INFLUENCE OF TREE SHELTERS AND WEED MATS ON GROWTH AND SURVIVAL OF BACKCROSSED CHESTNUT SEEDLINGS ON LEGACY MINELANDS IN EASTERN KENTUCKY[1]

Kenton Sena[2], Hannah Angel, and Chris Barton

Abstract. Surface mining for coal has contributed to wide-scale deforestation and forest fragmentation in the eastern United States. Over the last thirty years, mine reclamation practices involving heavy compaction and introduction of non-native vegetation have produced large areas of reclaimed land, which exist in a state of arrested natural succession, termed legacy sites. These sites were reclaimed to standards of the day, which usually involved compaction of the surface material and seeding an aggressive ground cover to control erosion. These sites are typically dominated by non-native grasses and legumes (e.g., tall fescue and lespedeza) and are often colonized by invasive and undesirable woody shrubs (e.g., autumn olive). Interest in restoring native hardwood forest on these sites has grown over the past decade. The development of techniques to mitigate the unfavorable soil and vegetative conditions on these legacy sites is essential to forest restoration in Appalachia. In addition to representing a good opportunity for native hardwood reforestation in Appalachia, legacy sites present a unique opportunity to reintroduce improved blight resistant American chestnut across much of its native range. This study investigated the impacts of tree shelters and weed mats on the growth and survival of planted American chestnuts on legacy mine sites in eastern Kentucky. Shelters significantly reduced browse pressure from deer and therefore improved growth and survival in most instances. Weed mats did not significantly influence tree growth or survival and were unpredictable in their effect on herbaceous biomass. This study demonstrates that properly prepared legacy mine sites can support the establishment of improved American chestnuts.

Keywords: Forestry Reclamation Approach, reforestation, compaction mitigation

---

[1] Paper submitted to JASMR, 2014. R.I. Barnhisel (Ed.) Published by ASMR, 3134 Montavesta Rd., Lexington, KY 40502.

[2] Kenton L. Sena is a Graduate Research Assistant at the University of Kentucky Department of Forestry, 218 T. P. Cooper Bldg, Lexington, KY 40546. Hannah Angel is a Graduate Research Assistant at Stephen F. Austin State University in Nacogdoches, Texas. Chris Barton is the Professor of Forest Hydrology and Watershed Management at the University of Kentucky, Department of Forestry, Lexington, KY.

Journal American Society of Mining and Reclamation, 2014 Volume 3, Issue 2 pp 41-63
DOI: http://doi.org/10.21000/JASMR14020041

Introduction:

Surface mining for coal has contributed to the dramatic alteration of a vast land area in the Appalachian region of the United States. During the surface mining process, richly diverse Appalachian forest is cleared from the site, leading to direct loss of habitat as well as fragmentation of remaining habitat (Wickham et al., 2013). If mined sites are not reclaimed appropriately, they can remain as highly disturbed patches in the landscape for an indefinite period (i.e., arrested natural succession). More than 40 years ago, reclamation on surface mined lands in the Appalachian region was somewhat loosely regulated by state agencies. On unreclaimed sites with favorable soil conditions, native forest systems were able to slowly regenerate, but many sites were characterized by unstable mine soils prone to erosion and landslides. When a national surface mine law (SMCRA) was passed in 1977, it was focused on improving mine soil stability and reducing erosion. Over time, operators and regulators have interpreted SMCRA stipulations as requiring heavily compacted mine soils and rapid vegetative groundcover establishment (Burger et al., 2005).

Unfortunately, it became apparent that native hardwood trees did not thrive on these post-SMCRA reclaimed sites. Heavy compaction associated with post-SMCRA reclamation techniques led to physically unfavorable soil conditions – soils with high bulk density (BD) and poor infiltration capacity (Conrad et al., 2002; Zipper et al., 2011). High soil BD limits root growth, inhibits drainage, and reduces soil aeration. In addition to poor soil physical conditions, mine soils were composed primarily of crushed overburden, rather than salvaged and replaced topsoil. These mine soils were often characterized by high pH and salinity, making them chemically unfavorable to native tree growth (Haering et al., 2004; Shrestha and Lal, 2011). Over time, reclamation experience demonstrated that trees survived and grew poorly on these sites; therefore, post-SMCRA mined lands were almost exclusively reclaimed to pasture and hay lands dominated by non-native grasses and legumes (Burger, 2011). Further, because of soil conditions and severe competitive pressure from the planted herbaceous species, the unused or unmanaged reclaimed grass and hay lands do not naturally allow recolonization of native trees (Burger, 2011). This trend has led to the development of unique and undesirable vegetative and faunal communities which increase habitat fragmentation in the highly bio diverse Appalachian region (Wickham et al., 2013).

To combat this alarming pattern, a team of reclamation scientists and concerned regulators developed a series of recommendations to guide successful regeneration of native forest on mined sites. This Forestry Reclamation Approach (FRA) stipulates 1) use of the best available growth medium, 2) minimal soil compaction, 3) minimal vegetative competition, 4) both early- and late-successional tree species, and 5) proper planting techniques (Burger et al., 2005). Experimental reclamation trials utilizing FRA techniques have been successful in several states (e.g., Wilson-Kokes et al., 2013; Sena, 2014). In addition, operators in mine reclamation are currently employing FRA techniques across the region.

Use of FRA techniques appears to be an effective way to help restore native Appalachian forest ecosystems to mined sites. However, the vast area of previously mined lands already reclaimed to grassland poses a slightly different problem, because of excessive soil compaction and heavy vegetative cover (Burger, 2011; Franklin et al., 2012). Reforestation efforts on these sites that do not overcome those barriers will most likely not succeed. Thus, research investigating the effectiveness of compaction and vegetative competition mitigation techniques is ongoing.

In a study across West Virginia, Ohio, and Virginia, researchers found that tilling to reduce compaction improved growth of hybrid poplar, but had no significant effect on native hardwoods or white pine after the first year (Casselman et al., 2006). However, after five years, ripping significantly improved growth and survival for most species and site combinations (Fields-Johnson et al., 2014). In Kentucky, compaction mitigation by deep ripping with a dozer successfully improved survival of black locust, northern red oak, and white oak, and improved growth of green ash (Michels et al., 2007). In a West Virginia study, Skousen et al. (2009) reported that ripping compacted reclaimed grasslands improved survival of black cherry, red oak, and yellow poplar and improved growth of these and black walnut and white ash. Finally, in Ohio, McCarthy et al. (2010) found that mechanical treatments (cross ripping, surface plowing and disking, and cross ripping followed by surface plowing and disking) significantly improved survival and growth of blight-resistant American chestnut. These studies are consistent with general reports that native hardwoods do not survive or grow well in heavily compacted mine spoils.

In addition to reducing compaction on mined sites, reclamationists are tasked with reducing competition associated with groundcover species. Unfortunately, these species are typically highly competitive and are planted specifically to attain rapid groundcover and erosion control. A number of studies have found treatment with herbicides can significantly reduce vegetative competition and improve planted tree survival (Sweeney et al., 2002; McCreary et al., 2011). Other studies have considered weed control by weed mats or mulch treatments. Appleton et al. (1990) found that soil coverings (various fabrics and films, as well as mulching) tended to mitigate soil temperature extremes and improve soil moisture, suggesting that use of weed barriers may have additional micro-climatic effects. Several researchers have reported improved growth and/or survival of planted trees with treatment by weed mats or mulch (Sweeney et al., 2002; Geyer, 2003; Chaar et al., 2008; and Ingels et al., 2013).

In contrast, Stange and Shea (1998) reported that northern red oak seedlings treated with landscape cloth for weed suppression were apparently targeted for browsing by deer. Red oak seedlings with weed mats suffered a higher rate of browse and higher mortality than seedlings not protected by mats. These studies demonstrate that weed mats can reduce negative impacts on growth and survival of planted trees associated with vegetative competition. However, specific site characteristics must be taken into consideration to identify potential issues with browse and other forms of herbivory.

The use of tree shelters to improve temperature and moisture conditions around the tree and to provide protection from browsing has been fairly well documented. Tree shelters have been found to improve tree growth and/or survival in northern red oak (Stange and Shea, 1998), Garry oak (Clements et al., 2011), blue oak (McCreary et al., 2011), and Englemann spruce (Jacobs 2012). In most cases, shelter effects on survival and growth are thought to be driven by reduction of browse and other herbivory. For example, ungulate browsing pressure was a major factor influencing Garry oak survival in the study by Clements et al. (2011) in the Pacific Northwest. Tree shelters both reduced browse and improved survival in the study.

Also of interest in the Appalachian region is restoration of the American chestnut (*Castanea dentata*). *C. dentata* was once a primary constituent of eastern hardwood forest (Russell, 1987). It was highly valuable for its wood and was an important food source for animals and humans because of its nutritious, reliable mast. However, due to the introduction of a fungal pathogen (*Cryphonectria parasitica*) which causes chestnut blight, American chestnut was essentially extirpated from the eastern U.S. by the middle part of the 20th century. In response to this devastation, the American Chestnut Foundation has supported a massive undertaking to breed natural blight resistance from Chinese chestnut into native American chestnut, all the while preserving as much of the desirable traits of American chestnut as possible. This breeding program has produced several lines of chestnut with 7/8 to 15/16 American chestnut heritage. These hybrids have demonstrated improved blight resistance compared to native American chestnut, but have retained the desirable American chestnut phenotypic traits (Diskin et al., 2006, Hebard 2006).

The Appalachian coalfields lie in the heart of the American chestnut historic range. Because of this and the need for reforestation on mined land, blight resistant American chestnuts have been tested for growth and survival on mined sites. McCarthy et al. (2010) found good first-year survival of chestnuts planted into compaction-mitigated minesoils. French et al. (2007) found good first year survival of chestnut seedlings planted into end-dumped (low compaction) mine soils. Skousen et al. (2013) reported 60% fourth-year survival for chestnuts planted as seeds directly into fresh mine soil. Additionally, some studies have investigated potential effects of weed mats and tree shelters for use in chestnut plantings. For example, Skousen et al. (2013) found that tree shelters did not significantly impact first-year growth and survival of chestnuts, and they removed the shelters during the second year to reduce observed heat stress effects. McCarthy et al. (2010), however, found that tree shelters significantly reduced predation of trees planted as seeds as well as post-sprouting browsing by herbivores. Use of weed mats in chestnut plantings is widespread, but appears somewhat under-researched (McCarthy et al., 2010; Angel et al., 2012; Bauman et al. 2013).

This study was initiated to evaluate the use of tree shelters and weed mats on survival and growth of backcrossed 15/16 American chestnuts on three compaction-mitigated legacy mine sites in eastern Kentucky.

Methods:

Three post-bond (i.e., considered fully reclaimed) mine lands in eastern Kentucky originally reclaimed as hay lands were single ripped by a dozer to mitigate compaction. The Pine Branch site was located in Perry County, the Rooster site in Whitley County, and the York site in Morgan County (Fig. 1). All three sites exhibited a dense herbaceous groundcover prior to ripping, but each differed with respect to species composition. The Pine Branch site had been actively managed for cattle grazing with tall fescue (*Schedonorus phoenix*) as the dominant vegetation species. The

Rooster site exhibited a thick stand of the invasive legume sericea lespedeza (*Lespedeza cuneata*), which was likely used in the original reclamation process. The York site also exhibited some tall fescue and sericea lespedeza, but also contained the invasive Chinese silver grass (*Miscanthus sinensis*) and native broomsedge (*Andropogon virginicus*). In addition, the York site exhibited a variety of native herbaceous species adapted to moist and constantly wet soils such as: nodding bur marigold (*Bidens cernua*), mist flower (*Eupatorium coelestinum*), sneezeweed (*Helenium autumnale*), bulrush (*Scirpus validus*), common rush (*Juncus effuses*), and cattail (*Typha latifolia*). The presence of these wetland species at York is likely attributed to poor drainage from compaction of the spoil.

At each site, four browse and vegetative competition mitigation treatments were replicated three times in a completely randomized design (4 treatments x 3 replicates = 12 plots per site). The treatments were as follows: 1) control (C), 2) weed mats only (M), 3) tree shelters only (S), and 4) tree shelters plus weed mats (MS). VisPore© weed mats were provided by Conservation Services, Inc., and measured 1 x 1 m. Tubex© plastic tree shelters (122 cm high) were also provided by Conservation Services, Inc. At each site, 25 bare root (1-0) seedlings were planted into each plot (n = 12, for a total of 300 seedlings per site. Backcrossed chestnut seedlings used in this study were 15/16 American (B2F3), progeny of a 7/8 American chestnut (SA417) open pollinated with a pure American chestnut. Seed was obtained from the Meadowview Research Farms of the American Chestnut Foundation and raised at the Kentucky Division of Forestry's Morgan County Nursery. Seedlings were planted using dibble bars in late March (Rooster) and early April (York and Pine Branch) 2010 while still dormant.

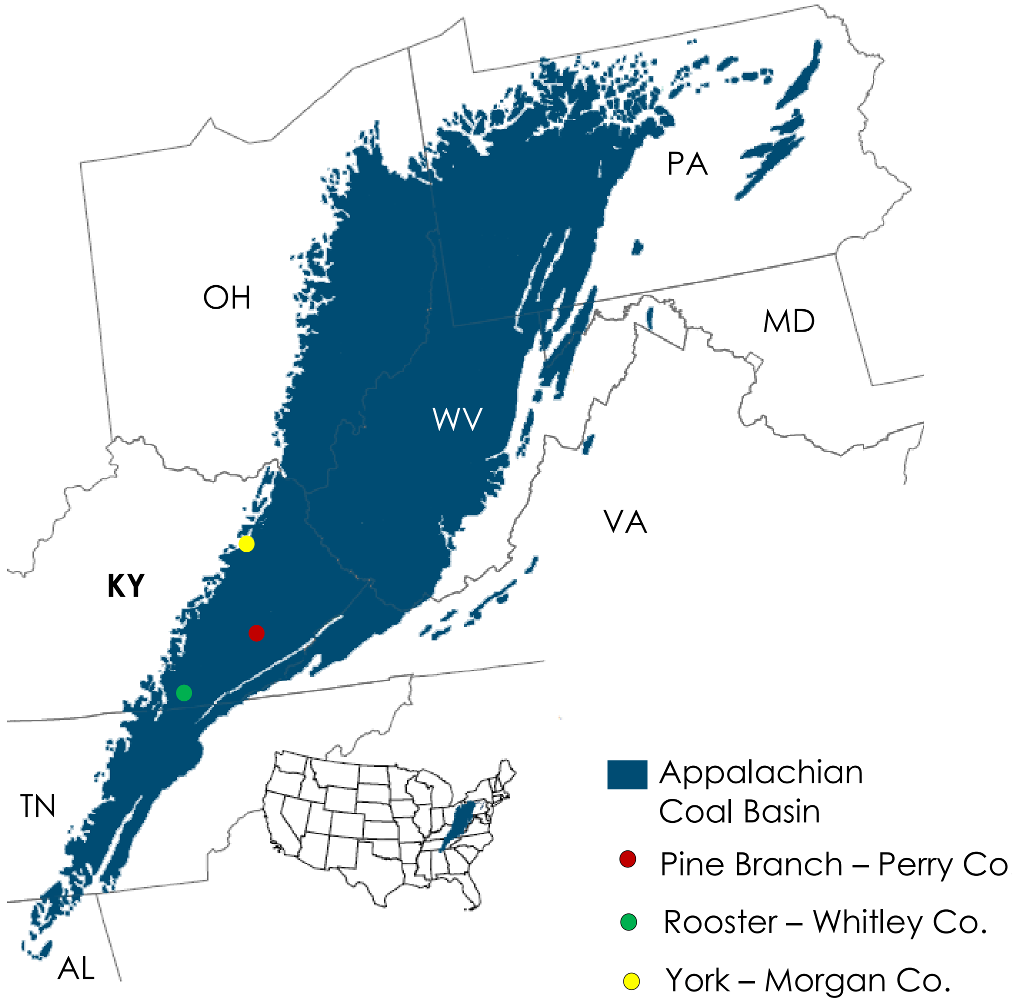

Figure 1: Three post-bond legacy mine sites in eastern Kentucky were deep-ripped with a dozer and planted with blight-resistant American chestnuts.

Seedling height (from the ground to the terminal bud) was measured during the summers of 2010, 2011, and 2012. Browse was visually assessed and assigned a score according to the following metric: (0) = No browse, (1) = 1-25% of branches browsed, (2) = 26-50% of branches are browsed but seedling not hedged, (3) = >50% of branches browsed, (4) = >50% of branches are browsed, seedling hedged, <3 cm tall. Biomass of surrounding vegetation was assessed as a proxy for vegetative competition. Within each plot, two 1 x 1 m grids were placed randomly around a centrally located seedling. Positioning the plot this way allowed for an assessment of the weed mat (if present) as it would be positioned within the 1 x 1-m grid. All herbaceous material within the grid was harvested just above the soil surface using grass clippers and hedge clippers, dried to 60°C and weighed.

At each site, soil was sampled to a depth of 10 cm within each plot using a shovel. Samples were mixed in the field to generate one composite sample per site. Samples were analyzed by the University of Kentucky Regulatory Services Soils Laboratory. Soil pH was determined in a 1:1 soil water solution (Thomas, 1996). P, K, Ca, Mg, and Zn were analyzed by Mehlich-III extraction (Soil and Plant Analysis Council, 2000). Cation exchange capacity (CEC) was determined by the ammonium acetate method (Summer and Miller, 1996). Plant available water, field capacity water, and wilting point water were calculated using a pressure plate method as described in Topp et al. (1993).

Statistical Methods

Mean height and browse scores were calculated for each plot (n = 12) in each site (n = 3) for each year (n = 3). Means were analyzed by ANOVA using PROC MIXED (SAS 9.3), with site, year, and treatment modeled as fixed effects and replicates modeled as a random effect. The repeated measures statement (year) was modeled using the covariance structure SP(POW). Significant differences identified by ANOVA were followed up using pairwise differences of LSMEANS (pdiff statement).

Survival proportions were calculated for each plot (n = 12) in each site (n = 3) for each year (n = 3). Survival proportions were analyzed by ANOVA using PROC GLIMMIX (SAS 9.3), with treatment, site, and year modeled as fixed effects and replicate modeled as a random effect. Significant differences detected by ANOVA were followed up using pairwise differences of LSMEANS (pdiff statement).

Results:

Soil pH was closest to native soil conditions (4.89-5.65; Cotton et al., 2012) at the York site (Table 1). Soil pH was much higher than native soil conditions at both Rooster (7.16) and Pine Branch (8.14) and is likely to be chemically unfavorable to native tree growth. In addition, higher CEC and water holding capacity at York is indicative of more advance soil development or suggests that the spoil material used contained native topsoil or was highly weathered. Retention of base cations and moisture by the soil matrix is important for maintaining favorable plant growth conditions. Given these observations, the soil at York is likely to be the most favorable to chestnut growth, while soil at Pine Branch will likely be less favorable.

Table 1: Mean soil chemical characteristics of ripped legacy mineland sites in eastern Kentucky.

	Rooster	York	Pine Branch
Soil-water pH	7.16	5.25	8.14
P (mg/kg)	5	22.5	4
K (mg/kg)	129	94.5	81
Ca (mg/kg)	2,004	1,378	3,566
Mg (mg/kg)	376	276	278
Zn (mg/kg)	11.6	4.8	4.0
CEC (cmol/kg)	12.25	15.74	8.88
Plant Avail Water (%)	10.34	14.73	11.2
Field Capacity Water (%)	22.41	32.26	21.6
Wilting Point Water (%)	12.07	17.54	10.4

The tree shelter effect on survival was strongly interacted with site ($p < 0.0001$). First year tree survival was similar across treatments at each site (Table 2) varying from a low of 72% at Rooster to a high of 97% at Pine Branch. Treatment effects began to be apparent in Pine Branch by the second study year (2011), with much higher survival in sheltered treatments (96% in MS and 99% in S) than either untreated (C at 8%, $p < 0.0001$) or mats-only (M at 19%, $p < 0.0001$). These differences in survival continued through the third year. At the Rooster site, we observed a single treatment effect on second-year survival, with MS (81%) higher survival than C (52%, $p = 0.0155$). By the third year, however, no treatment effect on survival was detected at the Rooster site. Finally, at the York site, we observed higher second-year survival in all treatments (M, $p = 0.0355$; MS, $p = 0.0206$; and S, $p = 0.0012$) compared to C. By the third year, survival was higher in M (71%) than either MS (46%, $p = 0.0461$) or C (41%, $p = 0.0155$) but similar to S (65%). These data indicate that shelter and mat effects on tree survival can be complicated and site-dependent.

Table 2: Percent Survival means and standard errors are compared among year, site, and treatment. Means with the same letter are not significantly different ($p > 0.05$) among treatments within site and year. (C = no mats or shelters; M = mats only; MS = mats and shelters; S = shelters only.)

	C	M	MS	S
		Rooster		
2010	77a ± 15	95a ± 4	72a ± 6	76a ± 10
2011	52b ± 17	59ab ± 8	81a ± 17	65ab ± 17
2012	39a ± 14	32a ± 10	37a ± 7	33a ± 13
		York		
2010	83a ± 6	95a ± 4	84a ± 5	92a ± 2
2011	51b ± 5	76a ± 5	79a ± 9	91a ± 4
2012	41c ± 8	71a ± 12	47cb ± 10	65ab ± 7
		Pine Branch		
2010	88a ± 4	92a ± 5	97a ± 3	97a ± 1
2011	8b ± 5	19b ± 3	96a ± 2	99a ± 1
2012	8b ± 5	19b ± 3	96a ± 2	99a ± 1

Shelter effects on browse were more consistent among sites, with trees in sheltered treatments experiencing significantly reduced damage from browsing compared to trees in unsheltered treatments ($p < 0.0001$, Table 3). At both Rooster and York, we observed that browse tended to decrease in unsheltered trees while at the same time increasing in sheltered trees throughout the study. For example, browse in the York C treatment group decreased from 1.78 to 0.75, while browse in the York S group increased from 0.12 to 0.45. This is to be expected; as trees grow above their shelters (see height data, Table 4), they will be more vulnerable to browse (Fig. 2). Also, we expect that competing vegetation would reduce browse pressure, both by providing alternate forage and by visually obstructing trees. In contrast to York and Rooster, we observed that browse scores were highly consistent within sheltered trees at Pine Branch throughout the study; however, browse scores in unsheltered treatments decreased slightly (3.06 to 1.86) throughout the study. Note that high browse scores in unprotected trees at Pine Branch corresponds closely to low survival (Table 2). The relationship of browse to survival at York and Rooster is less clear from these data.

Table 3: Browse score means and standard errors for chestnuts planted into ripped legacy mine in eastern Kentucky. Means with the same letter are not significantly different (p > 0.05) among treatments within site and year. (C = no mats or shelters; M = mats only; MS = mats and shelters; S = shelters only.)

	C	M	MS	S
		Rooster		
2010	1.8a ± 0.3	1.7a ± 0.4	0.2b ± 0.1	0.1b ± 0.1
2011	1.3a ± 0.5	1.6a ± 0.6	0.2b ± 0.1	0.3b ± 0.1
2012	0.8a ± 0.2	0.7a ± 0.2	0.3a ± 0.1	0.4a ± 0.2
		York		
2010	1.5a ± 0.1	1.1a ± 0.2	0.0b ± 0.0	0.0b ± 0.0
2011	1.0a ± 0.2	1.1a ± 0.2	0.1b ± 0.0	0.1b ± 0.0
2012	1.4a ± 0.2	1.5a ± 0.1	0.7b ± 0.1	0.4b ± 0.0
		Pine Branch		
2010	3.1a ± 0.0	3.0a ± 0.0	0.2b ± 0.1	0.1b ± 0.0
2011	1.9a ± 0.1	1.7a ± 0.2	0.1b ± 0.0	0.1b ± 0.0
2012	1.9a ± 0.1	1.3a ± 0.5	0.1b ± 0.0	0.1b ± 0.0

We also observed a clear effect of sheltering on height, with sheltered trees demonstrating increased height growth compared to unsheltered trees ($p < 0.0001$, Table 4). Although sheltering effects on height differed among sites during the first two years, note that sheltered trees (S and MS) were consistently higher than unsheltered trees across all three sites by the third year.

For herbaceous biomass, the interaction between treatment and site was not significant ($p = 0.6864$, Table 5). While mats only (728g) had no significant effect on herbaceous biomass compared to control (794g), shelters only (1,006g) had significantly more herbaceous biomass than mats ($p = 0.0984$) or mats and shelters (601g, $p = 0.0073$) treatments. In addition, herbaceous biomass varied significantly with site (Fig. 3-5), and was highest at York (1016g), followed by Rooster (717g, $p = 0.0078$) and Pine Branch (613g, $p = 0.0008$), probably related to soil properties as noted above.

Figure 2: As chestnuts in sheltered treatments grew above the height of the plastic tree shelters, they became vulnerable to browse, as was this individual at the Pine Branch site.

Table 4: Height (cm) means and standard errors of chestnuts planted into ripped legacy mineland in eastern Kentucky. Means with the same letter are not significantly different ($p > 0.05$) among treatments within site and year. (C = no mats or shelters; M = mats only; MS = mats and shelters; S = shelters only.)

	C	M	MS	S
		Rooster		
2010	56.3b ± 2.1	59.5b ± 2.2	99.5a ± 10.8	92.2a ± 6.0
2011	67.4b ± 5.9	60.0b ± 7.3	104.7a ± 13.6	126.3a ± 20.6
2012	81.8b ± 12.7	70.2b ± 14.3	123.1a ± 36.8	150.5a ± 14.1
		York		
2010	64.6b ± 1.9	63.4b ± 5.3	85.6ab ± 7.9	94.3a ± 4.7
2011	68.9c ± 5.2	70.9bc ± 3.2	99.6ab ± 3.8	110.2a ± 5.8
2012	62b ± 5.3	64.6b ± 2.3	115.5a ± 7.2	121.6a ± 4.0
		Pine Branch		
2010	72.7b ± 5.6	75.1ab ± 3.2	102.6a ± 8.6	95.4ab ± 4.8
2011	63b ± 2.0	63.1b ± 10.0	131a ± 10.8	111.3a ± 3.2
2012	63.9b ± 2.0	63.1b ± 10.0	128.8a ± 9.6	111.3a ± 3.2

Table 5: Means (g) and standard errors of vegetative biomass in ripped legacy mineland sites in eastern Kentucky. (C = no mats or shelters; M = mats only; MS = mats and shelters; S = shelters only.)

	C	M	MS	S
Rooster	612 ± 175	837 ± 132	432 ± 258	988 ± 122
York	946 ± 22	962 ± 47	828 ± 54	1,326 ± 255
Pine Branch	627 ± 47	582 ± 131	542 ± 79	703 ± 73

Figure 3: American chestnut seedlings planted into deep-ripped legacy mineland at the Pine Branch site in Perry County. Sheltered trees are clearly visible in the background, while mats-only treatments are visible in the foreground. Compared to the York and Rooster sites, Pine Branch had the lowest herbaceous biomass. (Photo taken September 2010)

Figure 4: American chestnuts planted into deep-ripped legacy mineland at the Rooster site in Whitley County. Vegetative biomass at this site was intermediate between Pine Branch and York. (Photo taken September 2010)

Figure 5: American chestnut seedlings were planted into deep-ripped legacy mineland at the York site in Morgan County. Herbaceous biomass was highest at York; tree shelters (1 m high) are barely visible above the thick vegetation in this figure. We hypothesize that vegetative competition was more intense at York and Rooster than at Pine Branch. (Photo taken September 2010)

When 2012 data for browse, survival, and height were averaged across sites (Table 6), the general trend for all sites supports our site-specific observations. Sheltered treatments (MS and S) consistently improve tree height and reduce browse. Mats (M), on the other hand, appear to have no effect on growth, browse, or survival.

Table 6: Mean (and standard error) data for height, browse, and survival of American chestnut planted into dozer ripped legacy mine land. Values are averaged across sites by treatment. (C = no mats or shelters; M = mats only; MS = mats and shelters; S = shelters only.)

	Height (cm)	Browse	Survival
C	69.2 ± 6.3	1.3 ± 0.3	0.3 ± 0.1
M	66.0 ± 2.2	1.2 ± 0.2	0.4 ± 0.2
MS	122.5 ± 3.9	0.4 ± 0.2	0.6 ± 0.2
S	127.8 ± 11.7	0.3 ± 0.1	0.7 ± 0.2

Discussion:

The variation in survival and browse among our three sites indicates that reforestation success on legacy mine sites is complex and governed by multiple factors. First-year survival ranged from 72-97% across sites, which is comparable to first year survival in Virginia (83%, Fields-Johnson et al., 2012) and West Virginia (80-85%, Skousen et al. 2013), both of which were loose-graded spoil studies. By the third year, however, survival had declined across treatments, from 87% in 2010 to 49% in 2012. A strong site*year interaction is evident, with survival remaining very high in sheltered treatments at the Pine Branch site (96-99%) through three years. Within treatments, survival of sheltered trees was lowest at the Rooster site (33-37%) and highest at the Pine Branch site (96-99%), while unsheltered tree survival was lowest in Pine Branch (8-19%) and highest in the York site (41-71%). Third-year survival of unsheltered trees planted into loosely graded spoil in West Virginia was high (73-74%, Skousen et al., 2013), and third-year survival of sheltered trees planted into compaction mitigated minesoil in Ohio was even higher (79-85%, McCarthy et al., 2010). Although third-year survival across our sites was quite variable, > 95% survival in sheltered trees at Pine Branch demonstrates that reforestation efforts on legacy minedland can be highly successful through the first few years. Continued monitoring into the future will provide valuable data on longer-term development of reforested systems on ripped legacy mineland.

Pressure from browsing was higher at Pine Branch than the other sites; consequently, the effect of sheltering on browse prevention was more extreme in that site than others. We observed that browse was significantly reduced in sheltered trees compared to unsheltered trees for nearly all site*year combinations. In Ohio, McCarthy et al. (2010) found that chestnuts planted as seeds without protection experienced a higher rate of herbivory than seeds planted with shelters. In contrast, Skousen et al. (2013) reported that shelters had no effect on tree growth and survival, identifying little to no pressure from browsing. These results suggest that browsing can heavily influence tree growth and survival (e.g., Pine Branch site); thus, if herbivory is a potential factor (i.e., deer, elk, horses, or small mammals could be present on the site); shelters should be used to minimize potential damage to trees. On highly disturbed sites that are sufficiently removed from habitat for small mammals and deer (e.g., Skousen et al., 2013), the risk of browse damage may be low enough to permit planting without shelters.

Tree height was consistently influenced by shelter treatments across sites, with sheltered trees significantly higher than unsheltered trees in nearly all site*year combinations. Height after three years was slightly higher than third-year height reported by McCarthy et al. (2010) with chestnut seedlings in compaction mitigated soils, ranging from 76-100 cm. Height in our study was also greater than third-year height of seedlings planted into loosely graded spoil in West Virginia, ranging from 68-112 cm (Skousen et al., 2013). It is important to note that third-year height of trees in sheltered treatments is only slightly above the height of the tree shelters. As sheltered trees grow above their shelters, they become vulnerable to browse; this effect is observable in the slightly increasing trend in browse scores in sheltered treatments through the third year. Continued monitoring will be essential to identify whether more extensive browse protection is required for long-term growth and survival on these sites.

While sheltering effects were clearly important for reducing browse, increasing height, and improving survival, weed mats appeared to have no effect. In nearly all cases, trees treated only with weed mats grew to similar heights and suffered similar browse pressure as trees without mats or shelters. Similarly, trees with both mats and shelters were similar in most cases to trees with only shelters. We surveyed herbaceous biomass across treatments and sites to see if weed mats were effective at reducing vegetative competition by reducing weeds. However, we found that mats did not significantly reduce herbaceous biomass.

The site gradient in biomass is likely responsible for our observations of variance in shelter effect on browse and survival. At Pine Branch, where biomass was lowest, the effect of shelters was most pronounced. Planted trees are more visible and thus more vulnerable to browse when they are not surrounded by thick vegetation. In addition, lower overall biomass was related to lower vegetative competition, making the site as a whole more favorable to tree growth. Thus, sheltered trees were less limited by either browse or competitive pressure. Conversely, at York, where herbaceous biomass was heaviest, the effect of shelters on tree survival and browse was least pronounced. Trees were less visible among the heavy biomass; thus, browse was not a significant pressure on survival at the site.

Among the three sites, we identified the soil at York as the most chemically favorable site. Thus, we expected this site to have the highest tree survival and growth. However, judging by the heavy herbaceous biomass, weedy vegetation responded well to soil conditions at the expense of planted trees. Low survival at the Rooster site was likely caused by a combination of mediocre soil quality and vegetative competition. Interestingly, although we identified Pine Branch as having the poorest soil, we found sheltered tree survival to be highest there. These data suggest that vegetative competition may be more important than soil quality in influencing improved American chestnut survival and growth on legacy sites. Also of note, these data indicate that 15/16 American chestnut seedlings can successfully establish in slightly alkaline soils (pH 7-8), despite being much less acidic than soils in native American chestnut range.

Conclusions:

Survival and growth of blight resistant American chestnuts on deep-ripped legacy mineland is clearly variable and governed by overall site quality, herbaceous competition, and browse. However, if site conditions can be improved such that they are favorable for tree growth, legacy mineland sites represent a significant opportunity for establishment of native forest systems on highly disturbed sites, and especially restoration of blight-resistant American chestnut to much of its native range. In our study, the effects of browse could be substantially mitigated by simply installing plastic tree shelters. We recommend that tree shelters be used on all restoration sites where herbivory is a potential influence. While tree shelters were effective at reducing herbivory, we found weed mats unsuccessful at reducing herbaceous competition. Thus, we do not recommend expenditure of time and money on the installation of weed mats. Unfortunately, vegetative competition appeared to be a major factor influencing survival and growth of planted trees in our study. We recommend continued research to develop practical methods for site-scale weed control on legacy mineland.

Acknowledgments:

The authors would like to thank the American Chestnut Foundation and Michael French for the chestnuts. We would also like to acknowledge Lee Bryant with Flying Rooster (Whitley Co), Larry York (Morgan Co) and Robert Ray with Pine Branch Coal Company (Perry Co) for allowing us to conduct the experiment on their land. Assistance with land preparation and planting was provided by Jacob Stewart and staff of the Kentucky Department of Fish and Wildlife Resources. Special thanks to Patrick and Tess Angel for assistance in several phases of the project.

References:

Angel, P.N., J.A. Burger, C.E. Zipper, and S. Eggerud. 2012. Reforesting unused surface mined lands by replanting with native trees. *In*: D.L. Haase, J.R. Pinto, and L.E. Riley, technical coordinators. National Proceedings: Forest and Conservation Nursery Associations-2011. Fort Collins (CO): USDA Forest Service, Rocky Mountain Research Station. Proceedings RMRS-P-68. 10-15. Available at: http://www.fs.fed.us/rm/pubs/rmrs_p068.html

Appleton, B.L., J.F. Derr, and B.B. Ross. 1990. The effect of various landscape weed control measures on soil moisture and temperature and tree root growth. Journal of Agriculture 16:264-268.

Bauman, J.M., C.H. Keiffer, S. Hiremath, B.C. McCarthy, and P. Kardol. 2013. Soil preparation methods promoting ectomycorrhizal colonization and American chestnut (Castanea dentatata) establishment in coal mine restoration. Journal of Applied Ecology 50:721-729. http://dx.doi.org/10.1111/1365-2664.12070.

Burger, J.A., D. Graves, P. Angel, V. Davis, and C. Zipper. 2005. The Forestry Reclamation Approach. USDOI Office of Surface Mining. Forest Reclamation Advisory No. 2. http://arri.osmre.gov/fra.htm.

Burger, J.A. 2011. Sustainable mined land reclamation in the eastern U. S. coalfields: A case for an ecosystem reclamation approach. Proceedings America Society of Mining and Reclamation, 2011 pp 113-141. http://dx.doi.org/10.21000/JASMR11010113.

Casselman, C.N., T.R. Fox, J.A. Burger, A.T. Jones, and J.M. Galbraith. 2006. Effects of silvicultural treatments on survival and growth of trees planted on reclaimed mine lands in the Appalachians. Forest Ecology and Management 223:403-414. http://dx.doi.org/10.1016/j.foreco.2005.12.020

Chaar, H., T. Mechergui, A. Khouaja, and H. Abid. 2008. Effects of treeshelters and polyethylene mulch sheets on survival and growth of cork oak (Quercus suber L.) seedlings planted in northwestern Tunisia. Forest Ecology and Management 256:722-731. http://dx.doi.org/10.1016/j.foreco.2008.05.027.

Clements, D.R., S. Luginbill, D.A. Jordan, R.V. Dragt, and P.K. Pelant. 2011. Techniques to promote garry oak seedling growth and survival in areas with high levels of herbivory and competition. Northwest Science 85:172-181. http://dx.doi.org/10.3955/046.085.0208.

Conrad, P.W., R.J. Sweigard, D.H. Graves, J.M. Ringe, and M.H. Pelkki. 2002. Impacts of spoil conditions on reforestation of surface mined land. Mining Engineering 54:39-46.

Cotton, C., C. Barton, J. Lhotka, P.N. Angel, and D. Graves. 2012. Evaluating reforestation success on a surface mine in eastern Kentucky. *In*: Haase, D.L.; J.R. Pinto, and L.E. Riley. tech. coords. National Proceedings: Forest and Conservation Nursery Associations-2011. Proc. RMRS.

Diskin, M., K.C. Steiner, and F.V. Hebard. 2006. Recovery of American chestnut characteristics following hybridization and backcross breeding to restore blight-ravaged Castanea dentata. Forest Ecology and Management 223:439-447. http://dx.doi.org/10.1016/j.foreco.2005.12.022.

Fields-Johnson, C.W., J.A. Burger, D.M. Evans, and C.E. Zipper. 2012. American chestnut establishment techniques on reclaimed Appalachian surface mined lands. Ecological Restoration 30:99-101. http://dx.doi.org/10.3368/er.30.2.99

Fields-Johnson, C.W., J.A. Burger, D.M. Evans, and C.E. Zipper. 2014. Ripping improves tree survival and growth on unused reclaimed mined lands. Environmental Management 53:1059-1065. doi: 10.1007/s00267-014-0261-7. http://dx.doi.org/10.1007/s00267-014-0261-7

Franklin, J.A., C.E. Zipper, J.A. Burger, J.G. Skousen, and D.F. Jacobs. 2012. Influence of herbaceous ground cover on forest restoration of eastern US coal surface mines. New Forests 43:905-924. doi:10.1007/s11056-012-9342-8. http://dx.doi.org/10.1007/s11056-012-9342-8.

French, M.E., C. Barton, D. Graves, P.N. Angel, and F.V. Hebard. 2007. Evaluation of mine spoil suitability for the introduction of American Chestnut hybrids in the Cumberland Plateau. Proceedings America Society of Mining and Reclamation, 2007 pp 249-258. http://dx.doi.org/10.21000/JASMR07010249.

Geyer, W.A. 2003. Weed barriers for tree seedling establishment in the central great plains. *In* Van Sambeek, J.W., J.O. Dawson, F. Ponder, Jr., E.F. Loewenstein, and J.S. Fralish., (eds.) 2003. Proceedings, 13th Central Hardwood Forest Conference; 2002 April 1-3; Urbana, IL. Gen. Tech. Rep. NC-234. St. Paul, MN: U.S.

Haering, K.C., W.L. Daniels, and J.M. Galbraith. 2004. Appalachian mine soil morphology and properties: Effects of weathering and mining method. Soil Sci. Soc. of Am. J. 68:1315-1325. http://dx.doi.org/10.2136/sssaj2004.1315.

Hebard, F.V. 2006. The backcross breeding program of the American Chestnut Foundation. *In*: Steiner, K.C.; and J.E. Carlson., (eds.) Restoration of American Chestnut to Forest Land, proceedings of a conference and workshop, May 4-6 2004, The North Carolina Arboretum, Asheville. Natural Resources Report NPS/NCR/CUE/NRR - 2006/01. Washington D.C.: National park Service; 61-67.

Ingels, C.A., T. Lanini, K.M. Klonsky, and R. Demoura. 2013. Effects of weed and nutrient mangement practices in organic pear orchards. *In* Granatstein et al., (eds.) Proc. 2nd International Organic Fruit Symposium, ISHS, 2013.

Jacobs, D.F. 2011. Reforestation of a salvage-logged high elevation clearcut: Engelmann spruce seedling response to tree shelters after 11 growing seasons. Western Journal of Applied Forestry 26:53-56.

McCarthy, B.C., K.E. Gilland, J.M. Bauman, and C.H. Keiffer. 2010, Factors affecting performance of artificially regenerated American chestnut on reclaimed mine sites, Proceedings America Society of Mining and Reclamation, 2010 pp p 582-597. DOI: 10.21000/JASMR10010582 http://dx.doi.org/10.21000/JASMR10010582

McCreary, D.D., W. Tietje, J. Davy, R. Larsen, D. Flavell, M. Doran, and S. Garcia. 2011. Tree shelters and weed control enhance growth and survival of natural blue oak seedlings. California Agriculture 65:192-196 http://dx.doi.org/10.3733/ca.v065n04p192.

Michels, A., C. Barton, T. Cushing, P. Angel, R. Sweigard, and D. Graves. 2007. Evaluation of low spoil compaction techniques for hardwood forest establishment on an eastern Kentucky surface mine. Proceedings America Society of Mining and Reclamation, 2007 pp 492-503. http://dx.doi.org/10.21000/JASMR07010492

Russell, E.W.B. 1987. Pre-blight distribution of Castanea dentata (Marsh.) Borkh. Bulletin of the Torrey Botanical Club 114:183-190. http://dx.doi.org/10.2307/2996129.

Sena, K. L. 2014. Influence of Spoil Type on Afforestation Success and Hydrochemical Function on a Surface Coal Mine in Eastern Kentucky. (M.S. Thesis, University of Kentucky) Retrieved from http://uknowledge.uky.edu/forestry_etds/16 .

Shrestha, R.K., and R. Lal. 2011. Changes in physical and chemical properties of soil after surface mining and reclamation. Geoderma 161:168-176. doi:10.1016/j.geoderma.2010.12.015. http://dx.doi.org/10.1016/j.geoderma.2010.12.015

Skousen J, T. Cook, L. Wilson-Kokes, and E. Pena-Yewtukhiw. 2013. Survival and growth of chestnut backcross seeds and seedlings on surface mines. Journal of Environmental Quality 42:690-695. doi:10.2134/jeq2012.0368. http://dx.doi.org/10.2134/jeq2012.0368

Skousen, J., J. Gorman, E. Pena-Yewtukhiw, J. King, J. Stewart, P. Emerson, and C. Delong. 2009. Hardwood tree survival in heavy ground cover on reclaimed land in West Virginia: Mowing and ripping effects. Journal of Environmental Quality 38:1400-1409. http://dx.doi.org/10.2134/jeq2008.0297.

Soil and Plant Analysis Council. 2000. Chapter 7. Major cations (potassium, calcium, magnesium, and sodium). In: Soil analysis handbook of reference methods. Soil and Plant Analysis Council, Inc., CRC Press, Boca Raton, FL.

Stange, E.E., and K.L. Shea. 1998. Effects of deer browsing, fabric mats, and tree shelters on Quercus rubra seedlings. Restoration Ecology 6:29-34. http://dx.doi.org/10.1046/j.1526-100x.1998.00614.x.

Summer, M.E., and W.P. Miller. 1996. Cation exchange capacity and exchange coefficients. Pages 1201-1230 *In* D. Sparks and J.M. Bartels, (eds.) Methods of Soil Analysis. Part 3. Chemical Methods. ASA and SSSA, Madison, WI.

Sweeney, B.W., S.J. Czapka, and T. Yerkes. 2002. Riparian forest restoration: Increasing success by reducing plant competition and herbivory. Restoration Ecology 10:392-400. http://dx.doi.org/10.1046/j.1526-100X.2002.02036.x

Thomas, G.W. 1996. Soil pH and soil acidity. Pages 475-790 *In* D.L. Sparks and J.M. Bartels, (eds.) Methods of Soil Analysis. Part 3. Chemical Methods. ASA and SSSA, Madison, WI.

Topp, G.C., Y.T. Galganov, B.C. Ball, and M.R. Carter. 1993. Chapter 53. Soil water desorption curves. *In*: M.R. Carter (ed.) Soil sampling and methods of analysis. Canadian Society of Soil Science. Lewis Publishers, Boca Raton, FL.

Wickham, J., P.B. Wood, M.C. Nicholson, W. Jenkins, D. Druckenbrod, G.W. Suter, M.P. Strager, C. Mazzarella, W. Galloway, and J. Amos. 2013. The overlooked terrestrial impacts of mountaintop mining. Bioscience 63:335-348. http://dx.doi.org/10.1525/bio.2013.63.5.7.

Wilson-Kokes, L., P. Emerson, C. DeLong, C. Thomas, and J. Skousen. 2013. Hardwood tree growth after eight years on brown and gray mine soils in West Virginia. Journal of Environmental Quality 42:1353-1362. http://dx.doi.org/10.2134/jeq2013.04.0113.

Zipper, C.E., J.A. Burger, J.M. McGrath, J.A. Rodrigue, and G.I. Holtzman. 2011. Forest restoration potentials of coal-mined lands in the eastern United States. Journal of Environmental Quality 40:1567-1577. http://dx.doi.org/10.2134/jeq2011.0040