribution.
Conclusions

This study demonstrates that significant differences in the spatial patterning of an early-successional community, following a severe anthropogenic disturbance event, exist and vary with respect to species identity, fine-scale microtopographic position, and existing vegetation cover. I observed here a set of spatial patterns similar to those expected on other types of successional systems (i.e., an aggregated distribution of stems during early forest succession) as well as evidence of competitively-structured spatial distribution (i.e., overdispersion of some species possibly as a result of existing groundcover vegetation). It should be noted that these patterns are being observed during an extremely early time period following an extreme disturbance, and the patterns observed here may ultimately be somewhat transient as succession proceeds. However, the patterns observed here do indicate that biotic and abiotic factors are at work shaping the plant communities developing on experimentally-reclaimed mine sites.

Literature Cited


Table 5.1: Results of Chi-square tests for woody seedling distribution in sampled plots at JHWMA. A sub-set of quadrats were compared within each plot based on relative mound positions (“top”, “side”, “swale”) created by the FRA method of reclamation.

<table>
<thead>
<tr>
<th>Sampling Unit</th>
<th>$X^2$</th>
<th>df</th>
<th>$P$</th>
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<tr>
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<tr>
<td>Plot 2</td>
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</tr>
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</tr>
<tr>
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<td>83.9</td>
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</tr>
<tr>
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<td>53.9</td>
<td>2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Plot 7</td>
<td>77.9</td>
<td>2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Plot 8</td>
<td>53.9</td>
<td>2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Plot 9</td>
<td>69.9</td>
<td>2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>Plot 10</td>
<td>85.9</td>
<td>2</td>
<td>&lt; 0.001</td>
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</table>
Table 5.2: Results of Chi-square tests for woody seedling distribution in sampled plots at JHWMA. A sub-set of quadrats within each plot were compared based on the presence or absence of a mat of herbaceous vegetation.

<table>
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<td>Plot 10</td>
<td>46.9</td>
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</table>
Figure 5.1: The Forestry Reclamation Approach (FRA) as implemented at Jockey Hollow Wildlife Management Area. The method results in a series of large mounds with water-holding “swale” areas between. Plots were sampled near the center of the reclamation (adjacent to the road visible in the foreground) and the remaining forest edge in the background.
Figure 5.2: Transformed Ripley’s L functions for all woody individuals in plots at JHWMA. The solid black line indicates the observed L(r)-r for the sampled plots, the dashed line indicates a theoretical completely spatially random distribution, and the solid gray area indicates 95% confidence bands generated from 100 Monte Carlo simulations based on a random distribution.
Figure 5.3: Stem-maps and transformed Ripley’s functions for *Platanus occidentalis* on both plot classes at JHWMA. Plots 1–5 represent sampling units adjacent to a remaining forest edge while plots 6–10 represent sampling units adjacent to a road through the center of the reclaimed area. The solid black line indicates the observed $L(r)$ for the sampled plots, the dashed line indicates a theoretical completely spatially random distribution, and the solid gray area indicates 95% confidence bands generated from 100 Monte Carlo simulations based on a random distribution.
Figure 5.4: Stem-maps and transformed Ripley’s functions for *Populus deltoides* on both plot classes at JHWMA. Plots 1–5 represent sampling units adjacent to a remaining forest edge while plots 6–10 represent sampling units adjacent to a road through the center of the reclaimed area. The solid black line indicates the observed $L(r)$-r for the sampled plots, the dashed line indicates a theoretical completely spatially random distribution, and the solid gray area indicates 95% confidence bands generated from 100 Monte Carlo simulations based on a random distribution.
CHAPTER 6 : CONCLUSION

With this work I have attempted to provide answers to two broad questions: 1) What can be expected during restoration plantings of American chestnuts on experimentally-reclaimed mine lands, and 2) What types of plant communities are becoming established on these sites during the initial natural colonization phase of succession that is occurring there?

Direct-seed introduction of the species may be a feasible option but appropriate precautions need to be taken and may end up being more labor-intensive and costly than planting bare-root seedlings. I observed acceptable rates of germination and initial survival in direct-seeded individuals but these results were not observed after the first year. High rates of browse seemed to suppress individuals not protected with a tall growing tube to a constant height not much taller than the 30 cm protective tube used at planting (unprotected seed planting was also found to not be a viable introduction method). Application of a tall protective tube during direct-seeding is likewise not advisable as seedlings showed sub-optimal survival after the first year with this method.

This research shows that the morphological fidelity of the late-generation hybrid chestnuts desired by the American Chestnut Foundation’s breeding program seems to stay true under harsh open-grown field conditions. However, the differences observed among the hybrid genetic lines were disappointing in that the Chinese parents and earlier hybrid genetic lines exhibited superior performance on the study site employed here. The exact mechanism for this performance disparity was preliminarily explored here in chapter three but a further inquiry into photosynthetic performance and the water
relations among the lines would aid greatly in further developing the hybrid chestnut for large-scale restoration.

The results seen in chapter four strongly support the limiting effect of traditional methods of mine reclamation on forest community recovery. Areas sampled that had been reclaimed using traditional methods had no woody species colonization and exhibited abiotic soil conditions that most likely are inhibiting colonization by woody plants (Chapter 4).

Edge influences seem to play an extremely important role in woody plant establishment on FRA-reclaimed strip mines. I saw evidence in both planted individuals (Chapter 2) and naturally-colonizing seedlings that survival adjacent to the forest edge was significantly lowered. Possible explanations include the light regime at these locations, possible fungal pathogen infection, and selective predation. These results are somewhat confounded, however, by the findings in chapter 5 where there was not a significant difference in seedling density between plots sampled adjacent to the remaining forest edge and those located in the center of the reclaimed area. Although the plots sampled in chapter 5 were twice as large as those utilized in chapter 4 and the results from chapter 2 were based on individual tree survival, the plots sampled plots in chapters 4 and 5 covered most of the microsite heterogeneity that can be found at the site eliminating that as a possible explanation. It is possible that the remnant road through the center of the site that the “center plots” were sampled along provides an avenue of dispersal but the wind-dispersed nature of the individuals sampled does not support this
idea as the major cause for this observed equality of distribution. Alternatively, it is possible higher predation rates or limited light is suppressing seedlings closer to the forest edge leading to equalities in density measurements despite an increased seed rain closer to the forest edge. This conflict highlights the need for further study of this type of system through the ecological lens of plant community development versus the more traditional agronomic framework employed in mine-land reclamation.

American chestnut appears to be a viable candidate for mine-land reclamation projects where the Forestry Reclamation approach has been implemented. However, the work conducted in this research has led to a host of other questions that now need to be addressed as chestnut restoration moves forward. Direct interspecies comparisons need to be conducted to specifically determine the desirability of American chestnut as a reclamation species over other confamilial species (i.e., northern red, chestnut, and white oak). A further examination of the mechanisms that may be driving the differences in performance between hybrid genetic lines could be pursued in order to better inform the American Chestnut Foundation’s breeding program as it moves forward. The Forestry Reclamation Approach appears to be situated to allow for a return to forest cover on former mine sites but further inquiry into the mechanisms of natural succession should be investigated as the sites age in order to determine the level of intervention at the site that may be necessary to achieve desired outcomes. Particularly, do the microsite conditions observed in the studies presented here continue to be relevant to community composition as the site ages?