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Morgantown, West Virginia
April 6–8, 2022

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Abstract

This proceedings from the 2022 Central Hardwood Forest Conference in Morgantown, WV contains 19 papers and extended abstracts pertaining to research conducted on the conifer component, forest health, growth and yield, the herbaceous component, restoration, silviculture, social and economic dimensions, and soils of the Central Hardwood region. The Central Hardwood region is broadly defined as the part of the continental United States where deciduous hardwood trees dominate the species composition. This region, which includes many physiographic provinces, supports a variety of culturally and economically important tree species.

Keywords: silviculture, oak-hickory forests, stand dynamics, natural regeneration, artificial regeneration

Cover Art

Central Hardwood forest in springtime along the Cheat River, near Parsons, WV. USDA Forest Service photo by Melissa Thomas-Van Gundy.

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22nd
CENTRAL HARDWOOD FOREST CONFERENCE

Managing Future Forests Today

Proceedings of a conference held at
West Virginia University, Morgantown, West Virginia
April 6–8, 2022

Compiled by:
Melissa Thomas-Van Gundy, USDA Forest Service,
Northern Research Station

Mary Beth Adams, USDA Forest Service, Northern Research Station

Sponsored by:
West Virginia University
Appalachian Hardwood Center
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USDA Forest Service, Northern Research Station

FOREWORD

The Central Hardwood Forest Conference is a series of biennial meetings intended to bring together forest managers and scientists to discuss research and issues concerning the ecology and management of forests in the Central Hardwood region. Beginning in 1976, universities and Research Stations of the USDA Forest Service located in this region have served as hosts of the meetings. The 22nd conference was planned for March 2020; however, the global SARS-CoV-2 pandemic interrupted the usual biennial schedule. Rather than holding a conference in 2021, virtual or in person, the steering committee decided to postpone the conference until 2022, putting the community back on schedule.

This, the 22nd conference, included presentations pertaining to forest regeneration, forest products, ecology and forest dynamics, human dimensions and economics, forest biometrics and modeling, silviculture, forest health and protection, and soil processes. There were lively Q&A sessions after the talks, and during the poster sessions, in which there were 14 poster presentations. The conference consisted of 42 oral presentations resulting in the 19 papers and extended abstracts published here. Excellent plenary talks by Tom Inman and Ann Bartuska prompted participants to think about the role of change and challenges in the hardwood industry affecting international wood markets, and to envision the future of forestry in North America. The next conference will be hosted by the University of Kentucky in 2024.

Melissa Thomas-Van Gundy and Mary Beth Adams

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REVIEW PROCESS

Manuscripts for oral presentations were assigned to one of the compilers and peer-reviewed by at least two professionals. Reviews were returned to authors to revise their manuscripts. Revised manuscripts were then submitted to the Northern Research Station, USDA Forest Service for final editing and publishing.

The Steering Committee is grateful to the following for providing peer review of manuscripts for the proceedings: Bryce Adams, Heather Alexander, Pat Brose, Matt Bumgardner, Stacy Clark, Adam Coates, Dan Dey, Sian Eisenhut, Songlin Fei, Kathryn Gazal, Todd Hutchinson, John Kabrick, Charlie Kelly, Tara Keyser, Benjamin Knapp, Jeff Kochenderfer, Shaneka Lawson, John Lhotka, Dacoda Maddox, Joe McNeel, Brenden McNeil, Randy Morin, Thomas Ochuodho, Deborah Page-Dumroese, Scott Schlarbaum, Jamie Schuler, Callie Schweitzer, Rebecca Snell, Susan Stout, Jeff Stringer, Scott Torreano, Lance Vickers, and Marcella Windmuller-Campione.

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The Steering Committee thanks plenary speakers Tom Inman and Ann Bartuska for their remarks to set the stage for the conference, and West Virginia University staff and graduate students who helped with logistics. We thank the following for serving as session moderators: Heidi Harmala, Logan Ferguson, Aliza Fassler, and Matthew Marsh.

HISTORY OF THE CENTRAL HARDWOOD FOREST CONFERENCE

This conference is the 22nd in a series of biennial meetings that have been hosted by numerous universities and USDA Forest Service Research Stations in the Central Hardwood forest region:

1976 Southern Illinois University

1978 Purdue University

1980 University of Missouri

1982 University of Kentucky

1985 University of Illinois

1987 University of Tennessee

1989 Southern Illinois University and the North Central Forest Experiment Station

1991 Pennsylvania State University and the Northeastern Forest Experiment Station

1993 Purdue University and the North Central Forest Experiment Station

1995 Northeastern Forest Experiment Station and West Virginia University

1997 University of Missouri and the North Central Forest Experiment Station

1999 University of Kentucky and the Southern Research Station

2002 University of Illinois, Urbana-Champaign and the North Central Research Station

2004 Ohio State University, Wooster, and the Northeastern Research Station

2006 University of Tennessee, Knoxville, and the Southern Research Station

2008 Purdue University and the Northern Research Station

2010 University of Kentucky and the Northern Research Station

2012 West Virginia University and the Northern Research Station

2014 Southern Illinois University and the Northern Research Station

2016 University of Missouri and the Northern Research Station

2018 Purdue University and the Northern Research Station

2020 *canceled due to global pandemic*

2022 West Virginia University and the Northern Research Station

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Forest Health

Forest Disturbance Trends in Central Hardwood Forests

Randall S. Morin, Thomas C. Goff, Greg C. Liknes, Mark D. Nelson, and KaDonna C. Randolph[1]

Abstract

Recent studies indicate that disturbance regimes impacting forests have increased in intensity and frequency in many parts of North America. We analyzed U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis permanent plot data from current and previous time periods to quantify trends in biotic and abiotic disturbances spatially and temporally for forest land in the Central Hardwood region. We found that the percentage of forest land experiencing disturbance increased over the past 10 years, from 14 percent to 21 percent for forest land in the Central Hardwood region, and the occurrence of disturbance relative to cutting increased from double to more than triple. The primary type of disturbance in Central Hardwood forest land was insects, with outbreaks increasing from 1.3 million acres (1 percent) in 2009 to 8.8 million acres (9 percent) in 2019. We then identified plots impacted by disturbance events within Central Hardwood forests to assess which tree species were impacted to help interpret the disturbance trend results. Quantifying trends and relative impacts of the different types of disturbance agents on forests can inform planning and treatment needs for land managers, forest planners, and forest health specialists. Additionally, further analyses of these data may help to address the susceptibility of Central Hardwood forests to disturbance and quantify the economic impact of losses to tree damage and mortality.

INTRODUCTION

Natural disturbances play an important role in forest stand dynamics. Canopy gaps created by the death or damage of overstory trees alter light regimes and understory microclimates, which in turn impact biodiversity (Thom and Seidl 2016) and forest succession (Abrams and Scott 1989, Holzmueller et al. 2012). The canopy gaps also create the mosaic of forest structure and stand-age classes across the landscape necessary for forest resilience and the sustainable production of forest products (Pickett and White 1985). The extent of natural disturbances varies considerably and can range from just a few trees damaged to thousands of acres damaged. Likewise, the effects can be general (all species affected) or targeted (one or a few species affected) and moderate or intense, such as broken limbs instead of complete windthrow (Frelich 2002). Although some effects of natural disturbances are positive, many are less desirable, such as the spread of invasive plants (Daniels and Larson 2020).

Recent studies indicate that forest disturbance regimes have increased in intensity and frequency in many parts of North America over the past few decades (Thom and Seidl 2016). Examples include fires in the western United States (Westerling et al. 2006),

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insect outbreaks in the Rocky Mountains (Meddens et al. 2012), and wind disturbances in the Midwest (Stueve et al. 2011). In addition to the perceived increase in extreme weather events in the Central Hardwood region (CHR) of the eastern United States, new introductions of invasive forest insects and pathogens continue to occur, and existing invaders continue to spread (Liebhold et al. 2013). There is also evidence that warming temperatures and extreme moisture conditions can lead to increased activity of forest pests and pathogens that are currently active and spreading in the CHR, including native species, such as southern bark beetle (*Dendroctonus frontalis*) (Bentz and Klepzig 2014) and nonnative species, such as hemlock woolly adelgid (*Adelges tsugae*) (Paradis et al. 2008).

Although the focus of this study is natural disturbance, we have included data on harvesting or tree cutting for comparison. Like natural disturbances, cutting modifies forest stand dynamics and impacts vary in terms of extent, specificity, and intensity. As such, they are considered "human-caused" disturbances.

The goals of this study were (1) to estimate trends by disturbance type within the CHR, (2) to examine variability in disturbances in forest land across four subregions within the CHR, and (3) to utilize damage observations at the tree level to provide further information about the types of tree damages in the CHR, particularly for plots with recorded disturbances.

METHODS

The USDA Forest Service, Forest Inventory and Analysis (FIA) program operates an extensive national forest inventory designed to inventory and monitor changes in forests across all land in the United States using a network of permanent field plots (Bechtold and Patterson 2005). FIA plots are distributed at a spatial intensity of approximately one plot for every 6,000 acres across the country. Site-level attributes, such as forest type and major disturbance, and tree-level variables, such as diameter at breast height (d.b.h.), species, and damage, are recorded at each forested plot location.

Plots are visited on an ongoing basis such that each plot in the eastern United States typically is remeasured once every 5 to 7 years and each plot in the western United States, once every 10 years (Bechtold and Patterson 2005). If a disturbance has occurred on a plot between assessments, FIA field crews record the disturbance agent and estimate the year of the event. The threshold for "disturbance" is at least 1 acre in size resulting in mortality or damage to 25 percent or more of all trees or 50 percent or more of a single species. Disturbance agents include insect, disease, fire, animal, weather, vegetation, human-induced (excluding silvicultural treatments), geological, and other. Similarly, silvicultural treatments (e.g., cutting and thinning) are recorded if they affect an area at least 1 acre in size. We estimated acres of "cutting" based on observances of the cutting treatment code. Detailed description of disturbance and treatment field measurement protocols are available within the FIA field guide (USDA Forest Service 2022a). Although up to three disturbances and three treatments can be recorded for each plot, we included only the first (most important) recorded disturbance and treatment.

To assess disturbance trends in the CHR, we geographically limited our study area to FIA survey units (aggregations of counties; Fig. 1A) corresponding to a map published by Pederson et al. (2014), resulting in a total of 23,508 forested plots. We distinguished two

subcategories of forest within the CHR: "Central Hardwood forest" (CHF) and "other forest" (OF). The CHF included FIA forested conditions in one of the seven U.S. National Vegetation Classification (USNVC) fifth-level macrogroups that represent characteristics of the CHF: Central Midwest oak forest, woodland, and savanna (1.B.2.Na.M012); southern and south-central oak-pine forest and woodland (1.B.2.Na.M016); Appalachian-northeastern oak-hardwood-pine forest and woodland (1.B.2.Na.M502); central Midwest mesic forest (1.B.2.Na.M882); Appalachian-interior-northeastern mesic forest (1.B.2.Na.M883); central hardwood forest floodplain forest (1.B.3.Na.M029); and central hardwood swamp forest (1.B.3.Na.M503)[2]. The OF included forested conditions in all other USNVC fifth-level macrogroups, typically mixed hardwood and softwood plots or softwood-dominated plots. Approximately 81 percent of the forested plots in the CHR were categorized as CHF (Fig. 1B) and approximately 19 percent as OF (Fig. 1C).

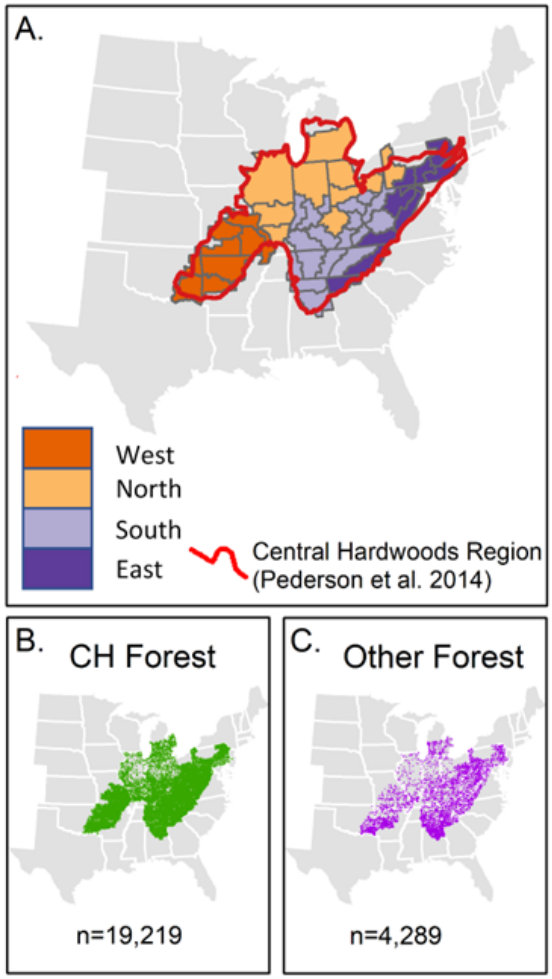

Figure 1.—(A) Forest Inventory and Analysis estimation units in the Central Hardwood region (CHR) and two study subregions: (B) Central Hardwood forest (CHF) plots and (C) other forest (OF) plots. Plot locations are approximate. Map A is reproduced from Pedersen et al. (2014).

[2] Personal communication from ecologist Don Faber-Langendoen, NatureServe, and research ecologist Jennifer Costanza, USDA Forest Service, Southern Research Station.

Using the FIA online estimation tool EVALIDator (USDA Forest Service 2022b), we generated estimates of disturbed forest land (total acres and proportional acres) within the study area previously defined for the most recent available year (2019) and two previous time steps at 5-year intervals (2014 and 2009). We also spatially partitioned results from our study area into four geographic subregions that reflect the character of forests in the CHR (Dyer 2006): oak-hickory (West), maple-beech-basswood (North), mesophytic/ Nashville basin (South), and Appalachian oak (East) (Fig. 1A).

In addition to these site-level observations of forest disturbance, up to three damages are recorded for individual live trees 5 inches d.b.h and larger. Tree damages are typically recorded in general categories such as fire, insect, disease, or decay, although some regions collect more-specific agents. The degree of damage must meet thresholds that indicate likely mortality within 2 years, reduced growth, or negatively impacted forest products (USDA Forest Service 2019). These agents are usually collapsed into general categories for reporting the incidence of tree damage (e.g., Morin et al. 2016). We used the FIA Tree Damage Dashboard (Morin 2022) to examine proportional occurrence of tree damage, by species, among all plots in the CHF with the insect condition-level disturbance.

RESULTS

Trends in the CHR indicate that natural forest disturbances increased from 14 percent to 21 percent of total forest land between 2009 and 2019, affecting more than 24 million acres in 2019; the relative occurrence and temporal pattern of disturbance were similar between CHF and OF within the study area. However, the occurrence of cutting was proportionally twice as high for OF within the study region when compared to CHF. Cutting decreased slightly across CHF and OF (Fig. 2). Since the focus area of our study was the CHF, more-specific analyses by disturbance type and subregion hereafter will focus on CHF.

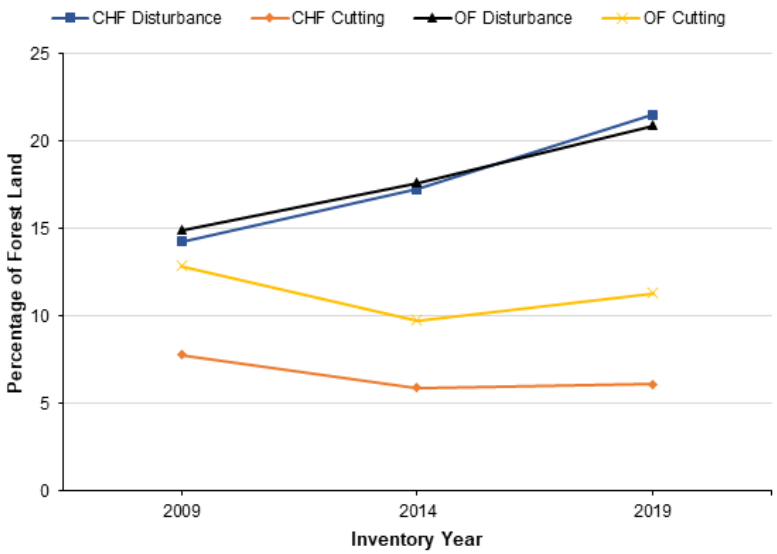

Figure 2.—Proportion of forest land area with disturbance or cutting in the Central Hardwood forest (CHF) and other forest (OF) by inventory year.

Insects were the primary type of disturbance in CHF after an increase in outbreak acreage from 1.3 million acres (1 percent) in 2009 to 8.8 million acres (9 percent) in 2019. As a disturbance category, insects overtook animals and weather, which were the highest two disturbance categories in 2009. Area disturbed by fire and disease doubled between 2009 and 2019, but these disturbance types impacted only 2.3 million acres (3 percent) and 1 million acres (1 percent) of CHF, respectively, in 2019. Weather disturbance peaked in 2014 at 4.5 million acres (5 percent). The area of forest land with disturbance by animals remained stable over the study period while the area cut decreased slightly (Fig. 3).

When disturbance and cutting trends were estimated at the subregion level, some variations emerged. Insect disturbances in CHF increased substantially between 2009 and 2019 for the East, North, and South subregions. Insect disturbances in the North and South increased to 3.1 million acres and over 3.5 million acres, respectively. Though weather disturbances decreased across all subregions, weather events were still a major disturbance factor in 2019, particularly in the South and West. In 2014, 2.4 million acres of forest land (10 percent) experienced a weather disturbance in the West region although the area disturbed decreased substantially by 2019 (2 percent). Fire disturbance doubled in both the East and West between 2009 and 2019, but this type of disturbance was a substantial factor only in the West subregion. Disease disturbance increased substantially between 2009 and 2019 for the East and North subregions, particularly in the East, where disease disturbance reached nearly 0.5 million acres (2 percent) in 2019 (Fig. 4). However, the occurrence of disease disturbance was much lower than both insect and weather disturbance across the CHR overall (Fig. 3). Finally, cutting remained stable across the North and South subregions and decreased slightly in the East and substantially in the West subregions (Fig. 4).

Estimates of number of live trees and number with observed damage reveal that nearly 30 percent of all trees had a damage recorded, and decay/disease was the dominant damage category. Insect damage was low overall (3 percent), but species-level estimates highlight several species with high rates of insect damage, including eastern hemlock (*Tsuga canadensis*, 23 percent), black oak (*Quercus velutina*, 18 percent), and white ash (*Fraxinus americana*, 17 percent). The incidence of tree-level insect damage by species from plots where insect disturbance was also recorded indicate that of the most important species (1.5 percent of total trees or more), green ash (*Fraxinus pennsylvanica*, 76 percent), white ash (75 percent), and eastern hemlock (73 percent) have exceedingly high rates of tree-level insect damage (Fig. 5).

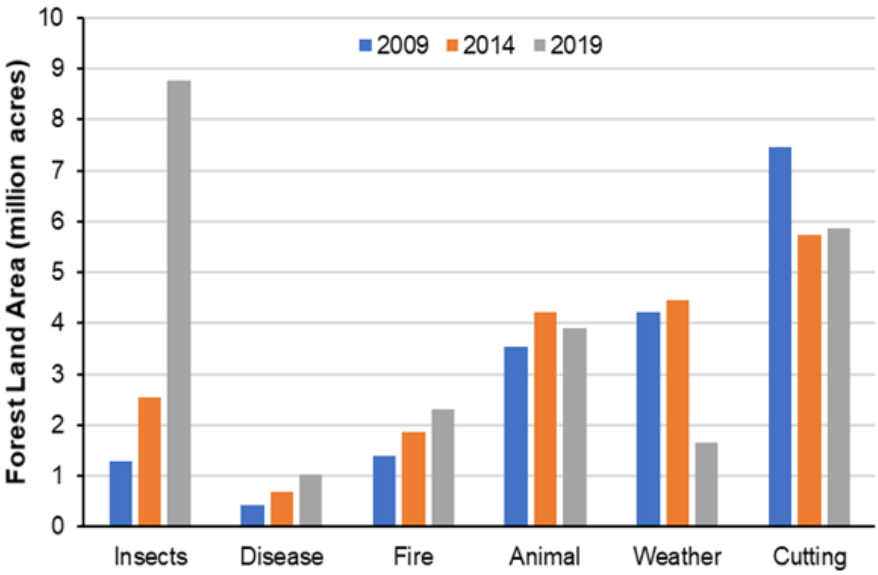

Figure 3.—Area of forest land in the Central Hardwood forest (CHF) by disturbance type and inventory year.

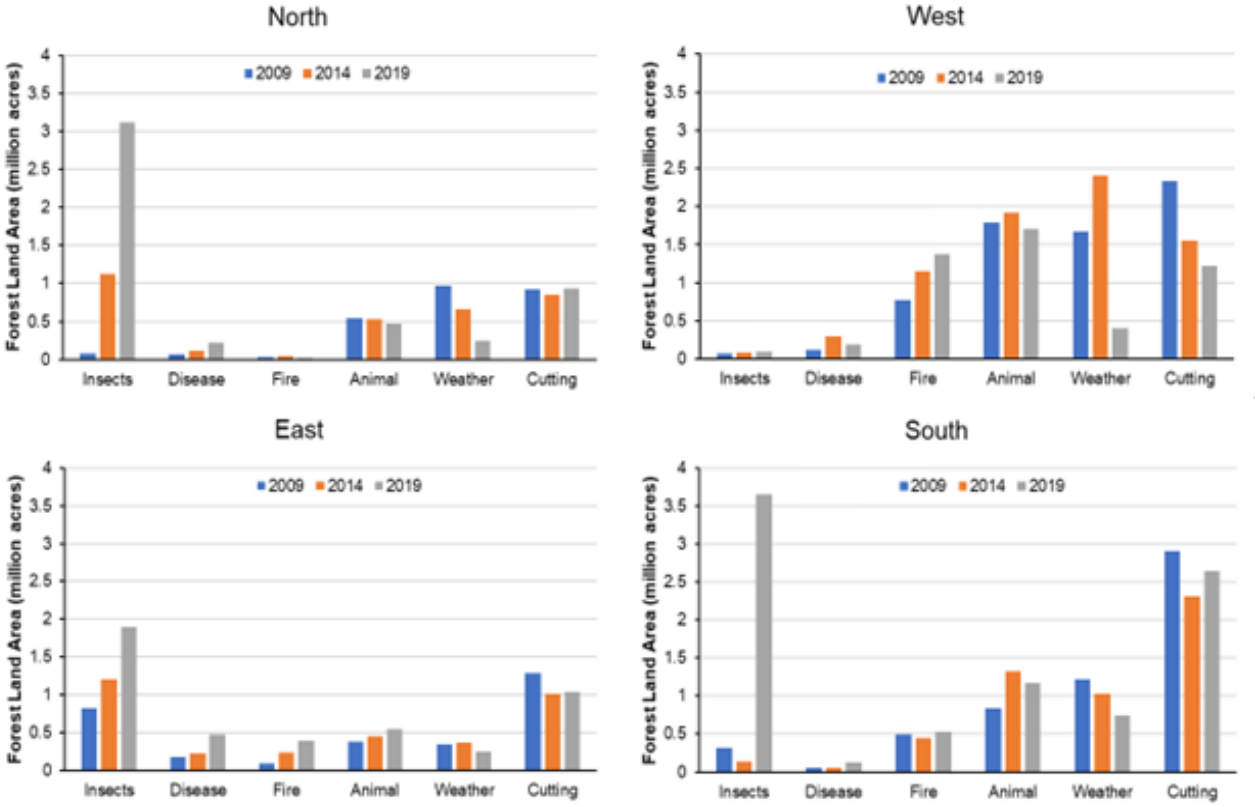

Figure 4.—Area of forest land with disturbances and cutting recorded in Central Hardwood forest (CHF) subregions, North, West, East, and South, by inventory year.

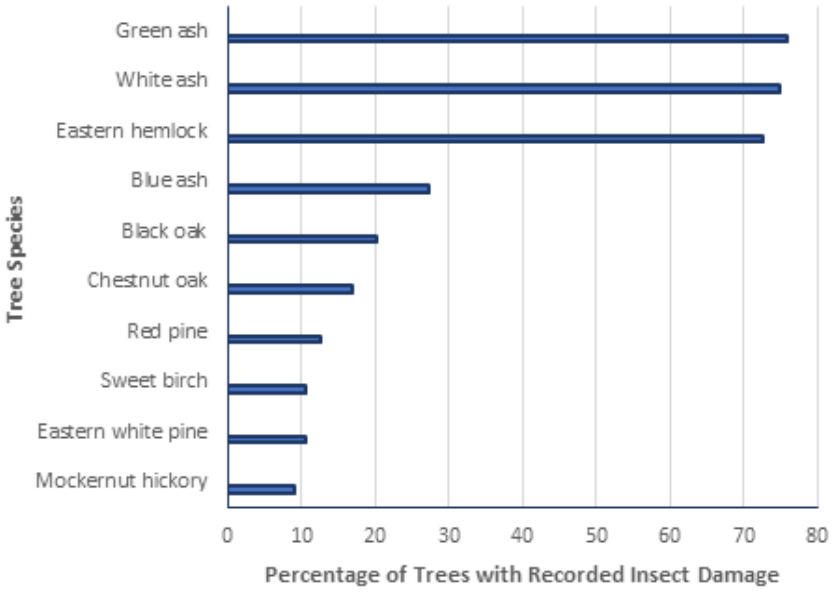

Figure 5.—Percentage of trees with recorded tree-level insect damage from plots with recorded condition-level insect disturbance for tree species with the 10 highest occurrences of insect damage including all species that compose at least 1.5 percent of all trees.

DISCUSSION

The United Nations Intergovernmental Panel on Climate Change (IPCC) has reported that a warming climate is likely to cause increased frequency and severity of extreme weather events like hurricanes, tornadoes, derechos, and ice storms that can cause damage to forests (IPCC 2023) and allow pests to spread and survive in areas that were previously too cold or increase the number of insect pest generations possible in a growing season (Logan et al. 2003). The results of our study indicate that forest disturbances, not including cutting, have increased by approximately 50 percent over the last decade in the CHR (Fig. 2) and for all U.S. forest land (results not shown) (USDA Forest Service 2022b). Insects were the primary driving factor of this increase within the CHF (Fig. 3), whereas insects, fire, and disease accounted for most of the increase for all U.S. forest land during the same period (2009 to 2019) (USDA Forest Service 2022b). The ratio of forest acres disturbed to acres cut in the CHF increased from double in 2009 to more than triple in 2019 (Fig. 2).

Examination of tree damage rates within disturbed plots can add information about the tree species that are likely to be suffering the most impacts of those disturbances. This method appears to work well for insect disturbances in the CHF where several well-known invaders have been impacting tree health of specific tree species. The high occurrence of tree-level insect damage on ash species and eastern hemlock indicates that most of the insect disturbance at the plot condition level is likely due to emerald ash borer (*Agrilus planipennis*) and hemlock woolly adelgid. Furthermore, a recent forest health condition report indicates that emerald ash borer and hemlock woolly adelgid are the most detected damaging agents in aerial detection surveys of the CHF region (Potter and Paschke 2022). Thus, reporting both plot condition disturbance and tree damage provides complementary information based on data collected at the same plot locations, but at two different scales of observation.

Our results show that weather disturbance decreased for our study period, but one thing to consider is that the 5- to 7-year remeasurement cycle for FIA plots means that estimates of disturbed forest land are on average 2.5 to 3.5 years old. Reports from the National Oceanic and Atmospheric Administration (NOAA) identified 2017 and 2020 as two of the biggest years for "billion-dollar weather and climate disasters" ever recorded (NOAA 2021). Many of the billion-dollar disasters in 2020 were extreme storm and wind events that occurred in the western half of the CHR (NOAA 2021). Because most plots in this study were measured before 2017, a relatively small portion of these large events in 2017 and 2020 are captured here. Hence, it is likely that the incidence of weather disturbance on forest land could increase, particularly in the West and South subregions, which have the highest proportional amount of weather disturbance within the CHR (Fig. 4). Further analyses that include multiple disturbance and treatment codes could reveal some additional disturbances that were not included in this study due to our methods. An example would be plots with insect damage recorded as the primary disturbance that also had weather recorded as a secondary disturbance.

CONCLUSIONS

As global temperatures rise, disturbance agents that cause minor to catastrophic damage to forests, such as extreme weather, insects, disease, and fire, are expected to increase in frequency and intensity. Disturbance data from the Forest Inventory and Analysis program can be used to quantify the impact from different disturbance types and identify broad-scale trends in disturbance regimes over time. Our results indicated that insect disturbance has increased substantially across the forest land of the Central Hardwood region over the last decade.

Disease disturbance also increased, but the occurrence is relatively low. In contrast, weather disturbance decreased during the study period, but it still made up an important component of overall disturbance. Overall disturbance has emerged as a more important factor than cutting as a driver of stand dynamics across the Central Hardwood region.

Further analyses of these data may address the susceptibility of Central Hardwood forests to different types of damaging agents based on tree species composition, recorded damages on individual trees, or other ecological or biophysical characteristics. These data may be of value for quantifying the economic impact of losses by assessing the value of damaged trees and disturbed sites.

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The Kinzua Quality Deer Cooperative: Balancing White-tailed Deer Populations and Their Browsing Impact to Forests over Two Decades of Local Land Stewardship

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Abstract

The Kinzua Quality Deer Cooperative (KQDC) was established in 2000 to determine whether coordinated management of forest habitat and white-tailed deer (*Odocoileus virginianus*) populations in northwestern Pennsylvania could improve forest ecosystem health and deer herd attributes. Across a 30,000 ha project area, the KQDC monitored how deer populations, browsing pressure, and understory plant diversity and abundance responded to vigorous adaptive hunting pressure made possible through additional antlerless tags, concurrent hunting seasons, and sustained hunter engagement. The approach was largely successful. Deer densities were quickly reduced by 50 percent and then sustained for much of the 21-year period. Concomitantly, browsing impact on woody regeneration decreased while tree regeneration became more frequent throughout the study area. Despite these successes, deer densities rebounded in the past 5 years, partly due to withdrawing hunter incentives and reducing tag limits in State-level harvest regulations. These increasing deer populations and resulting intensified browsing pressure threaten to negate the gains made in habitat and herd health.

INTRODUCTION

White-tailed deer (*Odocoileus virginianus*) population densities are currently at or above precolonial estimates of 4 to 8 deer per sq. km throughout much of the 110 million ha of forest land in the eastern United States (Hanberry and Hanberry 2020, McCabe and McCabe 1997). The consumptive and nonconsumptive impacts of high-density deer herds often diminish plant richness and abundance, degrade habitat, disrupt forest management, and hasten the spread of native and exotic plant invaders (Bernes et al. 2018). In Pennsylvania, high deer densities throughout much of the 20th century drastically altered forest understory vegetation composition and impeded sustainable forest management (Royo and Stout 2019). By the late 1960s, intense browsing influence over forest regeneration was so pervasive that even-aged regeneration harvests failed to regenerate trees. Instead, forest land shifted to fern- and herb-dominated areas. Where tree regeneration occurred, it was often dominated by the few species least susceptible to browsing, such as black cherry (*Prunus serotina*) and American beech (*Fagus grandifolia*) (Horsley et al. 2003). Thus, by the turn of the 21st century, forest managers routinely fenced regeneration harvests to exclude white-tailed deer, at considerable expense,

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to successfully regenerate diverse forests that would sustain a broad suite of ecosystem function and services.

In 2000, a group of private landowners, public land managers, scientists, hunters, and others, with the support and guidance of the Sand County Foundation (SCF), came together to form the 30,000 ha Kinzua Quality Deer Cooperative (KQDC) in northwestern Pennsylvania. The KQDC leadership team leveraged new regulations and deer management programs available through the Pennsylvania Game Commission (hereafter, Game Commission), coordinated Game Commission programs for managing habitat and deer, and developed outreach efforts. The goal of these actions was to improve habitat quality, enhance deer herd attributes, and sustain hunter engagement. To assess potential changes in habitat quality in response to lowered deer herds, the team used an extensive monitoring program to track changes in deer density, woody regeneration abundance, and browsing pressure. The KQDC capitalized on novel programs offered by the Game Commission, including concurrent antlerless and antlered deer seasons and the Deer Management Assistance Program (DMAP), which is a program that allowed landowners to request and distribute additional antlerless tags to focus hunting on specific land areas (Rosenberry et al. 2009).

In this study, we examine woody understory plant species responses to reduced white-tailed deer densities. We tested the hypothesis that reductions in deer populations will decrease overall browsing pressure on woody regeneration, thereby allowing woody regeneration to become more abundant across broad spatial and temporal scales. We further examined whether any reductions in browse pressure or gains in seedling abundance were eroded over the past 5 years as deer densities rebounded in response to lower hunting pressure resulting, in part, from a relaxation of game management policies.

METHODS

The KQDC is located within the unglaciated Allegheny High Plateau section of the Appalachian Plateau Province in northwest Pennsylvania, and has an average elevation of 613 m (range: 494 to 689 m; McNab and Avers 1994). Forests in this region are generally second-growth Allegheny hardwood, northern hardwood, and mixed oak stands that regenerated following the turn of the century exploitative harvests (Royo et al. 2021). Major species include black cherry, American beech, sugar maple (*Acer saccharum*), red maple (*A. rubrum*), black birch (*Betula lenta*), yellow birch (*B. alleghaniensis*), striped maple (*A. pensylvanicum*), white ash (*Fraxinus americana*), hemlock (*Tsuga canadensis*), and northern red oak (*Quercus rubra*).

In 2000, we randomly selected 26 blocks, measuring 259 ha (1610 m × 1610 m) each, within the 30,000 ha KQDC landscape as focal areas for data collection efforts. Beginning in 2002, we estimated deer densities yearly within each block following the fecal pellet group count method described by deCalesta (2013). Pellet group estimation methods can be inexpensively deployed across large landscapes and provide a measure of relative deer abundance (Forsyth et al. 2022). Briefly, the deCalesta (2013) method utilizes five 1,610 m long transects spaced 300 m apart to span the entire 259 ha block. Along each transect, observers count the number of pellet groups containing 10 or more fecal pellets in 1.2 m radius plots spaced at 30 m intervals. When adjusted for mortality, these surveys provide a relative estimate of deer densities present in the spring of that year.

In tandem with the pellet surveys, we estimated woody regeneration abundance and browsing impact in alternating plots along the transect (i.e., every 60 m). These surveys assess whether any tree regeneration exists on the plots and determine the frequency of occurrence (i.e., proportion of plots) and browsing intensity on six woody species: red maple, black cherry, American beech, striped maple, hemlock, and birch species. Vegetation surveys began in 2002, and birch (black and yellow, combined) was integrated into the protocol beginning in 2005. Following protocols established by Pierson and deCalesta (2015), we assessed browsing on individual stems 15 cm tall or taller but less than 1.8 m. The height range ensures that stems are accessible to white-tailed deer (Waller and Alverson 1997). Browsing impact is scored categorically, by species, as: zero (0 percent of stems browsed); low (1 to 50 percent of stems browsed); moderate (more than 50 percent of stems browsed); heavy (more than 50 percent of stems browsed and saplings were hedged); severe (more than 50 percent of stems browsed and saplings were hedged; less than 15 cm in height). For the purposes of this paper and ease of interpretation, zero and low were collapsed into a "Low" category and heavy and severe were collapsed into a "High" category.

We examined how overall tree regeneration abundance (i.e., proportion of plots containing regeneration of any tree species) varied in response to changes in deer densities across the 21-year period. Additionally, we tracked how abundance (i.e., frequency) and browsing intensity varied for each of the five focal species that we observed during all 21 years. We used repeated measures analyses of variance (ANOVAs) in a randomized complete block structure. Values for the block-level analyses used the mean value for each metric calculated from the five transects within a block. For analyses on deer density, overall stocking, and browse impact, year was considered as a fixed treatment variable and block was considered as a random variable. Analysis of individual species (browsing impact, frequency) included year, species, and year × species as fixed variables and added species (block) as a second random variable. Deer density, browsing intensity and impact, and overall stocking were modeled using a normal distribution. Abundance of the core species was modeled using a lognormal distribution. The distribution of each variable was assessed prior to analyses, and we further examined homoscedasticity and normality of the residuals using boxplots and Levene's test to ensure we met ANOVA assumptions. Correlations between years were modeled using an autoregressive first order covariance structure amended to the heterogeneous extension of the autoregressive first-order structure when necessary to adjust for heterogeneous residuals among years. When significant year effects were found, we simplified interpretation using contrasts to examine significant differences among three distinct deer management eras rather than look at pairwise differences among all 21 years (i.e., 210 comparisons). These eras were classified as before DMAP (pre-DMAP; 2002 through 2004), when harvest regulations were in the process of being implemented and hunting pressure intensified; DMAP (2005 through 2017), when regulations were sustained; and relaxed DMAP (2018 through 2022), when a combination of lowered harvest allocations, fewer DMAP tags, and abbreviated concurrent seasons combined to reduce hunting pressure on the landscape.

RESULTS

Deer populations averaged 10.4 deer per sq. km in the pre-DMAP era and then dropped by half to 5.2 deer per sq. km in the DMAP era. When harvest regulations loosened, deer

densities increased to an average of 8.3 deer per sq. km in the relaxed DMAP era (Fig. 1). As deer densities dropped, woody regeneration frequency steadily increased from 39 percent in the pre-DMAP era to 48 percent in the DMAP era and almost 54 percent in the relaxed DMAP era (Table 1). American beech was the most frequently encountered species, and its abundance increased by about 30 percent in the DMAP era before leveling off. Striped maple, the second most frequent species at the onset of the study, remained static over time. Black cherry nearly doubled in abundance in the DMAP era only to decline during the relaxed DMAP era to levels observed at the beginning of the study. Red maple steadily increased in frequency throughout the course of the study, from 3 percent of plots in the pre-DMAP era to nearly 16 percent in the relaxed DMAP era, becoming the second most abundant of the five tracked species. Finally, hemlock, the rarest species (fewer than 1 percent of plots), increased following reduction in deer densities in the DMAP era before leveling off.

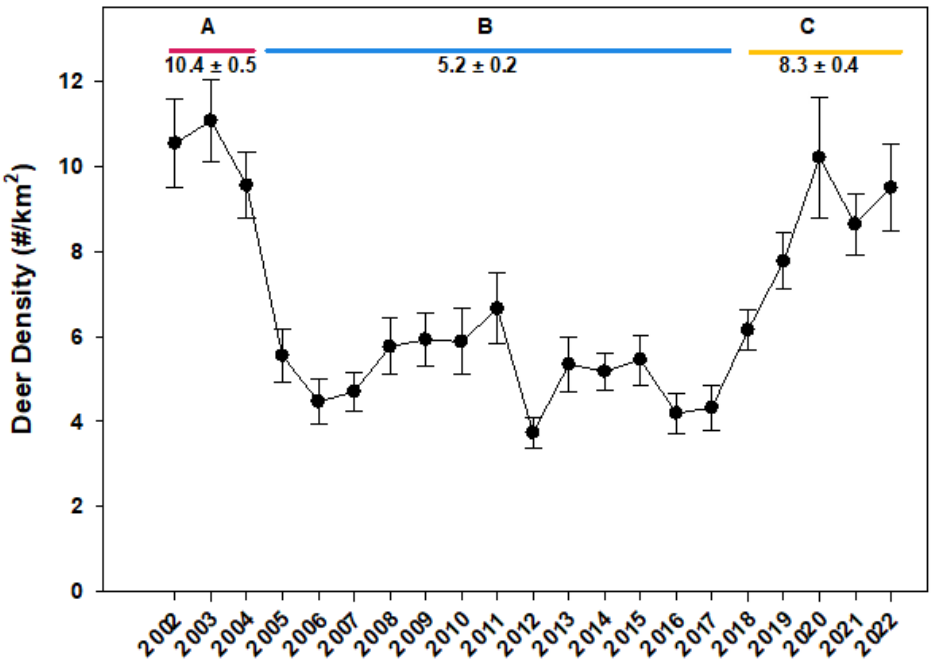

Figure 1.—Estimated deer densities (deer per sq. km) across the 21-year project period as assessed by springtime fecal pellet surveys, Pennsylvania. Error bars indicate ±1 standard error around the estimate. Years are grouped into three distinct deer management eras and indicated by the horizontal lines at the top of the graph. The pre-Deer Management Assistance Program (DMAP) era (2002–2004), prior to the institution of enhanced doe permit availability and concurrent antlered and antlerless harvest, is labeled A. The DMAP era (2005–2017), when game management policies were in place, is labeled B. The relaxed DMAP era (2018–2022), during which extra doe permits were available, is labeled C. Values under the horizontal lines indicate average deer densities for each era.

Table 1.—Average woody regeneration frequency ± 1 standard error throughout the Kinzua Quality Deer Cooperative landscape in northwestern Pennsylvania, as assessed by springtime browse impact surveys. Yearly data are summarized as the averages of each of three deer management eras: pre-Deer Management Assistance Program (DMAP) era (2002–2004), the DMAP era (2005–2017), and relaxed DMAP era (2018–2022). Different lowercase letters denote significant differences among the three eras.

Species	Pre-DMAP	DMAP	Relaxed DMAP
Any regeneration	39.3% ± 1.7% a	48.4% ± 0.8% b	53.6% ± 1.1% c
American beech	16.9% ± 1.5% a	22.2% ± 0.6% b	23.8% ± 1.0% b
Striped maple	7.5% ± 0.9%	9.0% ± 0.4%	7.4% ± 0.5%
Black cherry	3.4% ± 0.5% a	5.9% ± 0.3% b	3.5% ± 0.6% a
Red maple	3.0% ± 0.6% a	10.5% ± 0.7% b	15.9% ± 1.2% c
Hemlock	0.6% ± 0.1% a	2.1% ± 0.2% b	2.2% ± 0.2% b
Birch species	Not measured	7.8% ± 0.4% a	10.4% ± 0.6% b

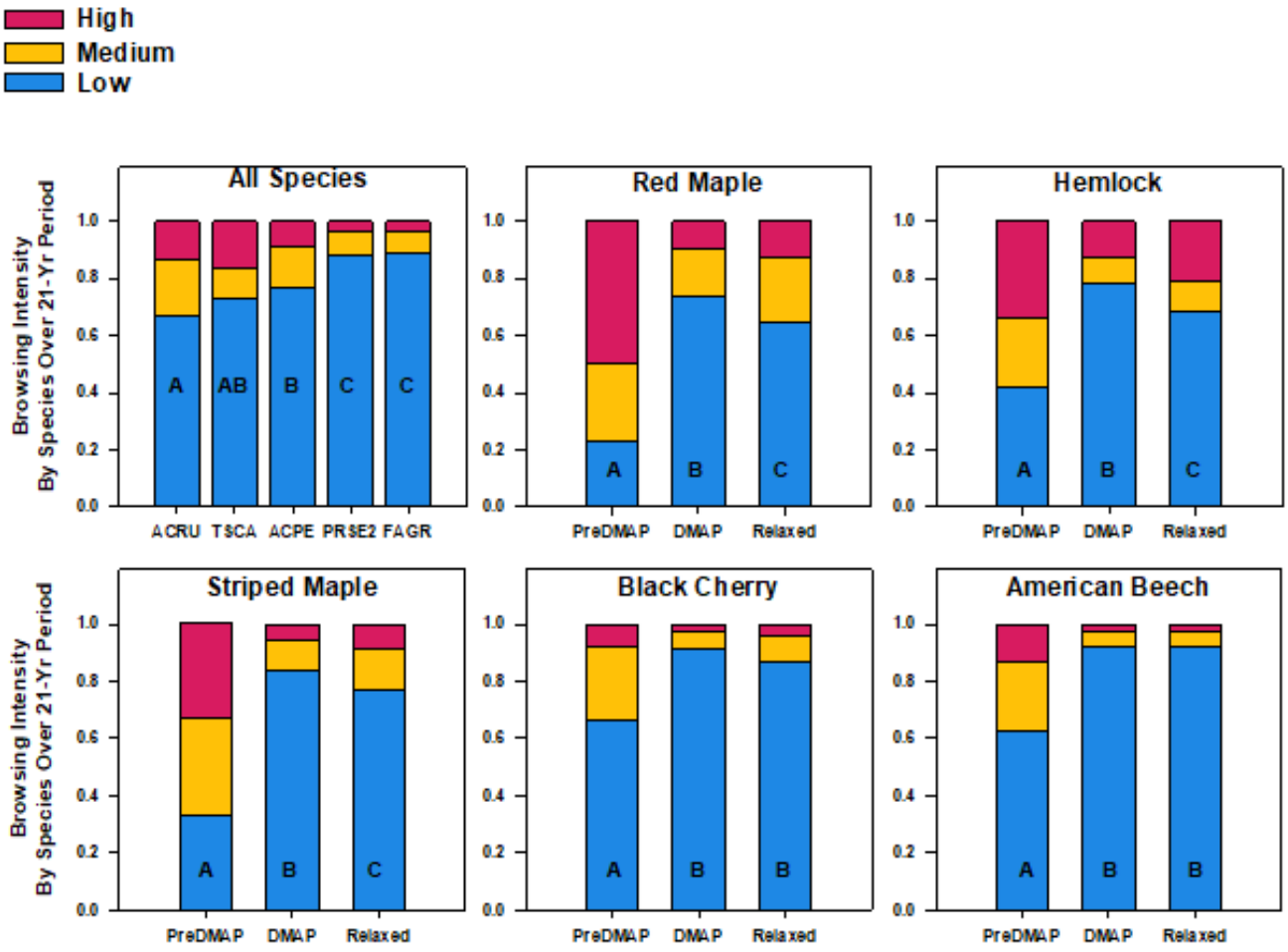

Figure 2.—Average browsing intensity throughout the Kinzua Quality Deer Cooperative landscape, Pennsylvania, as assessed by the springtime surveys for each Deer Management Assistance Program (DMAP) era, for all species combined and each of five focal species: red maple (ACRU), hemlock (TSCA), striped maple (ACPE), black cherry (PRSE2), and American beech (FAGR). Browsing is summarized by category: Low is 50 percent or fewer stems browsed; Medium is more than 50 percent stems browsed but seedlings are not hedged; and High is more than 50 percent of stems browsed and browsing has prevented growth into sapling size class. Yearly data are summarized as the average in each of three deer management eras: pre-DMAP era (2002–2004), DMAP era (2005–2017), and relaxed DMAP era (2018–2022). Different uppercase letters on bars denote significantly different browse susceptibilities.

The focal tree species differed in browse susceptibility (intensity) when averaged across the 21-year period (Fig. 2). Red maple and hemlock were the species most susceptible to browsing, striped maple had intermediate susceptibility, and black cherry and beech were the least susceptible species (Fig. 2). When examined individually, browsing pressure on every species declined in the DMAP era. The proportion of plots rated as experiencing low browse pressure increased by between 37 percent (black cherry) and 215 percent (red maple) in the DMAP era relative to the pre-DMAP era (Fig. 2). However, as deer densities rebounded in the relaxed DMAP era, browsing pressure increased on red maple, hemlock, and striped maple (Fig. 2).

DISCUSSION

Over 21 years, deer populations within the KQDC have waned and waxed as deer management policies changed. Beginning in 2000, the Game Commission enacted a series of policy changes that caused a rapid and substantial reduction in deer populations statewide. These changes, including the availability of additional doe tags through the DMAP and the 2-week concurrent season, concentrated hunting pressure on antlerless deer (Rosenberry et al. 2009, Wildlife Management Institute 2010). As deer densities and associated browsing pressure declined within the KQDC project area, woody regeneration of various focal species, as well as regeneration overall, became more abundant across the landscape. Thus, this study demonstrates that reducing deer population over time to levels within the historical range of variation (4 to 8 deer per sq. km; Hanberry and Hanberry 2020), enables forest understory recovery. These results extend findings from small-scale exclosure experiments that similarly find elimination of browsing pressure enhances tree regeneration (reviewed by Bernes et al. 2018). Other studies document that woody regeneration rapidly responds positively to reductions in browse pressure relative to other plant life forms (Habeck and Schultz 2015). This differentiation occurs as reproductive individuals remain on-site and can continue to supply propagules that fuel recruitment, whereas herbaceous layer species are often locally extirpated or at low abundance (Habeck and Schultz 2015).

Our monitoring protocols determined that the five focal taxa separated into three palatability bins. We found red maple and hemlock, species reported as moderately to highly preferred or intolerant to browsing, or both (Frelich and Lorimer 1985, Lesser et al. 2019), were most likely to suffer browsing, and both increased in abundance (i.e., frequency) as deer populations declined in the DMAP era. In contrast, our monitoring efforts ranked black cherry and American beech as least susceptible to browsing, corroborating findings by other researchers (Champagne et al. 2021, Healy 1971). Although our rapid browsing assessment is less detailed than other published methods (Morellet et al. 2001, Potvin 1995, Waller et al. 2017), it proved remarkably accurate in discerning palatability rankings among species. Such coherence with relatively low-resolution data is likely possible through the use of long-term records collected across large areas scaled to the average home range size of white-tailed deer (e.g., Tierson et al. 1985). This aggregate approach likely reduces the high temporal and spatial variation often observed in browse impact surveys; the observed variation may be attributable to shifts in deer populations and forage composition and availability, among other factors (see review by Russell et al. 2001).

Following a decade of sustained low deer densities, some hunting regulations that had allowed effective control over deer populations were changed. Specifically, overall harvest allocations for the wildlife management unit (WMU) encompassing the majority of the KQDC were reduced by as much as 50 percent, DMAP antlerless tags were decreased, and the 2-week concurrent antlered and antlerless season was shortened by a week. Consequently, deer populations within the KQDC increased by 60 percent in the relaxed DMAP era relative to the prior DMAP era. This 5-year average masks the true magnitude of the increases as the deer densities have increased steadily since 2017 and are currently almost as high (9.49 deer per sq. km) as they were before the DMAP era (about 10 deer per sq. km). Concomitantly, our results clearly demonstrate that browsing has intensified on woody regeneration, particularly on the most palatable species. The proportion of plots categorized as experiencing low browsing for red maple, hemlock, and striped maple dropped by 11, 13, and 8 percent, respectively, between the DMAP era and the relaxed DMAP era. Despite increases in deer densities and the subsequent increase in browse pressure, during the relaxed DMAP era, frequencies of the most palatable species have remained constant (hemlock, striped maple) or even increased (red maple). Other studies have found that changes in stem densities often lag behind shifts in deer densities as demographic inertia in tree populations, including recruitment, growth, and mortality, are not immediately impacted by browsing (Bradshaw and Waller 2016, Rooney et al. 2002). However, although both black cherry and American beech were not highly preferred, frequencies for both species changed significantly over time, likely because of forest health issues. American beech abundance in forest understories has increased regionally by as much as 350 percent from increased root suckering as mature trees decline from beech bark disease (Garnas et al. 2011). Black cherry is the dominant hardwood in the Allegheny forest canopy and has suffered declines of up to 76 percent in the past 20 years, putatively from diminished nitrogen deposition and increased fungal virulence (Royo et al. 2021).

CONCLUSIONS

Over its 21-year history, the Kinzua Quality Deer Cooperative project has become a world-class demonstration of how adaptive forest and game management, coupled with extensive outreach and communication, can improve both forestry outcomes and hunting outcomes throughout a 306 sq. km managed landscape of public and private ownerships (Reitz et al. 2004, Royo and Stout 2019, Stout et al. 2013). The coordinated approach, coupled with game management policies, was successful in sustaining effective hunting pressure over the resident herds, a task that proves increasingly difficult as the active hunter population drops (Diefenbach et al. 2022). Results from the project have shaped the Pennsylvania Game Commission decisions on deer management over two decades and informed forest monitoring guidelines for the state forests of the Commonwealth of Pennsylvania. Although our results document increased woody regeneration when deer densities were sustained at lower levels, research in browse legacy forests strongly suggests herd reductions on decadal timescales are necessary to regain diverse understory plant communities (McGarvey et al. 2013, Nuttle et al. 2014, Royo et al. 2010, Tanentzap et al. 2012). Hence, the current trajectory of rising deer populations places the gains made in the KQDC area at risk. In fact, during the era of the Deer Management Assistance Program, use of operational-scale fences to protect regeneration dropped drastically throughout the KQDC region (Stout et al. 2013). However, in Pennsylvania, use of fencing has re-emerged as a viable management option to protect and ensure diverse regeneration (Parker et al.

2020). Encouragingly, recent changes to season and bag limits in Pennsylvania expanded Sunday hunting opportunities and reinstituted the statewide 2-week antlered and antlerless season (Pennsylvania Game Commission 2021). Further monitoring will be required to assess whether these policies help curb browsing impacts and what effect, if any, rising deer populations over the past 5 years had on overall plant community diversity.

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Nitrate Assimilation by Maples and the Herbaceous Layer in a Central Appalachian Hardwood Forest

Sian E. Eisenhut and Kirsten Stephan[1]

INTRODUCTION

Forest plants that can assimilate nitrate may act as a nitrate sink and consequently reduce leaching losses of nitrate from watershed ecosystems (Rothstein et al. 1996). Deciduous forests of the northeastern United States have experienced high levels of nitrogen (N) deposition for many years (National Atmospheric Deposition Program 2021), prompting a large body of research on N saturation and nitrate leakage (e.g., Aber et al. 1989, Bobbink et al. 2010, Galloway et al. 2008, Rose et al. 2015). In addition, watershed tree species composition was found to correlate with nitrate leaching measured as stream water nitrate (Peterjohn et al. 2015). Other factors that are linked to increased concentrations of soil nitrate are the recent removal of vegetation (Likens and Bormann 1974, Likens et al. 1970, Martin et al. 1986, Pardo et al. 1995), old growth (versus second growth) (Goodale and Aber 2001, Ollinger et al. 2002), N fertilization (Binkley et al. 1999, Gundersen et al. 2006), or a northerly (versus southerly) aspect (Gilliam 2010, Gilliam et al. 2015). However, only a few studies have quantified nitrate use of mature temperate deciduous trees, and even fewer have quantified nitrate use of herbaceous layer species in these forests (Eddy et al. 2008, Olsson and Falkengren-Grerup 2003, Tang et al. 2012).

This study investigated nitrate reductase activity (NRA), an indicator of plant nitrate uptake and N assimilation, of mature red maple (*Acer rubrum*), sugar maple (*A. saccharum*), and the surrounding herbaceous layer species under conditions conducive to different levels of soil N availability. These conditions comprise an older versus a younger second growth forest, fertilized versus unfertilized, and a north versus south aspect.

METHODS

This study took place in the temperate montane hardwood forest of the USDA Forest Service's Fernow Experimental Forest located in the central Appalachian Mountains of West Virginia. The study area comprised four adjacent watersheds (see Figure 1 in Smith and Stephan 2021) that naturally regenerated after heavy cutting around 1910 (Kochenderfer 2006). Watersheds differ in aspect, fertilization treatment, stand age (due to more recent disturbance history), and relative basal area of dominant tree species (see Table 1 in Eisenhut et al. 2022). Briefly, watershed (WS) 10 and WS13 remained undisturbed after the 1910 cutting; WS10 faces due south and WS13 faces due north. WS3 and WS7 were clearcut again over multiple years in the 1970s. WS3 has been fertilized three times annually with a total of 35 kg N per ha per year via aerial applications of granular ammonium sulfate ($(NH_4)_2SO_4$) between 1989 and 2019 (Kochenderfer 2006).

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We quantified NRA in red and sugar maple trees and surrounding common herbaceous layer species at the tissue (foliage, roots) and plot level. NRA measurements were conducted in 18 plots (5 m radius) per watershed in summer 2018 and summer 2019 (three sampling campaigns) and in 9 plots per watershed in spring 2019 (one sampling campaign). To evaluate the relative importance of trees and herbaceous layer plants in nitrate uptake on a per-area basis (referred to as NRAA, in μmol per sq. m per ha), NRA values (μmol per g per ha) were multiplied by tissue biomass per area of ground (g per sq. m). In the statistical analyses, NRA of (1) summer overstory maple foliage, (2) summer herbaceous layer foliage, (3) maple and herbaceous layer roots, (4) maple and herbaceous layer roots versus foliage, and (5) herbaceous layer foliage in spring versus summer, respectively, was the dependent variable in individual analyses of variance (ANOVAs). Analyses were conducted in watershed pairs: (i) fertilized WS3 and unfertilized WS7 with a similar stand age of approximately 50 years (testing for a fertilization effect); (ii) WS7 and WS13 with differing stand ages (50 versus 110 years, respectively) and cutting history (testing for a stand age or time since disturbance effect); and (iii) WS13 and WS10 with a similar stand age of approximately 110 years but differing in watershed aspect (north versus south, respectively; testing for an effect of aspect). Independent variables in most analyses comprised watershed-level treatment and plot-center maple species, and their interaction (Eisenhut et al. 2022).

RESULTS

In all species, NRA was detected in both foliage and roots. Comparing the mature maple species, mean sugar maple foliar NRA was approximately 60 percent greater than mean red maple foliar NRA (Fig. 1A). However, NRA of mature maples did not vary significantly among the watersheds. Some herbaceous layer species had much higher NRA values than the maples (Fig. 1). In the herbaceous layer, differences in summer foliar NRA between watersheds were detected for individual species. In particular, tissue from violets (*Viola*), Christmas fern (*Polystichum acrostichoides*), and intermediate wood fern (*Dryopteris intermedia*) had greater NRA in the fertilized watershed (WS3) than in the other watersheds (Fig. 1B,D). Herbaceous layer species NRA did not differ whether the species were growing under red or sugar maple. When comparing overstory and herbaceous layer root NRA of fertilized WS3 and unfertilized WS7, across all six species or groups studied (red maple, sugar maple, blackberry [*Rubus* spp.], common greenbrier [*Smilax rotundifolia*], ferns, violets), root NRA was greater in fertilized WS3 than in unfertilized WS7 ($P < 0.05$). Foliar NRA was significantly greater than root NRA for sugar maple and for each of the four herbaceous layer species ($P < 0.05$), but results were the opposite for red maple. In the analysis of herbaceous layer foliar NRA across one spring campaign and three summer campaigns in fertilized WS3 and unfertilized WS7, spring foliar NRA was greater ($P < 0.05$) than all summer NRA campaigns for blackberry and common greenbrier. Foliar NRA of star chickweed (*Stellaria pubera*) and violets was greater in spring than in two of the three summer campaigns. Combining NRA with biomass, the herbaceous layer NRA at the plot level (NRAA) accounted for 9 to 41 percent of the total (tree + herbaceous layer) foliar NRAA and between 21 and 29 percent of total (foliar + fine root) NRA during the growing season.

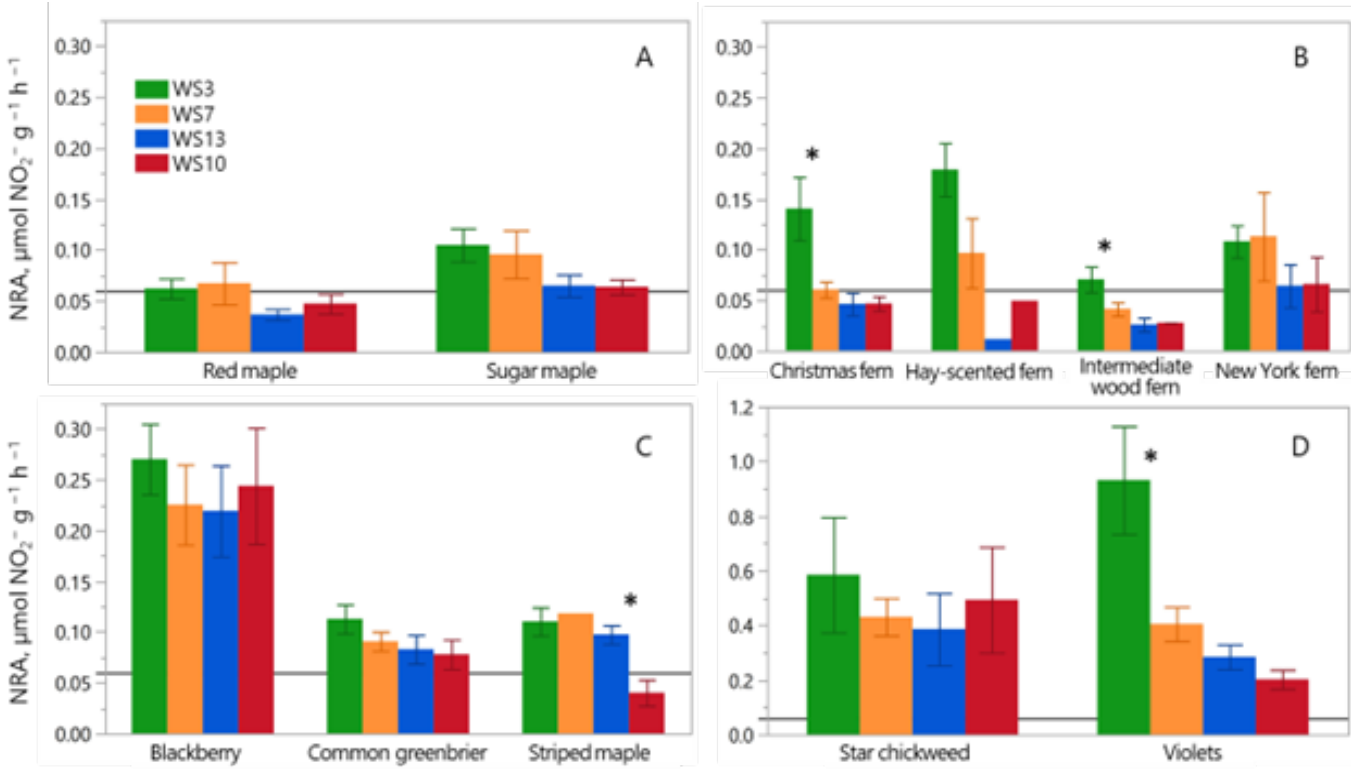

Figure 1.—Mean summer foliar nitrate reductase activity (NRA) in each watershed (WS) of overstory maple species (A), herbaceous layer ferns (B), shrubs and vines (C), and common herbaceous species (D). Data were collected at the Fernow Experimental Forest from fertilized WS3 (50-year-old stand), and unfertilized WS7 (50-year-old stand), WS13 (110-year-old stand, north facing), and WS10 (110-year-old stand, south facing). Note the different scales of the y-axes. The horizontal reference lines show the mean of all red and sugar maple foliar NRA (0.06 µmol NO2- per g per hour). Error bars represent 1 standard error based on n=27 for overstory maples (nine trees per maple species in each of three sampling campaigns in each watershed) and maximum n=18 for herbaceous layer species (i.e., the number of plots per watershed; NRA was averaged across the three summer sampling campaigns before calculating the watershed mean). Asterisks indicate significant differences in NRA between the two watersheds closest to the asterisk. In most comparisons, species had higher NRA in fertilized WS3 than unfertilized WS7. Modified from Eisenhut et al. (2022).

DISCUSSION AND CONCLUSIONS

This study corroborates that mature temperate tree species may not respond to nitrogen fertilization via an increase in their rate of nitrate assimilation (Rothstein et al. 1996, Tang et al. 2012). It is therefore not surprising that presumably more-subtle differences in soil N availability due to stand age (or disturbance history) or watershed aspect in this study did not translate into discernable differences in NRA of mature red and sugar maples. NRA of herbaceous layer plants also did not respond to stand age or watershed aspect. However, in the herbaceous layer some species appeared to respond to fertilization with greater foliar NRA or root NRA, or both. Similar results were reported for seedlings of some northeastern U.S. tree species; some had elevated NRA with fertilization while other species did not (Downs et al. 1993). These findings suggest genetic determination of the ability of a species to respond (perhaps depending on developmental stage) with greater foliar NRA to greater soil N availability, as opposed to blanket environmental induction across species (Tang et al. 2012). In addition, this study indicates that herbaceous plants may also contribute to nitrate retention in the spring through greater NRA in spring than summer. This would expand the vernal dam hypothesis on the role of herbaceous plants in N retention (Blank et al. 1980, Muller and Borman 1976), which suggests that spring ephemeral plants (completing their life cycle in spring) temporarily retain N before trees

are active. While the relatively low NRA values measured in this study confirm a low affinity of many forest species for soil nitrate, the large biomass of nitrate-assimilating tissues likely prevents significant amounts of nitrate from leaching. Disproportionate to herbaceous layer biomass in the forest, nitrate assimilation of the herbaceous layer highlights the outsize importance of this stratum in forest nutrient dynamics.

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Silviculture

Occurrence of Oak Stump Sprouts in Pennsylvania and Their Fate in the Post-harvest Environment

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INTRODUCTION

In 2000 the Forestry Sciences Labs at Irvine, PA, and Morgantown, WV, combined forces to begin updating the SILVAH decision-support system to be more applicable to mixed-oak (*Quercus*) forests in the mid-Atlantic region. A major part of this update was the installation and repeated measurements of a large-scale, long-term oak regeneration project (Study 80) that followed the survival and growth of oak reproduction (seedlings and stump sprouts) after the final harvest of two- and three-cut shelterwood sequences. The objective of this research was to develop dominance probabilities for the oak reproduction based on preharvest variables that could easily be measured by foresters preparing the timber sale.

METHODS

Study 80 took place in 32 mixed-oak stands on Pennsylvania Bureau of Forestry (PBF) lands that were harvested from 2002 through 2006. Prior to the final harvests, we established up to twenty 0.1 acre (37.2 foot radius) plots in each stand. For assessing oak stump sprouting rates, we tagged three to six healthy oak trees marked for harvest in each plot. We identified each tagged tree to species: black (*Quercus velutina*), chestnut (*Q. montana*), northern red (*Q. rubra*), scarlet (*Q. coccinea*), and white oak (*Q. alba*). We measured the diameter at breast height (d.b.h., 4.5 feet above ground) of each tagged tree to the nearest inch. We grouped the oaks into four d.b.h. classes as follows: less than 6 inches (sapling), 6 to 11 inches (pole), 12 to 16 inches (small sawlog), and more than 16 inches (medium and large sawlog); these classes had been previously used by Sander et al. (1984). Additionally, we collected topographic data in each plot to classify site quality (defined as site index at age 50, or SI_{50}) as low (oak SI_{50} less than 63 feet), medium (oak SI_{50} 63 to 74 feet), or high (oak SI_{50} greater than 74 feet) based on existing guidelines (Brose et al. 2008). Also prior to the final harvests, PBF foresters rated each stand for browsing impact by white-tailed deer (*Odocoileus virginianus*) using accepted procedures (Brose et al. 2008). Those stands rated as moderate, high, or extreme deer impact were subsequently fenced with an 8-foot-high woven-wire fence to exclude deer while those rated as low or very low were left unfenced.

RESULTS, DISCUSSION, AND CONCLUSIONS

We tagged a total of 2068 oak trees at the start of the project. Chestnut oak was the most tagged species (*n*=804) followed by northern red (579), white (298), black (248), and scarlet (139). The most frequently tagged d.b.h. class was small sawlog oaks (*n*=705)

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followed by pole (580), medium and large sawlog (495), and sapling (288). Most tagged oaks were on low-quality sites (*n*=953) followed by medium-quality sites (868) and high-quality sites (247) (Table 1).

Initial sprouting rates ranged from 0 to 100 percent with the extremes in this range being caused by small sample sizes in some of the combinations of site, species, and d.b.h. class. When combinations with fewer than 20 tagged oaks are excluded, sprouting trends are similar to those reported by Sander et al. (1984). Overall, chestnut oak displayed the highest sprouting rate (88 percent), followed closely by northern red oak (85 percent). White oak displayed the lowest sprouting rate (61 percent). Smaller diameter oaks were more likely to sprout than larger diameter oaks, especially when the d.b.h. of the parent tree exceeded 16 inches. Site quality did not appear to influence initial stump sprouting trends.

Table 1.—Distribution of oak stump sprouts by species, parent tree diameter (d.b.h.) class, and site class (site index [SI] at age 50). The three-number sequence for each combination is the number of trees initially tagged, the number of tagged trees that initially sprouted, and the number of sprouts that were dominant or codominant 15 years later, respectively. The number in parentheses is the percentage of stumps that produced dominant or codominant sprouts 15 years after cutting.

Oak species on low-quality site (oak SI_{50} <63 feet)	**Diameter class: <6 inches**	**Diameter class: 6 to 11 inches**	**Diameter class: 12 to 16 inches**	**Diameter class: >16 inches**
Black	19/13/4 (21)	28/27/14 (50)	41/29/6 (15)	21/13/4 (19)
Chestnut	72/58/32 (44)	152/139/70 (46)	141/121/53 (38)	55/41/20 (36)
Northern red	42/39/22 (52)	55/52/38 (69)	70/66/41 (59)	33/25/9 (27)
Scarlet	21/17/13 (62)	20/18/10 (50)	33/28/9 (27)	7/4/0 (00)
White	19/18/4 (21)	70/47/12 (17)	42/28/3 (07)	12/6/0 (00)
Oak species on medium-quality site (oak SI_{50} 63 to 74 feet)	Diameter class: <6 inches	Diameter class: 6 to 11 inches	Diameter class: 12 to 16 inches	Diameter class: >16 inches
Black	13/13/1 (08)	20/14/4 (20)	59/29/12 (20)	38/15/0 (00)
Chestnut	49/44/10 (20)	102/94/33 (32)	117/90/26 (22)	63/48/3 (05)
Northern red	21/18/8 (38)	30/28/6 (20)	73/61/13 (18)	81/58/4 (05)
Scarlet	4/4/2 (50)	15/11/5 (33)	27/19/6 (22)	14/6/0 (00)
White	15/13/1 (07)	56/35/8 (14)	53/25/5 (09)	18/6/0 (00)
Oak species on high-quality site (oak SI_{50} >74 feet)	Diameter class: <6 inches	Diameter class: 6 to 11 inches	Diameter class: 12 to 16 inches	Diameter class: >16 inches
Black	4/4/3 (75)	1/1/1 (100)	2/0/0 (00)	2/0/0 (00)
Chestnut	6/6/0 (00)	20/19/6 (30)	18/15/2 (11)	7/6/0 (00)
Northern red	3/2/1 (33)	8/8/3 (38)	27/21/8 (30)	136/90/6 (04)
Scarlet	None found	None found	None found	None found
White	0/0/0 (00)	3/3/0 (00)	2/0/0 (00)	8/2/0 (00)

After 15 years, the influence of site quality became apparent, with 38 percent of the tagged oaks on the low-quality sites achieving canopy dominance or codominance while that success rate was 17 and 12 percent, respectively, for medium- and high-quality sites. This result agrees with Olson and Della-Bianca (1959), who showed oaks were much more competitive when SI_{50} was less than 63 feet. It also agrees with Weigel and Peng (2002), who found over time a decreased competitiveness of oak stump sprouts on higher quality sites. These differences in oak stump sprout competitiveness among sites is likely due to differences in non-oak competitors (red maple [*Acer rubrum*], sweet birch [*Betula lent*a], and yellow-poplar [*Liriodendron tulipifera*] on medium- to high-quality sites; blackgum [*Nyssa sylvatica*], hickory [*Carya*], and pine [*Pinus*] on low-quality sites) and periodic short-term summer droughts. Among species, scarlet oak stump sprouts were the most likely to be dominant or codominant (32 percent) although this rate is likely inflated as the species was not found on high-quality sites. Of the four oak species found on all site classes, northern red oak sprouts were at least codominant 27 percent of the time, followed by black and chestnut oak (each 20 percent) and white oak (11 percent). These rates are markedly less than those reported by Sander et al. (1984), but the discrepancy could be due to differences in non-oak competitors, site quality, and study duration between the two studies. The order of species regarding sprout success differs between the two studies but again may be due to differences in the aforementioned factors. However, both studies consistently report white oak as being the least successful sprouter and both show a trend toward decreasing sprout success as d.b.h. of the parent tree increases.

These results will be used to develop dominance probability coefficients for the SILVAH decision-support system.

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Flexibility of the Cut-Stump Herbicide Application Window at Controlling Beech Root Sprouts

Patrick H. Brose and Gary W. Miller[1]

INTRODUCTION

The development of recalcitrant layers of shade-tolerant vegetation is one of the major obstacles to regenerating many of the commercially valuable tree species in the Central Hardwood region (Lorimer 1993, Lorimer et al. 1994). These layers are especially problematic on mesic sites, where production of high-quality timber is often a primary management goal. The canopy layers arise primarily through natural successional processes occurring in the absence of disturbance and form over decades of little or no active management. Selective browsing by white-tailed deer (*Odocoileus virginianus*), exclusion of periodic surface fires, and atmospheric deposition also contribute to the formation of these dense understory layers (Royo and Carson 2006). Layers of shade-tolerant vegetation in forest understories cause several problems for desirable tree species, most of which are shade intolerant or shade intermediate. First and foremost, this understory vegetation creates a dense shade (less than 5 percent of full sunlight) that is below the light compensation points for desirable species, so their seedlings cannot survive for more than just a year or two. Second, the understory vegetation, with its well-developed root systems, competes with desirable seedlings for nutrients and water, further exacerbating the plight of the seedlings. Finally, it provides cover for small mammals that consume the seeds cast by desirable tree species and also kill seedlings via gnawing or clipping the stem. Of the many species classified as problematic understory vegetation, American beech (*Fagus grandifolia*) is one of the most troublesome (Tubbs and Houston 1990). This native tree is very shade tolerant, is not preferred/is resistant to deer browsing, and readily regenerates via sprouting from the root systems of mature trees. Sprouting occurs from mature live beech, resulting in dense thickets of hundreds of beech sprouts around a single parent tree. Beech bark disease caused by the scale insect *Cryptococcus fagisuga* and the fungi *Neonectria ditissima* and *N. faginata* further stimulates formation of beech root sprouts by slowly killing mature beech.

Glyphosate herbicides are commonly used to control beech root sprouts (Jackson and Finley 2005, Senseman 2007). On large areas that are suitable for machinery, herbicides are applied with commercial equipment. If the area is too small or unsuitable for such machinery, herbicides are applied by individual-stem techniques (Kochenderfer et al. 2012). One of these methods, the cut-stump technique, lends itself well to controlling beech root sprouts (Chedzoy and Smallidge 2011, Jackson and Finley 2005, Kochenderfer et al. 2006). With this method, the mature beech (the parent tree) is felled during summer or fall and herbicide is applied to the freshly cut stump. The herbicide is translocated throughout the root system and kills the attached root sprouts.

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However, this technique has one caveat: published guidelines recommend that the herbicide be applied immediately after felling the parent tree (Jackson and Finley 2006, Kochenderfer et al. 2012). This constraint creates an operational safety issue. Either fellers must apply the herbicide, a practice they may not be licensed to do, or the herbicide applicator must be present during the felling, which is a safety concern. If this application window can be expanded to several days without a loss in herbicide efficacy, then the safety issue becomes a moot point.

To date, only two studies have examined this application timing issue connected to cut-stump herbicide treatments. In the first, Kochenderfer and Kochenderfer (2009) found waiting 4 hours to be just as effective as waiting only 1 hour regardless of whether the glyphosate herbicide was full strength or a 50 percent solution. In the second, Kochenderfer et al. (2013) tested whether a 50 percent solution by volume of Razor®Pro (Nufarm Americas, Alsip, IL) could be applied up to 5 days after felling of beech parent trees. They found little decrease in efficacy through 4 days and a decrease in efficacy in day 5, while still significantly more than not using herbicide. However, their study was done on a small scale—40 trees—at 1 location in West Virginia, and at just 1 time (mid-September) under cool, humid conditions.

In 2014 and 2015, we repeated the West Virginia study to verify its findings. Additionally, we incorporated weather conditions, months of application, and influence of neighboring mature beech trees into our study to understand how these variables influenced the results. Finally, we wanted to determine the extent of the area affected by the herbicide. This information will help foresters and forest landowners who want to use the cut-stump herbicide technique to control dense beech understories.

METHODS

This study was conducted in northwestern Pennsylvania in three hardwood stands located on the Bradford Water District (BWD), Clear Creek State Forest (CCSF), and State Game Lands 143 (SGL143). Each stand was 40 to 50 acres, was fully stocked, contained a mixture of tree species common to the area (American beech, black cherry [*Prunus serotina*], northern red oak [*Quercus rubra*], red maple [*Acer rubrum*], sugar maple [*A. saccharum*], white oak [*Q. alba*], and yellow-poplar [*Liriodendron tulipifera*]), and had a dense understory dominated by beech root sprouts.

In spring 2014, 120 mature beech were selected and numbered in each stand. Selected trees were at least 12 inches diameter at breast height (d.b.h.), were healthy (little or no evidence of beech bark disease), had at least 10 beech sprouts near their bases, and were located at least 100 feet from each other. Around each tree, a 0.01-acre circular plot (11.8-foot radius) was installed. All beech sprouts between 1 foot tall and 5-inch d.b.h. were tallied. Beech stems greater than 5-inch d.b.h. that were determined to be "in" according to a 10-factor prism were measured for d.b.h. and rated for crown health using established guidelines (Brose et al. 2008). Finally, the location of each tree was mapped using global positioning system coordinates to facilitate relocating the trees for applying treatments and recording subsequent results.

The mapped trees were organized into clusters of five. Each cluster was randomly assigned to a month of herbicide application (July, August, or September) and time interval (1 hour, 8 hours, 1 day, 2 days, 3 days, 4 days, or 5 days since felling and an untreated control). The

marked beech were manually felled with chainsaws in the appropriate month by foresters from the USDA Forest Service, Northern Research Station or the cooperating agency. Once each beech was felled, the stump was cut level to avoid herbicide runoff, and the cutting time was recorded. At the appropriate time after felling, a 50 percent solution of RazorPro (41.0 percent active ingredient) and water was applied to the outer 2 inches of the stump at a rate of 1.0 tablespoon per inch of stump diameter. Lastly, ambient air temperature and relative humidity were recorded once each afternoon during the 6 days it took to complete the study.

In late summer 2015, approximately 11 to 13 months after treatment, each plot was re-inventoried for living beech sprouts and trees. If a sprout or tree was more than 75 percent dead relative to its pretreatment condition, it was considered to be dead. In addition, the distance from the stump to the farthest dead sprout or tree in each of the four cardinal directions was measured.

The differences between the 2014 and 2015 beech sprout and tree data were determined by subtracting the post-treatment counts of each plot from the corresponding pretreatment tallies. These differences were analyzed using a randomized complete block design and mixed-effects analysis of covariance model (SAS Institute 2011). Month of herbicide application, time interval, and their interactions were the fixed effects in the model and stand was the random effect. The pretreatment number of beech sprouts and neighboring beech greater than 5-inch d.b.h. and the mean air temperature and relative humidity for each combination of month and time since felling were used as covariates. We evaluated the differences among fixed effects by the Tukey-Kramer least square means test at an alpha of 0.05 and residuals were examined to ensure that model assumptions were met.

RESULTS

At the beginning of the study, differences in beech sprout and tree densities existed among the three study sites. CCSF and SGL143 had more overstory beech per plot (3.6 and 3.8 trees, respectively) than did BWD (2.4 trees). CCSF and SGL143 also had more beech sprouts per plot (48.0 and 49.0 stems, respectively) than did BWD (41.0 stems). However, no differences in beech sprout and tree densities existed within the three sites among the combinations of month and time interval of treatment ($P > 0.25$).

Initial analysis showed the weather variables to be nonsignificant covariates ($P > 0.40$), so they were dropped from the analyses. Regarding beech sprout mortality, pretreatment beech sprout density was a strongly significant covariate ($P < 0.0001$). After accounting for pretreatment differences in sprout density, all herbicide applications killed more beech sprouts than not using herbicide ($P < 0.0001$) regardless of month or time interval. Regarding month of treatment, applying herbicide in September killed more beech sprouts per plot (31) than did the July and August applications: 27 and 26 sprouts per plot, respectively ($P < 0.01$). Regarding time interval, the intervals formed two groups: within the first day and after 3 days. Applying herbicide within 1 day killed more beech sprouts per plot, 34 to 37, than did applying herbicide after 3 days, 26 to 28 sprouts per plot ($P < 0.01$). The 2-day application interval killed 32 sprouts per plot and this number was not significantly different from either of the two groups ($P > 0.25$).

Examining month of application and time interval together (24 possible combinations, including controls) revealed 6 distinct groupings (Fig. 1). Treating beech stumps in September and within 24 hours killed the most beech sprouts, approximately 40 stems per plot (more than 85 percent). The next most lethal combination was treating within 24 hours in August, which killed 36 to 38 stems per plot (78 to 82 percent). The same time period for July, the 2-day August application, and the 2-day and 3-day September sprayings all killed approximately 33 stems per plot (71 percent). The 2- to 4-day intervals in July, the 3- and 4-day intervals in August, and the September 4-day combinations formed a group that killed from 28 to 30 beech sprouts per plot (60 to 65 percent). The 5-day interval for all 3 months formed another group that accounted for 23 to 26 dead beech sprouts per plot (50 to 57 percent). The untreated control in each month constituted the least lethal group, with beech sprout mortality being less than 2 per plot (4 percent).

No differences in mortality of nearby overstory beech were detected among months, among time intervals, or among their combinations ($P > 0.25$). On average, all treatments, other than the controls, killed 1 nearby overstory beech. Likewise, no differences were found among the main factors and their combinations regarding area affected by the herbicide. On average, herbicide effects occurred within about 15 feet of the treated stump (approximately 700 square feet).

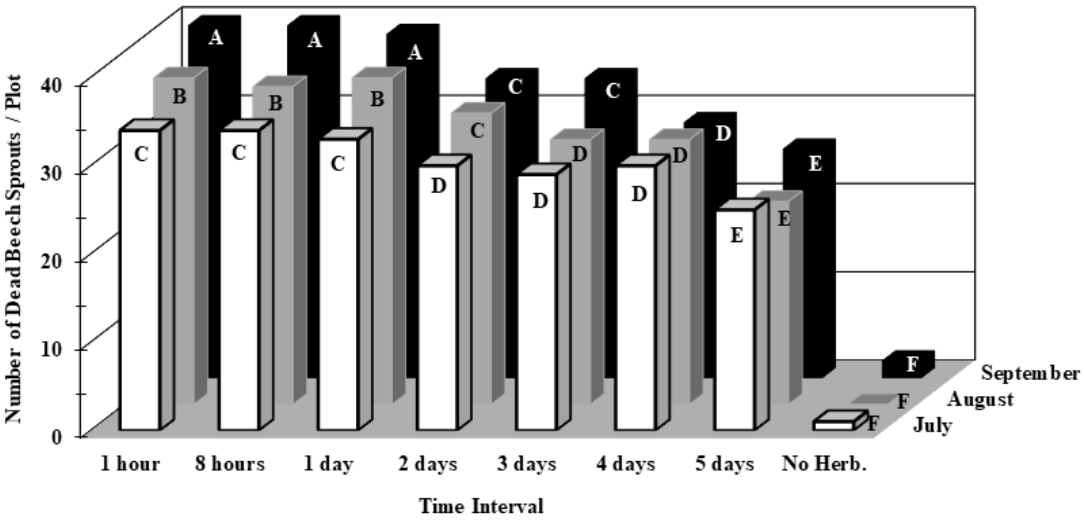

Figure 1.—The adjusted means of American beech root sprouts killed by glyphosate herbicide application to the mature mother tree via the cut-stump method, Pennsylvania. "Month" indicates when the mature mother tree was felled and "time interval" indicates how long after the felling that the herbicide was applied to the surface of the stump. Inventory of sprouts was taken again 1 year post-felling. Columns with the same letter are not significantly different at the alpha = 0.05 level.

DISCUSSION AND CONCLUSIONS

The cut-stump herbicide technique has widespread acceptance as a method for controlling problematic woody species like American beech that reproduce and spread via root sprouts (Jackson and Finley 2005, Kochenderfer et al. 2012). The one weakness of the technique is the recommendation to apply glyphosate herbicides "immediately" following felling of the parent tree. This limitation creates operational and safety issues. Loggers typically do not want to take time to treat freshly cut stumps or they may not be licensed to do so, depending on state-level herbicide application laws. Having the herbicide applied by a licensed applicator, via a contract, means that person must be on-site during timber harvesting, an obvious safety concern. The findings from this study, indicating that glyphosate herbicides can be applied up to 5 days after cutting, resolve these issues. Loggers do not have to apply herbicides to the stumps and licensed applicators can do the treatments a few days after the timber harvesting has finished or moved to another part of the stand.

These results are consistent with prior herbicide research on controlling beech root sprouts. Horsley and Bjorkman (1983) identified August and September as the optimal months for controlling beech understories with the minimum amount of glyphosate applied via broadcast spraying. We found these same months to be best when using the cut-stump method. Kochenderfer et al. (2013) found a 50 percent solution of RazorPro could be applied up to 5 days after felling overstory beech and could still achieve more than 50 percent control of beech root sprouts. All herbicide stump treatments resulted in at least 50 percent mortality of beech root sprouts.

Unlike the West Virginia study (Kochenderfer et al. 2013), our study detected differences among the various combinations of month and time interval. This is likely due to the differences in sample size and the number of months that treatments were applied. The West Virginia study included just 40 trees while our study used 360 trees. The West Virginia study applied treatments only in September. Our findings that all herbicide combinations killed one additional overstory beech and impacted an area of approximately 700 square feet are not surprising because we did not vary the herbicide concentration and we applied a consistent amount of RazorPro, based on diameter of the stump. Additionally, the sizes of the felled mature beech trees were relatively uniform. If we had used an undiluted glyphosate herbicide, varied the application rate, or treated beech of markedly different diameters, we probably would have found differences in the area impacted by the herbicide and the number of neighboring overstory beech that were killed.

Initially, we had expected that weather would influence the results; however, it was not a significant covariate. Afternoon temperature and humidity varied little during summer 2014. Additionally, the study was done in mature unharvested forests so beech stumps were shaded nearly all the time, thereby ameliorating changes in air temperature and humidity. Finally, RazorPro contains a surfactant (14 percent by volume), which would facilitate absorption of the herbicide into a stump even though its surface may have somewhat dried since felling of the parent tree.

These results allay concerns about the time interval recommendation of cut-stump herbicide treatments to control beech root sprouts. Optimal control (more than 80 percent) is achieved by applying the glyphosate herbicide within 24 hours of cutting in September. However, reasonable control of 50 to 70 percent can be obtained in July and

August and up to 5 days after felling beech parent trees. It should be noted that these results are likely to be contingent on using the same herbicide formulation at the same rate tested (50 percent solution of RazorPro applied at 1 tablespoon per inch of stump diameter). Moreover, these results apply only to American beech; other sprout-producing species such as blackgum (*Nyssa sylvatica*) and black locust (*Robinia pseudoacacia*) may respond differently.

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Planted White Oak and Natural Regeneration Response to Even- and Uneven-aged Silviculture in the Ridge and Valley Ecoregion of Tennessee

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BACKGROUND

Upland oak (*Quercus*) species are difficult to sustain in many North American stands because the regeneration process is failing. Loss of oak species will affect ecosystem services such as hydrology and wildlife food sources and important local and international economic markets (Dey 2014, Luppold and Bumgardner 2018). The objective of this study was to determine the early efficacy of two silvicultural treatments (two-aged shelterwood and an irregular shelterwood) and a competition control treatment on natural oak regeneration and planted white oak (*Q. alba*).

METHODS

A two-aged shelterwood with reserves (SWR; residual basal area of 17 square feet per acre) was compared to the first entry of an irregular shelterwood (also known by its German name, femelschlag; Raymond et al. 2009), which was implemented initially as 0.5-acre gaps, with no residual overstory, hereafter referred to as "gaps." Three replications of each silvicultural treatment were installed in mature oak-dominated stands from 2013 to 2014 on the University of Tennessee's Forest Resources AgResearch and Education Center near Oak Ridge, TN. Site index for white oak was 70 feet (base age 50; USDA Natural Resources Conservation Service 2022). Two gaps, separated by at least 500 feet, were created in each gap treatment stand. The SWR and gap treatments were split into two equal-sized areas to test a competition control treatment (i.e., split plot). Competition control consisted of two entries within the same experimental unit: preharvest herbicide injections to all midstory stems (1 to 6 inches diameter at breast height [d.b.h.], excluding oak and hickory species) and a post-harvest (year 5) herbicide injection to competition to treat undesirable species, predominantly red maple (*Acer rubrum*) and yellow-poplar (*Liriodendron tulipifera*), within a 45-degree cone of interference around natural regeneration or planted oak seedlings. Advance oak reproduction (greater than 3 feet tall, less than 1.5 inches d.b.h.) was tallied 7 years after treatment (2020) using 0.005-acre regeneration plots in each SWR unit (10 plots per SWR) and in the center of each gap (8 plots per gap center), edge of gap (4 plots per gap edge), and forest matrix outside the gap (10 to 12 plots per gap matrix extending 46 to 135 feet from gap edge). Regeneration plots in gaps were placed along four directions that represented the shortest and widest axis from gap center. In 2015, we planted 90 seedlings in each SWR unit and 72 seedlings in and around gaps on a 12-foot

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spacing using handheld machine augers with a 6-inch bit. White oak seedlings (2-0) from local sources were grown in the nursery to maximize overall size (Kormanik et al. 1994) and were visually graded (Clark et al. 2000) before planting (average height of 3.6 feet). Seedlings were assessed for survival and total height 6 years after planting (2020). We analyzed data using generalized linear mixed modeling to determine differences in natural regeneration density and planted seedling survival and height between silvicultural and competition control treatments (we did not use data from gap edge or matrix plots). We also tested differences among gap locations (center, edge, and outside) and interactions with competition control.

RESULTS

Advance oak reproduction 7 years after treatment implementation consisted primarily of chestnut oak (*Q. montana*), white oak, scarlet oak (*Q. coccinea*), and northern red oak (*Q. rubra*) (data not shown). Collectively, these oak species were 9 to 100 times more abundant in SWR stands (597 to 870 trees per acre) than in gap centers (8 to 67 trees per acre) ($F_{1,4}$ = 19.61, P = 0.01) (Table 1).[2] Differences in oak reproduction densities among the three gap locations (center, edge, matrix) were not significant ($F_{2,50}$ = 0.61, P = 0.55). Competition control did not affect oak density in the SWR ($F_{1,4}$ = 0.09, P = 0.77), but effects of competition control were nearing significance in gap centers (59 fewer oak stems per acre; 67 versus 8) with competition control ($F_{1,4}$ = 4.92, P = 0.09) (Table 1). Competition control did not affect density in gap edges ($F_{1,50}$ = 0.02, P = 0.88) or in the gap matrix ($F_{1,50}$ = 0.41, P = 0.52).

The 15 percentage point difference in 6-year survival of white oak seedlings planted in the SWR versus the gap centers was not significant (70 percent versus 55 percent; $F_{1,4}$ = 1.8, P = 0.25) (Table 1). Survival differences among gap edges (65 percent), gap centers (55 percent), and outside gaps (45 percent) were relatively large, but also not significant ($F_{2,8}$ = 3.6, P = 0.08). Competition control reduced seedling survival by 14 percentage points across the two silvicultural treatments, but this difference was not significant (70 percent versus 56 percent; $F_{1,4}$ = 6.34, P = 0.07) (Table 1). In gap centers, competition control decreased seedling survival by 27 percentage points, nearing significance ($F_{1,4}$ = 7.63, P = 0.05) (Table 1). Competition control had no effect in the SWR ($F_{1,4}$ = 0.20, P = 0.68). Survival on gap edges was 27 percentage points higher with competition control, but this effect was also not significant ($F_{1,8}$ = 4.27, P = 0.07). Competition control had no effect in the gap matrix ($F_{1,8}$ = 0.29, P = 0.61).

Six-year heights of planted seedlings were similar between the SWR and gap centers ($F_{1,4}$ = 1.95, P = 0.23) (Table 1). Seedling height was affected by specific gap location (center, edge, matrix) ($F_{2,8}$ = 9.03, P = 0.01), and planted seedlings were tallest in gap centers (6.1 feet) compared to outside gaps (3.4 feet) and gap edges (4.6 feet). Across the two silvicultural treatments, height mean differences between competition control (7.0 feet) and no herbicide plots (6.1 feet) were approaching significance ($F_{1,4}$ = 4.89, P = 0.09) (Table 1). Specific mean height differences between herbicide and nonherbicide plots, however, in both the SWR ($F_{1,4}$ = 2.11, P = 0.22) and gap centers ($F_{1,4}$ = 3.01, P = 0.16) were not significant. Similarly, competition control did not affect seedling height on gap edges ($F_{1,8}$ = 0.59, P = 0.46) or in the gap matrix ($F_{1,8}$ = 1.05, P = 0.34).

[2] Selected results are provided in Table 1; unless otherwise indicated with a reference to Table 1, data are not shown.

Table 1.—Least-squares means for natural oak advance reproduction density (trees greater than 3 feet tall, less than 1.5 inches diameter at breast height) and planted white oak seedling survival and height in shelterwood with reserve (SWR) and irregular shelterwood (gaps) stands treated with or without competition control. Means followed by different lowercase letters denote mean differences among the treatment combinations. Means followed by different uppercase letters denote differences between the main effects of silvicultural (SWR versus gap) treatment or competition control (Tukey's honestly signficant difference, $P < 0.05$).

Treatment	Density (trees per acre), SWR	Density (trees per acre), gap (center plots only)	Density across all plots (trees per acre)
Competition control	870 a	8 b	85 A
No competition control	597 ab	67 ab	200 B
Overall	720 A	24 B	
Treatment	**Survival (percent), SWR**	**Survival (percent), gap (center plots only)**	**Survival across all plots**
Competition control	69 a	41 a	56 A
No competition control	71 a	68 a	70 A
Overall	70 A	55 A	
Treatment	**Height (feet), SWR**	**Height (feet), gap (center plots only)**	**Height (feet) across all plots**
Competition control	7.4 a	6.7 a	7.0 A
No competition control	6.8 a	5.4 a	6.1 A
Overall	7.1 A	6.0 A	

DISCUSSION

The residual overstory in the SWR and slash may have moderated and varied the light environment and ground flora (Miller et al. 2006, Runkle 1987), resulting in reduced competition from shade-intolerant species like yellow-poplar and greatly improved advance oak reproduction densities compared to the gap centers. Planted seedlings had marginally better outcomes, although not statistically significant, in the SWR compared to gap centers.

Oak regeneration is typically clustered (Christensen 1977), and the 273 stem per acre increase (597 versus 870; Table 1) from competition control in the SWR may be biologically meaningful despite its lack of statistical significance. In contrast, competition control reduced natural oak reproduction densities and planted seedling survival in gap centers, albeit not significantly. The high-light environment of gap centers probably favored germinant poplar, particularly where stump sprout competitors were removed prior to the harvest, and oak seedlings were quickly overtopped, negating any effect of the second herbicide treatment. Our results contrasted with others who found that gap edges improved the natural oak regeneration process (Patterson et al. 2022). In contrast, planted seedlings on gap edges had similar survival to those in gap centers, but did obtain heights desirable for advance regeneration (between 3 and 4 feet).

The future of oak regeneration in the irregular shelterwood (gaps) is questionable because there was sparse advance reproduction on edges and forest matrix despite having 1,600 to 4,400 oak seedlings per acre less than 3 feet tall prior to treatment implementation (data not shown). Planted seedlings increased the advance oak reproduction pool by 50 to 180 stems per acre in gap edges, which is where planted trees will benefit the most from future

gap expansion (Puettmann et al. 2009). Planting outside gaps resulted in low survival and height that rarely exceeded browse line (less than approximately 4.5 feet), consistent with other studies (Berg 2004).

MANAGEMENT IMPLICATIONS

After 6 years, the most efficacious silvicultural treatment for natural reproduction was the shelterwood with reserve, which also has the advantage of being more economically feasible in the short term. After 7 years, planting was more beneficial in gaps due to their low natural oak reproduction. We used high-quality, locally adapted seedlings that probably contributed to their success, even following a severe drought in 2016. The trees were difficult to plant because of their large root systems, but planting fewer trees in targeted areas would improve efficiency. Competition control had minor positive effects on natural regeneration and on planted seedling height but tended to reduce survival. Competition control was only marginally beneficial for planted seedlings on gap edges. Gap expansions will be implemented in 2022 by creating 0.3-acre openings that adjoin existing gaps, and we will determine whether the irregular shelterwood can be used to retain oak in this experiment over successive entries.

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Growth and Yield

Estimating Sustainable Harvest Levels Within the Northern Minnesota and Ontario Peatlands (NMOP) Section Using a Fully Regulated Age-Class Distribution Approach

Curtis L. VanderSchaaf [1]

Abstract

To help the Minnesota Department of Natural Resources (DNR) achieve its goal of harvesting 800,000 cords annually, a planning process called Section/Subsection Forest Resource Management Planning (SFRMP) was used. Planning is based at the section and subsection level of the DNR ecological classification system. As part of this planning process, ecological sections were modeled separately using the program Woodstock to determine their potential contribution to the statewide target. Woodstock uses linear programming along with landscape-level inputs and growth and yield estimates to project the landscape subject to management constraints while attempting to maximize financial returns of harvested volumes when discounted back to year 0. To verify projections from Woodstock, a separate analysis was conducted assuming the forested landbase was in a fully regulated condition. By assumption, for each cover type, an equal number of acres existed for each 1-year age-class up to some biological rotation age. For each cover type, a cover type-specific constant number of acres of equal productivity was harvested annually.

The Woodstock linear programming and fully regulated harvesting techniques were applied in 2015 during the SFRMP for the Northern Minnesota and Ontario Peatlands (NMOP) ecological section. Harvests within this section accounted for much of the annual harvest target. Woodstock was used to examine four scenarios that differed in the amount of desired future forest conditions, resulting in different average annual harvests over the next 50 years. These estimated harvest amounts, in cords, were 239,048; 257,720; 254,534; and 266,332. The fully regulated approach estimated 214,087 cords annually. It appears that the Woodstock scenario estimates are reasonable depictions of harvest levels that could occur over the next 50 years.

INTRODUCTION

Around 2015, the Minnesota Department of Natural Resources (DNR) Division of Forestry (DOF) managed close to 2.79 million acres of timberland. Products produced included timber for paper, pulp, oriented strand board, lumber, and occasionally logs for log homes, as well as fiber for wood-burning biomass facilities, and even decorative products such as balsam boughs and spruce tops. During 2015, the DOF had a goal of offering for sale annually 800,000 cords of timber. This goal was based on historical harvest levels and on timber supply assessments. The long-term ability to meet that target relied upon timber supply assessments largely dependent on yield tables. Planning allows the DNR to strategically and sustainably manage state forest land over time.

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In the past, the DNR directed timber harvesting on administered lands through 5- to 10-year forest resource management plans developed for each of its administrative forestry areas. Opportunities for public involvement were limited regarding the development and review of these timber management plans.

In response to growing public interest in DNR timber management planning, the DNR Section/Subsection Forest Resource Management Plan (SFRMP) process was developed around the early 2000s to provide a more standardized formal process and opportunities for increased public involvement. In addition, planning is based at the section and subsection levels of the DNR ecological classification system (ECS) rather than DNR administrative areas as in the past (i.e., DNR Administrative Area forestry boundaries). As part of the SFRMP process, an assessment of the Northern Minnesota and Ontario Peatlands (NMOP) ECS section (Fig. 1) was conducted around 2015 (Minnesota DNR 2014). The NMOP section contains two ECS subsections: the Agassiz Lowlands and the Littlefork-Vermillion Uplands.

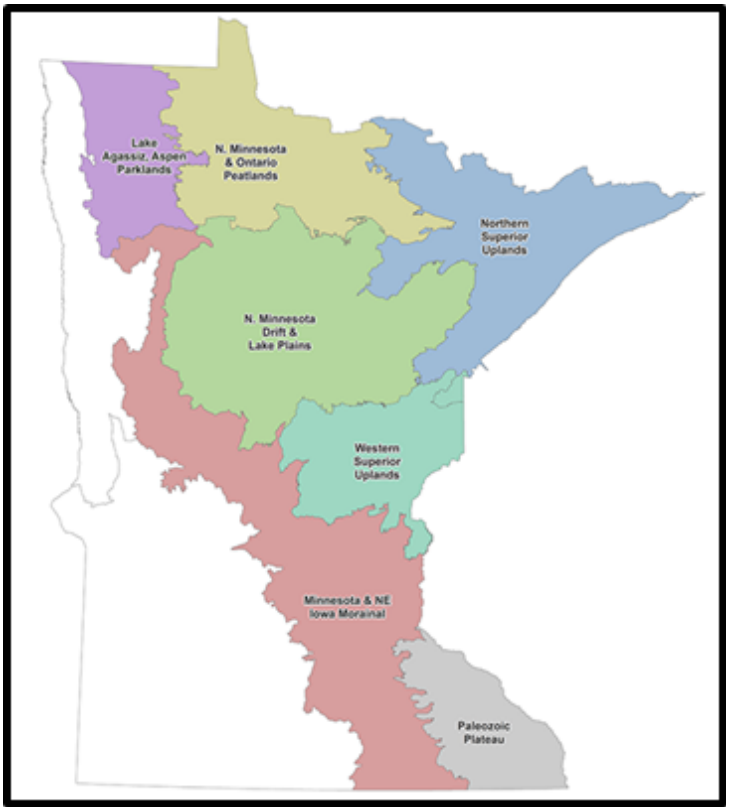

Figure 1.—Map of Minnesota showing the Northern Minnesota and Ontario Peatlands (NMOP) section, as classified by the Minnesota Department of Natural Resources (DNR). The DNR Ecological Classification System divides this section into two subsections: the Agassiz Lowlands and the Littlefork-Vermillion Uplands. Source: Minnesota DNR (2023), reprinted with permission.

The NMOP ECS section covers about 5.3 million acres in north-central Minnesota adjacent to the Canadian border. Land cover to a great extent in the NMOP is identified as lowland conifer deciduous forest, upland deciduous forest, and aquatic environments (swamp, wetlands). This region is characterized by flat and poorly drained topography, and about half of the ecological section consists of clayey deposits from Glacial Lake Agassiz. There is an extensive amount of peatland that supports forest ecosystems such as tamarack (*Larix laricina*) and black spruce (*Picea mariana*) swamps and spruce bogs, and other forest ecosystems including mesic and wet forests of mixed conifer and boreal hardwood species, northern white-cedar (*Thuja occidentalis*), or black ash (*Fraxinus nigra*). On uplands, aspen-birch ((*Populus-Betula*) and fire-dependent jack pine (*Pinus banksiana*) or red pine (*Pinus resinosa*) forests are found.

A little over 2.9 million acres (55 percent) of land in the NMOP is in public ownership (Federal, county, State), of which State ownership accounts for approximately 2.3 million acres. A SFRMP was conducted for the 2.3 million acres administered by the DNR Division of Forestry and Division of Fish and Wildlife to specify on what acres a treatment prescription is to be conducted over the next 10 years (10-year stand exam list).

The SFRMP results presented in this document are from the second SFRMP prepared for these specific DNR lands in northern Minnesota. The original SFRMPs were prepared in 2002 for the Agassiz Lowlands subsection and in 2010 for the Littlefork-Vermillion Uplands. The 2002 and 2010 SFRMPs provided direction for vegetation management from 2002 through 2015. The results from this second NMOP SFRMP are being used to provide initial direction for vegetation management starting in Fiscal Years 2016 through 2025. The term "initial direction" is used because efforts have been made and continue to be made to accommodate new information relative to emerging issues and the need for monitoring and potentially adaptive management.

As part of the 2015 SFRMP, landscape modeling was conducted by using the software package Remsoft® Spatial Planning System (RSPS; Remsoft, Fredericton, NB). Woodstock is a component of RSPS that allows users to examine how various land uses, management alternatives, and social policies will affect timber supply at a strategic level, given the existing forest/cover types and stand inventories. "Strategic" means at a large scale, and ignores the spatial relationships between individual stands.

Within DNR strategic-level plans, the planning horizon was 150 years, but the NMOP planning team analyzed only the initial 50 years. For this analysis, the objective function in Woodstock was to maximize discounted revenues to the present time. Woodstock uses linear programming to find an optimal solution when trying to maximize discounted revenues, which is merely a mathematical operation. Accordingly, if no mathematical constraints are included, most stands will be harvested at year 150. Of course, linear programming has no concept of the future beyond the 150-year planning period. Hence, the additional 100 years helps to provide a more realistic depiction of how stands will be managed near the end of the initial 50-year period. A projection period of 5 years was used.

For the Woodstock analysis conducted during the NMOP SFRMP, four scenarios were tested. These scenarios had various levels of desired future forest conditions (DFFCs) and hence forest management constraints (Table 1), resulting in different total annual harvested volume.

Table 1.—Summary of desired future forest conditions (DFFCs – parameter) for the four modeled scenarios

	Scenario			
DFFC – parameter	A	B	C	D
Even flow[a]	Tight 5%	Moderate 20%	Relaxed 40%	Relaxed 40%
LCOG[b] designation	High 10%	Moderate 5%	High 10%	Low 1.5%
Cover type change[c]	Climate change response[d]	Original SFRMP[e]	Original SFRMP[e]	No change
Older forest[f] (if needed for certain cover types or groupings of types)	More	Some	More	None

[a] Refers to the amount of variability from one projection period to another in terms of average annual harvest levels.
[b] Lowland conifer old growth (such as black spruce lowland and tamarack cover types). These stands represent the latter stages of succession in forested ecosystems of a particular cover type.
[c] The amount of transition from the current cover type to a more desired cover type.
[d] Refers to the use of cover type conversion goals taking into consideration climate change.
[e] Original Section Forest Resource Management Planning (SFRMP) does not consider climate change when specifying cover type conversion goals.
[f] Refers to the amount of acreage of age-classes by even-aged managed cover types exceeding the Minnesota Department of Natural Resources "normal" rotation age (essentially biological rotation age). Older forest is not necessarily consistent with old growth.

As a means of verifying the projections from the Woodstock analysis, a fully regulated approach was also used to project the current forested DNR landscape into the future. For this analysis, a fully regulated forest assumes that every age-class to some optimum (usually biological) rotation age contains an equal number of acres, site qualities are at a constant proportion in each age-class, reforestation occurs immediately after harvest and harvested cord productivity levels remain constant indefinitely, and a constant amount of acreage is harvested annually (Clutter et al. 1983, Davis 1966). Thus, every year, we should obtain a constant cord harvest level. For the aspen (*Populus spp.*) and balm-of-Gilead (*Populus ×jackii*) (ABg) cover type, for example, biological rotation age commonly ranges from around age 40 to 50 (which is a function of when annual increment [MAI] is greatest). Hence, to produce the greatest amount of volume into infinity, we would want to harvest every acre at age 45.

The Woodstock scenario analyses are estimated using an objective function of maximizing discounted revenues. Thus, verification of these estimates using the fully regulated harvesting approach can help to determine whether an objective function of maximizing discounted revenues still results in reasonable, and attainable, average annual cord harvest levels.

To conduct the fully regulated approach, initially the total amount of timberland acres of a cover type needs to be determined. This acreage is then divided by an average biological rotation age (essentially the same as the DNR "normal rotation age" [NRA]), and the

estimated annual sustainable harvest acreage is multiplied by a site-index weighted cord per acre estimate from established yield tables at or near the biological rotation age. This approach is valid for estimating long-term annual sustainable harvest levels when assuming the landbase is managed under a fully regulated age-class distribution. For the DNR, an NRA is defined as the rotation age set by the SFRMP team for even-aged managed cover types. It is based on the culmination of mean annual increment (CMAI), other available data related to forest productivity that also consider wood quality, and local knowledge.

The objective of this assessment was to compare the fully regulated annual harvest estimate with those from the four Woodstock scenario estimates. If the fully regulated annual harvest estimate is close to Woodstock scenario estimates, then it can be inferred that the Woodstock scenario estimates provide a reasonable representation of the amount of annual harvest over the next 50 years.

METHODS

Shapefile for the Northern Minnesota and Ontario Peatlands Section

To conduct a landscape-level harvest scheduling analysis, the landbase must be quantified as to cover-type acreage by age and site productivity and potential management restrictions/actions that can occur on those acres. The most recent (May 15, 2014) DNR Forest Inventory Module (FIM) database of shapefiles (ArcGIS, Esri, Inc., Redlands, CA) was queried. Cover type is determined based on internal DNR algorithms, site index is calculated based on measurement of dominant trees within the field and appropriate equations, and age is based on field measurements or records. For more information on the Forest Inventory Module (FIM), visit https://gisdata.mn.gov/dataset/biota-dnr-forest-stand-inventory. The landbase for the commonly managed timber types in the NMOP section includes only harvestable stands as defined by the DNR, in other words, "timberlands" (hence excluding old-growth and other nonharvestable stands) (Table 2).

Estimated Harvest Yields

Site-quality weighted average cords per acre by cover type were adjusted to account for DNR stand alterations/deferrals and for residual/leave tree requirements of 5 percent; alterations and deferrals often result in timber sales not occurring within that year for all the stands prescribed to be harvested and found on the exam list (Table 3). The number of acres available for final harvest after considering biological constraints (see next section), annual acres harvested, and total annual yield by cover types were estimated (Table 4).

Reductions in Harvestable Timberland Acres Due to Biological Considerations

Within SFRMP planning there are site quality restrictions placed on certain cover types to help better represent the reality of marketing stands that will result in timber sales. Table 5 shows the restrictions based on site quality implemented for certain cover types which reduce the number of harvestable timberland acres reported in Table 4 as compared to Table 2. For example, for the ash cover type (MN_CTYPE = 1), due to the restrictions in Table 5, the amount of acreage has been reduced from 50,484 acres (Table 2) to 23,095 acres (Table 4).

Table 2.—Manageable (i.e., excluding old-growth and other nonharvestable area) cover type acreages within the Northern Minnesota and Ontario Peatlands Section Forest Resource Management Planning dataset

Cover type (MN_CTYPE; Minnesota Department of Natural Resources)	Cover type name	Number of stands	Area (acres)
1	Ash	2,845	50,484
9	Lowland hardwoods	252	5,968
12	Aspen	16,568	338,332
13	Birch	368	5,868
14	Balm-of-Gilead	1,500	23,242
20	Northern hardwoods	131	1,927
30	Oak	26	396
301	Oak – high slope	--	--
51	White pine	99	997
52	Red pine natural	265	3,042
521	Red pine plantation	1,068	15,864
53	Jack pine	2,984	44,399
61	White spruce	251	3,107
611	White spruce plantation	639	10,856
62	Balsam fir	1,385	23,070
71	Black spruce lowland – high site quality	2,435	53,170
710	Black spruce lowland – low site quality	2,686	83,638
711	Black spruce lowland – medium site quality	5,521	165,430
72	Tamarack – high site quality	2,960	98,594
721	Tamarack – low site quality	2,892	139,420
74	Upland black spruce	126	1,768
Total		45,001	1,069,572

Table 3.—Average cords per acre at "normal/biological" rotation age by cover type

Cover type (MN_CTYPE; Minnesota Department of Natural Resources)	Number of stands[a]	Total area at NRA (acres)	Total yield at normal/biological rotation age (cords)	Average cords per acre[b]	Ratio to account for alterations and deferrals	Mod1 (cords per acre)[c]	Ratio to account for 5% leave	Final estimate (cords per acre)[d]
1	15	505	7,219	14.31	0.80	11.45	0.95	10.87
9	12	60	917	15.36	1.00	15.36	0.95	14.59
12	13	7,518	127,415	16.95	0.78	13.22	0.95	12.56
13	15	117	1,489	12.69	1.00	12.69	0.95	12.05
14	14	516	6,580	12.74	1.00	12.74	0.95	12.10
20	11	19	410	21.29	1.00	21.29	0.95	20.23
30	8	4	62	15.76	1.00	15.76	0.95	14.97
51	8	10	248	24.90	1.00	24.90	0.95	23.66
52	12	30	1,024	33.66	0.99	33.32	0.95	31.65
521	14	227	9,096	40.14	0.99	39.74	0.95	37.75
53	5	9	235	26.58	1.00	26.58	0.95	25.25
61	13	44	762	17.18	1.00	17.18	0.95	16.32
611	22	361	7,392	20.45	1.00	20.45	0.95	19.43
62	13	461	4,620	10.01	0.64	6.41	0.95	6.09
71	9	664	13,121	19.75	0.84	16.59	0.95	15.76
710	2	697	5,768	8.28	0.84	6.95	0.95	6.60
711	2	1,654	22,888	13.84	0.84	11.62	0.95	11.04
72	11	1,408	19,069	13.54	0.70	9.48	0.95	9.00
721	4	1,549	11,374	7.34	0.70	5.14	0.95	4.88
74	10	35	467	13.22	1.00	13.22	0.95	12.55

[a] Number of stands near NRA that were used to produce the estimate of average cords.

[b] Weighted average across all site qualities at NRA.

[c] Mod1 is a percentage of the average yields per acre, which better accounts for alterations and deferrals.

[d] To account for residual/leave requirements, 95 percent is taken of the Mod1 cords per acre value (0.95 × Mod1) to produce the final averagecords per acre.

Table 4.—Harvestable cover type acreages within the Minnesota Department of Natural Resources Forest Inventory Module dataset[a], "normal" rotation age, acreage harvested annually, average cords per acre harvested (see Table 3), and annual total yields

Cover type (MN_CTYPE; Minnesota Department of Natural Resources)	Cover type name	Area (acres)	Normal rotation age (years)	Annual harvested area (acres)	Cords per acre	Total yield (cords)
1	Ash	23,095	100	231	10.87	2,511
9	Lowland hardwoods	4,063	100	41	14.59	593
12	Aspen	338,332	45	7,518	12.56	94,424
13	Birch	5,868	50	117	12.05	1,415
14	Balm-of-Gilead	23,242	45	516	12.10	6,251
20	Northern hardwoods	1,042	100	10	20.23	211
30	Oak	396	100	4	14.97	59
51	White pine (WP)	997	100	10	23.66	236
52	Red pine natural (RPN)	3,042	100	30	31.65	963
521	Red pine plantation (RPP)	15,864	70	227	37.75	8,555
53	Jack pine	44,399	50	888	25.25	22,422
61	White spruce	3,107	70	44	16.32	724
611	White spruce plantation (WSP)	10,856	60	181	19.43	3,515
62	Balsam fir	23,070	50	461	6.09	2,809
71	Black spruce lowland – high site quality	53,170	80	665	15.76	10,473
710	Black spruce lowland – low site quality	83,638	120	697	6.60	4,603
711	Black spruce lowland – medium site quality	165,430	100	1,654	11.04	18,264
72	Tamarack – high site quality	98,594	70	1,408	9.00	12,681
721	Tamarack – low site quality	139,420	90	1,549	4.88	7,564
74[b]	Upland black spruce	1,768	50	35	25.25	893
Total		1,039,393		16,289		199,165
Thinning yields (ages of thinnings in parentheses)						
52	RPN (25, 35, 45, 55, 65, 75)	2,935		176	10	1,761
521	RPP (25, 35, 45, 55)	15,726		899	10	8,986
51	WP (25, 35, 45, 55, 65, 75)	926		56	10	556
611	WSP (25, 40)	10,856		362	10	3,619
Total						14,922
Total						214,087

[a] Excludes stands classified as old growth or stands with other harvesting restrictions, and excludes other timberland acres based on constraints thought to result in greater likelihood of successful timber sales (see Table 5).

[b] Upland black spruce is often managed along with the jack pine cover type (MN_CTYPE = 53).

Table 5.—Constraints placed on particular cover types thought to increase the likelihood of successful timber sales: only stands meeting or exceeding a minimum site index (feet, base age 50) were considered for harvesting

Cover type (MN_CTYPE; Minnesota Department of Natural Resources)	Cover type name	Site index (feet, base age 50)
1	Ash	≥45
9	Lowland hardwoods	≥45
20	Northern hardwoods	≥55

Thinning Harvest Amounts

Simple assumptions were made to estimate the amount of thinned volume (Table 4). For red pine natural and white pine on site indices of 45 feet and greater (base age 50), six thinnings were assumed to occur prior to the final harvest age of 100 years (ages 25, 35, 45, 55, 65, and 75). For red pine plantations on site indices of 45 feet and greater (base age 50), four thinnings were assumed to occur prior to the final harvest age of 70 years (25, 35, 45, and 55). For white spruce plantations, two thinnings were assumed to occur prior to the final harvest age of 60 years (25 and 40)—there are no site quality restrictions. For these cover types, each thinning generated 10 cords per acre, a reasonable assumption on average (VanderSchaaf and Deckard 2012, VanderSchaaf et al. 2017). Given the site-index restriction of 45 feet and greater (base age 50) on the ability to conduct a commercial thinning, the number of acres reported in Table 2 for red pine plantations is reduced from 15,864 acres to 15,726 acres, for natural stands from 3,042 acres to 2,935 acres, and for white pine stands from 997 acres to 926 acres.

It must be remembered that a first thinning in a 100-year-old white pine rotation occurs only once, a second thinning occurs only once, a third thinning occurs only once, and so on. Thus, for a particular acre the number of thinned cords over a rotation should be 60 cords (six thinnings × 10 cords per acre); 60 cords should then be divided by 100 years. It can be somewhat confusing thinking that mathematically a first thinning should be conducted every 25 years on an acre; however, this is mathematically incorrect for the fully regulated approach because the first thinning is not mutually exclusive from a final harvest at age 100. To clarify, for red pine plantations for example, 225 acres annually should receive a first thinning (15,726 acres divided by 70 years) because in a fully regulated age-class forest, annually 225 acres would be 25 years old.

RESULTS AND DISCUSSION

According to the fully regulated timber harvest analysis, the yield estimates from Woodstock are reasonable. In fact, for total harvested volume, Woodstock produces greater values than the fully regulated assessment. When using Woodstock, the average annual cords harvested over the next 50 years for scenarios A, B, C, and D (Table 1) are 239,048 cords, 257,720 cords, 254,534 cords, and 266,332 cords, respectively (Fig. 2). Based on the restrictive level of the desired future forest conditions (e.g., Parameter), the amount of total cords harvested annually differs between the scenarios. The more restrictive scenario A results in the fewest average annual cords being harvested, but the most consistent harvest levels across time (e.g., Tight Even Flow in Table 1),

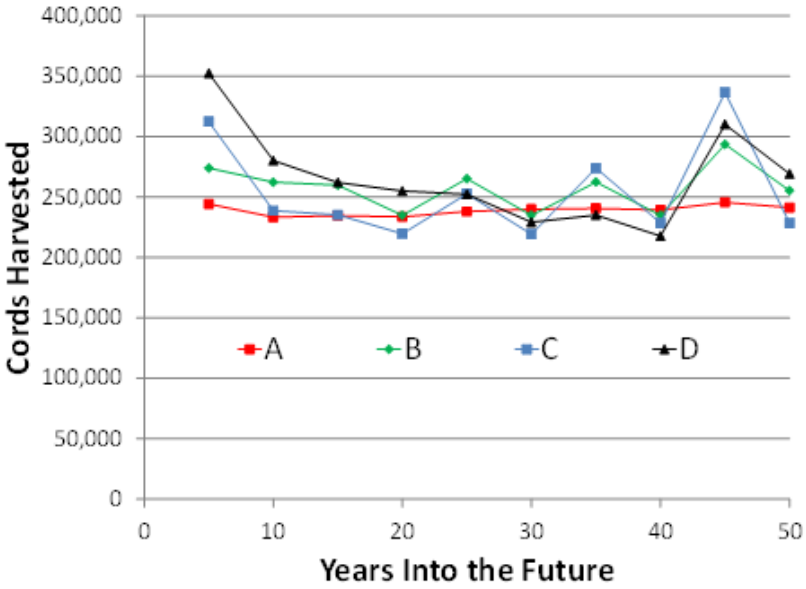

Figure 2.—Estimated harvested timber volume (cords) by scenario and 5-year projection period across all cover types and species, Northern Minnesota and Ontario Peatlands. See Table 1 for details about the four modeled scenarios (A through D).

while the more flexible scenario D results in the greatest number of cords harvested annually on average but a substantial amount of variability annually across the 50-year projection. The annual harvest level estimate of 214,087 cords when using the fully regulated approach shown in Table 4 is justifiably somewhat low, because in reality the DNR often harvests stands greater in age than NRA and for most cover types standing yields continue to increase past NRA. Thus, the total harvest estimate provided in Table 4 could potentially be increased arbitrarily based on experience with timber supply assessments by up to 20 percent to account for this factor.

Woodstock estimates are much more sensitive to actual age-class distributions and allow for more sophisticated constraints to be included. Harvest estimates are much more sensitive to the actual site quality of the stands selected for treatment in a particular year (projection period in practice) and the differences per acre in yield due to site quality. Additionally, the Woodstock linear programming approach can alter and adjust the annual cover type harvest levels across time, and hence has greater flexibility to, in a sense, take advantage of the productivity of certain cover types at particular times (e.g., Fig. 3).

The ABg cover type is an extremely important cover type in the NMOP ECS section. This cover type has the greatest acreage of any cover type in this ECS section (Table 2: MN_CTYPE = 12 and MN_CTYPE = 14). Although black spruce lowland and tamarack cover types also have substantial acreages, the yields per acre are generally less relative to ABg and rotation lengths are longer (Table 4). The red pine cover type acres, for plantations in particular, are often the most productive and valuable cover type on a per acre basis. However, they have substantially less acreage (Table 2). When using Woodstock, for the ABg cover type, the average annual harvest over the next 50 years for scenarios A, B, C, and D (Table 1) are 124,372 cords, and 122,410 cords, 123,480 cords, and 137,423 cords, respectively (Fig. 3). The fully regulated approach is much more constrained, and results in a lower average annual harvested cord estimate of 100,675 cords for the ABg cover type.

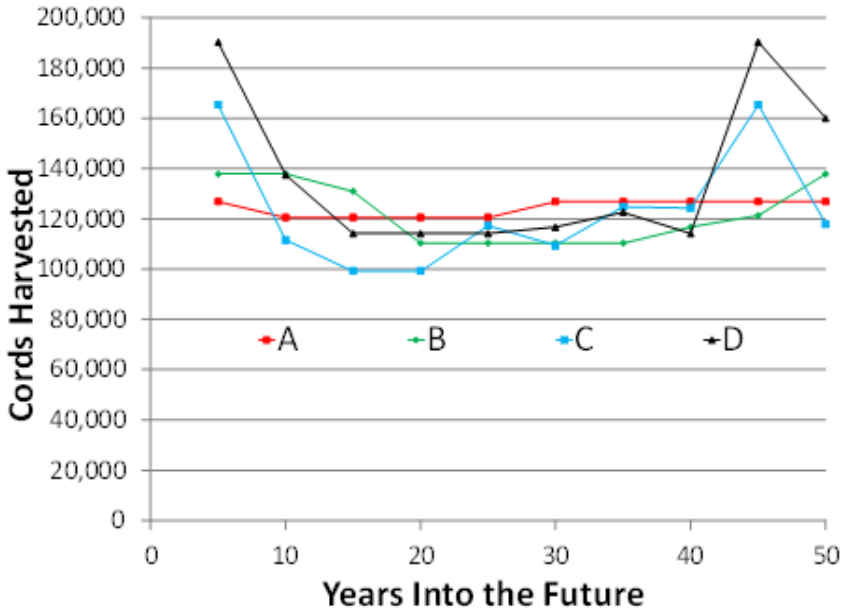

Figure 3.—Estimated harvested volume (cords) of aspen (MN_CTYPE = 12 in Table 2) and balm-of-Gilead (MN_CTYPE = 14 in Table 2) cover type (ABg) by scenario and 5-year projection period. See Table 1 for details about the four modeled scenarios (A through D).

CONCLUSIONS

Thus, it is reasonable to assume that the Woodstock harvest level estimates across time are superior estimates. Additionally, the Woodstock estimates are superior because the linear programming approach can better account for the impacts of forest management constraints (e.g., Table 1) on projections. It is more difficult to include desired future forest condition (DFFC) constraints when using the fully regulated approach. Nonetheless, this fully regulated analysis was conducted to help verify the estimates produced by the linear programming software package Woodstock during the NMOP SFRMP. In conclusion, the objective function of maximizing discounted revenues within Woodstock linear programming software did not result in illogical estimates of annual harvested cords.

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The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

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Site Index Equations for Bottomland Hardwood Forests in the Western Half of the Southeastern United States

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Abstract

Dominant height and site index (base age 50) equations were developed for the swamp chestnut oak-cherrybark oak (*Quercus michauxii-Q. pagoda*) forest type as defined by the USDA Forest Service Forest Inventory and Analysis (FIA) program. Separate equations were developed for either swamp chestnut oak or cherrybark oak and dominant height was defined as the average height of the tallest 50 percent of living trees for a particular species. FIA plots of all ownerships within the entire states of Arkansas, Louisiana, Mississippi, and Tennessee, and limited counties within western Alabama, were considered for inclusion. Only plots age 20 and greater where basal area per acre was at least 30 percent of that species were included. Trees with a broken top were excluded. For cherrybark oak and swamp chestnut oak, plots used in model fitting had to have a minimum of either 4 or 2 trees per plot, respectively, meeting these criteria. The Schumacher anamorphic model form was selected for both species. For cherrybark oak, 44 plots were included and the R2 was 0.4107; for swamp chestnut oak, 13 plots were included and the R2 was 0.5565. For the same numerical site index (base age 50), cherrybark oak exhibits greater early height growth, but around age 50 swamp chestnut oak starts to exhibit greater height growth. Cherrybark oak predictions were compared to four other equations that could be considered applicable to this geographic region. Predictions very nearly mirrored those from Schultz et al. (2010).

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INTRODUCTION

Bottomland hardwood forests store floodwater and reduce associated risks and severity of flooding, thereby protecting downstream communities (Mahaffey and Evans 2016). These forests also improve water quality by filtering and flushing nutrients, processing organic wastes, and reducing sediment load in streams. Bottomland forests are of relatively high wood productivity and financial value (Banzhaf et al. 2016, Self 2020); they are important stores of carbon, and many forest products are produced from trees in these forests

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(Banzhaf et al. 2016, Oswald et al. 2017). Bottomland hardwood forests provide habitat for a diversity of animals ranging from beetles (Order Coleoptera) to black bears (*Ursus americanus*) (Mahaffey and Evans 2016, Rudis and Tansey 1995, Self 2020, Ulyshen et al. 2004). Many bird species depend on southeastern U.S. bottomland hardwood forests (Pashley and Barrow 1993). Bottomland forests are also premium hunting grounds and command high values for leases (Hussain et al. 2007).

There are a variety of nomenclatures and forest type designations of bottomland hardwood forests (e.g., Allen et al. 2004, Clatterbuck et al. 1985). This study presents site index equations for the swamp chestnut oak-cherrybark oak (*Quercus michauxii-Q. pagoda*) forest type as defined by the USDA Forest Service Forest Inventory and Analysis (FIA) program (Burrill et al. 2021). FIA defines this forest type to be found within alluvial flood plains of major rivers, on all ridges in the terraces, and on the best fine sandy loam soils on the highest first bottom ridges. Common associates include Shumard oak (*Q. shumardii*), white ash (*Fraxinus americana*), various hickories (*Carya* spp.), white oak (*Q. alba*), black tupelo (*Nyssa sylvatica*), sweetgum (*Liquidambar styraciflua*), southern red oak (*Q. falcata*), elms (*Ulmus* spp.), yellow poplar (*Liriodendron tulipifera*), and American beech (*Fagus grandiflora*). This forest type is also similarly defined by the Society of American Foresters as stated in Allen et al. (2004). Sweetgum is often a major component in these forests, particularly in terms of tree density, but these forests are most often dominated by bottomland oaks, such as swamp chestnut oak and cherrybark oak (Lockhart et al. 2010). Mahaffey and Evans (2016) define bottomland hardwoods as forested wetlands found in broad, river and stream floodplains. These forests occur primarily in alluvial (waterborne soil) floodplains and are also described as floodplain forests. Mahaffey and Evans (2016) further state that cherrybark oak and swamp chestnut oak are desirable and suitable species on ridges and high flats of both major and minor river bottoms. Allen et al. (2004) have a similar definition of bottomland hardwoods.

REVIEW OF EXISTING SWAMP CHESTNUT OAK AND CHERRYBARK OAK SITE INDEX CURVES AND EQUATIONS

Broadfoot (1961) developed guides to estimate site productivity of forests based on cherrybark oak site index for stands found throughout southwestern Alabama, western Tennessee, Mississippi (except coastal regions), central and eastern Louisiana, and the Delta and southeastern regions of Arkansas. Broadfoot (1961) also graphically presented "smoothed" anamorphic site index curves developed from ring counts along the stem (stem analysis). Site index was defined to be the average height of dominant and codominant trees in well-stocked stands that have not undergone any modifying influence or treatment. The base age was 50 years. Stands inventoried were located on soils defined to be Mississippi Valley alluvium by Carmean et al. (1989). Carmean et al. (1989) used the graphically presented site index curves from Broadfoot (1961) to estimate parameters of a modified Chapman-Richards site index equation. They did not state whether the portions of Alabama that Broadfoot (1961) included were also included when they fit their own regression equation using the curves of Broadfoot (1961). The site index curves presented by Broadfoot (1961) and the equation presented by Carmean et al. (1989) are commonly used by managers. The site index equation developed by Carmean et al. (1989) is used for cherrybark oak-dominated forests as part of the FIA inventory (Vissage et al. 2019) and the Forest Service Forest Vegetation Simulator (FVS) software system (FVS Staff 2021).

Within FVS, cherrybark oak and swamp chestnut oak site indices (base age 50) can range from lows of 30 and 35 to highs of 125 and 95, respectively. FVS is one of the few growth and yield model systems available for bottomland hardwoods in the western half of the southeastern United States.

Clatterbuck (1987) presented a site index curve for cherrybark oak from natural, even-aged stands on the minor bottoms of central Mississippi. However, he found that predicted height development for cherrybark oak from his curves differed meaningfully from that of Broadfoot (1961), which was the standard at that time for all bottomland hardwood sites. He suggested that the difference could potentially be due to growth differences among trees within major river bottoms and loessial hills, as these site types made up the majority of plots used for site index estimation within Broadfoot (1961). In contrast, most of the data that Clatterbuck (1987) used were from trees within minor river bottoms.

Iles (2008) and Schultz et al. (2010) presented an anamorphic Chapman-Richards site index equation. Plots were located within minor bottoms of essentially eastern Mississippi and a small part of western Alabama that were partially composed of cherrybark oak and swamp chestnut oak. Site index was defined to be only dominant and codominant red oaks. Hence, since red oak was the predominant overstory species group, they considered dominant and codominant red oak trees to be the driver of growth and yield model systems. A base age of 50 years was selected.

McTague et al. (2008) presented polymorphic site index equations for hardwood stands across the South. Dominant height was defined as the 40 tallest trees per acre irrespective of species. A base age of 25 years was selected. It is difficult to tell exactly what proportion of the data is from bottomland hardwood sites within the states of Alabama, Arkansas, Louisiana, Mississippi, and Tennessee and how much cherrybark oak or swamp chestnut oak accounts for the trees considered to be dominant. Lockhart (2013) summarized and described various techniques to determine site index of bottomland hardwoods, including cherrybark oak and swamp chestnut oak. Oswald et al. (2017) developed separate anamorphic site index equations for willow oak (*Q. phellos*) and cherrybark oak in East Texas bottomland forests. These forests were defined as located in minor bottoms and having periodic standing water or existing on lower upland slopes along smaller streams and creeks (or both conditions). Site index was defined to be dominants and codominants, and they presented site index curves using a base age of 50 years. However, they actually modeled individual trees classified as dominants and codominants and not plot-level averages of trees classified as dominants and codominants.

Based exclusively on the criterion of the geographic location of the model fitting data, within the region of Arkansas, Louisiana, Mississippi, Tennessee, and western Alabama, there are only four site index curves or equations available where site index is determined based on existing site trees for stands dominated by cherrybark oak (Broadfoot 1961, Carmean et al. 1989, Clatterbuck 1987, Schultz et al. 2010). No site index curves are available for swamp chestnut oak. Within the FVS, the site index curve (Broadfoot 1963) of water oak (*Q. nigra*) is used to represent dominant height growth of swamp chestnut oak (FVS Staff 2021). The cherrybark oak equation presented in Oswald et al. (2017) for East Texas could potentially be used in this region as well. In a sense, it could be argued that McTague et al. (2008) is applicable to some degree specifically for cherrybark oak or swamp chestnut oak in this region.

Objective

An objective of this study was to review the current status of site index equations for swamp chestnut oak and cherrybark oak. Since there are no equations for swamp chestnut oak, an equation was developed for bottomland hardwood stands classified by FIA as the swamp chestnut oak-cherrybark oak forest type. To determine whether a site index equation for cherrybark oak developed using data from a broader geographic range is needed, an equation developed using the same geographic range as Broadfoot (1961) was produced. Only plots located within Arkansas, Louisiana, Mississippi, Tennessee, and those western counties of Alabama consistent with Broadfoot (1961) were selected. Outside of the datasets already used in modeling as reported here, few to no long-term and repeatedly measured plots exist for this forest type and these two species that would allow for an independent validation analysis of observed data. Therefore, predictions from the site index equation developed here for cherrybark oak are used as a means to verify predictions through time of the other available equations, and predictions from the equation for swamp chestnut oak are used as a means to verify the use of a water oak equation to represent swamp chestnut oak as FVS is currently doing.

METHODS

Model Fitting Data—Plots

FIA (Burrill et al. 2021) plots designated as the swamp chestnut oak-cherrybark oak forest type (FORTYPCD = 601 in the FIA Condition table) within Arkansas, Louisiana, Mississippi, and Tennessee, plus those western counties of Alabama consistent with Broadfoot (1961), were selected (Fig. 1). A limited number of counties in west-central and southwest Alabama were selected consistent with Broadfoot (1961), and similar in range as used by Schultz et al. (2010). Across all of the five states, this geographic range is essentially consistent with that of Broadfoot (1961). Only naturally regenerated stands were selected (STDORGCD = 0 in the FIA Condition table) and all plots were selected regardless of whether they were defined as timberland by FIA, hence FORESTLAND (SITECLCD from 1 to 7 in the FIA Condition table). According to FIA (USDA Forest Service 2022), there are 76,934 acres of this FIA-defined forest type in the applicable counties of Alabama, consistent with those included by Broadfoot (1961). For the entire area of the four other states, there are 256,708 acres in Arkansas, 211,926 acres in Louisiana, 323,206 acres in Mississippi, and 82,292 acres in Tennessee of this FIA defined forest type.

The data used in model development were obtained from FIA annual surveys. The FIA panels used varied by state. Panels from 2013 to 2020, 2014 to 2019, 2009 to 2018 (excluding 2010), 2012 to 2019, and 2012 to 2017 were used for Alabama, Arkansas, Louisiana, Mississippi, and Tennessee, respectively. Data were obtained from the FIA database website (USDA Forest Service 2021). All ownerships were included (OWNCD in the FIA Condition table was not used to filter the data).

Model Fitting Data—Individual Trees

Only living trees were included (STATUSCD = 1 in the FIA Tree table), and all tree classes were included (TREECLCD within the FIA Tree table was not used to filter the data). Only trees whose heights were actually measured were included (HTCD = 1 in the FIA Tree table). Additionally, to be as consistent as possible with the often desired characteristic of no crown deformities (Clatterbuck 1987, Oswald et al. 2017), trees with broken tops were

removed. Hence, only trees whose HT (FIA Tree table) equaled its ACTUALHT (FIA Tree table) were included.

Plot Selection and Elimination

Only plots of age 20 years and greater were included, consistent with Schultz et al. (2010) and Oswald et al. (2017). This minimum plot age is essentially consistent with Clatterbuck (1987) as well, although he did not specify a particular minimum age. The minimum age of 20 years takes into account the slower growth rate of cherrybark oak relative to other species up to and at this age, but by age 20 the height growth of cherrybark oak will surpass many tree species and hence dominance of individual cherrybark oak trees can be better identified.

Additionally, only plots with a basal area per acre (diameter at breast height [d.b.h.] of 1 inch and greater) composed of at least 30 percent (Schultz et al. 2010) of cherrybark oak were selected for the fitting of the cherrybark oak equation. Similarly, for swamp chestnut oak, only plots with a basal area per acre (d.b.h. of 1 inch and greater) composed of at least 30 percent of swamp chestnut oak were selected for the fitting of the swamp chestnut oak equation. In addition, each selected plot by species had to have a minimum tree amount of that species. In general, there are a greater number of cherrybark oak trees per plot; therefore, only plots that contained four or more cherrybark oak trees were selected.

For swamp chestnut oak, only plots that contained two or more swamp chestnut oak trees were selected, as there are generally fewer trees per plot. Pairing the number of trees per plot and a minimum basal area ratio helps to ensure that plots are more reasonably stocked, and in some ways ensures that the plots used in model development are more consistent with previous modeling efforts where it is stated trees were selected only from within well-stocked stands (Clatterbuck 1987, Oswald et al. 2017, Schultz et al. 2010). Using this minimum number of trees per plot to quantify dominant height is consistent with Hall et al. (2020).

Prior to any model fitting, general trends in dominant height were examined to ensure that as much as possible observations exhibit biologically reasonable behavior. For cherrybark oak, all plots meeting the preceding criteria were used in model fitting. Thus, a total of n=44 plots were used in model fitting. However, for swamp chestnut oak, based on the protocol expressed earlier, one plot exhibited illogical behavior in terms of stocking and a realistic dominant height. This plot was reported to be 58 years old and had only two trees, and the one tree (the upper 50 percent) used to measure dominant height was 22 feet tall. Additionally, the observed basal area per acre was 23.6 square feet. Thus, this plot was removed. All other plots meeting the criteria expressed earlier exhibited logical behavior. Thus, a total of n=13 plots were used in model fitting of swamp chestnut oak. The two species did not have any plots in common based on these criteria for plot selection (Fig. 1; Tables 1 and 2).

Definition of Dominant Height

Dominant height of a particular plot for model development for each species was defined as the average height of the tallest 50 percent of all cherrybark oak or swamp chestnut oak surviving trees, which is consistent with some definitions (Boyer 1983, Golden et al. 1981) (Figs. 2 and 3). Others have used different percentages (Lynch et al. 2016, McTague 2008).

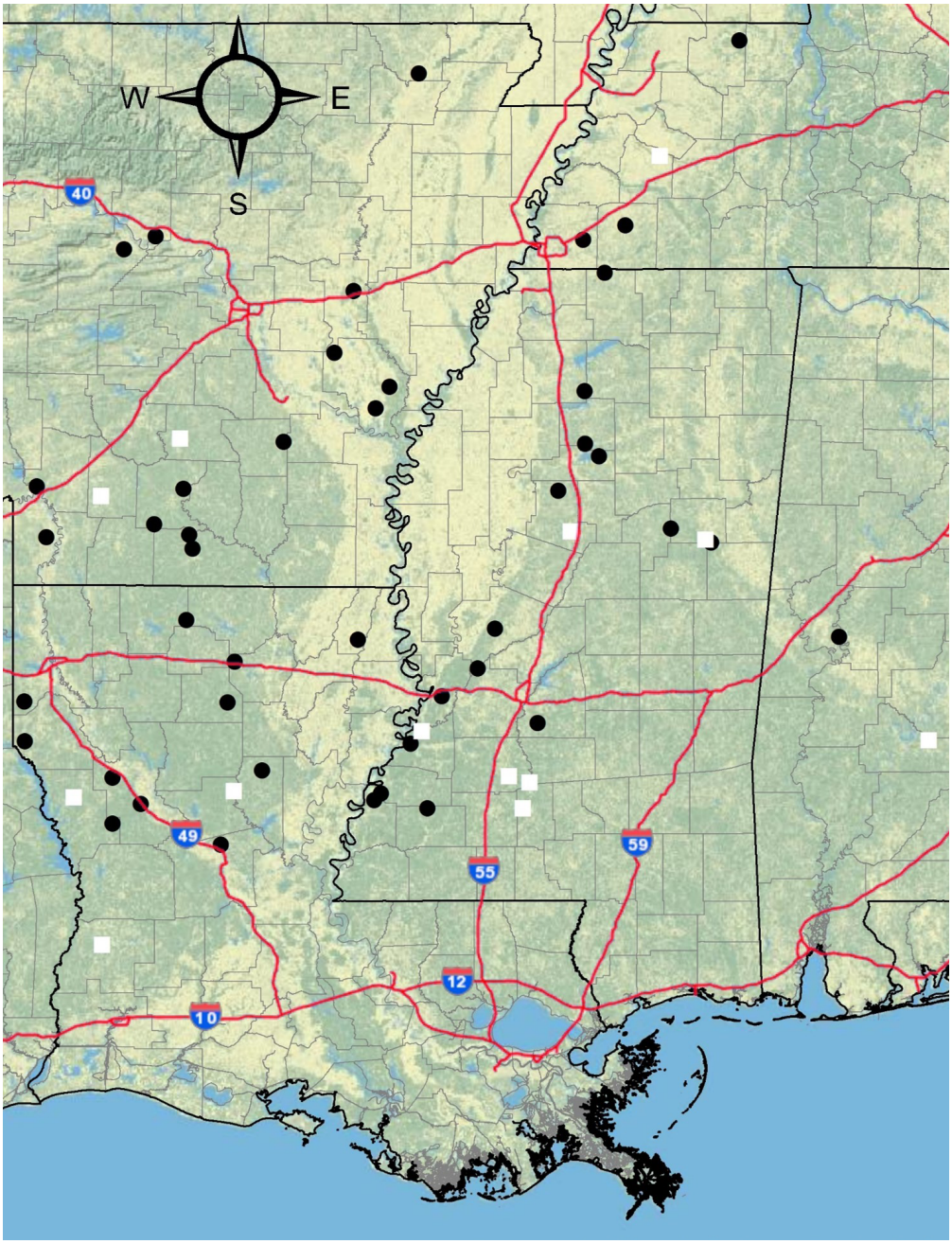

Figure 1.—Spatial location of the *n*=13 Forest Inventory and Analysis (FIA) plots (white squares) used to fit the swamp chestnut oak dominant height equation (equation [1] in text) and the spatial location of the *n*=44 FIA plots (black circles) used to fit the cherrybark oak dominant height equation. Plot locations are approximate.

Table 1.—Summary statistics by state and across the entire model-fitting dataset for the USDA Forest Service Forest Inventory and Analysis plots (*n*=13) used to fit the swamp chestnut oak dominant height equation (equation [2] in text)

State	*n*	Mean dominant height (feet)	Maximum dominant height (feet)	Minimum dominant height (feet)	Standard deviation	Mean stand age (years)	Maximum stand age (years)	Minimum stand age (years)	Standard deviation
Alabama	1	75	75	75	-	40	40	40	-
Arkansas	2	76	78	75	2	66	74	58	11.3
Louisiana	3	107	118	89	16	65	78	43	19.4
Mississippi	6	95	130	57	25	68	84	30	21.0
Tennessee	1	97	97	97	-	64	64	64	-
All	13	94	130	57	21	65	84	30	17.7

State	*n*	Mean basal area per acre (square feet)	Maximum basal area per acre (square feet)	Minimum basal area per acre (square feet)	Standard deviation	Mean stand age (years)	Maximum stand age (years)	Minimum stand age (years)	Standard deviation
Alabama	1	58	58	58	-	26	26	26	-
Arkansas	2	95	114	75	28	42	62	23	27
Louisiana	3	143	192	97	47	56	84	40	24
Mississippi	6	107	144	50	42	44	74	22	21
Tennessee	1	152	152	152	-	68	68	68	-
All	13	113	192	50	43	47	84	22	21

Table 2.—Summary statistics by state and across the entire model fitting dataset for the USDA Forest Inventory and Analysis plots (*n*=44) used to fit the cherrybark oak dominant height equation (equation [3] in text)

State	*n*	Mean dominant height (feet)	Maximum dominant height (feet)	Minimum dominant height (feet)	Standard deviation	Mean stand age (years)	Maximum stand age (years)	Minimum stand age (years)	Standard deviation
Alabama	1	122	122	122	-	84	84	84	-
Arkansas	14	89	122	70	16	55	75	30	15.4
Louisiana	11	87	118	67	16	46	78	20	19.3
Mississippi	15	99	142	64	26	56	84	22	18.3
Tennessee	3	84	98	74	12	45	59	33	13.1
All	44	92	142	64	20	53	84	20	17.8

State	*n*	Mean basal area per acre (square feet)	Maximum basal area per acre (square feet)	Minimum basal area per acre (square feet)	Standard deviation	Mean stand age (years)	Maximum stand age (years)	Minimum stand age (years)	Standard deviation
Alabama	1	167	167	167	-	68	68	68	-
Arkansas	14	132	207	87	32	64	159	37	31
Louisiana	11	108	160	71	29	56	87	34	16
Mississippi	15	129	258	58	51	60	101	24	24
Tennessee	3	213	297	157	74	120	204	71	74
All	44	131	297	58	48	65	204	24	32

Definition of Dominant Height

Dominant height of a particular plot for model development for each species was defined as the average height of the tallest 50 percent of all cherrybark oak or swamp chestnut oak surviving trees, which is consistent with some definitions (Boyer 1983, Golden et al. 1981) (Figs. 2 and 3). Others have used different percentages (Lynch et al. 2016, McTague 2008). Other available site index equations for cherrybark oak use a definition of dominants and codominants (Broadfoot 1961, Carmean et al. 1989, Clatterbuck 1987, Oswald et al. 2017).

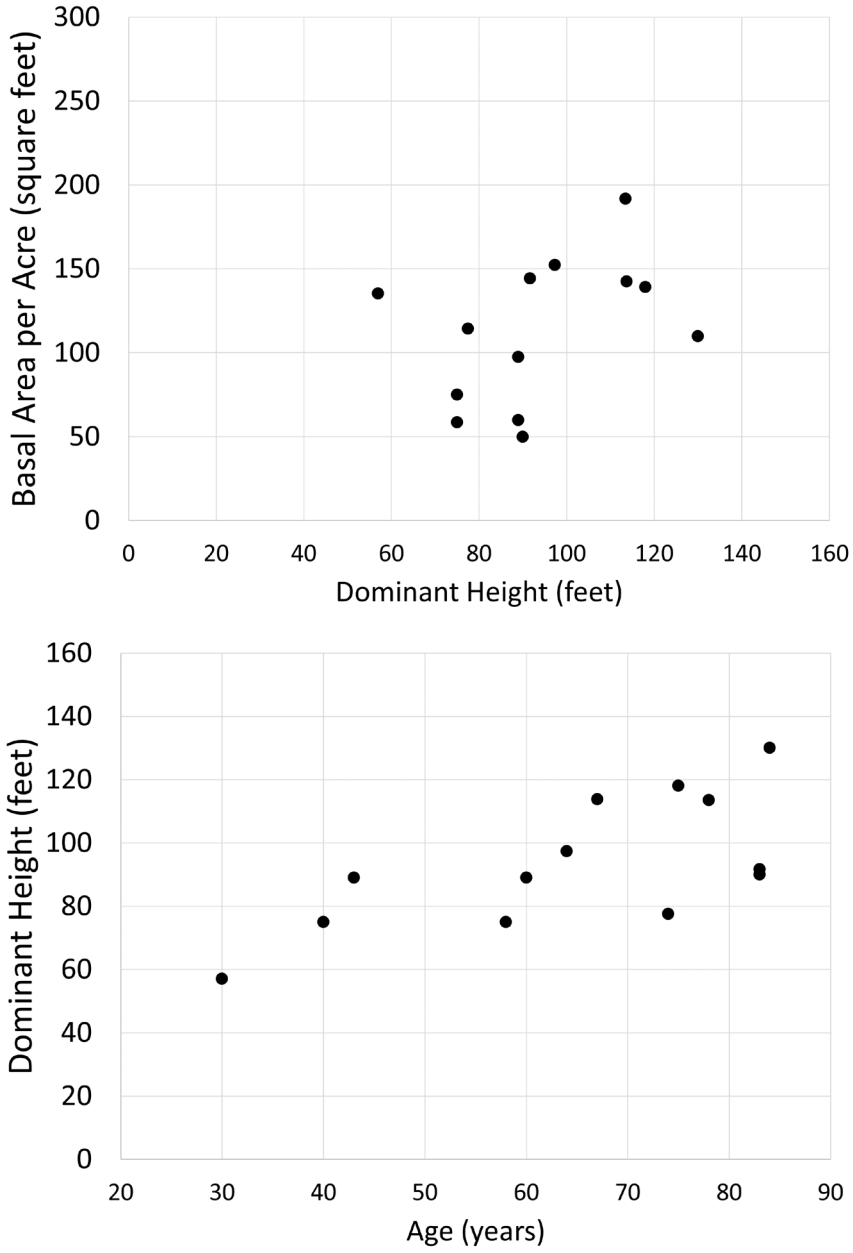

Figure 2.—Swamp chestnut oak basal area per acre plotted against dominant height (defined as the average height of the tallest 50 percent of living trees) (upper panel) and dominant height plotted against stand age (lower panel); *n*=13 plots.

There are advantages and disadvantages to both types of definitions (either a fixed percentage or dominants and codominants based on visually estimated crown class). Given the FIA database, there is no way to definitively specify dominant or codominant trees (or both) because this information is not collected for each individual tree when plots are measured in the field.

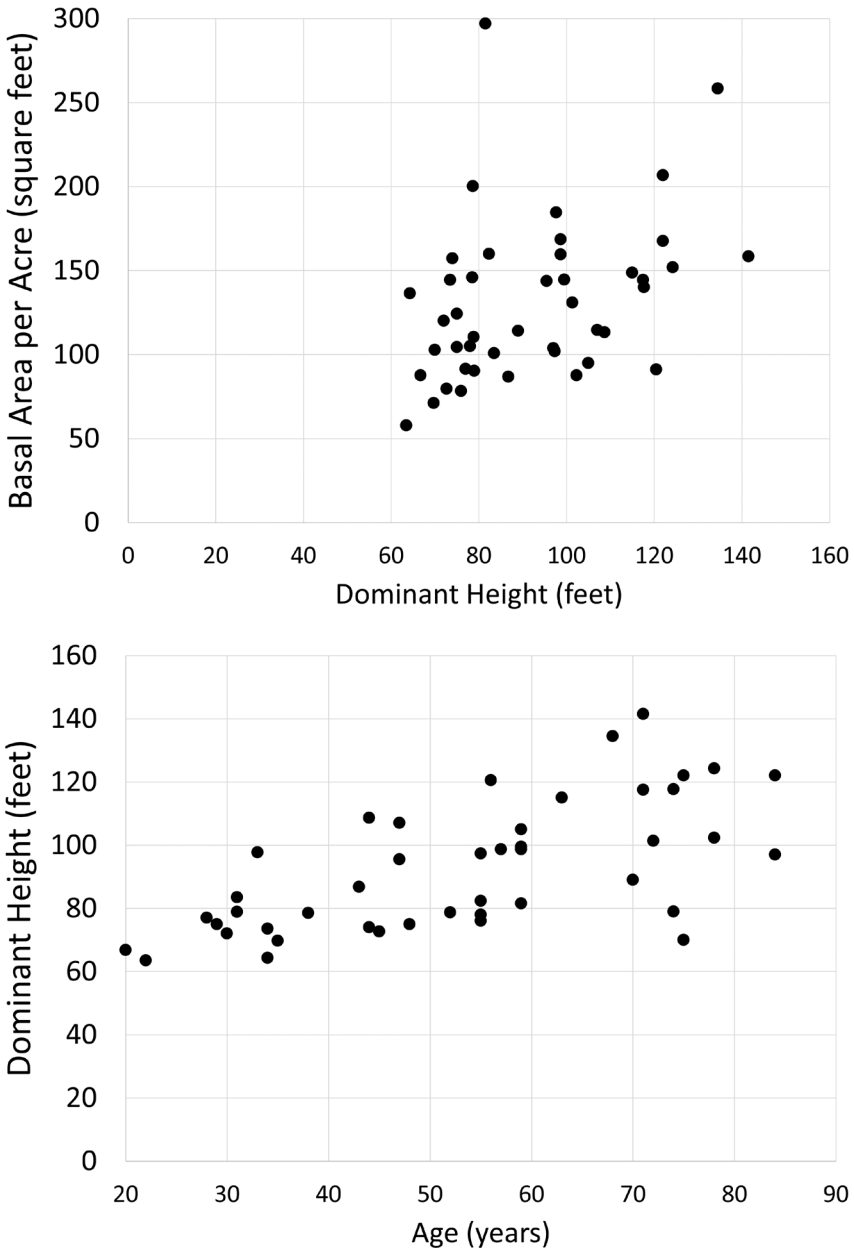

Figure 3.—Cherrybark oak basal area per acre plotted against dominant height (defined as the average height of the tallest 50 percent of living trees) (upper panel) and dominant height plotted against stand age (lower panel); *n*=44 plots.

Dominant Height Equation

A common equation form used to produce anamorphic site index curves is the Schumacher model (Clutter et al. 1983, Hall et al. 2020):

$$\ln Ht_D = b_0 + b_1 A^{-1} + \varepsilon \quad [1]$$

Where:

Ht_D is the dominant height (feet), defined as the average height of the tallest 50 percent of living trees for a particular species, within an FIA plot,

A is plot age (STDAGE in the FIA Condition table), and

b_0, b_1 are parameters to be estimated.

In addition, another commonly used equation form to produce anamorphic site index curves is the Chapman-Richards equation (Hall et al. 2020, Schultz et al. 2010). However, for both species, statistically meaningful results were not obtained when using the Chapman-Richards equation and hence only model fitting results when using equation [1] are presented.

RESULTS AND DISCUSSION

Swamp Chestnut Oak

After model fitting of equation [1], equation [2] was produced:

$$\ln Ht_D = 4.965748 - 26.3807 A^{-1} \quad [2]$$

The R^2 value was equal to 0.5565 (n=13).

Cherrybark Oak

After model fitting of equation [1], equation [3] was produced:

$$\ln Ht_D = 4.833462 - 15.295 A^{-1} \quad [3]$$

The R^2 value was equal to 0.4107 (n=44).

For both species, the parameter estimates had $p < 0.0001$ except for the slope estimate for swamp chestnut oak, for which $p < 0.0034$. On the logarithmic scale, the residuals showed no seriously adverse trends for swamp chestnut oak or cherrybark oak (Fig. 4).

Following transformation back to the original units, and algebraic manipulation as demonstrated within Clutter et al. (1983), an equation for each species was obtained to estimate dominant height. For swamp chestnut oak, the equation is:

$$Ht_D = SI e^{\left(50^{-1} - A^{-1}\right)^{26.3807}} \quad [4]$$

and for cherrybark oak, the equation is:

$$Ht_D = SI e^{\left(50^{-1} - A^{-1}\right)^{15.295}} \quad [5]$$

Site index equations (base age 50) can also be produced; for swamp chestnut oak, the

equation is:

$$SI = Ht_D \, e^{\left(50^{-1} - A^{-1}\right)^{-26.3807}} \quad [6]$$

and for cherrybark oak, the equation is:

$$SI = Ht_D \, e^{\left(50^{-1} - A^{-1}\right)^{-15.295}} \quad [7]$$

Where:
SI is the dominant height (average height of the tallest 50 percent of living trees for a particular species) at a reference base age of 50 years, and others as previously defined for equation [1].

For swamp chestnut oak, the R^2 value of 0.5565 is within the range observed for other model fittings of bottomland hardwood oak site index equations (Oswald et al. 2017, Schultz et al. 2010). For cherrybark oak site index equations, Schultz et al. (2010) reported an R^2 value of 0.61 and Oswald et al. (2017) reported an R^2 value of 0.63. Oswald et al. (2017) reported an R^2 value for willow oak of 0.52. For cherrybark oak the observed R2 value of 0.4107 is lower than that observed from other studies. Contributing factors could be that Oswald et al. (2017) modeled individual trees and both Schultz et al. (2010) and Oswald et al. (2017) had more limited geographic ranges.

Although the R^2 value for cherrybark oak is lower relative to other available site index equations, the equation presented has the advantage of being developed by using data from a wider geographic range (Fig. 1). Thus, technically, this equation has greater geographic range of inference. Data were obtained from the states of Arkansas, Louisiana, Mississippi, Tennessee, and a part of Alabama. This area is a wider range than that of Clatterbuck (1987), which was developed by using data only from central Mississippi, Schultz et al. (2010) which was developed by using data essentially from east-central Mississippi and a small part of west-central Alabama, and Oswald et al. (2017), which was developed by using data only from central East Texas. Broadfoot (1961) also used the geographic range of Arkansas, Louisiana, Mississippi, and Tennessee, and the similar range of counties within Alabama. Thus, it can be assumed that Carmean et al. (1989) did so as well because their equation was developed based on the site index curves presented by Broadfoot (1961). However, Carmean et al. (1989) did not state that Alabama was included in their fitting of the curves presented by Broadfoot (1961).

It is somewhat difficult to absolutely determine whether the FIA plots used in model fitting of equation [1] were obtained from major bottoms (majority of Broadfoot (1961) and hence Carmean et al. (1989)) or minor bottoms (Clatterbuck 1987, Schultz et al. 2010, majority of Oswald et al. 2017). As seen in Figure 1, it could be interpreted that the majority of plots are from minor bottoms, as they are not spatially close to major rivers. However, this interpretation is not entirely sound because all reported FIA plot latitude and longitude coordinates are "fuzzed," and some FIA plot coordinates are "swapped." "Fuzzed" means the reported spatial coordinates of most plots within the FIA database are randomly relocated within 0.5 mile of their actual coordinates, with the remainder relocated up to 1 mile. This means that the actual plot location is generally masked within a 500-acre area.

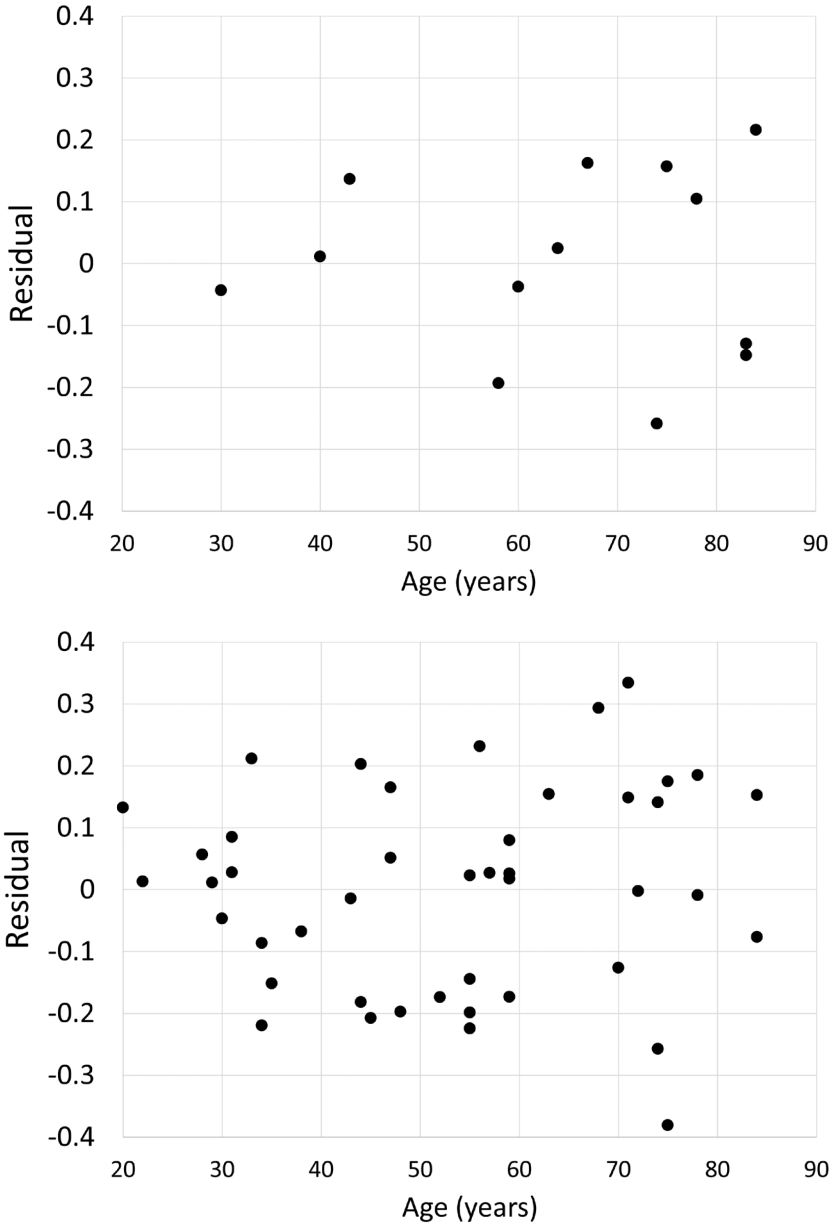

Figure 4.—Residuals from the fitting of equation [1] (in text) for swamp chestnut oak (upper panel) dominant height (defined as the average height of the tallest 50 percent of living trees) over stand age (*n*=13 plots) (a) and cherrybark oak (lower panel) dominant height (defined as the average height of the tallest 50 percent of living trees) over stand age (*n*=44 plots).

When plot coordinates are "swapped," the reported spatial coordinates of up to 10 percent of the FIA plots, but only on private land, within a county are randomly selected to be swapped with other FIA plots within that county, resulting in perhaps 25 percent of the plots having some adjustment in their reported spatial coordinates due to "swapping." Unfortunately, there is no way for the casual user of FIA inventory plots to determine specifically which plots were "swapped." For swamp chestnut oak and cherrybark oak, 92.3 percent and 88.6 percent of the observations were defined to be privately owned (OWNCD = 46 in the FIA Condition table), respectively. Hence, it is probable that some of the plot spatial coordinates were "swapped."

For swamp chestnut oak, when applying equation [6] to the observed data, the average site index (base age 50) is 86 feet, the minimum site index was 65 feet, and the maximum was 105 feet. This range is consistent with that reported in Lockhart (2013) in relation to Baker and Broadfoot (1979); the reported site index minimum was 65 feet and the maximum was 110 feet. The maximum of 105 feet slightly exceeds the maximum of 95 feet within the Forest Vegetation Simulator (FVS) program (FVS Staff 2021).

For cherrybark oak, when applying equation [7] to the observed data, the average site index (base age 50) is 94 feet, the minimum site index was 63 feet, and the maximum was 129 feet. This range is consistent with that reported in Lockhart (2013) in relation to Baker and Broadfoot (1979); the reported site index minimum was 70 feet and the maximum was 125 feet. This observed range in site index is also similar to that reported by Schultz et al. (2010), who reported a minimum of 67 feet and a maximum of 133 feet. The maximum of 129 feet is essentially the same as the maximum of 125 feet within the FVS program (FVS Staff 2021).

Predictions of dominant height development over time for site indices of 65, 85, 105, and 125 feet (base age 50) are presented in Figure 5. A site index of 125 feet likely exceeds the maximum observable site index for swamp chestnut oak (Baker and Broadfoot 1979, Lockhart 2013, FVS Staff 2021), but the site index is included for illustrative purposes of cherrybark oak. Cherrybark oak develops faster than swamp chestnut oak (Frey et al. 2018). For younger ages, cherrybark oak for sites at the preceding lower site index (e.g., cherrybark oak site index of 105 feet compared to swamp chestnut oak site index of 125 feet) has greater dominant height growth relative to swamp chestnut oak; this ceases to occur across all site qualities around age 30. However, for all site qualities, swamp chestnut oak starts to exceed cherrybark oak at around age 50. The temporal consistency across all site qualities of the timing of the crossover between the two species is most likely due to the anamorphic nature (and hence behavior) of the site index curves.

In a graph of dominant height estimates across time for cherrybark oak using equation [5] and equations presented in four earlier publications, curves from Carmean et al. (1989) exhibited slower early height growth relative to Schultz et al. (2010) and equation [5] for three site indices (base age 50) (Fig. 6). Carmean et al. (1989) estimates exhibited slightly greater growth than the other curves from around ages 35 to 50, and then had substantially less growth beyond age 50. Similar results were reported by Oswald et al. (2017) in a similar comparison to Broadfoot (1961). As noted by Clatterbuck (1987), Broadfoot (1961) overestimated early height growth but underestimated height growth after age 50.

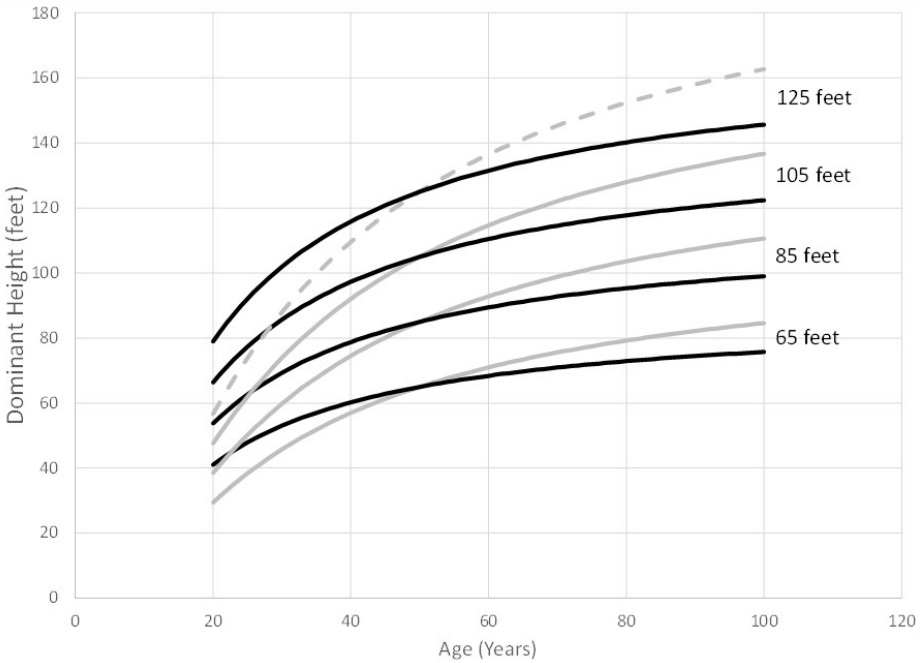

Figure 5.—Predicted dominant height (defined as the average height of the tallest 50 percent of living trees, in feet) by site index (base age 50) for cherrybark oak (equation [5]—black curves) and swamp chestnut oak (equation [4]—gray curves). It should be noted that the USDA Forest Service Forest Vegetation Simulator (FVS) growth and yield simulator has a maximum site index (base age 50) of 95 feet for swamp chestnut oak (FVS Staff 2021). Lockhart (2013) in relation to Baker and Broadfoot (1979) reported a site index maximum of 110 feet, whereas the maximum observed in this study after applying equation [6] was 105 feet. Please see text for equations.

Since the equation presented by Carmean et al. (1989) was developed using the curves found within Broadfoot (1961), it also exhibits the same behavior. Clatterbuck (1987) essentially concluded that the differences were because the majority of data used to produce Broadfoot (1961) came from major river bottoms and loessial hills. It should be noted that only the curve from Carmean et al. (1989) possesses a polymorphic nature, and thus perhaps the predictions of the four other curves are somewhat restrictive because of their anamorphic nature.

The equations found in Clatterbuck (1987) and Oswald et al. (2017) have nearly identical projections. Both datasets are largely from minor bottoms, but differ substantially in geographic range. Clatterbuck (1987) data are from central Mississippi and Oswald et al. (2017) data are from central East Texas. Predictions from both of these equations differ substantially from equation [5] and Schultz et al. (2010). Clatterbuck (1987) and Oswald et al. (2017) both predicted smaller dominant heights at younger ages and substantially exceeded predictions of the three other equations beginning around age 60. Differences in the definition of dominant height obviously could lead to substantially different predictions, and differences of equation form could also contribute to the variation in predictions among equations.

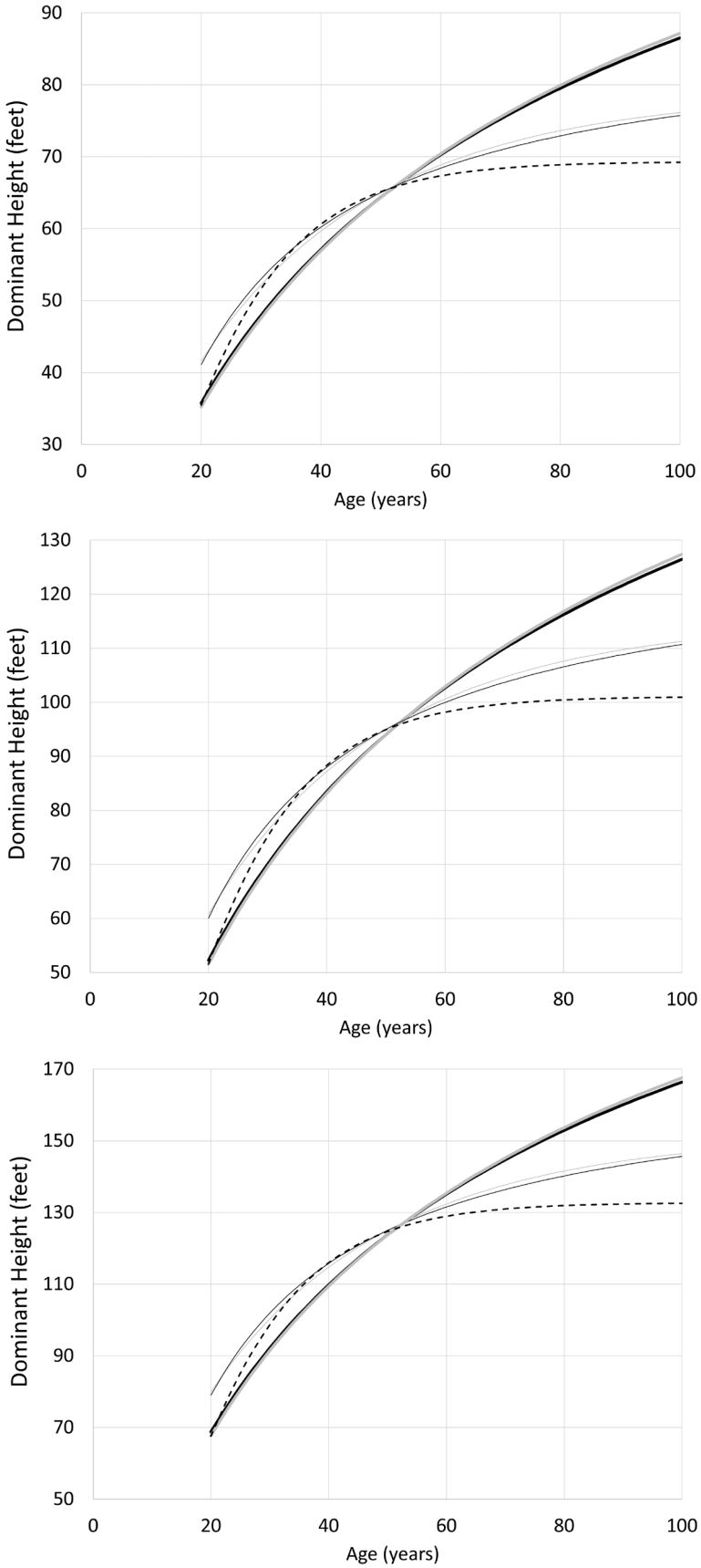

Figure 6.—Predicted dominant height by site index (SI; base age 50) for cherrybark oak for site indices of 65, 95, and 125 (upper to lower panels). Equation [5] (in text)—black line, Schultz et al. (2010) —gray line, Oswald et al. (2017)—bold black line, Clatterbuck (1987)—bold gray line, and Carmean et al. (1989)—dashed black line.

Across all site indices, equation [5] and the equation from Schultz et al. (2010) had nearly identical projections. In fact, across all three site indices and from ages 20 to 100, the average absolute percent difference was only 0.82 percent; the maximum difference at any age was 1.56 percent. This result is not necessarily validation, but it is a means of verification. There are a few inferences that could be drawn from this result. First, Schultz et al. (2010) could be applied beyond essentially eastern Mississippi to other parts of the western half of the southeastern United States. Second, it verifies that the predictions from equation [5] are biologically reasonable and that the site index equation could be applied with confidence. Third, site index curves developed using FIA data, with proper plot and tree selection, appear to provide reasonable dominant height predictions.

Figure 7 presents dominant height estimates across time for swamp chestnut oak using equation [4], and the equation presented in Carmean et al. (1989) for water oak, based on the site index curves presented in Broadfoot (1963), for three site indices (base age 50). The curve from Carmean et al. (1989) exhibited faster early height growth relative to equation [4]. However, similar to Carmean et al. (1989) for cherrybark oak relative to the four other cherrybark oak equations, the water oak curve from Carmean et al. (1989) had substantially less growth beyond age 50 relative to equation [4]. Based on Figure 7, the use of the water oak site index equation from Carmean et al. (1989) by FVS to represent the behavior of swamp chestnut oak may not be appropriate. The use of the water oak equation may produce biased stand-level predictions. It should be noted that the curve from Carmean et al. (1989) possesses a polymorphic nature, and thus perhaps the predictions of equation [4] are somewhat restrictive because of their anamorphic nature.

CONCLUSIONS

Based on observation with other available cherrybark oak site index equations, the predictions obtained from equations [5] and [7] can be used with confidence within applicable ranges of the states of Arkansas, Louisiana, Mississippi, and Tennessee, and the western half of Alabama. As expected, there are differences in dominant height development between cherrybark oak and swamp chestnut oak. Dominant height and site index estimates for swamp chestnut oak when using equations [4] and [6] appear reasonable relative to other assessments of swamp chestnut oak.

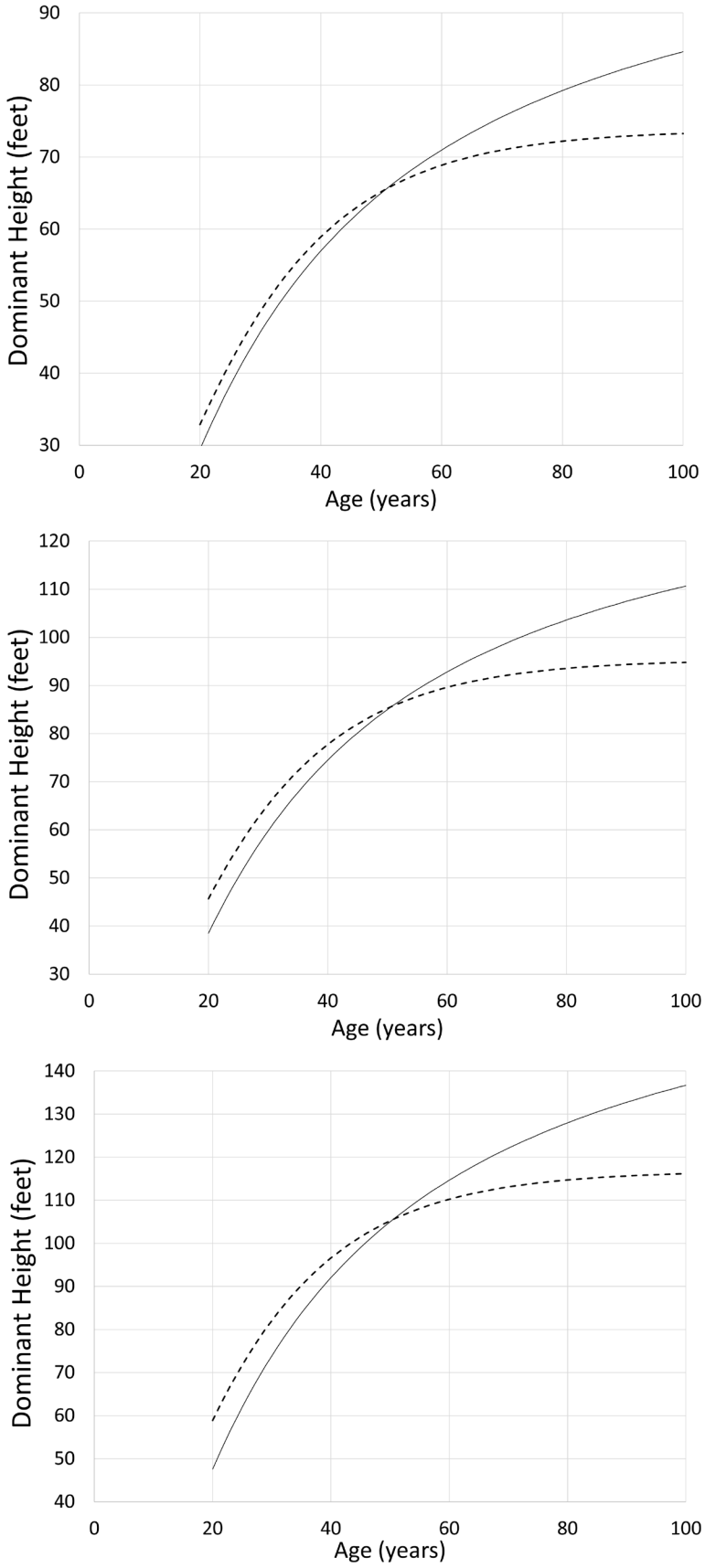

Figure 7.—Predicted dominant height by site index (SI; base age 50) for swamp chestnut oak for site indices of 65, 95, and 105 (upper to lower panels). Equation [4] (in text)— black line and Carmean et al. (1989)—dashed black line.

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The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

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Growth Projections in SILVAH: Do They Measure Up?

Matthew C. Marsh, Bryce T. Adams, Todd E. Ristau, and Jamie L. Schuler[1]

INTRODUCTION

Much of the northeastern United States is covered in forests, which provide many ecosystem services such as timber products, carbon sequestration, water filtration and storage, wildlife habitat, and opportunities for recreation (Brockerhoff et al. 2017, Thompson et al. 2013). These services are highly valued by society; therefore, proper and sustainable forest management is highly important (Schuler et al. 1993). However, forest management is a complex undertaking that requires the combination of ecological and operational knowledge on different harvesting practices to satisfy societal needs (Twery and Weiskittel 2013). Because forest-level management is complicated, tools that reduce future uncertainty of stand conditions are of value (Shifley et al. 2017, Twery and Weiskittel 2013).

One management tool that has had consequential effects on improving management outcomes is forest growth and yield models (Schuler et al. 1993). SILVAH (SILViculture of Allegheny Hardwoods), a computer program developed by the USDA Forest Service in the late 20th century, generates growth and yield projections for Allegheny Plateau forest types and Central Hardwood forests dominated by oak (*Quercus*) (Marquis and Ernst 1992, Stout and Brose 2014). SILVAH uses a model that is categorized as a distance-independent, individual-tree growth model (Marquis and Ernst 1992). Among a set of other functions, SILVAH can predict basal area, number of trees, and volume over a specified projection period by relying on the following predictors: diameter, species, site index, and relative stand density (Marquis and Ernst 1992). The model is useful for sensitivity of future product yields to varying initial stand conditions, rotation length, and other factors.

The SILVAH growth and yield simulator is an iterative process that starts by categorizing tree-level data into five species groups, then applies mortality functions to these groups. The species groups are (1) sugar maple (*Acer saccharum*); (2) American beech (*Fagus grandifolia*) and eastern hemlock (*Tsuga canadensis*); (3) black cherry (*Prunus serotina*), white ash (*Fraxinus americana*), and yellow-poplar (*Liriodendron tulipifera*); (4) red maple (*Acer rubrum*) and others; and (5) oak species. For each of the four non-oak species groups, there are four mortality equations determined by relative stand density (RSD) level. RSD is a measure of tree density developed for use in mixed species stands. It is similar to percent stocking, but RSD was developed specifically for use in forests of the Allegheny Plateau (Stout et al. 1987). RSD differs from absolute measures of density like basal area in that it is calculated with species-specific equations that more accurately capture site utilization in a single-cohort mixed-species hardwood stand (Brose et al. 2008, Nyland et al. 2020, Stout et al. 1987).

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SILVAH categorizes RSD into four classes or groups: less than 50 percent, between 50 and 80 percent, between 81 and 95 percent, and greater than 95 percent. The equations determine a percent mortality that is used to reduce the basal area of each tree, which is a method of mortality application similar to other probabilistic methods of mortality application in other growth and yield models (Weiskittel et al. 2011). Other forest growth and yield models, such as the Forest Vegetation Simulator (FVS), apply mortality by multiplying the expansion factor for a given tree by the predicted chance of survival (Weiskittel et al. 2011). Following the application of percent mortality, SILVAH then projects growth in a similar fashion with the same five species groups, but within each species group there are only three different RSD equations. The oak species group is projected using growth equations from Hilt (1985).

SILVAH, as well as other growth and yield models, relies on statistical relationships between commonly sampled forest inventory variables such as diameter, site quality, species, and stand density to predict future growth and mortality. Although these models are widely used, little testing has been done to validate growth and yield estimates, especially over longer projection periods. A few studies have tested SILVAH predictions, but no long-term projections (greater than 50 years) have been tested (Bankowski et al. 1995, Brooks and Miller 2011, Schuler et al. 1993). The vast majority of growth model validation focuses on short-term projections (less than 20 years). We tested the projection accuracy of trees per acre (TPA) projections and basal area per acre (BAAC) projections in SILVAH over different time periods with an emphasis on long-term (more than 50 years) projections. The results of this study will guide future research into the improvement of the regression equations used in the SILVAH growth simulator.

METHODS

Five datasets comprising 129 plots were used for this paper (Table 1). Study sites are located in northwestern Pennsylvania and northern West Virginia. Datasets are from long-term Forest Service research and West Virginia University studies. Sampled stands consisted of predominantly even-aged Allegheny Hardwoods. In this research we were specifically interested in looking at differences in prediction error under different relative stand densities; therefore, we decided to aggregate all of the studies given that most studies (all but one) were in the same research forest. The following procedure is based on work done by Brooks and Miller (2011). For each plot, BAAC and TPA were calculated for each measurement and remeasurement period.

Table 1.—Plot-level statistics across all remeasurement periods for trees per acre (TPA) and basal area per acre (BAAC) by study

Study	Minimum TPA	Maximum TPA	Minimum BAAC (square feet per acre)	Maximum BAAC (square feet per acre)	Plots (n)	Length of projection time available (years)	Plot size (acres)
Bear Tree	342	848	55.64	136.31	15	15, 30, 60	0.5
Weeding	160	390	1.56	17.91	48	15, 30	0.1
Thinning study	91	1,055	45.18	132.24	31	15, 30	0.6
Yield study	460	2,040	12.77	147.7	14	30, 60	0.1
West Virginia Continuous Forestry Inventory	95	420	55.13	192.52	21	15	0.2

Stand-level summaries for thinned plots were performed after a thinning took place and before a second thinning took place. Tree data for each plot were entered into SILVAH and projected for either 15, 30, or 60 years. Because of the nature of the different datasets, linear interpolation was required for plots that did not match 5-year remeasurement periods. Projections in SILVAH were then compared to the field-sampled data from the projected timeframe. Bias and root mean squared error (RMSE) were calculated for TPA and BAAC based on these comparisons. Bias was calculated to determine whether SILVAH is more likely to overpredict or underpredict BAAC or TPA. RMSE provides an estimate of error unaffected by changes in the algebraic sign of the comparison between the measured data and the projections. SILVAH does not include ingrowth in its projections, nor do our analyses in the stand-level totals.

RESULTS

Basal Area

Basal area projection error varied over projection length and RSD group, ranging from 4 to 56 percent of measured values (Fig. 1). On average, group 3 was the most accurately predicted RSD group (deviation percentage of 11 percent), while group 1 had the greatest departure from the true values (46 percent at 60 years; Fig. 1). Additionally, broad differences in error were seen across RSD groups. For example, RSD group 3 had relatively small projection errors even for longer projection periods. SILVAH was more likely to underpredict BAAC than overpredict it (Fig. 2). This was true for plots with large and small error rates but became more pronounced at higher (above 31 percent) prediction error rates (Fig. 2).

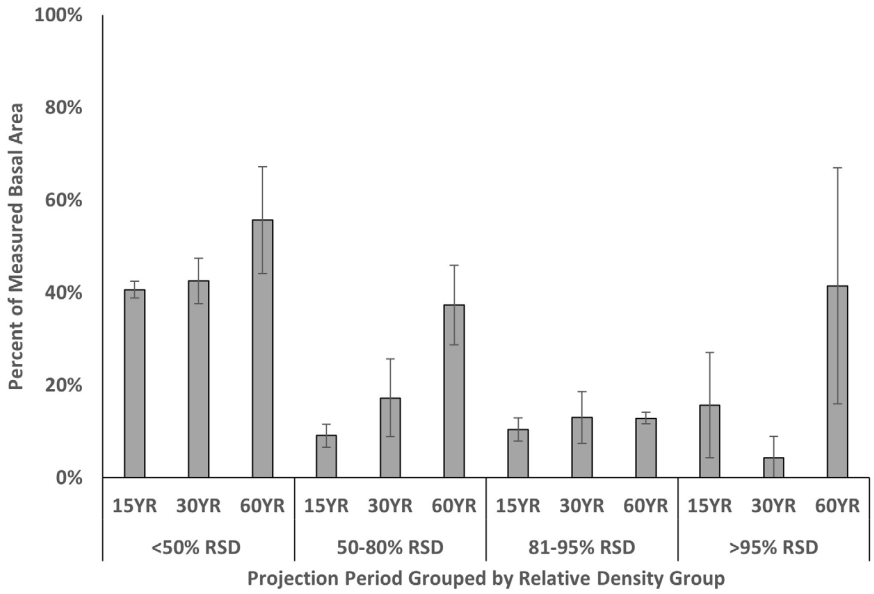

Figure 1.—Projection error for basal area per acre by SILVAH over different projection periods grouped by SILVAH relative stand density (RSD) groups, shown with standard error. Number of plots (n) per group: 57, 37, 20, and 15, respectively.

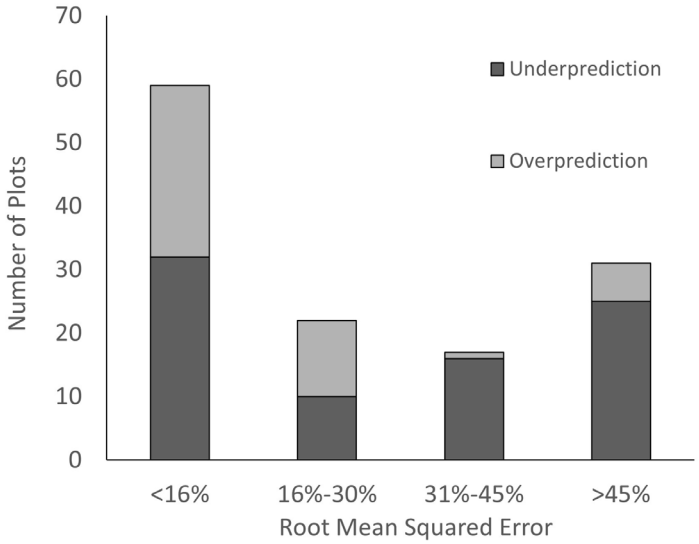

Figure 2.—Frequency distribution of prediction root mean squared error by SILVAH between observed and predicted basal area per acre with bias results. All projection periods are included.

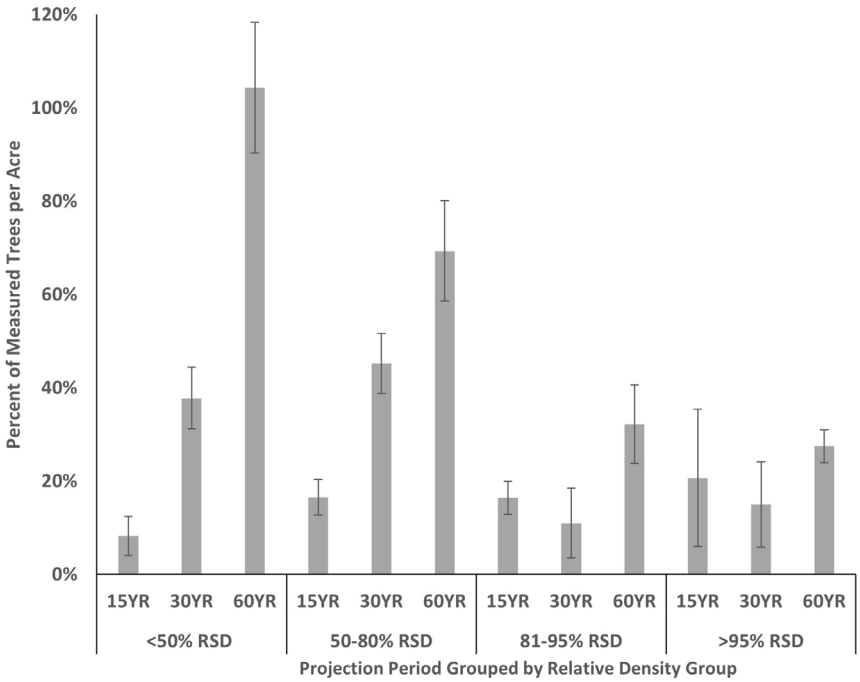

Figure 3.—Projection error for trees per acre by SILVAH over different projection periods grouped by SILVAH relative stand density (RSD) groups, shown with standard error. Number of plots (n) per group: 57, 37, 20, and 15, respectively.

Trees per Acre

TPA projection error tended to increase as projection length increased. However, there was no clear trend seen among the 15- and 30-year projections (Fig. 3). Errors in predicted TPA were generally larger than those for BAAC. In some cases (i.e., group 1 at 60 years), the difference between the true and predicted value was more than 100 percent. In contrast to the BAAC predictions, SILVAH tended to overpredict TPA (Fig. 4). When projection error was greater than 45 percent of measured TPA, SILVAH exclusively overpredicted TPA (Fig. 4).

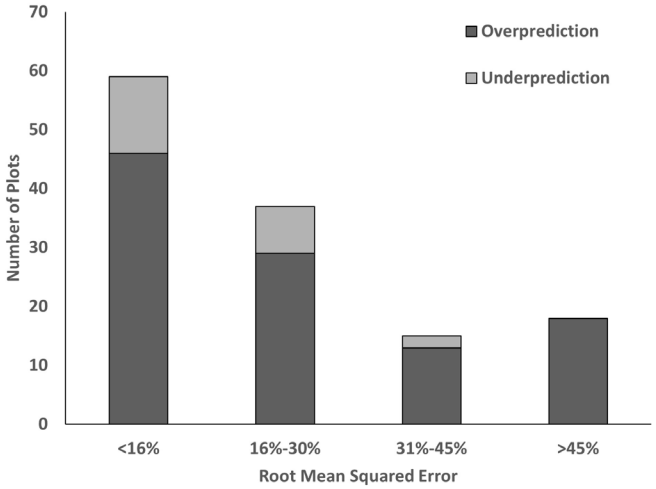

Figure 4.—Frequency distribution of root mean squared error by SILVAH between observed and predicted trees per acre with bias results. All projection periods are included.

DISCUSSION AND CONCLUSIONS

The results of this study show that the SILVAH growth and yield model is more accurate at predicting basal area per acre and trees per acre over shorter projection periods (15 years) and at higher relative stand densities (more than 50 percent relative stand density). These results are directly applicable to northwestern Pennsylvania and the Allegheny Hardwood forest type but are in line with other SILVAH validation tests that were more geographically expansive (Bankowski et al. 1995, Brooks and Miller 2011, Schuler et al. 1993). For example, previous work has found that SILVAH basal area predictions can depart up to 25 percent from actual values and SILVAH has been found to make relatively large prediction errors in unthinned young stands, which is not surprising given our findings about predictive performance and RSD (Brooks and Miller 2011, Schuler et al. 1993). Differences in SILVAH projection accuracy were seen over varying projection periods and across RSD groupings. This discrepancy can be accounted for by two different factors. First, little written information about the original development of SILVAH is available; however, the results of this study are consistent with there being a lack of low relative density stands among those used to develop the models in SILVAH. Because the growth and mortality models were likely calibrated from datasets that lacked stands with low densities, the relative prediction error was higher in stands of low tree density. Furthermore, stands with low tree densities are often more variable, making accurate predictions more difficult. In addition, errors compound over time when modeling growth and yield, so it is unlikely that differences in error across projection periods will ever be eliminated (Lotka 2017). However, other research has suggested that whole-stand estimates of volume within 10 to 15 percent of actual stand volume is a realistic goal in Appalachian stands (Brooks and Miller 2011). Our results suggest that improvement of the growth and yield model in SILVAH is feasible given the advancements with modern computer capabilities and modern statistical techniques like mixed-effects modeling.

Furthermore, additional data from long-term studies, even those used to develop the existing SILVAH growth models, have become available. These results can guide future efforts to update the growth and yield model in SILVAH.

ACKNOWLEDGMENTS

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The content of this paper reflects the views of the author(s), who are responsible for the facts and accuracy of the information presented herein.

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Soils

New Digital Soil Tools Available for Forest Scientists and Managers

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Abstract

Soil information is essential for identifying appropriate tree species to manage, selecting appropriate silvicultural methods, and planning timber harvests. Traditionally, soil information was available in paper copies of soil surveys, and more recently in digital form via Web Soil Survey. However, newly available soil information tools provide interactive access to digital soil information and interpretations. We describe several different soil information tools that provide access to soil survey information and interpretations in an interactive framework, including SoilWeb, SoilWeb Earth, and the Soil Properties viewer. Each tool is an interactive web-based computer application that facilitates access to and use of soil survey data on personal computers, tablets, or phones.

INTRODUCTION

Foresters recognize that soil information is essential for managing forest land. Soil maps and associated soil property information are needed for delineating stands, identifying tree species suited for the site, selecting appropriate regeneration methods, and protecting soils from compaction, rutting, and erosion during harvesting operations. However, despite the recognized importance of soil information in forestry, accessing and using soil information has historically been challenging for non-soil scientists due to the inherently complex, multidimensional nature of soils and limitations with conveying soil information in traditional paper soil surveys.

The most comprehensive inventory of soils in the United States is the Soil Survey Geographic database (SSURGO). Historically, access to SSURGO was limited to paper copies of soil surveys published by county or national forest. More recently, the Web Soil Survey (USDA Natural Resources Conservation Service 2017) was developed to deliver a wider array of soil information digitally. This powerful software can be used to produce soil maps and reports about specific soil properties and identify the suitability of soils for various land use practices. It also provides the means for downloading SSURGO data to be used in other applications such as with geographic information systems (GIS) or in other databases. However, this software requires users to possess considerable technical knowledge about soil terminology and about SSURGO for it to be used effectively.

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Recently, several new software tools have been developed that provide an interface between SSURGO and soil information users. These new tools include SoilWeb, SoilWeb Earth, and the Soil Property Viewer (O'Geen et al. 2017). There are additional applications (apps) including the Soil Data Explorer and the Soil Extent Viewer that run within them or stand alone. These software tools provide access to the SSURGO database in a user-friendly, intuitive, and interactive way which helps convey complex soil information more effectively than a traditional soil survey or the Web Soil Survey. The objective of this paper is to describe these different tools, illustrate how they can be used to better inform foresters, and show how they can be applied in forestry applications.

SOILWEB

SoilWeb (https://casoilresource.lawr.ucdavis.edu/soilweb-apps/) is an app which provides an interactive interface to SSURGO that can be used with a web browser on a desktop or laptop computer, a tablet, or a cellular phone. There also is a native SoilWeb app for Android and iOS devices (users can check the app store on their device for SoilWeb produced by the University of California, Davis). When used on a phone or tablet, the app displays one or more soil profiles based on the user's location with profile sketches that are generated from the best available data in SSURGO. The browser version displays interactive soil maps and can also be used on a phone, tablet, or computer. The phone app version lacks these maps but includes access to all other SSURGO data and a link to the browser version.

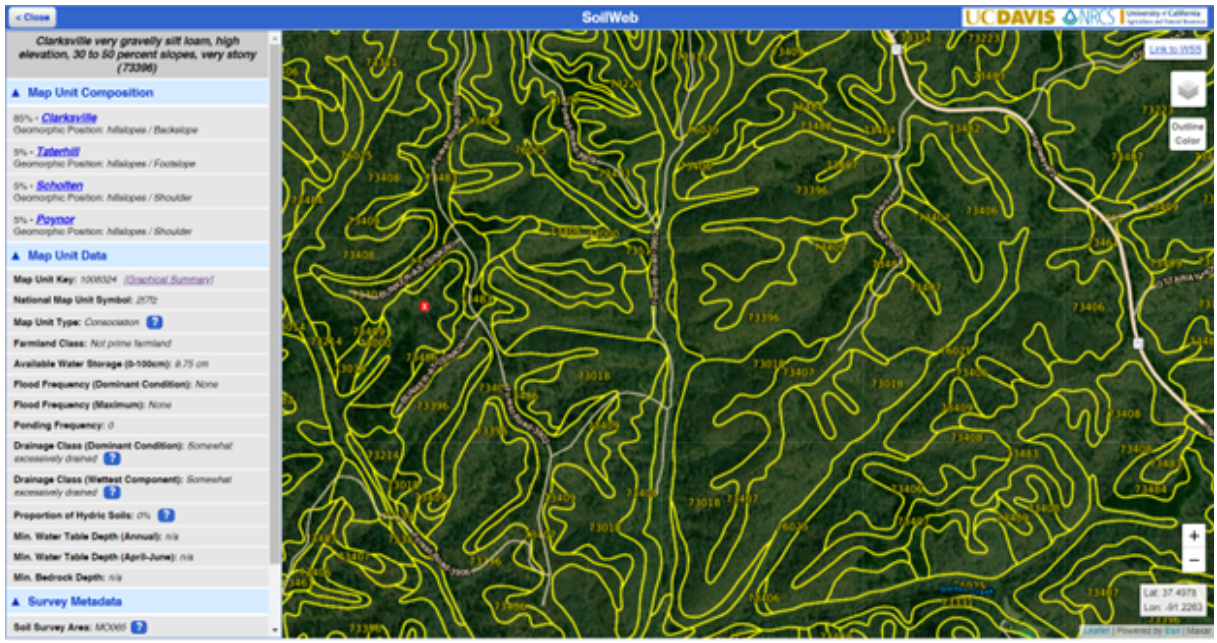
The following SoilWeb description and figures are focused on the browser version because this version includes the interactive soil maps. With the browser version, the Leaflet web mapping library is used to provide an interactive means for exploring and querying soil maps. This interface allows users to select one of the following three ways to locate a point on the map: using web browsing software to geolocate the user's computer; typing in an address, ZIP Code™, or landmark; or entering the latitude and longitude. Stable links to specific locations (WGS84 coordinates) and zoom levels are provided in the navigation bar or copied to the clipboard when the user right-clicks on the map.

Several base maps can be displayed, including satellite imagery with map unit (soil polygons) superimposed and an option to display road names and natural features like rivers and streams (Fig. 1A). There also are options to select U.S. Geological Survey (USGS) or Esri (Esri, Inc., Redlands, CA) topographic base maps with soil unit boundaries superimposed (Fig. 1B).

By selecting a location on the soil map, a summary is displayed that provides information about the map unit name, symbol, primary soil phase, slope class, and expected soil components found within. A soil component is typically named for a soil series[2] but can represent more generic concepts such as a family or subgroup taxon from Soil Taxonomy.

[2] A soil series concept encompasses a relatively narrow range of soil physical, chemical, and morphologic properties (as a suite of genetic horizons) that give rise to similar use and management considerations. The soil series is also the lowest category in Soil Taxonomy.

A



B

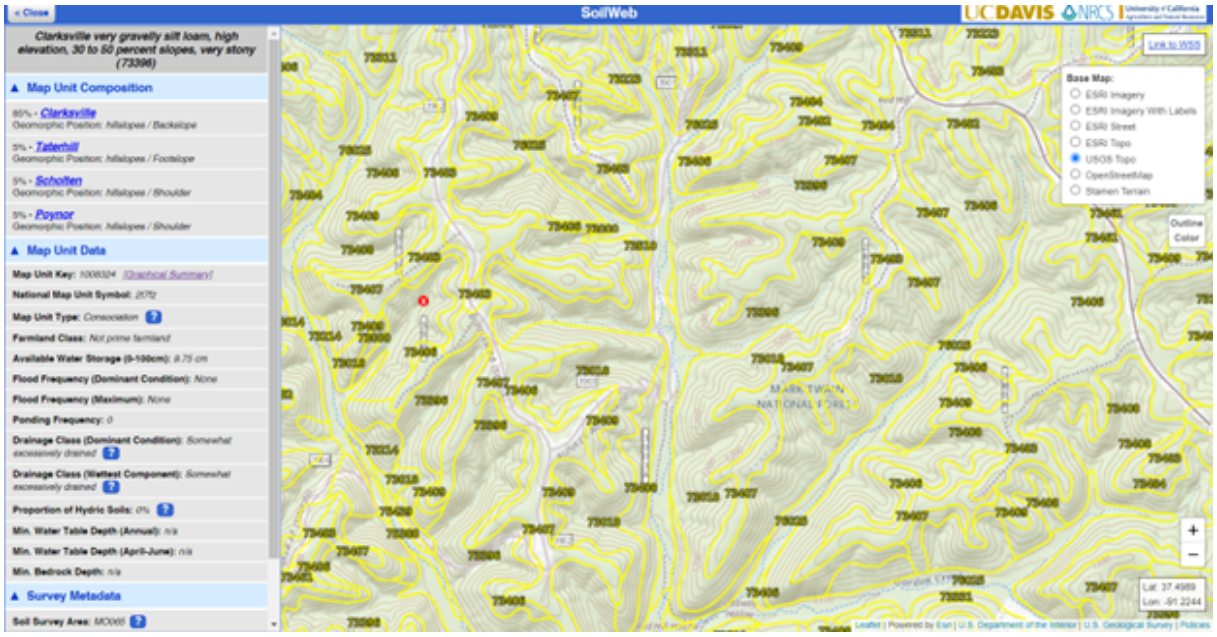


Figure 1.—Base map display including satellite imagery (A) and topographic map (B).

This summary also includes the typical arrangement of components by geomorphic or hillslope position, estimated areal proportions, and the map unit type—a classification of spatial complexity relative to mapping scale (Fig. 2). The ability to show multiple soil components, their expected proportion, subgroup level classification, geomorphic clues, and drainage class is an exceptionally important feature because it conveys soil variability in a very intuitive way. By selecting the drop-down menus, additional information about the map unit is displayed.

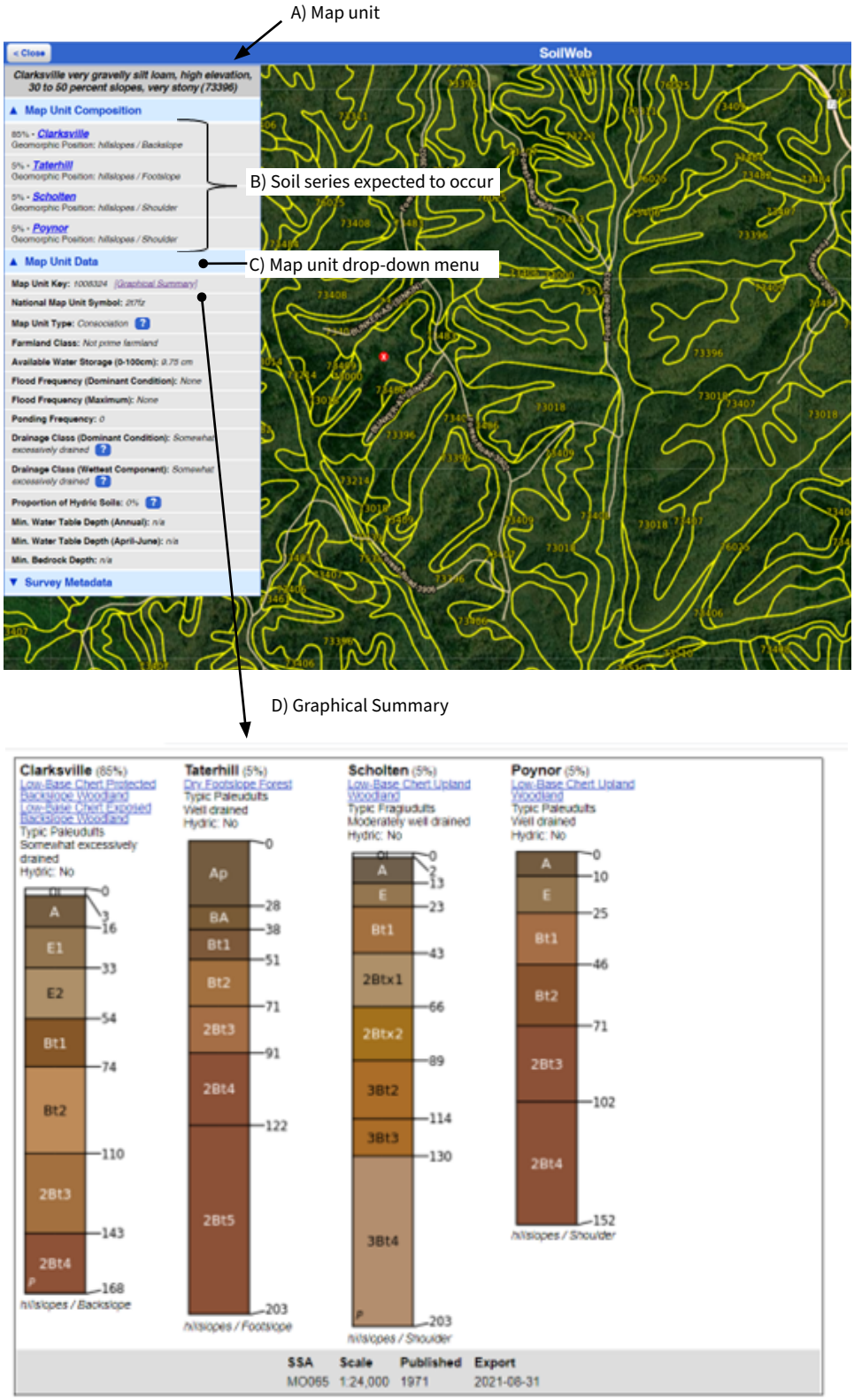
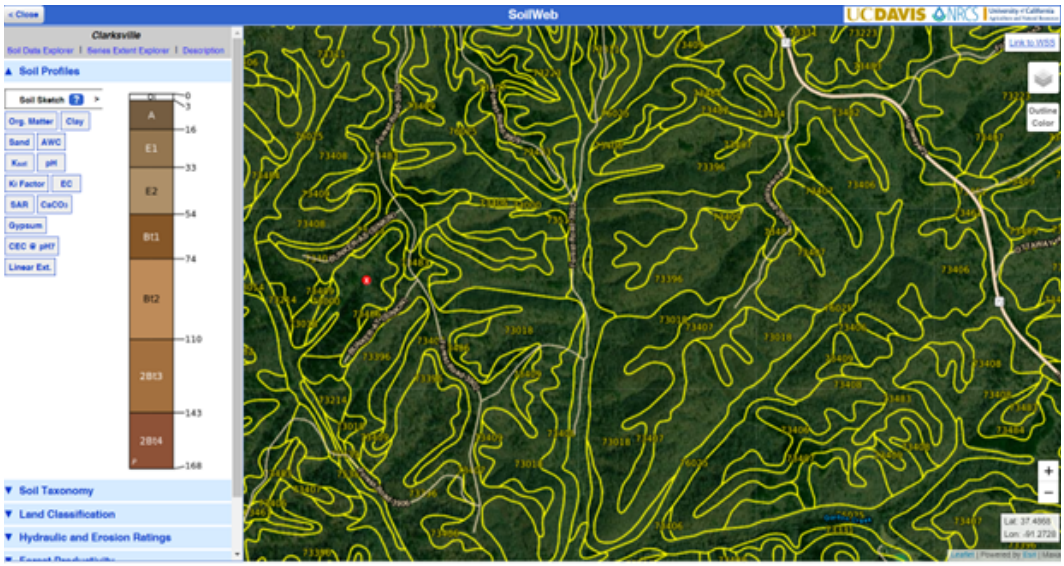


Figure 2.—Map unit (soil polygon) summary (A) of map unit components, which are often correlated to soil series (B) in the drop-down menu (C), and graphical summary (D).

When a single soil component is selected, a stylized soil profile "sketch" is displayed to illustrate soil horizons and their colors (Fig. 3A). This is particularly useful in forestry applications for identifying the types of genetic horizons within the profile (indicative of soil formation processes), the presence of root-restricting layers (e.g., a fragipan; Fig. 3B), or drainage restrictions indicated by the presence of gleyed (gray-colored) horizons. Soil profile sketches are generated from field descriptions when available (also called component pedons); otherwise, sketches are generated from Official Series Descriptions. Sketches for higher-taxa components (e.g., subgroup-level classification) are created directly from SSURGO records, which do not contain representative soil colors. A suite of commonly used soil properties can be selected and displayed graphically or in table form. Hyperlinks provide additional information about the soil sketches.

A



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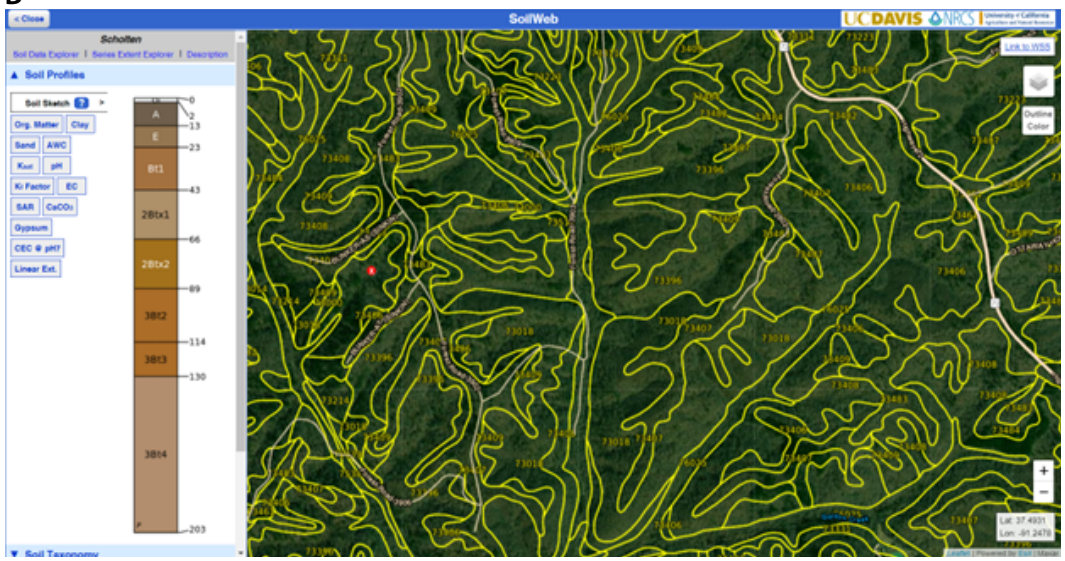

Figure 3.—Soil profile sketches (A and B). A soil with a fragipan or fragic conditions is indicated by the "Btx" horizons (B). Hyperlinks are used to access additional information.

Other pull-down menus on the left (Fig. 4A,B) provide access to important information for practicing foresters. Key menus include the:

- **Soil Taxonomy** menu, which provides classification information to the series level for including in forest management plans.

Figure 4.—Drop-down menus for **Soil Taxonomy** and **Land Classification** (A) and **Forest Productivity** and **Soil Suitability Ratings** (B).

- **Land Classification** menu, which provides the Ecological Site Description and information about expected plant communities associated with the soil map unit as well as common vegetation states under different land uses and management practices. In the eastern United States, Ecological Sites are analogous to the Ecological Land Types developed by the USDA Forest Service (Iverson et al. 2019). More information about Ecological Sites can be found by selecting the hyperlink. This menu also gives organic carbon pools, which can be useful for understanding potential impacts of forest operations.
- **Hydraulic and Erosion Ratings** menu, which includes the erosion tolerance (T-Factor), runoff rates, soil drainage class, hydric rating, and the Hydrologic Group (used in runoff models). The total Plant Available Water rating is also included, which provides an estimate of water storage, and is an important metric needed for predicting forest regeneration potential (Kabrick et al. 2014), stand development patterns, and total productivity (Kabrick et al. 2008). On the browser version, hyperlinks provide access to more information about these ratings.
- **Forest Productivity** menu, which provides site index information associated with individual soil series where suitable site index trees were available (for suitability criteria, see Carmean et al. 1989). Productivity estimates (cubic feet per acre per year) are also provided and are similar to those by Schnur (1937).
- **Soil Suitability Ratings** menu, which provides links to various land uses, including Forestry ratings related to planted seedling performance; rutting and erosion associated with harvesting, road construction, and log landing use; and risk associated with windthrow, diseases, and droughts. Information about seasonal rutting hazard is also included and this information can help determine the best seasons for logging operations. Criteria for developing these ratings are available in the "National Forestry Manual" (Natural Resources Conservation Service Staff 1998). For the Central Hardwood region, there is a Black Walnut Suitability Index that can be used for identifying suitable soils for growing this species. This index was developed using published information about site factors known to affect black walnut (*Juglans nigra*) growth, including soil texture, drainage class, flood frequency, soil depth, pH, plant available water, and historical vegetation (Wallace and Young 2008). A suitability rating using a similar framework is planned for improved northern red oak (*Quercus rubra*) stock.
- **Details** menu, which provides additional information about soil properties and a link to the National Cooperative Soil Survey Data Mart for downloading laboratory characterization data.

The same pull-down menus can be found in the SoilWeb phone app by selecting a soil component and then selecting the "details" link on the bottom of the screen.

SOIL DATA EXPLORER

The **Soil Data Explorer** (Fig. 5A–D) provides links to several tabs with important soil information for foresters. The first tab provides a link to the **Official Series Description** (OSD; Fig. 5A). The OSD provides a succinct narrative about parent material, drainage, and permeability, as well as a detailed description of a representative pedon for the selected series. It also provides context by identifying other soils that are typically mapped with the featured soil and the key properties distinguishing it from other soil series. The OSD is the most comprehensive information source for learning about a specific soil series.

The **Lab Data** tab (Fig. 5B) provides a link to aggregated data for selected chemical and physical properties for all pedons correlated to the selected soil series. It also provides the median, the 5th percentile, and the 95th percentile by depth for each property, indicating range of variation that can be expected for each of the soil properties. Here foresters can quickly get estimates of the amount of stored carbon, the pH and base saturation, and the cation exchange capacity by depth for entire soil profiles. These properties are important for assessing nutrient storage potential and depletion risk caused by harvesting, prescribed burning, and other forest management activities where trees and other vegetation biomass are consumed or removed.

The **Water Balance** tab (Fig. 5C) was introduced in 2022, and provides average monthly precipitation, potential evapotranspiration (PET), and soil plant available water storage, and the resulting monthly expected soil water availability. Precipitation data are based on PRISM 30-year averages and PET is estimated by Thornthwaite's (1948) method. The water storage estimate is from SSURGO. Available water storage is estimated using a "leaky bucket" style model (Arkely and Ulrich 1962) that accounts for soil drainage in addition to net changes in soil water due to inputs from precipitation and losses from evapotranspiration. The water budget provides an estimate of the plant water availability for the soil during a typical year and identifies the time of year when water deficits are expected and when recharge normally occurs.

The **Sibling Summary, Competing Series,** and **Shared Subgroup** tabs provide information about how the selected soil series is related taxonomically to other soil series, which may be of more interest to pedologists and other soil classifiers than to foresters. However, the **Block Diagrams, Map Units,** and **Extent** tabs provide insight into where in the landscape a series is expected to occur, the map units associated with the series, and where the series is mapped (Fig. 5D). This information is particularly useful for foresters working on large districts or with private landowners where first-hand knowledge and experience with forests on specific groups of soils can be transferred elsewhere in the region where the same or similar soils occur.

SOILWEB EARTH

SoilWeb Earth (https://casoilresource.lawr.ucdavis.edu/soilweb-apps/) runs on the Google Earth platform to overlay soil map unit boundaries on satellite imagery. It has similar functionality as the browser version of SoilWeb and has the added advantage of providing access to point data from sampled pedons stored in the National Cooperative Soil Survey Soil Characterization Database. Other benefits of SoilWeb Earth are its utilization of Google Earth's high-resolution satellite imagery, its three-dimensional rendering of the landscape, and access to SSURGO data from map unit to regional scales.

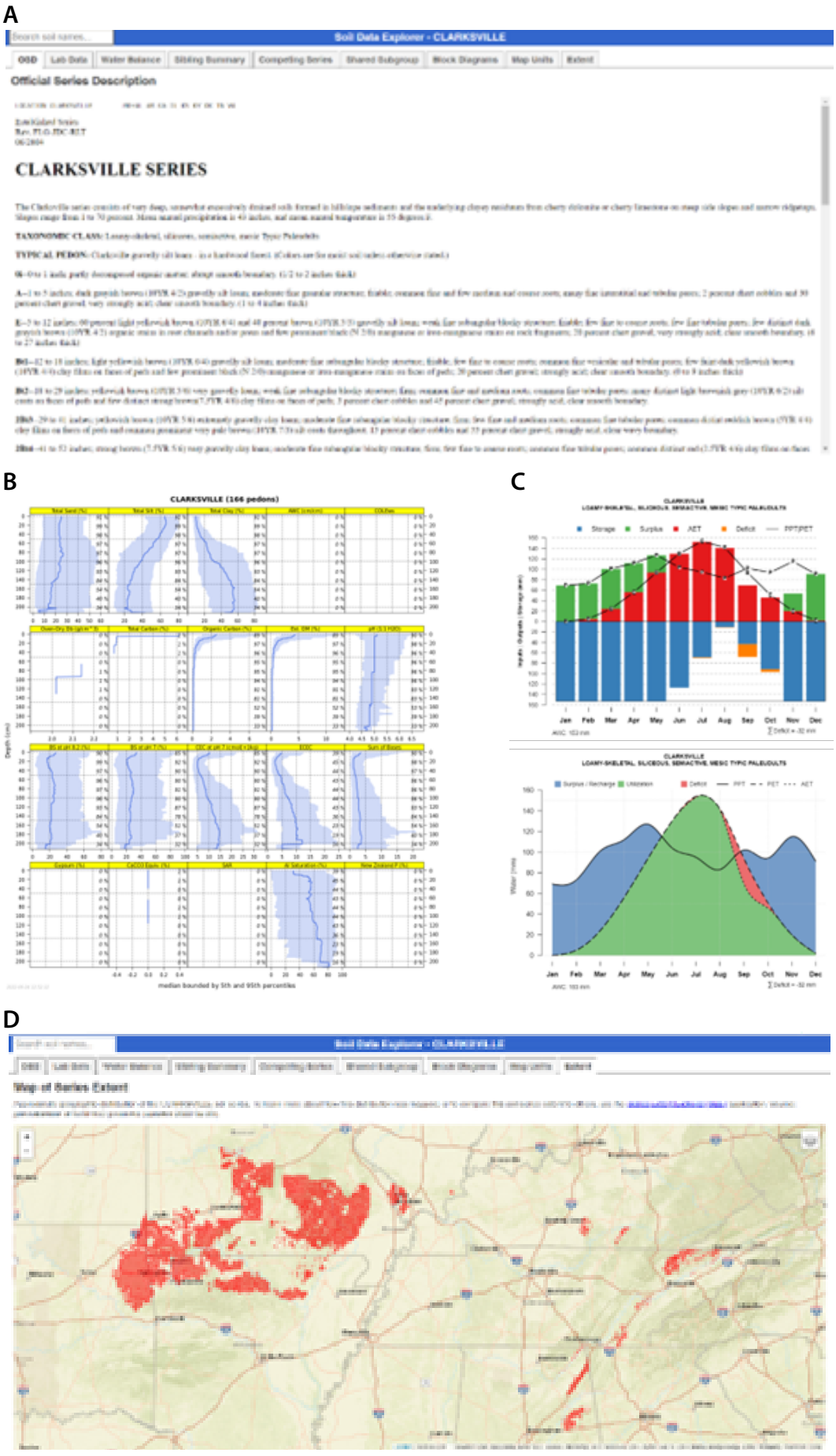

Figure 5.—Soil Data Explorer tabs for the Official Series Description (A), Lab Data (B), Water Balance (C), and Extent (D).

SOIL PROPERTIES VIEWER

The Soil Properties viewer (https://casoilresource.lawr.ucdavis.edu/soil-properties/) (Fig. 6A,B) allows users to examine chemical and physical soil properties from local to national scales. The soil properties are derived by aggregating map unit-scale data from SSURGO to create spatial datasets at a 2,624 foot (800 meter) grid resolution. The grids use an Albers Equal Area projection centered on the continental United States and the NAD83 datum (EPSG:5070).

The information provided by the viewer is particularly useful for examining regional and national patterns in soil properties. The soil property data can also be downloaded in a GeoTIFF format and, with appropriate software, linked to other data to examine relationships between soil properties and forest vegetation. For example, Figure 6C shows the negative relationship between soil pH and the abundance of black birch (*Betula lenta*) in the eastern United States. The abundance of black birch was derived from USDA Forest Service Forest Inventory and Analysis (FIA) data, scaled to the same grid resolution as the GeoTIFF of soil pH.

DISCUSSION AND CONCLUSIONS

SoilWeb and associated applications are an intuitive and interactive suite of tools for accessing SSURGO data. They provide a powerful means for assisting foresters to quickly identify and obtain technical information about soil map units, soil properties, and interpretations. These tools offer alternative means for accessing SSURGO data compared to Web Soil Survey, an app which can be used to produce customized reports for specific locations. In contrast, SoilWeb is a data-focused interface for exploring soils and soil properties. It uses integrated data panes to provide soil properties in graphic or tabular form along with the soil map in the background. This facilitates a rapid understanding of soil information, which often is complex and difficult to visualize.

SoilWeb source data are based on an annual snapshot of SSURGO, updated October 10 each year. However, the apps themselves are continually being updated and data products are rapidly evolving, with new components and applications being developed regularly. Because the tools can be used on a variety of platforms, they can be used in the office or in the field, wherever an internet or data connection is available.

Throughout the Central Hardwood region, SSURGO data are available and accessible with SoilWeb and associated apps. A comprehensive list of soils networks and resources (including those described here) can be found in Appendix B of "Forest and Rangeland Soils of the United States under Changing Conditions" (https://www.fs.usda.gov/treesearch/pubs/61256).

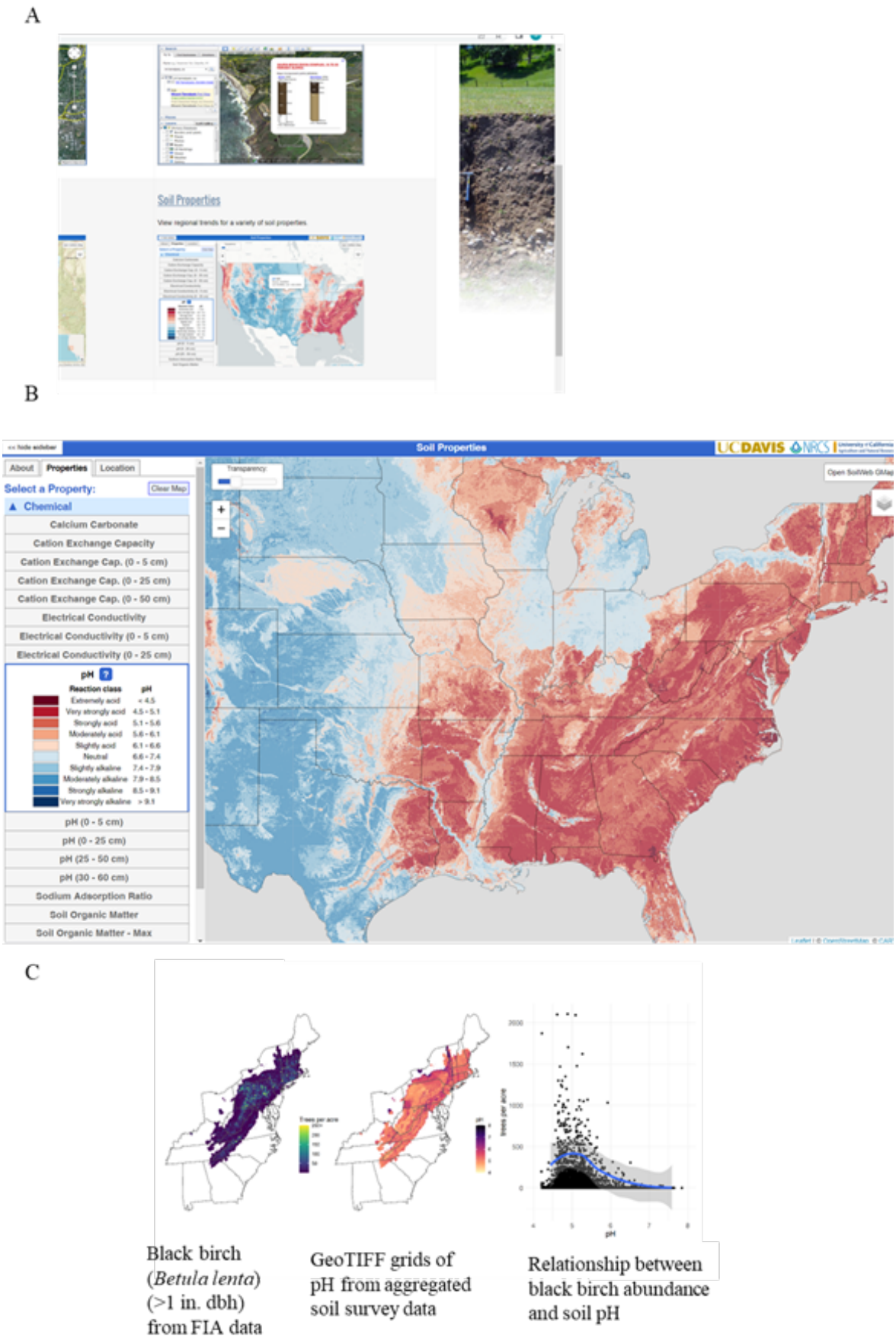

Figure 6.—Soil Properties viewer location on the UC Davis California Soil Resource Lab website (A); regional trends in soil pH shown in the viewer (B). Downloaded soil data from the Soil Properties viewer were used to examine relationships between soil pH and black birch (*Betula lenta*) abundance (C); dbh is diameter at breast height and FIA is the Forest Service's Forest Inventory and Analysis program.

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Soil-Site Relationships with Heartwood/Color Formation in Sugar Maple in Southern Ontario

Peter A. Williams and E.P. (Ted) Taylor[1]

INTRODUCTION

Sugar maple (*Acer saccharum*; Mh) is an economically important species in northern hardwood forests and is usually the most valuable, although values vary over time. The minimum criterion for higher-grade veneer logs and grade 2 or better sawlogs is heartwood less than 1/3 of log diameter. Mh is valued for its "white" wood, and logs are usually grade 2 or lower with the presence of heartwood/discoloration greater than 1/3 of the log diameter at either end of the log. As the high-grade sawlogs, export logs, and veneer logs can be many times more valuable than grade 2 or lower logs, industry should pay more for higher grade Mh, and landowners should receive more. The ability of buyers, foresters, and landowners to predict with some reliability where trees will have high proportions of heartwood and where trees will meet the standards for high-quality sawlogs and veneer, would improve returns to landowners. This capability would also help foresters target management strategies that foster low proportions of heartwood to sites most likely to benefit from those strategies.

Research before 2000 found that heartwood formation was associated more often with injury to the trees such as skidding or other trunk damage, branch or structural damage, or compaction than with other causes (Erickson et al. 1992, Nyland 1999). Since 2000, more site-related studies have been conducted (discussed later). Researchers have found that large trees, slower-growing trees, and trees from more-acidic sites tend to have larger hearts while trees that are faster growing or growing on sites with higher pH (i.e., within the pH range 4.8 to 6.0) tend to have smaller hearts (Dey et al. 2017, Hugget et al. 2007, Yanai et al. 2009).

Industry (e.g., timber buyers, mills, loggers) can generally predict where smaller Mh hearts can be found in a local area, although there is always some guesswork. It has been difficult to identify factors that reliably predict the amount of heartwood in trees. In our experience, education and training for practitioners reflect pre-2000 recommendations. The premises of these earlier recommendations are that injuries to Mh result in increasing heartwood/discoloration and that those injuries or external appearances (i.e., tree grade) reflect value potential; accordingly, pre-2000 recommendations call for marking to foster or harvest that value, without consideration for site characteristics.

BACKGROUND

Mixed results with afforestation and woodlot management initiatives by forest extension staff in the study area of southern Ontario, led to recognition of the need to understand tree species suitability to soil and site conditions (Taylor and Jones 1985, 1986). To address this need, the Soil Interpretations for Forestry program was initiated in 1980 to measure

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species performance and relate it to readily recognizable soil features, develop forest interpretations for key soil series (for forest planning purposes), and train forest staff to identify key soil properties on-site and apply the interpretive guides. Species performance by soil type was presented in terms of "site class," which ranges from 1 (best) to 5 (worst) based on site index data for each species.

Field research included the description and measurement of forest, soil, and site features. The key attributes of the study were site index (height over age at age 50) of commercially valuable trees in fully stocked, even-aged stands (conifer plantations and polewood hardwood stands) and readily recognizable soil physical properties. Co-author Ted Taylor was a principal in this work; Peter Williams was associated with the project.

Since 1996, the corresponding author has planned and managed about 900 timber harvests in deciduous and mixed-wood forests. Most of these harvests were for private landowners in southwestern (SW) Ontario, but some were farther north up to Bracebridge and east to Peterborough (Fig. 1). Most were handled from stand assessment through harvest monitoring by Williams & Associates. During the harvest inspections, the stumps, logs, and growth were often scrutinized, and there were regular discussions with the loggers, mills, and veneer buyers regarding volumes and grade of the logs.

Based on pre-2000 research (Erickson et al. 1992, Nyland 1999), it was initially expected that trees with large stump scars (i.e., from a harvest 20 years earlier or tapping) would have big hearts or streaks running up or down from the wounds. However, observations of heartwood in Mh logs at the stump and on landings in SW Ontario differed from what was taught in forestry and tree-marking programs. When asked about the tree wounds and heartwood, the cutters reported that they did not lose much veneer from color extending up from wounds; they just cut the scarred butts off and the logs were fine. This response was counter to expectations, and we paid more attention, making regular observations of hearts, discoloration, and site conditions.

It was observed that at many of the better sites in the area, Mh heartwood/discoloration had not developed from wounds from harvesting equipment, broken branches, tapping, split or broken tops, and other causes. This finding differed from observations of logs from areas with different site conditions, such as sites limited by acidic parent materials (i.e., loose, geological materials from which soil profiles develop) and other site factors that limited available rooting space. These site factors included high water table, clayey sites, shallow soils over bedrock, or severely eroded sites (i.e., sites with free carbonates).

Over time, patterns became apparent and certain heartwood patterns were associated with site characteristics and physiography. Our experience in soil/site productivity described earlier helped us make these connections. Our discussions with harvest contractors and colleagues who worked in the areas confirmed that the associations between site characteristics and Mh heartwood development were reasonable. Indeed, the emerging literature revealed that these associations did not contradict but added to the current understandings (Dey et al. 2017, Germain et al. 2015, Kaminski et al. 2019, Yanai et al. 2009). In this paper we summarize our observations of approximately 900 timber sales over 25 years. Although our findings are observational in nature, we believe they can be useful for developing guidelines for landowners and managers and may lead to quantitative research, particularly on the influence of higher pH and site quality on heart characteristics and the form of heartwood we call balloon heartwood.

METHODS

Study Area

For the most part, southern Ontario has a landscape shaped by glaciation with distinct areas dominated by calcareous limestone soil parent materials and acidic soils derived from the granitic and gneissic bedrocks of the Canadian Shield (Chapman and Putnam 1951). Glacial advancement and retreat were intermittently and ultimately followed by the sequential, depositional forces of meltwater (glacio-fluvial), postglacial lakes (glacio-lacustrine), and aeolian (windblown) sands and silts—which, when combined, created landforms with drift (loose, unconsolidated materials) thicknesses exceeding 200 feet in places. Most of the productive agricultural croplands and associated farm woodlands of southern Ontario are located on these landscapes.

Forest site quality and productivity vary widely across southern Ontario, largely because of the physiographic and site history discussed in the preceding paragraph. Areas on the Canadian Shield (Fig. 1) are generally less productive because of the shallower, stonier, and more acidic soils. Where calcareous parent materials dominate, there tends to be higher agricultural and forest productivity.

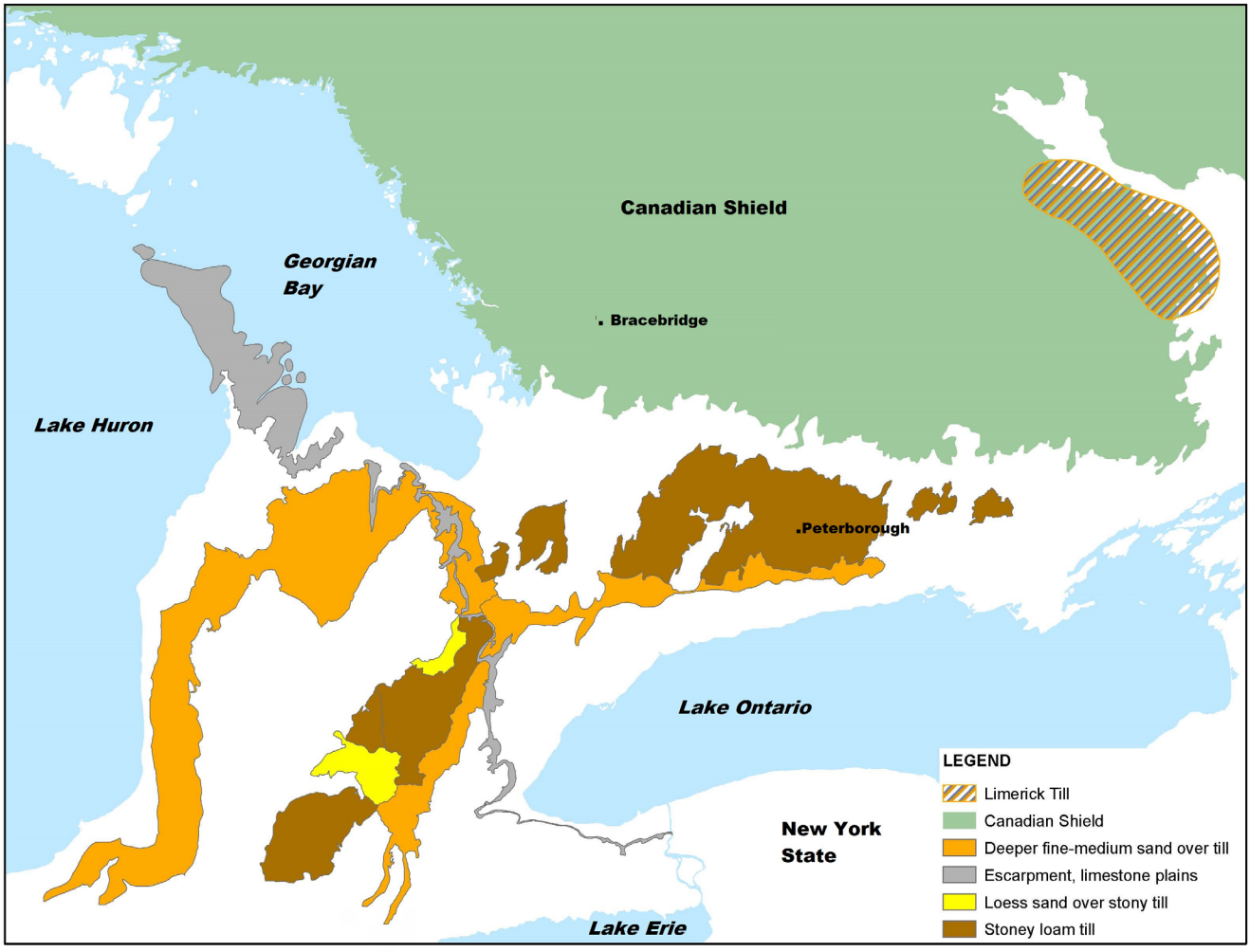

Figure 1.—Southern Ontario, Canada, with selected physiographic regions, aggregated by depositional mode, soil and site conditions, and the Canadian Shield.

During and after European settlement of the northeastern United States and Ontario, forests were periodically high-graded and the forests were retained when site conditions were not amenable to agriculture. Where sites seemed amenable to agriculture, the land was normally cleared for farming or for a settler to gain title to the land (or both). In areas settled for agriculture, land that seemed arable was mostly cleared for farming, and forests were either left where site conditions were not suitable for agriculture or partly retained to provide resources to support the population (e.g., fuelwood, lumber).

From 1900 through 1950, remnant forests in southern Ontario were intensively harvested for fuelwood for heating or fueling trains and even more heavily logged during the Depression and the two World Wars. Functionally, almost all forests in SW Ontario were harvested during settlement and have been commercially or scorched-earth clearcut one or more times since the late 1800s. Since the 1950s many have been subject to high-grading or diameter-limit harvests.

In our experience, residual forests in agricultural areas have functionally different characteristics than forests in forested landscapes. Rather than the repeated high-grading of forested landscapes over many cutting cycles, forests and woodlots in agricultural areas were largely on better sites (or they would not have been cleared) and utilized more intensely. In southern Ontario, these forests tend to have been cut more regularly, have been cut more heavily, have better growth than those in forested landscapes, and have even-aged structures.

Because of this history, there are very few old/undisturbed forests in southern Ontario. While many SW Ontario woodlands have large trees, they are 80 to 130 years and growing on good sites. Since the 1960s, the Province of Ontario has supported private-land forestry activities such as Advisory Services, tree marking and sale advertising, tree planting, enabling municipal tree-cutting bylaws, and, since 1995, tree marker training.

General Heartwood Observations

The heartwood at both ends of Mh butt logs and the site conditions where the logs came from were observed on the landings and harvest sites in SW Ontario over approximately 25 years of timber sale management. Heartwood observations made at the stump and on landings were organized into four general categories: normal large hearts, normal small hearts, balloon hearts, and streaks. Large hearts are areas of discoloration greater than 1/3 the log diameter at both ends and small hearts are less than 1/3 the log diameter at both ends. Balloon hearts are defined as areas where the butt end of the log had a small heart, but the top end had a much larger heart. Soil and site information was based on our experience with local landscape/physiography and soils, field observations, and provincial soils maps and physiography resources. Detailed soil examinations were not conducted.

RESULTS

The preceding discussion illustrates the complexity of soil and site conditions in the southwestern area because of the complex of geology, deposition by glaciers, and reworking of those deposits by wind and water. The higher site quality for Mh discussed earlier is important, and sometimes-subtle variations in the soil and landscape complex influence heartwood development.

Heartwood Observations

Larger hearts were observed predominantly on imperfectly to poorly drained sites with limited telluric (lateral) water movement, and soils with acidic (granite-based) parent material. The poorly drained sites were predominantly calcareous at depth, level, silty to clayey sites. The sites with acidic parent material were associated with the Canadian Shield with granite bedrock.

Normal, smaller hearts (i.e., less than 1/3 log diameter at both ends) were largely observed at sites with deeper soils with (1) good moisture-holding capability (e.g., fine sands to silt loams), or (2) coarser textures with hesitation layers (i.e., finer strata in or near the rooting zone, which are textural discontinuities that slow internal water drainage), or where there was moving, subsoil water seepage (telluric water movement, likely oxygenated), or (3) soils developed from calcareous parent materials. In general, small hearts were found on moist sites where there was windblown fine to very fine sand at varying depths from 2 to 10 feet over till, on floodplains, with aeolian deposits, and on loamy morainal sites where site quality was generally high. Interestingly, loggers reported that small hearts, similar to those in SW Ontario, were found in a few spots on the Canadian Shield. Comparing those harvest sites to a map of bedrock geology revealed that those areas happened to be where the bedrock had a significant marble component. Marble is metamorphosed limestone and soils developed with this mineral as parent material would tend to be calcareous.

Balloon hearts were observed on sites where trees had good growth and health but would suffer droughty conditions during some part of the growing season. Sites with balloon hearts included soils with drier gravelly deposits at depth; deep medium or coarse sands; well-drained, heavy clay sites; and some shallow to bedrock sites. Balloon hearts were also found on good sites with a broad range in soil texture, depth to bedrock, and drainage, not necessarily indicated by site quality (e.g., site index) or tree characteristics. In many cases, the trees were very healthy with very high standing-tree grades. The commonality was that trees with balloon hearts were observed on excellent sites that might have very droughty conditions at some point during the growing season, including (1) deep medium sands, (2) shallow, clayey textured soils over limestone bedrock, and (3) sites with deep gravelly subsoils. On these sites, the buyers or owner might expect a high proportion of veneer. The cutters may have also expected a higher veneer potential based on the small heart at the stump. However, some disappointment occurred at the landing when the tree was bucked up and the small-end heartwood size exceeded the 1/3 log diameter criteria.

Streaks or other wood discoloration associated with wounding (e.g., tapping, bark scrapes, or limb breakage) is well known. Convention suggests that damage can result in a column of discoloration that can run up and down the trunk. However, this discoloration seems to remain localized in trees in SW Ontario growing on deep soils with pH from 6.0 to 7.5, with reasonable drainage.

On high-quality sites with small hearts, discoloration associated with damage was compartmentalized within about 6 inches of the wounds (e.g., basal scars, tapping, top breakage). This resulted in higher proportions of high-grade sawlogs and veneer than expected by convention. Being able to recognize this tendency would have a clear positive effect on timber value and forest management profitability and benefit industry as well.

DISCUSSION AND CONCLUSIONS

Our findings revealed that an ideal site for providing soil moisture for Mh tree growth is one that is deep, stone free, and with a water table that fluctuates no higher than 3 feet from the soil surface; it consists of loamy soils with high silt and very fine sand contents (e.g., silt loams and very fine sandy loams). These factors are functionally incorporated into site index results. The study revealed that most site class 1 soils for Mh were deep, stone-free, well to moderately well drained loamy soils.

In SW Ontario, the complexity of site conditions (i.e., because of variable depths and overlapping of depositional layers) is very high over short distances. This complexity limits the reliability of general soil mapping units or other classification systems.

When related back to landform, our best Mh sites were found on aeolian (loess, or windblown silts; and very fine sands) covered calcareous glacial tills and lacustrine soils. The next best sites were found on deep loamy glacial tills and deep loamy lacustrine soils. Average sites for maple were found on moderately stony glacial tills (drumlins, ground moraines, end moraines), lacustrine clays and clay loams, most sites with imperfect (somewhat poorly drained) drainage, and sandy soils. Poor sites for maple included poorly drained soils, dry gravelly soils, shallow to bedrock sites, and acidic shallow to bedrock sites of the Canadian Shield.

Well-managed northern hardwood stands on good sites in southern Ontario can provide exceptionally high financial returns, largely supported by a high proportion of veneer in harvest volumes (Williams 2023). This demonstrates the value of thoughtful management on good sites. Higher base status (i.e., soils developed from calcareous parent material) soils and other excellent soil and site conditions have led to some larger Mh that are relatively young and have potential for high veneer yields on some sites. However, on some ostensibly good sites with Mh that have excellent growth and tree grade, the veneer yield is much lower than expected because of balloon and large hearts.

The knowledge that a site has good growth and quality is not the whole story. It is also important to be aware that the relationship between physiography/glacial deposition and soil characteristics can have significant effects on Mh growth and value. For example, it seems that sites can have a high site index and tree grade, but also have the internal "defect" of the balloon heart where there are periodic, severe moisture deficits at some point during the growing season. This can reduce the anticipated proportion of veneer in a harvest and be disappointing to buyers and landowners.

The retention of high-value individual trees until they are financially mature is a key to realizing the best returns for forest management. With consideration of financial maturity in tree marking, the veneer trees would reach financial maturity later than trees with only sawlog potential. There is a clear temptation for timber buyers to harvest veneer-potential trees as soon as they meet the grading criteria, perhaps leaving inferior but good quality individuals. Harvesting the veneer-potential trees prematurely significantly reduces long-term revenues and profitability of owning and harvesting timber.

The ability to identify site types that produce small and balloon hearts would help foresters manage for the highest value and help astute buyers to better estimate the proportion of veneer. Should these observations be validated, it may suggest a shorter rotation to

financial maturity for trees that may have a lower veneer potential and a longer rotation for the Mh with the potential on the truly superior sites.

Heart size is a primary criterion for higher grade Mh logs since veneer requires heart size to be less than 1/3 the log diameter at both ends. Recent research has found that Mh that were younger, growing more rapidly, growing on soils with pH up to 6.0, or assigned a higher tree grade were limited predictors of higher log quality and veneer potential (Dey et al. 2017). The unpredictability of heartwood size across site types is challenging and may have led veneer buyers in SW Ontario to abandon their efforts to buy standing timber directly. With local experience or sensitivity (or combination thereof) to the effects of soils and site on heartwood, veneer quality is likely to be more predictable.

Much previous research has found limited connection between heart size and site characteristics; however, none of the previous research mentioned heart characteristics described as balloon hearts. Perhaps our observations differ from previous research because the studies targeted less site-specific site characteristics, had less connection between the actual harvest and log yields, did not include a site/stump relationship in their research methods, or analyzed for the average heart diameter of both ends of the butt logs. For example, Germain et al. (2015) measured only dark heartwood on the stumps of harvested sugar maple in New York when comparing heart size with numerous site and landscape variables. In a study across several states in the northeastern and southeastern United States, heartwood was measured on both ends of logs in mill yards, but only landscape-level conditions were used in the analysis (Yanai et al. 2009). A study of high-quality trees in Wisconsin included more details at the site level in the findings but used average heart size for the assessment (Dey et al. 2017). In research to predict the size and shape of heartwood, Kaminski et al. (2019) sampled sites with different conditions but used the heart size averaged from both ends and collected soil samples only from the top 8 inches of the soil with no information on drainage or rooting space.

Sugar maple growing in SW Ontario on high-quality, neutral to higher pH sites can yield a significantly higher proportion of veneer and high-quality logs than reported from other areas. While the reduced heartwood formation described earlier was found on sites with calcareous parent materials, high-quality logs may potentially also be found on high-quality acidic sites.

The conundrum that balloon hearts were found on good sites which may present moisture deficits during the growing season is interesting. One suggestion is that a moisture deficit causes a break in the water column at some height up the trunk, and the tree interprets the "break" as injury, resulting in discoloration of the wood in the affected area.

Validating the balloon heart phenomena could provide forest managers guidance on management strategy and timing strategies for harvesting trees with varying financial maturity. An awareness of site conditions where balloon hearts may occur could help astute timber buyers avoid overspending on timber with the inciting site factors or foresters could adjust their management approach. Better information should be developed and incorporated into forest management strategies and training for foresters and tree markers. Such training should include the concepts of financial maturity of individual trees and the management of and for high-quality Mh on good sites. Awareness of the opportunities and concepts regarding financial maturity would be valuable to help landowners appreciate that their forests could rival or exceed other investments.

Ideas in this presentation should be considered in future research on Mh heartwood that gives more consideration to local/microsite conditions, soil parent material, characteristics of the deeper soil, and heartwood diameter at both ends of the logs. We will be more diligent and consistent in collecting detailed site and heartwood information in upcoming harvests.

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Social and Economic Aspects

U.S. Hardwood Lumber Exports: Changing Roles and Markets

Matthew S. Bumgardner and William G. Luppold[1]

Abstract

In the 1990s, increasing exports expanded the market for domestically produced hardwood lumber. Since the early 2000s, hardwood lumber exports have acted as a hedge against declining U.S. demand arising from the loss of domestic furniture manufacturing and then the Great Recession. In this paper, we discuss changes in hardwood lumber exports to major global regions and for important species at pivotal periods in time. While the oaks (*Quercus* spp.) continue to be the major U.S. export species by volume, walnut (*Juglans nigra*) has realized the largest increase in proportion of U.S. exports since 1990. Hardwood lumber exports peaked in 2017 and the combined China and Vietnam market (CHV) displaced North America (Canada and Mexico) as the predominant export region. U.S. hardwood lumber exports have declined since 2017 due to decreased shipments to CHV. Export markets are dynamic and periodic assessments are needed to better understand export trends given their importance to U.S. hardwood lumber production.

INTRODUCTION

Lumber exports can lead to increased profits for hardwood sawmills and serve as a hedge against changes in domestic economic conditions (Ifju and Bush 1993). Since 1990, exports have become an increasingly important part of the U.S. hardwood lumber market (Luppold and Bumgardner 2016). Exports increased in the early and mid-1990s, approaching 2.9 million cubic meters (cu. m) in 1997 (Fig. 1). During this period, exports accounted for 16 percent of the U.S. market for hardwood lumber graded for use in appearance applications. After 1997, exports trended upward, then declined between 2006 and 2009. In 2010, lumber exports started to rapidly increase, and in 2014, were 3.9 billion cu. m. In combination with relatively low levels of domestic consumption in 2014, exports then accounted for 37 percent of the U.S. appearance market. After a slight decline in 2015, exports approached 4.5 million cu. m in 2017, then declined to under 3.1 million cu. m in 2020.

Figure 1 shows the growth of U.S hardwood lumber exports over the last three decades but excludes information on global destinations and the volume of species exported. Export markets are dynamic given changes in regional economies and trade policies (Luppold and Bumgardner 2021). In addition, COVID-19 disruptions have impacted global supply chains. Given the importance of the export market to the hardwood industry, it is important to keep abreast of these trends. In this paper we assess changes in hardwood lumber exports to major global regions and important export species at pivotal periods in time. More extensive information, including value and imputed price analyses, can be found in Luppold and Bumgardner (2021).

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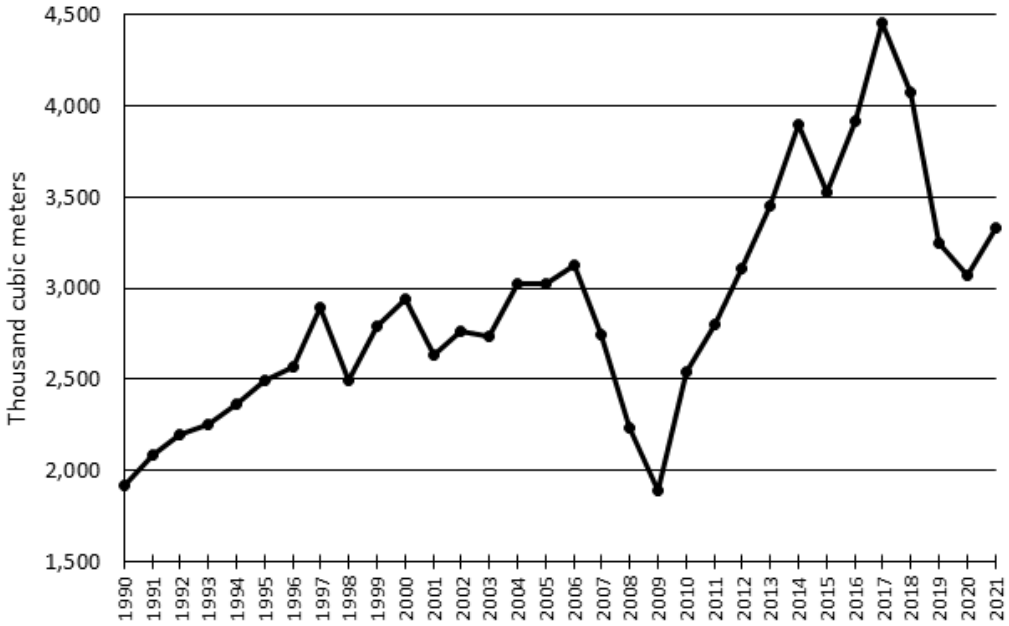

Figure 1.—Volume of U.S. hardwood lumber exports, 1990–2021.

METHODS

The data used in this study were developed using the U.S. Department of Agriculture (USDA), Foreign Agricultural Service (FAS), Global Agricultural Trade System (GATS) application (USDA FAS 2022). Figures for export volumes are presented in cubic meters. Percentages are used to show changes over time and regional market shares. In terms of conversion factors, 1,000 board feet (MBF) of lumber equals 2.36 cu. m (Jones 2016).

Because of errors in U.S. export data in the 1980s (Luppold 1995), 1990 was selected as the starting point for analysis. Exports grew by 50 percent between 1990 and 1997, fluctuated between 1997 and 2006, then decreased below 1990 levels in 2009 as a result of the global economic slowdown associated with the Great Recession (Fig. 1). Except for a relatively small downturn in 2015, the 2009 to 2017 period was one of unparalleled growth in hardwood lumber exports. This growth period was followed by a large decline from 2017 to 2020 with a relatively small uptick in 2021. This series of market fluctuations is analyzed by examining changes between five time periods: 1990 to 1997, 1997 to 2006, 2006 to 2009, 2009 to 2017, and 2017 to 2021.

Market Regions Examined

Since 1990, three global regions have received high volumes of U.S. hardwood lumber: Europe (the European Union and the United Kingdom), North America (Canada and Mexico), and East Asia. In the early 1990s, Japan, Taiwan, and South Korea (Korea) were the major East Asian importers of U.S. hardwood products. Improved trade relations with China in the early 2000s resulted in increased exports directly to this country or through Hong Kong intermediaries. Initially, a large portion of exports to China were to supply lumber to wood furniture manufacturers that migrated from Taiwan to China (Schuler and Buehlmann 2003). A trade dispute between China and the United States resulted in the substantial relocation of the bedroom furniture portion of the Chinese industry to

Vietnam (Luppold and Bumgardner 2011).

In this paper, Japan, Taiwan, and Korea (JTK) and China and Vietnam (CHV) are examined as two separate trading regions along with Europe and North America. Hong Kong is included in the CHV grouping because it acted as a trading intermediary for China. During the study period, the combined regions of Europe, North America, JTK, and CHV accounted for at least 90 percent of annual U.S. hardwood lumber export volumes (USDA FAS 2022).

Species Examined

Red oak and white oak (*Quercus* spp.), hard and soft maple (*Acer* spp.), and yellow-poplar (*Liriodendron tulipifera*) account for 65 percent of the hardwood sawtimber volume in the eastern United States (USDA Forest Service 2020) and have been exported in large volumes over the last 30 years (USDA FAS 2022). Three minor eastern species (by sawtimber volume) that also have been exported at relatively high levels are the ashes (*Fraxinus* spp.), cherry (*Prunus serotina*), and walnut (*Juglans nigra*) (USDA FAS 2022). Red alder (*Alnus rubra*), a species indigenous to the western United States, also was examined because it has been a major export species.

Between 1990 and 2006, maple exports were separated into three product categories at the 10-digit harmonized code level: hard maple, other maple, and dressed (surfaced and edged) maple. After 2006, maple was reported as hard maple and other categories. Because of these ambiguities, maple exports are reported in this paper as a single product at the six-digit code level (not separated by hard and soft maple). All species examined in this study (red oak, white oak, maple, yellow-poplar, ash, cherry, walnut, and red alder) are reported at the six-digit harmonized code level from the GATS database.

RESULTS

Regional Analysis

In 1990, Europe was the most important export market for U.S. hardwood lumber, receiving 673,000 cu. m (Fig. 2). North America and JTK also were major export markets, receiving 639,000 cu. m and 492,000 cu. m, respectively. The volumetric market shares in 1990 for Europe, North America, JTK, and CHV were 35, 33, 26, and 1 percent, respectively.

The increased volume of U.S. hardwood lumber exported between 1990 and 1997 was primarily due to increased exports to North America. Exports to CHV increased in the 1990s, and by 1997, were 161,000 cu. m (representing 6 percent of the volume that year).

Total hardwood lumber exports trended upward between 1997 and 2006 as a result of large increases in shipments to CHV and smaller increases to North America. Shipments to JTK declined during this period by 349,000 cu. m. European exports declined by 201,000 cu. m. In 2006, the relative market shares for Europe, North America, JTK, and CHV were 23, 39, 5, and 25 percent, respectively.

Lumber exports declined between 2006 and 2009 to all regions, but exports to CHV had the smallest percentage decline. Exports to JTK continued the decline that began in the late 1990s, with that region accounting for 62,000 cu. m (3 percent) of the volume of shipments in 2009. The volume of exports to CHV in 2009 was only 2,000 cu. m lower

than shipments to North America.

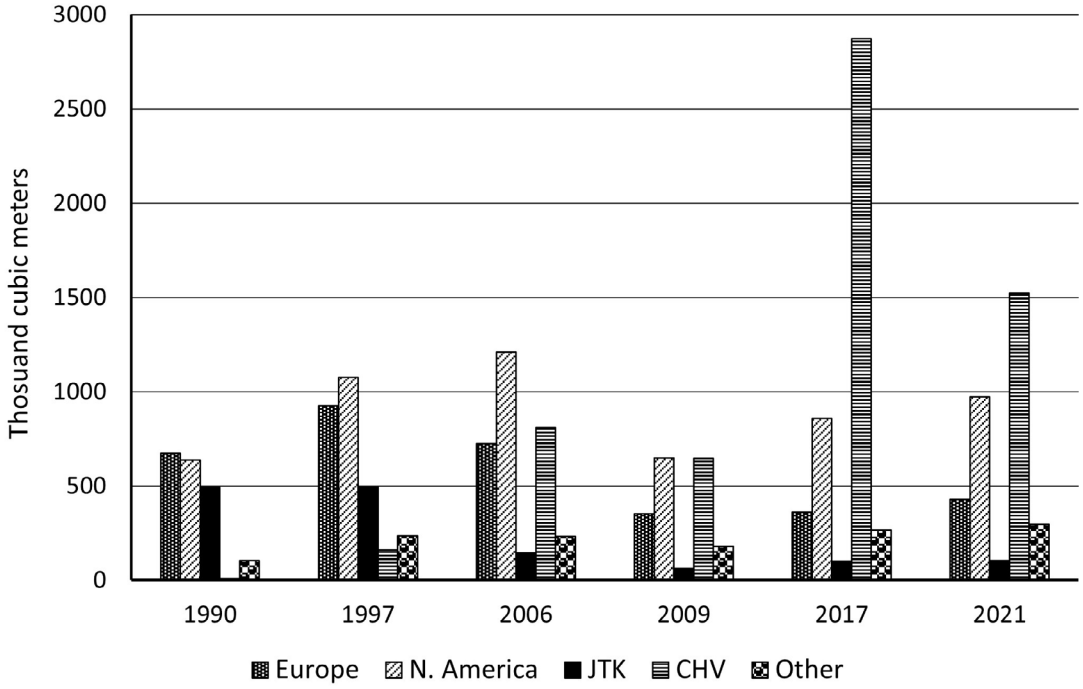

Figure 2.—Volume of U.S. hardwood lumber exported to Europe, North America, JTK region (Japan, Taiwan, and Korea), CHV region (China, Hong Kong, and Vietnam), and all other regions, selected years.

The increase in export volume between 2009 and 2017 was primarily the result of increased exports to CHV. In 2010, CHV displaced North America as the most important receiving region for U.S. exports (USDA FAS 2022). In 2017, CHV at 2.871 million cu. m accounted for 64 percent of the total volume of exports.

The overall decline in exports after 2017 primarily was the result of decreased exports to CHV, but this region still accounted for the plurality of U.S. hardwood lumber exports in 2021. However, the gap with North America closed considerably. The volumetric market shares for Europe, North America, JTK, and CHV in 2021 were 13, 29, 3, and 46 percent, respectively.

Species Analysis

In 1990, the volume of red oak and white oak lumber was 600,000 cu. m and 526,000 cu. m, respectively, and combined, accounted for 59 percent of total export volume (Table 1). Of the six other species identified in Table 1, ash, yellow-poplar, red alder, and maple had volumes approaching or exceeding 100,000 cu. m. The increase in exports between 1990 and 1997 was led by large increases in maple and red alder shipments combined with smaller increases in white oak and yellow-poplar. As a result, combined red oak and white oak market share of total export volume declined to 45 percent in 1997. Red alder exports peaked at 303,000 cu. m in 1997 (USDA FAS 2022).

Between 1997 and 2006, red oak and white oak exports declined as shipments of all other eastern species increased (Table 1). In 2006, combined red and white oak shipments were 1.090 million cu. m (35 percent of the total) while combined maple, yellow-poplar, ash, cherry, and walnut shipments were 1.233 million cu. m. The decline in exports between 2006 and 2009 occurred across all species, but the oaks had relatively smaller declines.

The growth of hardwood exports after 2009 occurred across all species, with the largest volumetric increase occurring in red oak and yellow-poplar (Table 1). The highest relative increases between 2009 and 2017 were for cherry, red oak, ash, and walnut with increases of 340 percent, 270 percent, 254 percent, and 232 percent, respectively.

Exports of all species declined between 2017 and 2021 with the exception of maple and walnut. While walnut accounts for less than 1 percent of the U.S. sawtimber resource (USDA Forest Service 2020), it was the fourth most important lumber export species by volume in 2021. The increase in maple was associated mostly with the increasing importance of the North American export market. In summary, over the study period, all species realized increases in volumes exported, ranging from 33 percent for white oak to 1,064 percent for walnut (Table 1).

In 2021, CHV held at least a 50 percent market share for every U.S. hardwood export except maple and white oak (Fig. 3). Most U.S. maple exports went to North America (70 percent). White oak was the least concentrated export species, having sizable market shares in CHV (34 percent), Europe (29 percent), and North America (17 percent).

Table 1.—Volume of U.S. hardwood lumber exports of important species and all species, selected years

Species	1990 volume (thousand cubic meters)	1997 volume (thousand cubic meters)	2006 volume (thousand cubic meters)	2009 volume (thousand cubic meters)	2017 volume (thousand cubic meters)	2021 volume (thousand cubic meters)	Percentage change in volume 1990–2021
Red oak	600	649	487	335	1,239	804	34
White oak	526	665	603	370	715	699	33
Maple	98	391	404	162	177	201	105
Yellow-poplar	107	204	388	326	757	369	245
Ash	140	152	180	124	439	228	63
Cherry	47	112	165	42	185	141	200
Walnut	22	20	96	65	216	256	1,064
Red alder	103	303	259	133	221	167	62
All species [a]	1,918	2,893	3,123	1,890	4,458	3,327	73

[a] Includes listed species plus birch, beech, hickory, tropical, and other species.

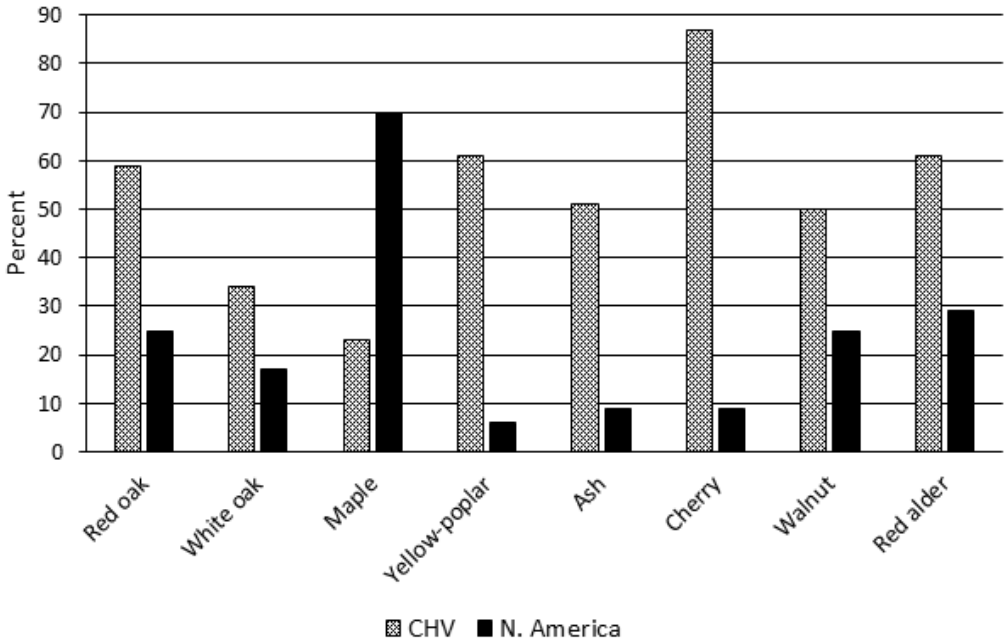

Figure 3.—China, Hong Kong, and Vietnam (CHV) and North America (Canada and Mexico) market shares of worldwide U.S. hardwood lumber exports by species, 2021.

CONCLUSIONS

Exports account for a relatively large proportion of the total demand for appearance-graded hardwood lumber produced in the United States. The increasing importance of export markets has coincided with several other changes in the hardwood market. The loss of the domestic furniture industry and reduced consumption of hardwood lumber used in construction markets since the Great Recession contributed to a decline in total demand for higher quality hardwoods. Without exports acting as a hedge, the decline in hardwood lumber production could have been larger.

Export markets have shifted since 1990, with notable increases in the importance of walnut as an export species and CHV as an export destination. However, by 2021, the gap between North America and CHV as U.S. hardwood lumber-receiving regions was again narrowing. This narrowing reflected, in part, a trend toward shortening supply chains given COVID-19 disruptions and other global uncertainties. An implication of this shift has been that, since 2017, maple has performed better than most species given its importance to the Canadian export market. The oaks remain the most important U.S. export species by volume, but they have lost market share since 1990.

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West Virginia Family Forest Owner Satisfaction with Timber Transactions

Jeffrey M. Lee, David W. McGill, Joseph F. McNeel, and Kathryn Arano Gazal[1]

Abstract

No two timber transactions are the same, due to variables such as site conditions, land uses, and landowner goals. For some landowners, selling timber will be very successful, while others may experience frustration, disappointment, and stress. The intent of this research was to study the perception of landowners about the timber harvesting operations, business relationships, and financial exchanges, and their degree of satisfaction throughout the entire timber transaction. We carried out a mail-based survey to landowners who recently harvested timber and evaluated the open-ended questions from the survey.

In general, respondents indicated that at the outset of the timber transaction, they were concerned mostly with the future condition of their property, financial benefits, and the various people who might be involved. Questions related to what was "best," what was "worst," financial concerns, and landowner suggestions about what they might do differently are also discussed.

INTRODUCTION

Forest industries in the eastern United States rely heavily on family forest owners to supply fiber needs for their mills. An estimated 82 percent of all timber harvests in the eastern United States, or 12.7 billion board feet, was removed from private lands in 2022. The remaining 18 percent comprises 9 percent (1.4 billion board feet) that was removed from combined Federal, State, and local government-owned lands and 9 percent that is considered as unavailable or unknown ownership (USDA Forest Service, Forest Inventory and Analysis Program 2022).

Private forests can be categorized into three groups based on ownership type: corporate (private corporate), family, and other private. Corporate forest owners are part of the forest products sector and include timber investment management organizations and real estate investment trusts (Butler and Wear 2013). Noncorporate or nonindustrial private forest owners are more commonly labeled family forest owners (FFOs) (Butler and Leatherberry 2004). FFOs include families, individuals, trusts, estates, family partnerships, and other unincorporated groups or individuals that own at least 1 acre of forest land that is 10 percent stocked. West Virginia FFOs are the focal point of this study and own more than half (52 percent) of the timberland in the State (West Virginia Division of Forestry 2020).

FFOs face important land management decisions that have long-term impacts on their properties. Some of these decisions are whether to harvest timber, lease their property, or

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sell part or all of their property for added income (Ma and Kittrridge 2011). To complicate matters, when an FFO decides to harvest timber, other parties become involved in the future conditions of the property. The timber sale may include loggers, consulting foresters, procurement foresters, state foresters, and others (Bowers and Punches 2007, Grotta 2014, Hoover and Seifert 2002, Rickenbach et al. 2018). All these parties have a role to play in the success of the sale and future of the land.

An example of the complexity of a timber transaction in West Virginia is illustrated in Figure 1, a graphical model that the authors have developed over the years from other research (Maltempie 2017; McGill et al. 2004, 2006) and direct contacts with foresters, loggers, and landowners. The figure lays out the process of a four-party timber transaction and serves as an educational tool in outreach activities to give a visual description and example of the situations that might arise during these complex transactions. This four-party scenario includes the FFO (landowner), a professional forester, a logging company, and the West Virginia Division of Forestry (WVDOF). The first phase is conceptualization, where some condition (e.g., medical bills, tuition, new roof) or stimulus (e.g., a "knock at the door") causes the FFO to decide to sell timber. The next phase is timber sale preparation, where contact is made with a professional forester who will help the FFO plan and start the sale by writing up a sale prospectus or a sale contract. Once the timber is sold, in this case to the logger, the WVDOF must be notified of the planned harvest within 3 days of starting the timbering operations. There are many different scenarios and parties that may be involved in timber transactions. Each timber transaction is distinct and unique and there are often no prior certainties about the "success" of the transaction.

Regardless of the type of industry in question, measuring customer satisfaction is an important part of improving quality of work and retaining customers for future business transactions (Cengiz 2010). In the case of the forest products industry, customer satisfaction experienced during timber transactions not only involves a simple exchange of money for logs but may involve perceptions and emotions around considerable logistical operations on private properties. These properties are owned by people who are often uninformed or misinformed about the complexities of these transactions.

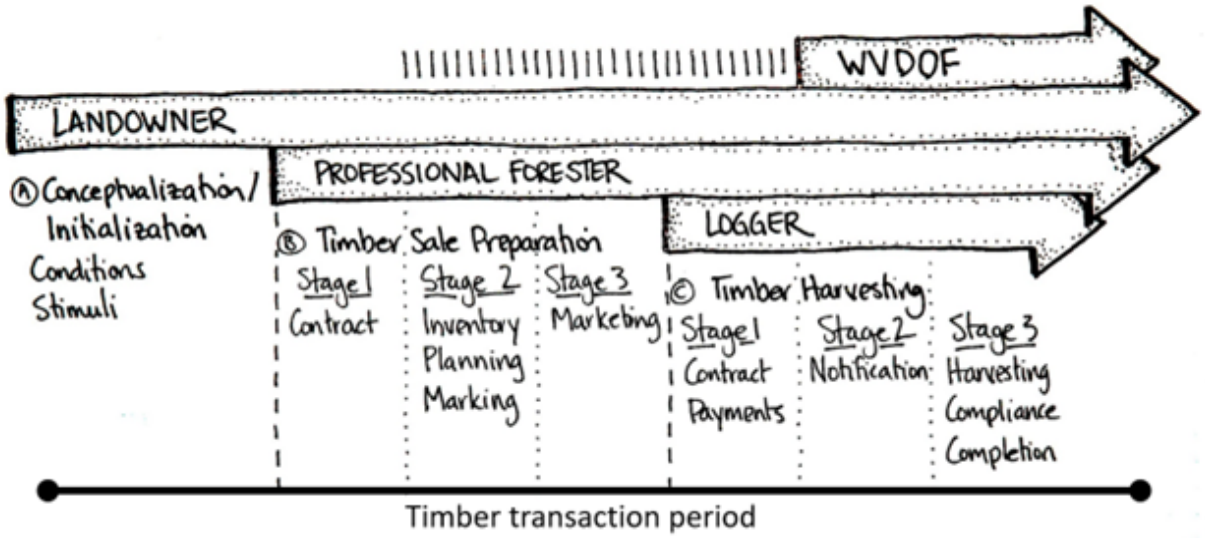

Figure 1.—Example of a four-party scenario of a timber transaction process. WVDOF = West Virginia Division of Forestry. Graphic by D. McGill.

Issues related to hiring a forester, the landowner's previous knowledge of what occurs during a harvest, and trusting the parties that are involved in the harvest can all affect the outcome of a harvest. Hiring a forester or being prepared for the sale could positively impact a timber sale, whereas not trusting the forester or the logging company might negatively impact the outcome of the timber sale (Maltempie 2017). Literature from many organizations and government agencies is available to guide the landowner to successful timber transactions (e.g., Bowers and Punches 2007, Grotta 2014, Hoover and Seifert 2002, Rickenbach et al. 2018). Despite these efforts, unsatisfactory timber sales still occur. Research on landowner satisfaction with timber transactions is limited but substantiates the claim that while most landowners are not averse to future transactions given recent sales, some have had bad experiences (Karha 2004, Maltempie 2017).

The objective of this paper is to examine the responses of FFOs to open-ended questions from Lee (2020) and to tease out satisfying and unsatisfying aspects of their timber sales. The intent of this effort is to benefit professionals who administer sales or carry out operations; this work may also benefit landowners interested in learning more about these complex transactions.

METHODS

Open-ended questions from a survey of West Virginia FFOs (Lee 2020) formed the basis of this paper. The survey was designed to explore how overall FFO satisfaction is influenced by events and conditions that occur during a timber sale transaction. A mailed questionnaire provided a sample of FFO experiences with timber sales by eliciting responses related to elements of timber sales. Six sections within the questionnaire aimed to elicit experiences regarding the property management, planning, operations, and post-harvest results, as well as demographics. The open-ended questions were included in the questionnaire to give respondents the chance to express specific reasons why they were or were not satisfied with these various elements of timber transactions.

A list of FFOs that recently harvested timber from their properties was obtained from the West Virginia Logging Operations Notification Inspection and Enforcement system (LONIE). The system is maintained by the West Virginia University Appalachian Hardwoods Center and the WVDOF (Spong 2013, WVDOF 2022). LONIE was developed as a tool for the WVDOF to comply with the West Virginia Logging Sediment Control Act (LSCA) by creating a registry of logging operations and inspection findings. Logging operators are required to submit a notification on LONIE within 3 days of beginning a harvest. The LONIE database includes data such as harvesting location, parties involved, logger's license number, client contact information, and when and where the harvest would occur (Spong 2013).

FFOs selected for this study were obtained from timber notification forms submitted to LONIE between November 2012 and November 2019. Landowner entries were filtered to remove non-FFOs. Duplicates were removed if the FFOs owned multiple properties or had performed multiple harvests within the time period of interest. Only individual FFOs were included in our mailings. We did not use any entry in the LONIE database that listed keywords such as trust, company, corporation, limited, or limited liability company. The LONIE file contained entries that went back as far as 2012. The entries were sorted by end date of the harvest to eliminate harvests that had not been completed. From the list of

FFOs, 1,349 were selected to conform to our desired sampling intensity for the quantitative portion (Lee 2020) of this survey based on Krejcie and Morgan (1970) estimates.

Lee (2020) described the questionnaire and the sections related to the elements of a timber sale. These sections were based on peer-reviewed and forestry extension-based technical publications. The questionnaire also included a set of open-ended questions to get more details on why respondents were satisfied or dissatisfied with the transaction. Open-ended questions were used in the questionnaire to elicit responses from FFOs that may have otherwise not been revealed within the more quantitative portion of the questionnaire. FFO responses were coded based on themes or patterns that emerged from each question. Emergent categories were determined by identifying themes or issues that recurred within the data. Categories were then defined after the data had been read and coded by two collaborating coders (Taylor-Powell and Renner 2003). Importantly, some respondents could have two or three themes in a single open-ended answer, which can cause the numbers observed in theme subcategories to be greater than the number of respondents.

RESULTS

In total, 1,349 questionnaires were sent to West Virginia FFOs with 257 completing the questionnaire. This resulted in a 19 percent response rate. The cooperation rate—the number of questionnaires filled out divided by the number returned (which includes some refusals and blanks)—was 75 percent (American Association for Public Opinion Research 2016). Ninety-four questionnaires were returned blank, 177 questionnaires were returned due to bad addresses, and 34 were deemed ineligible either because they were not an FFO or the FFO was deceased (Wiseman 2003).

The majority of FFOs (61 percent) described this as their first harvest. Eighty percent of the respondents were male. The largest age group was 65 to 74 (35 percent) and, in the most often reported education category, 35 percent had completed high school. Forty-eight percent lived on their property. Seventy-six percent did not lease any part of their property.

Guidance Prior to the Sale

Five themed categories resulted from 125 FFOs (48 percent) answering the question "Prior to deciding to sell timber, did you receive guidance or information on the best way to harvest timber? If yes, from where?" These five themes were foresters (39 percent), organizations (21 percent), other timber professionals (15 percent), family and friends (15 percent), and personal knowledge and research (11 percent) (Table 1). In the largest category labeled forester, most respondents listed a specific forester or just the job title "forester" (39 percent). There was no attempt in the questionnaire to guide responding landowners to detail the type of forester, whether governmental "county" forester, procurement forester, or consulting forester. The majority in this subcategory (65 percent) used general terms for forester like "independent," "consulting," "professional," "my forester," or just "forester." Others used their foresters' real names (14 percent), their organizations (12 percent), or both names and organizations (9 percent).

Twenty-one percent of the respondents listed guidance from governmental agencies (e.g., WVDOF, West Virginia Department of Natural Resources) or private organizations (e.g., "the logging company," "timber company") (Table 1).

Table 1.—Sources of prior guidance and information on the best way to harvest timber

Guidance from…	n	Percentage
Foresters	51	39
Organizations	28	21
Other professionals	19	15
Family/friends	19	15
Personal knowledge/research	14	11
Total	**131**	**100** [a]

[a]Total does not add to 100 because of rounding.

Those who listed other timber professionals as their sources of guidance (15 percent) used terms like "timber agent," "individual who had sold timber," "NRCS forester," and "person knowledgeable in timber sales" (note: NRCS is the USDA Natural Resources Conservation Service). Family and friends provided guidance and information to another 15 percent of respondents. Surprisingly, nearly a third (32 percent) of these landowners stated their friends and family associates had some forestry experience and some were foresters. Finally, approximately 1 in 10 respondents listed having prior knowledge or having done research about the best way to sell timber.

Main Concerns During Decision to Sell

Nearly 9 out of 10 respondents (89 percent) answered the key question "When you were deciding to sell timber, what were your main concerns?" Concerns were coded into three primary areas pertaining to: (1) condition of the property, (2) financial values, and (3) people involved (Table 2). A smaller "other" category comprised responses not closely related to any of the three primary categories and included the "no concerns" responses. Most were concerned with the condition of the property during or after the transaction (61 percent), while financial value (20 percent) and concerns about the people involved in the transaction (15 percent) were less frequently cited.

Just under half (47 percent) of those selecting "condition of property" as their main concern stated something about the "general condition" or residual stand characteristics, including aesthetics and best management practices (BMPs) (Table 2). A closely related subcategory was made up of responses that explicitly stated "damage" as part of their response. Twenty-eight percent of those who were concerned about the condition of their property specified damage as a main concern. The remaining subcategories each accounted for less than 8 percent of the responses but included concerns about the mess (trash and slash) that might be left, access to their property (or lack thereof due to damaged roads), and other topics—for example, concern that the logger might not take out all stumps when the landowner was interested in converting the forest to pasture.

Financial concerns were stated in 20 percent of the responses. Three-quarters (76 percent) of the respondents listing this concern were simply interested in getting the best financial deal from their sale of timber. General market conditions and marketing methods were put forward as concerns by 16 percent of these respondents. Costs for the transaction and taxes associated with the timber sale revenue were listed as financial concerns by 7 percent.

Table 2.—Respondents' "main concerns" prior to harvesting their timber

Concern and subcategories	n	Subcategory percentage
Condition of property (61%)		
General condition—residual stand, aesthetics, best management practices	82	47
Damage to property, timber	49	28
Trash and slash	12	7
Access	11	6
Other—dead ash, clearing for pasture, wildlife	10	6
Erosion and ruts	9	5
Subtotal	173	
Financial value (20%)		
Best price	42	76
Other (e.g., marketing method, market conditions)	9	16
Costs, taxes	4	7
Subtotal	55	
People (15%)		
Control (competence, doing it right, reputation)	19	41
General associates (e.g., foresters, timber buyers)	12	26
Loggers	10	22
Family and neighbors	3	7
Liability	2	4
Subtotal	46	
Other (4%) Note: No subcategories were coded for this concern group		

Respondents also specified people associated with the timber sale in 15 percent of the concern statements. Subcategories related to concerns about people included mentions of the general associates (26 percent), loggers (22 percent), and family and neighbors (7 percent). The single largest subcategory, however, was labeled "control" (41 percent) (Table 2); statements were labeled as such when landowners declared their concern about the reputation, competence, and forthrightness of the associates involved with the timber sale.

Best Things About the Timber Transaction

Apart from concerns related to their most recent timber transaction, respondents shared their impressions about the "best things" that had occurred in their timbering experiences. Eighty-four percent of the respondents answered this question, including 4 percent who claimed that there was nothing good about the transaction. On a more positive note, over half (58 percent) claimed either a financial (31 percent) or a forest productivity (27 percent) aspect as the best thing about their most recent transaction. The financial benefits were expressed in two ways. The first was simply "the money" (87 percent). The second

had to do with the way the transaction was structured. A few examples of these were "the company paid us in advance of the work," "payments to me were correct, documented and on time," and "they gave me an estimate of what I would get and gave me part of the money and when that was gone I would get a check each week till the harvest was done." The forest productivity category contained other subthemes including improved woods (62 percent), wildlife habitat (15 percent), management planning (9 percent), a general improved condition of the property (8 percent), and forest and wildlife together (6 percent).

Financial and forest productivity themes were followed by statements appreciating the trust that respondents had in their professionals—the "company," foresters, and loggers—who were associated with the transactions (18 percent). Three smaller categories were: (1) an "other" category (8 percent) with respondents who were surprised or thankful that the operations were simply just finished or had been completed very rapidly, 2) an access category (7 percent) listed by landowners grateful to have improved access to their property because of the logging operations, and (3) a land conversion category (6 percent) filled with claims about how new pastures or cleared land were great benefits of the transaction.

Worst Things About the Timber Transaction

As mentioned, only 4 percent of the respondents answered that there were no "best things" about their timber transactions, but on the subsequent question about the "worst things," 23 percent answered that there were no "worst things"; that is, they generally approved of or felt comfortable with all the activities they experienced during their timber transaction. Other responses contained five "worst things" themes: condition of property (42 percent), communications/contracts (13 percent), financial (9 percent), operations (7 percent), and other (7 percent).

The worst attributes of timber transactions related to the condition of respondents' properties following the timber sale and mirrored concerns they had prior to the timber sale. Conditions that were unsatisfactory to the respondents in this category ranged from damaged residual trees, to high and difficult-to-drive-over water bars, to "destruction of the beauty and the mess of treetops everywhere . . . so disheartening." Financial complaints were almost exclusively about revenue received that was less than expected or desired.

Frustration with operations were also diverse, with responses related to topics from the timing and completion of the operations to equipment failures, accidents, and abandonment. The "other" category was a catch-all for disappointing realizations, such as "I look at it now and wonder if I should have waited," "the rain and mud," and "worrying about bad things that never happened."

Financial Concerns

The questionnaire also specifically addressed financial concerns. FFOs were asked whether they had been concerned about any financial aspect "during the entire timber sale." This question was followed up by "If so, when and why were you concerned? If not, why not?" Most respondents (73 percent) did not have any financial concerns. Only one in four respondents (25 percent) stated concerns and 3 percent gave answers that were unclear as to whether it was a concern or not. Most FFOs that were not concerned gave no specific

reason (31 percent). However, the reasons from respondents who did give reasons fell into one of four categories: (1) associates, (2) perceived value, contract, or self-confidence/experience (Table 3).

Equal percentages (27 percent) of respondents lacking financial concerns had statements coded into the two top categories: associates, who were the people directly involved in the timber transaction in one way or another, and perceived value, which was related to both tangible and intangible values that had accrued to the responding landowners. Eighty-one percent listed one of three associates subcategories—loggers (46 percent), general log buyers and companies (23 percent), and foresters (13 percent)—as the main reason they had no financial concerns. They conveyed their confidence in or the professionalism shown by associates in statements like "talked with someone who used this logging company they were very satisfied," "everything went well . . . loggers were friendly, hardworking, and paid as the contract stated," and "we felt we were dealing with fair people and the final payments were close to the estimated amount of timber they said could be taken." The fourth subcategory—familiarity with associates/process—was a general category that included the remaining 19 percent of the responses in the associates category. These responses suggested the landowners answering the question had past experiences with timber harvesting on their properties and knew the associates involved or had researched transactions in depth to identify and contact those who could help them in a way to satisfy their goals.

Table 3.—Reasons why respondents had no financial concerns during the entire timber sale

Category (label and percentage)	n	%
Associates (27 percent)		
Logger	22	46
General log buyers/companies	11	23
Familiarity with associates/process	9	19
Forester	6	13
Subcategory total:	48	100
Perceived value (27 percent)		
Reasons other than income	19	40
Upfront payments	18	38
Good revenue	8	17
All aspects worked out	3	6
Subcategory total:	48	100
Contract (9 percent)	16	100
Self-confidence/experience (7 percent)	12	100
Total	**181**	**n/a**

In the category of perceived value, four subcategories were identified: (1) reasons other than income, (2) upfront payments, (3) good revenue, and (4) all aspects worked out (Table 3). Among those in the "reasons other than income" subcategory (40 percent), respondents stated that income was not their primary reason to sell timber. These respondents were more interested in accomplishing management goals. These goals included salvage logging, creating wildlife habitat, improving their timber stand, and creating new or better pasture or tillable lands. A nearly equal number of respondents were content with the upfront payments they received (38 percent) and another 17 percent thought the revenue they received was good. In fact, a couple of respondents made statements like "received $28,000 more than expected" and "considered pennies from heaven." The fourth subcategory— "all aspects worked out"—combined reflections of money, communications, and expectations. For example, one respondent stated that "we agreed upon all fees and were kept in the loop of selling of the timber."

Two other categories were less frequent (Table 3). Having a contract assuaged the concerns of 9 percent of the respondents as they reflected: "[T]he contract was very specific so no problems were encountered" and "I had a contract for agreed amount of final sale [and] also a contract that I found to protect my interests." The last category in this group of landowners—self-confidence/experience (7 percent)—comprised those stating they had no financial concerns due to past experiences with or training and education related to timber harvesting. A word cloud is shown in Figure 2 that helps interpret some of the most commonly used words in describing landowners' lack of financial concern. Terms like "paid," "contract," "good," "money," "company," "logger," and "forester" give a multidimensional view of factors related to landowners' financial comfort.

Figure 2.—Word cloud showing terms used most often to describe why respondents were not concerned about financial aspect during the entire timber sale. Size of font relates to word usage frequency (larger = more frequent). Word cloud set to use 75 percent of words in text file. Word cloud source: FreeWordCloudSource.com (2022).

Five themed categories emerged from the respondents who indicated that they did have a financial concern about their timber transaction. These five categories were related to payments (56 percent), incidental costs (15 percent), general concerns (13 percent), legal issues (8 percent), and lack of experience (8 percent).

Just over half (56 percent) had financial concerns about the payments. Thirty-six percent of these landowners had uncertainty or suspicion of whether the wood being removed from their property was being accounted for or paid for as agreed upon (or both). Some even used strategies to help monitor the removal of timber, as one respondent "was concerned about being paid for every load [and] utilized camera on road." Others seemed disheartened by not knowing how to monitor the trees cut from their property. As a few landowners claimed, "we were not aware of how or when a load was removed or when they were taken," and "through the whole process, I saw trucks leaving but no money coming in." Twenty-nine percent of these landowners were simply concerned about maximizing harvested timber value. Another one in four (25 percent) landowners concerned about payments had unfulfilled expectations. Most of these made statements similar to the one made by a landowner who described his concern as the "actual offered prices were somewhat lower than the estimated price." The least frequently cited subcategory—missing payments—consisted of landowners who believed they had evidence of logs taken, but not paid for, like the landowner who claimed, "after I was told what the logs would bring, I found out from the mill a different price and I also was told roughly how many loads there were and found out it was double and that I should have received more money."

If You Could Do It Again

Finally, landowners were asked, "If you could do it again, how would you do it to make it more successful?" A plurality of respondents stated they would not change anything about the sale (30 percent; Table 4). Another 12 percent either had no idea what they might do differently (5 percent) or would not or could not have another timber sale for one reason or another (7 percent). Suggestions from the remaining 58 percent included better contract/planning (16 percent), better people/companies (14 percent), better work/operations (11 percent), and better monitoring (9 percent).

Table 4.—Responses to question of how landowners would make their next timber harvest more successful

Next time…	N	%
Same way	60	30
Better contract/planning	32	16
Better people/companies	27	14
Better work/operations	22	11
Better monitoring	18	9
Better prices/income	16	8
Would not or cannot	13	7
Not sure/don't know	9	5
Total	197	100

DISCUSSION AND CONCLUSIONS

This evaluation of landowners' impressions of timber being sold and harvested from their properties corroborates anecdotal information found in forestry extension literature designed to inform people how to successfully sell timber (e.g., Bowers and Punches 2007, Grotta 2014, Hoover and Seifert 2002, Rickenbach et al. 2018). Some of the primary findings here help shed light on factors that have led to satisfaction with various parts of timber transactions. Other landowners' written testimonials have brought out points of contention and disappointment with conditions and the associates participating in timber transactions. All this feedback may help lead to improvements for future transactions.

Prior to harvesting timber, landowners from this study were primarily concerned with the condition of their property following the sale. Some were concerned with the financial aspect and the various people involved, but most stated concerns about the physical aspects of their property. They were concerned with damage to roads and residual trees, reduced aesthetic appeal, restricted access, and trash and slash that might be left. Despite these preharvesting concerns, most had no or little financial concern before or during the transaction. Still, nearly a third of the respondents listed financial benefits among the "best" things about the timber sale. Importantly, nearly an equal number of respondents stated that "improved forest productivity/habitat" was the best thing about the sale.

Maltempie (2017) highlighted themes common among a diverse group of FFOs that recently had timber transactions. Factors that had important consequences in successful transactions included having a forester, previous knowledge of what occurs during a harvest, and trusting the parties that were involved in the harvest. These themes were also found in this broader current study, brought out in the testimonials of responding landowners.

Ideas about what might work best to improve timber transactions were reflected in the FFO testimonials in this study. Given that every timber transaction is different, it is not possible to have a one-size-fits-all approach to guidance on successful timber transactions. However, our study corroborates extension literature that attempts to give guidance to FFOs about certain activities that occur during timber sales and that need to be managed during the entire transaction process. Extension literature recommends that FFOs be clear on their property restrictions, visit the site and communicate with parties who are involved, create a written contract with involved parties, and hire a forester, or at least communicate with other professionals for guidance (Bowers and Punches 2007, Grotta 2014, Grushecky et al. 2012, Heiligmann and Bratkovich 2002).

The final question about how timber sales might be improved takes a secondary look at what was not satisfactory. Apart from the 30 percent who said they would do nothing different, at the top of the list of suggestions is better contracts/planning. In these suggestions were statements demonstrating the lack of control that landowners felt during timbering operations as they specified items they would include in a future contract: "set expectations and hold the logger accountable, hold a stronger performance bond," "demand replanting of cutting areas," and "learn more about it, better contract and watch more carefully." Landowners do not like the sense of not having control of what happens on their property (McGill et al. 2008).

Again, no two timber transactions are the same. The goal of this research was to illuminate important aspects of successful and unsuccessful timber transactions. The landowners who answered these open-ended questions about who helped them, what was best, what was worst, and what they would do if they could do it again lived through one of these complex experiences. Their participation has resulted in direct and generalized testimonials that we hope will help both professionals and landowners to understand how future timber transactions might be more successful.

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Herbaceous Component

Deer Browsing Effects on Herbaceous Layer Diversity Following Shelterwood Harvests

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Abstract

Shelterwood harvests are a common silvicultural treatment to promote tree regeneration prior to final removal harvest. Although it is known that white-tailed deer (*Odocoileus virginianus*) are among the principal forces controlling tree establishment, growth, and composition early in succession in some locations, the degree to which browsing affects herbaceous-layer plant communities following a shelterwood preparatory cut is not known. This study explored how deer browsing affects herbaceous layer richness and abundance by manipulating deer access using fenced exclosures and unfenced control treatments at seven northern hardwood stands in Pennsylvania following preparatory cuts. Vegetation surveys showed that browsing reduced sapling recruitment by 43 percent. Browsing also reduced forb cover, but it did not affect the overall species richness, cover, or diversity of the herbaceous layer. The surveys also indicated that excluding deer indirectly limits forb richness due to competition from the dense sapling layer. Our results suggest maintaining moderate browse pressure, rather than eliminating it, may be more appropriate to sustain species-rich understory layers.

INTRODUCTION

Sustainable forest management requires a balance among economic timber and pulpwood values, conservation values, and ecological services provided by forest ecosystems (Nyland et al. 2016). Critical among these values is the diversity represented by the understory plant layer. This stratum typically holds an overwhelming proportion (80 percent) of plant species richness in temperate forests (Spicer et al. 2020). The impacts of forest harvesting on this stratum have long been debated (e.g., Duffy and Meier 1992, Roberts and Gilliam 2003). Recent reviews have noted that although effects can be negative (e.g., a decline in plant diversity post-clearcut in late successional forests) or positive (e.g., an increase in plant species richness), any effects are largely dependent on the severity of the harvest and, perhaps more importantly, the stage of succession during which monitoring occurs (Chaudhary et al. 2016, Duguid and Ashton 2013). This temporal consideration is critical because the early successional plant communities that develop following disturbance are often very diverse yet dynamic, and the forces that shape plant biodiversity in these initial years can leave a lasting legacy on forest structure and diversity (Nuttle et al. 2014).

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Among the principal forces shaping vegetation compositional patterns early in stand development after harvesting is ungulate herbivory. Ungulates, including white-tailed deer (*Odocoileus virginianus*), often preferentially forage in early successional communities during the spring and summer because plant biomass in these habitats is typically abundant, short statured, and nutritious (Johnson et al. 1995). These browsers are known to influence species composition, plant community structure, and productivity through their selective foraging of preferred species (Bernes et al. 2018, Côté et al. 2004). In early successional communities, deer often browse preferentially on more palatable shrubs and shade-intolerant trees, thereby delaying recruitment of trees into taller height strata (Bradshaw and Waller 2016). This browse-induced delay of transitioning to stem exclusion maintains higher understory light levels, effectively extending conditions favorable for a diverse understory plant community (DiTommaso et al. 2014, Faison et al. 2016, Parker et al. 2020, Royo et al. 2010a). Despite a generally positive effect of browsing on diversity, browsing may also pose a risk to herbaceous layer species, particularly those considered more sensitive to browsing such as painted trillium (*Trillium undulatum*), sessile bellwort (*Uvularia sessilifolia*), and Indian cucumber root (*Medeola virginiana*) (Huebner et al. 2010), and thereby negatively affect plant species diversity overall.

In a system that is heavily influenced by herbivory, the role that deer play in species reassembly following a disturbance remains unclear within northern hardwood forests. In this study, we asked to what degree deer browsing in operational shelterwood harvests affects the herbaceous layer. The shelterwood method is an even-aged system that removes the mature forest in two or more successive cuttings, leaving some mature trees (20 to 60 percent basal area removal) as a seed source and protecting the regeneration (Nyland et al. 2016). These less intensive harvests allow a diverse regeneration cohort to establish before the final overstory removal harvest. The shelterwood method is widely used in northern hardwood and oak communities throughout the deciduous forests of eastern North America (Hannah 1988). We predicted that excluding deer would cause greater richness, abundance, and diversity of herbaceous species and browse-sensitive taxa in particular. However, if tree succession is hindered by browsing, we alternatively predicted that by delaying succession (i.e., stand entering stem exclusion stage), deer browsing may promote greater herbaceous richness.

METHODS

Study Area

We conducted our study at seven northern hardwood forest sites distributed throughout a 6,500 sq. km area of northern Pennsylvania. Study sites have a mean elevation of 581 m; a humid, temperate climate with average daily temperatures of 9 °C; and average annual precipitation of 1067 mm (Bailey et al. 2005). Fecal pellet surveys at these sites conducted between 2013 and 2016 found average deer densities of 6.3 ± 0.3 deer per sq. km (levels that are moderate given historical trends in the region; Royo and Stout 2019). At all sites, managers conducted the initial cut (i.e., the seed cut) of a shelterwood sequence between 2005 and 2013. At all sites, managers applied broadcast herbicides (tank mix of glyphosate and sulfometuron methyl) to control interfering plant species 1 to 3 years (median = 1) prior to the establishment of the experiment. Typical interfering plants that often require control on the Allegheny Plateau include rhizomatous ferns (e.g., *Dennstaedtia punctilobula* and *Thelypteris noveboracensis*) and various graminoids (Horsley 1991).

Herbicide applications are commonly applied in shelterwood settings to mitigate interfering plants (Horsley 1990) and do not exert lasting negative effects on understory flora (Ristau et al. 2011).

Vegetation Surveys

Within each site, we established two 0.42 ha (60 m × 70 m) plots. One plot was randomly assigned a deer exclosure (fence) treatment and the other served as an unfenced control (Fig. 1). Exclosure construction occurred from June to September 2013 on the plots, which ranged between zero and eight growing seasons (median = one growing season) following the shelterwood cut. Within each plot we established a 40 m × 50 m sampling grid with a 10 m × 10 m spacing. We then surveyed tree regeneration (i.e., stem counts) and herbaceous layer abundance (percent cover of all nonarborescent vascular vegetation less than 1 m in height) by species in summer 2020 within each plot. Tree regeneration sampling occurred on 10.51 sq. m circular subplots centered on three of the nodes of the grid dispersed evenly on a diagonal across the plot (e.g., the northwest corner, the center, and the southeast corner of the plot). Regeneration surveys consisted of counts of individual trees by species. Each individual was assigned one of four height classes: less than 30 cm, 31 to 91 cm, 92 to 152 cm, and greater than 152 cm but less than 2.54 cm diameter at breast height (d.b.h.).

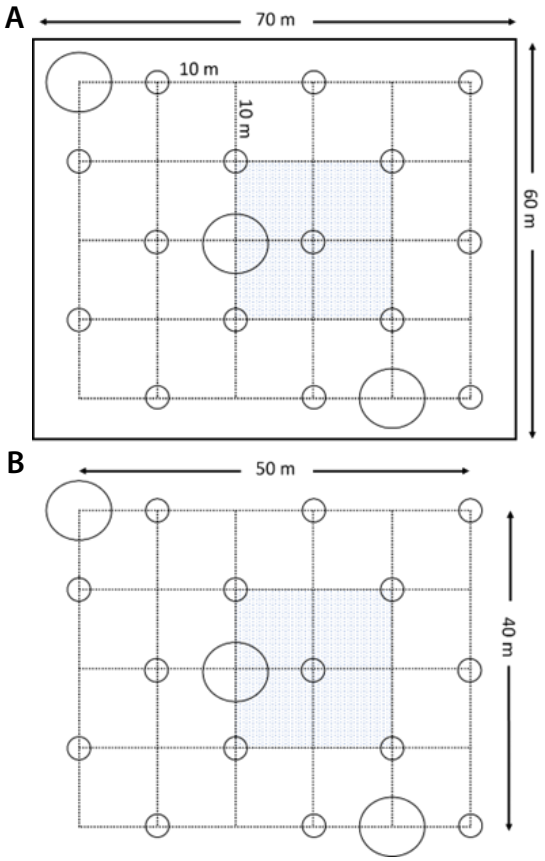

Figure 1.—Conceptual diagrams of experimental design: 70 m × 60 m exclosure (A) and unfenced control (B) at each site in Pennsylvania. Both depict 50 m × 40 m sampling grids, nodes at 10 m intervals, circular subplots where sampling occurred (large circles), circular quadrats where vascular plant cover surveys were conducted (small circles), and central areas where meander searches occurred (gray squares).

In addition, we used ocular estimation methods to estimate cover for each vascular plant species less than 1 m in height using 1 sq. m circular quadrats located at 15 nodes of the grid. All vegetation within a cylinder 1 m tall extending from the quadrat was included, whether rooted in the plot or not and regardless of overall plant height. Finally, to obtain a robust estimate of richness we conducted a meander search through the central 400 sq. m (20 m × 20 m) portion of each plot to account for any species missed on the subplots (Goff et al. 1982).

Statistical Analyses

We used generalized linear mixed models (PROC GLIMMIX; SAS Institute 2013) to examine the direct effect of deer browsing on the herbaceous layer community as well as its potential indirect effect on the herb layer mediated through browse suppression of the woody regeneration layer. We first tested the degree to which browsing limited woody regeneration. We then expanded the mixed model to include a covariate (ANCOVA) when we explored how deer browsing and sapling layer density (i.e., covariate) affected understory plant richness, cover, and diversity. For these analyses, we defined saplings as individuals greater than 152.4 cm in height but less than 2.54 cm d.b.h. as they overtop the understory stratum. Analyses of covariance typically test the unequal slopes assumption (i.e., treatment × covariate interaction; Littell et al. 2006). However, we did not test this interaction because our study encompassed 14 experimental units (2 exclosure treatments × 7 sites = 14) and because our replication level was inadequate to robustly test regression at the exclosure treatment level (Jenkins and Quintana-Ascencio 2020). Nevertheless, we tested the simple sapling density covariate across the pooled set of experimental units. We conducted all analyses at the p-value ≤ 0.10 significance level given the variability encountered across all sites with respect to land management ownership approaches and harvest operations.

We calculated overall stem density (N individuals per ha; all stems) and sapling density (N individuals per ha) from the average of the three regeneration subplots within each plot. For the understory stratum, we calculated overall mean vegetative cover and the Shannon Diversity Index (H') as the average from all herbaceous quadrats within a plot and calculated species richness using the full species list, which included species found in both the quadrats and the meander surveys in the plots. Species richness, diversity, and cover were examined for all vascular taxa (i.e., including tree species) and for nontree taxa alone. Plant functional groups are known to respond differently to browsing and competition (Royo et al. 2010a), so we further examined the effect of deer browsing and density of the sapling layer on the richness and cover of the fern, forb, graminoid, and shrub functional groups. Overall cover, nontree cover, and diversity (H') were normally distributed and modeled using a Gaussian distribution. Means of stem densities and richness were generally skewed from normal and were overdispersed. We modeled them using a negative binomial distribution and modeled percent cover data on plant functional groups using the beta distribution, which is recommended for percent cover data (Damgaard and Irvine 2019). Finally, because metrics such as richness or diversity (H') can mask underlying differences in species composition (Peet 1974), we performed a permutational analysis of variance (PERMANOVA) with fence treatment as the explanatory factor to determine whether browsing altered plant communities in terms of species composition and

abundances. This analysis was run by using the entire species matrix using Bray-Curtis multivariate distances using the adonis function of the vegan package in R (Oksanen et al. 2019).

RESULTS

The understory plant communities within these shelterwood sites were diverse. We found 137 taxa including tree (18.2 percent), fern and lycopod (7.3 percent), forb (47.4 percent), graminoid (13.1 percent), and shrub (13.8 percent) species or morphospecies. As in mature second-growth stands in temperate forest systems, the overwhelming bulk of the taxonomic diversity is found in species that complete their life cycle in the understory rather than in tree species (Spicer et al. 2020). The understory flora overwhelmingly consisted of native species (93.4 percent); the few exotic invasive species were rare and of low abundance (e.g., *Rosa multiflora, Berberis thunbergii, Frangula alnus;* Table 1).

At these shelterwood harvested sites, woody seedling recruitment was high and overall stem densities were unaffected by deer exclusion (Table 2). However, deer browsing limited recruitment into the taller sapling class by 43 percent. We found no effects on richness, cover, or diversity of the herbaceous layer when pooled across all vascular species or when analyses were restricted to nontree species (Table 2). Furthermore, the sapling density covariate exerted no effect on any of these metrics (Table 2). Finally, browsing did not alter plant community composition (PERMANOVA; $F = 0.56$, $R^2 = 0.05$, $p = 0.22$).

Table 1.—List of all species found across all study sites sorted alphabetically within each functional group. Each species is designated native (N) or exotic (E) (USDA Natural Resources Conservation Service 2012), and mean abundance in the control and fence plots is provided. Asterisks (*) denote species that were present within a treatment, but because they were found only on meander searches they lack an estimate of abundance. Dashes (–) represent species that were not found within a treatment.

Species	Group	Native or exotic	Abundance in control plots (percent)	Abundance in fence plots (percent)
Acer pensylvanicum	Tree	N	1.49	0.68
Acer rubrum	Tree	N	6.03	3.62
Acer saccharum	Tree	N	1.18	0.48
Amelanchier arborea	Tree	N	1.96	2.37
Betula allegheniensis	Tree	N	1.03	2.73
Betula lenta	Tree	N	9.49	3.50
Betula spp.	Tree	N	0.26	0.94
Carpinus caroliniana	Tree	N	–	*
Carya cordiformis	Tree	N	*	*
Cornus alternifolia	Tree	N	–	*
Fagus grandifolia	Tree	N	3.41	5.99
Frangula alnus	Tree	E	0.05	0.28
Fraxinus americana	Tree	N	4.30	2.70
Liriodendron tulipifera	Tree	N	*	1.14
Magnolia acuminata	Tree	N	0.43	0.50

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Table 1 (continued).—List of all species found across all study sites sorted alphabetically within each functional group. Each species is designated native (N) or exotic (E) (USDA Natural Resources Conservation Service 2012), and mean abundance in the control and fence plots is provided. Asterisks (*) denote species that were present within a treatment, but because they were found only on meander searches they lack an estimate of abundance. Dashes (–) represent species that were not found within a treatment.

Species	Group	Native or exotic	Abundance in control plots (percent)	Abundance in fence plots (percent)
Pinus strobus	Tree	N	*	–
Populus grandidentata	Tree	N	–	*
Prunus pensylvanica	Tree	N	0.08	0.23
Prunus serotina	Tree	N	0.78	1.27
Quercus alba	Tree	N	–	*
Quercus rubra	Tree	N	1.34	0.57
Sorbus americana	Tree	N	*	*
Tilia americana	Tree	N	0.13	*
Tsuga canadensis	Tree	N	*	*
Aralia spinosa	Shrub	N	0.29	0.08
Aralia spp.	Shrub	N	*	–
Berberis thunbergii	Shrub	E	*	–
Crataegus spp.	Shrub	N	*	–
Diervilla lonicera	Shrub	N	0.32	4.32
Hamamelis virginiana	Shrub	N	–	*
Ilex montana	Shrub	N	0.14	0.56
Lonicera spp.	Shrub	E	0.24	–
Rhus typhina	Shrub	N	–	*
Ribes spp.	Shrub	N	0.05	*
Rosa multiflora	Shrub	E	*	*
Rubus allegheniensis	Shrub	N	12.71	18.33
Rubus flagellaris	Shrub	N	0.10	1.00
Rubus hispidus	Shrub	N	9.32	7.33
Rubus ideaus	Shrub	N	5.42	1.15
Rubus occidentalis	Shrub	N	1.24	*
Rubus odoratus	Shrub	N	0.10	–
Rubus spp.	Shrub	N	0.10	–
Sambucus racemosa	Shrub	N	0.06	0.27
Sambucus spp.	Shrub	N	*	0.07
Sassafras albidum	Shrub	N	*	–
Sassafras albidum	Shrub	N	*	*
Agrostis capillaris	Graminoid	E	*	–
Agrostis spp.	Graminoid	N	*	*

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Table 1 (continued).—List of all species found across all study sites sorted alphabetically within each functional group. Each species is designated native (N) or exotic (E) (USDA Natural Resources Conservation Service 2012), and mean abundance in the control and fence plots is provided. Asterisks (*) denote species that were present within a treatment, but because they were found only on meander searches they lack an estimate of abundance. Dashes (–) represent species that were not found within a treatment.

Species	Group	Native or exotic	Abundance in control plots (percent)	Abundance in fence plots (percent)
Anthoxanthum odoratum	Graminoid	E	–	*
Brachyelytrum erectum	Graminoid	N	0.49	0.43
Carex communis	Graminoid	N	–	0.05
Carex intumescens	Graminoid	N	*	*
Carex spp.	Graminoid	N	8.57	6.34
Carex swanii	Graminoid	N	–	0.24
Cinna latifolia	Graminoid	N	0.53	0.13
Danthonia compressa	Graminoid	N	1.18	0.60
Dichanthelium clandestinun	Graminoid	N	0.77	–
Elymus hystrix	Graminoid	N	–	*
Glyceria melicaria	Graminoid	N	–	*
Glyceria striata	Graminoid	N	–	*
Juncus effusus	Graminoid	N	*	*
Panicum lanuginosum	Graminoid	N	0.05	*
Poa alsodes	Graminoid	N	–	0.05
Scirpus atrovirens	Graminoid	N	*	–
Aralia racemosa	Forb	N	–	*
Arisaema atrorubens	Forb	N	0.12	*
Aster acuminatus	Forb	N	0.10	*
Aster divaricatus	Forb	N	0.17	*
Aster ericoides	Forb	N	*	–
Aster prenanthoides	Forb	N	*	*
Aster puniceus	Forb	N	0.07	0.17
Aster spp.	Forb	N	0.05	*
Aster umbellatus	Forb	N	0.11	0.10
Cardamine pratensis	Forb	N	–	*
Caulophyllum thalictroides	Forb	N	–	*
Chelone glabra	Forb	N	–	*
Chrysosplenium spp.	Forb	N	–	*
Circaea lutetiana	Forb	N	*	–
Claytonia virginica	Forb	N	–	*
Dalibarda repens	Forb	N	*	0.40
Disporum lanuginosum	Forb	N	*	–

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Table 1 (continued).—List of all species found across all study sites sorted alphabetically within each functional group. Each species is designated native (N) or exotic (E) (USDA Natural Resources Conservation Service 2012), and mean abundance in the control and fence plots is provided. Asterisks (*) denote species that were present within a treatment, but because they were found only on meander searches they lack an estimate of abundance. Dashes (–) represent species that were not found within a treatment.

Species	Group	Native or exotic	Abundance in control plots (percent)	Abundance in fence plots (percent)
Eupatorium rugosum	Forb	N	0.12	*
Fragaria virginiana	Forb	N	0.05	0.70
Galium asprellum	Forb	N	0.14	*
Galium triflorum	Forb	N	0.28	0.52
Geranium robertianum	Forb	N	–	0.24
Geum spp.	Forb	N	–	*
Hepatica nobilis	Forb	N	–	*
Hieracium spp.	Forb	N	*	0.05
Impatiens capensis	Forb	N	*	0.88
Laportea canadensis	Forb	N	*	–
Lycopus spp.	Forb	N	*	–
Lysimachia quadrifolia	Forb	N	*	*
Maianthemum canadense	Forb	N	1.16	1.48
Maianthemum racemosum	Forb	N	*	0.05
Medeola virginiana	Forb	N	0.12	0.19
Mentha arvensis	Forb	N	*	–
Mentha spp.	Forb	N	0.85	–
Mitchella repens	Forb	N	0.26	1.30
Monotropa uniflora	Forb	N	*	*
Oxalis montana	Forb	N	0.19	*
Oxalis stricta	Forb	N	*	–
Packera aurea	Forb	N	*	–
Persicaria sagitata	Forb	N	–	*
Phytolacca americana	Forb	N	–	*
Podophyllum peltatum	Forb	N	–	*
Polygonatum biflorum	Forb	N	*	*
Polygonatum pubescens	Forb	N	*	*
Polygonum cilinode	Forb	N	0.64	0.30
Polygonum persicaria	Forb	E	*	0.10
Polygonum spp.	Forb	N	*	–
Potentilla simplex	Forb	N	0.10	*
Prenanthes alba	Forb	N	–	0.05
Prenanthes altissima	Forb	N	0.13	*

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Table 1 (continued).—List of all species found across all study sites sorted alphabetically within each functional group. Each species is designated native (N) or exotic (E) (USDA Natural Resources Conservation Service 2012), and mean abundance in the control and fence plots is provided. Asterisks (*) denote species that were present within a treatment, but because they were found only on meander searches they lack an estimate of abundance. Dashes (–) represent species that were not found within a treatment.

Species	Group	Native or exotic	Abundance in control plots (percent)	Abundance in fence plots (percent)
Prenanthes trifoliata	Forb	N	*	*
Ranunculus recurvatus	Forb	N	–	*
Solidago rugosa	Forb	N	0.62	1.67
Tiarella cordifolia	Forb	N	–	*
Trientalis borealis	Forb	N	0.22	0.12
Trillium erectum	Forb	N	*	0.05
Trillium undulatum	Forb	N	0.24	0.08
Tussilago spp.	Forb	E	–	0.14
Urtica dioica	Forb	N	*	*
Uvularis sessilifolia	Forb	N	0.10	–
Veronica officinalis	Forb	E	*	–
Veronica spp.	Forb	N	*	–
Viola spp.	Forb	N	1.71	1.24
Dennstaedtia punctilobula	Fern and allies	N	10.45	5.76
Diphasiastrum digitatum	Fern and allies	N	0.05	–
Dryopteris intermedia	Fern and allies	N	1.67	1.23
Dryopteris marginalis	Fern and allies	N	–	*
Lycopodium obscrurum	Fern and allies	N	0.84	3.29
Onoclea sensibilis	Fern and allies	N	–	*
Osmunda spp.	Fern and allies	N	*	–
Polystichum acrostichoides	Fern and allies	N	0.05	*
Pteridium aquilinum	Fern and allies	N	–	0.14
Thelypteris novoboracensis	Fern and allies	N	0.65	0.32
Richness			**105**	**111**
Total cover (percent)			**95.33**	**88.94**

Table 2.—Effects of deer exclusion (i.e., fence treatment) on total stem densities, sapling densities, richness, percent cover, and diversity (H'). Analyses of nontree taxa also include sapling density as a covariate. Total stem densities and sapling densities refer to tree species. Richness, mean cover, and diversity refer to all vascular taxa; nontree richness, cover, and diversity refer only to the forb, shrubs, ferns, and graminoids combined. Values in bold are significant effects ($p \leq 0.1$).

Variable	Treatment: control	Treatment: fence	Treatment: F-value, p-value	Sampling density covariate: F-value, p-value
Overall stem density (stems/ha)	129,491 ± 34,197	101,978 ± 26,931	$F_{1,6} = 0.46$, $p = 0.524$	–
Sapling density (stems/ha)	**5,289 ± 1,618**	**10,729 ± 3,282**	**$F_{1,6} = 0.72$, $p = 0.036$**	–
Richness (species/plot)	33.6 ± 5.7	39.8 ± 6.7	$F_{1,4} = 1.72$, $p = 0.260$	$F_{1,4} = 3.37$, $p = 0.140$
Mean cover (percent)	93.1 ± 8.6	90.1 ± 8.6	$F_{1,8} = 0.15$, $p = 0.715$	$F_{1,11} = 0.41$, $p = 0.533$
Diversity (H')	2.3 ± 0.2	2.4 ± 0.2	$F_{1,7} = 0.28$, $p = 0.611$	$F_{1,9} = 0.11$, $p = 0.751$
Nontree richness (species/plot)	27.0 ± 4.5	30.3 ± 5.0	$F_{1,8} = 0.56$, $p = 0.475$	$F_{1,9} = 1.11$, $p = 0.319$
Nontree cover (percent)	59.1 ± 8.7	65.4 ± 8.7	$F_{1,8} = 0.52$, $p = 0.491$	$F_{1,11} = 1.96$, $p = 0.189$
Nontree diversity (H')	1.8 ± 0.2	1.8 ± 0.2	$F_{1,8} = 0.34$, $p = 0.576$	$F_{1,11} = 0.01$, $p = 0.758$
Forb richness (species/plot)	7.6 ± 2.3	9.2 ± 2.8	$F_{1,9} = 0.41$, $p = 0.541$	**$F_{1,11} = 3.50$, $p = 0.088$**
Forb percent cover	**4.7 ± 2.2**	**7.9 ± 3.5**	**$F_{1,5} = 4.72$, $p = 0.078$**	$F_{1,6} = 1.45$, $p = 0.272$
Shrub richness (species/plot)	4.1 ± 0.9	4.1 ± 0.9	$F_{1,11} = 0.00$, $p = 0.963$	$F_{1,9} = 0.15$, $p = 0.709$
Shrub percent cover	22.1 ± 8.4	34.5 ± 10.8	$F_{1,8} = 2.55$, $p = 0.150$	$F_{1,10} = 2.22$, $p = 0.166$
Fern and allies richness (species/plot)	3.1 ± 0.7	3.5 ± 0.7	$F_{1,1} = 0.14$, $p = 0.776$	$F_{1,1} = 0.06$, $p = 0.843$
Fern and allies percent cover	12.1 ± 4.4	10.4 ± 4.0	$F_{1,9} = 0.67$, $p = 0.436$	$F_{1,9} = 0.10$, $p = 0.756$
Graminoid richness (species/plot)	–	–	–	–
Graminoid percent cover	9.6 ± 2.2	8.5 ± 1.9	$F_{1,7} = 0.17$, $p = 0.696$	$F_{1,9} = 2.27$, $p = 0.166$

Focusing our analyses on plant functional groups revealed some subtle treatment effects. As predicted, forbs had 68 percent greater cover when protected from browsing ($p = 0.08$; Table 2). Additionally, although the fencing treatment alone did not affect forb richness, richness declined as sapling densities increased across all plots (Fig. 2). No other plant functional group responded to the exclosure treatments (Table 2).

DISCUSSION

The primary goal of shelterwood harvests is to promote desirable tree species regeneration (Nyland et al. 2016). Our results show that even though the overall regeneration response is robust, browsing limits entry into the taller sapling size class and thereby slows the rate of succession (Bernes et al. 2018). This successional delay persists following the final overstory removal and can leave a long-term imprint on the maturing forest overstory even decades later (Parker et al. 2020, Reed et al. 2022). These results are in line with recent reviews of browsing impacts in temperate and boreal forests globally (Bernes et al. 2018, Habeck and Schultz 2015), which report that browsing strongly limits woody species abundance, particularly in the larger size classes.

Despite this strong browsing effect on tree regeneration, browsing did not directly affect the overall pooled herbaceous layer richness, abundance, or diversity. Moreover, indirect browsing impacts on these metrics mediated through variation in the sapling

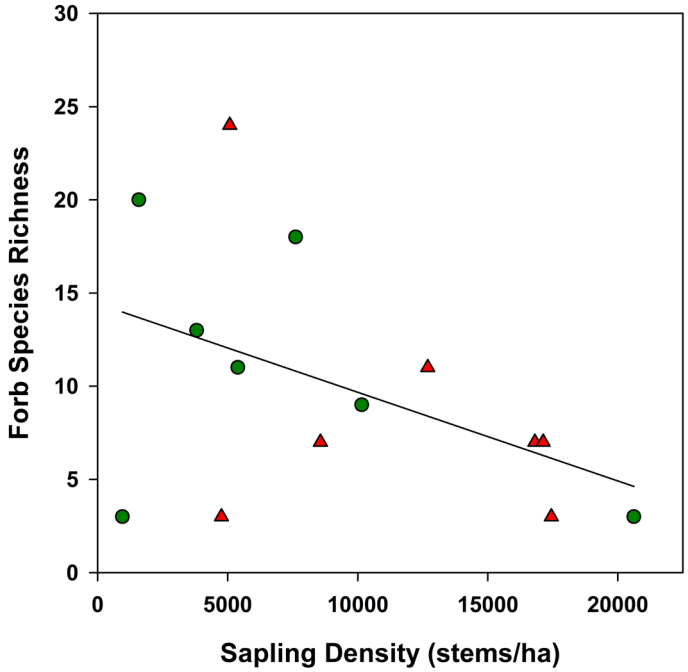

Figure 2.—Relationship between sapling density (N stems per ha) and forb richness. The analysis on forb richness found a significant covariate effect (p = 0.09); hence the regression is pooled across all plots, regardless of browsing treatment. Red triangles denote fenced plots. Green circles denote unfenced control plots.

densities were also not detected. These results are largely consistent with regional and global browsing syntheses, which have found that effects are often undetected when all understory taxa are pooled (Bernes et al. 2018, Habeck and Schultz 2015). Our results and these syntheses suggest that combining taxa that vary widely in their abundance, growth form, and browse- and shade-tolerance may mask responses to the potential direct or indirect effects of browsing.

We found that forbs as a group were directly and substantially limited by browsing. This result is consistent with other studies showing that deer browsing can limit forb abundance, especially for species that are more palatable (Huebner et al. 2010). Our results also suggest that browsing may indirectly promote forb richness by limiting sapling densities. These results suggest that forb communities in recently harvested stands are structured by the opposing direct and indirect effects of browsing. Namely, browsing directly reduces overall forb abundance, yet by also limiting sapling recruitment and thereby sustaining greater resource availability and heterogeneity, it indirectly promotes forb richness (Bernes et al. 2018, DiTommaso et al. 2014, Faison et al. 2016, Parker et al. 2020).

Despite these effects on forbs, community composition was unaffected by browsing. This finding was surprising given that deer often cause compositional shifts due to selective browsing pressure (Rooney 2001). We suggest that the absence of a detectable shift in species composition may arise from at least three nonexclusive mechanisms. First,

although deer densities regionally were high (greater than 12 deer per sq. km) throughout much of the 20th century, enactment of science-based hunting policies in 2003 lowered deer densities to more moderate levels (about 6.3 deer per sq. km at these sites) that are compatible with plant diversity (Horsley et al. 2003, Royo and Stout 2019). Second, shelterwood harvesting purposefully accelerates succession and shifts composition toward fast-growing, shade-intolerant species. In northern hardwood and Allegheny hardwood forest types, harvesting enhances *Betula lenta* recruitment (Royo et al. 2019), stimulates the proliferation of sedge (*Carex*) and *Rubus* (Hughes and Fahey 1991), and favors fern reinvasion (e.g., *D. punctilobula*; Ristau et al. 2011). These taxa currently dominate (greater than about 50 percent) the understory at our sites in both fenced and unfenced areas (Table 1) and, apart from *Rubus allegheniensis*, are relatively unpreferred browse or have high tolerance to browsing, or both (Horsley et al. 2003, Parker et al. 2020), making them unresponsive to moderate browsing pressure. Finally, the historical decadeslong overbrowsing of these sites may have led to understory communities that remain depauperate even when deer densities are reduced (Royo et al. 2010b, Webster et al. 2005). Harvesting and fencing alone may not be sufficient management actions to restore understory species diversity to sites with a long history of overbrowsing.

No other plant functional group responded to the exclosure treatments. We were surprised by the lack of a browsing effect on shrubs because these taxa are often preferentially browsed (Parker et al. 2020, Royo et al. 2010a). At our sites, the shrub layer was largely dominated by *Rubus allegheniensis*, a taller shrub species that is highly preferred by deer (Healy 1971), and *R. hispidus*, a species with a short trailing growth habit that makes it less accessible to deer, which usually browse within the 0.2 to 1.8 m stratum (Waller and Alverson 1997). Closer examination of their respective abundances shows that *R. allegheniensis* is less abundant in areas open to browsing (18.3 and 12.7 percent in fence versus control, respectively), whereas *R. hispidus* displayed the opposite pattern (9.3 and 7.3 percent in control versus fence, respectively) (Table 1). Hence, the lack of a response in shrubs and in other species groups may be masked in analyses that merge species with varying susceptibility to browsing. Shrubs and graminoids were unaffected by browsing or the sapling layer. In our study, neither functional group increased in areas open to browse pressure.

CONCLUSIONS

Our results suggest that in the context of an operational shelterwood, browsing effects on the herbaceous community were generally constrained and largely anticipated albeit ecologically important. Our results found browsing directly limited forb abundance and indirectly enhanced forb richness by slowing tree succession. Whether this browse-induced lengthening of the early seral forb community is ecologically meaningful is unknown, particularly given the simultaneous negative impacts of browsing on forb cover and tree seedling growth. Nevertheless, although fencing is often necessary to protect tree regeneration and sustain the associated ecosystem services of early seral communities where deer populations are high (Redick and Jacobs 2020), the zero-browsing conditions created within exclosures rarely have a historical analogue in landscapes where deer were present and thus may hasten succession and alter composition in ways that may be inimical to the overall maintenance of plant diversity (Ristau and Horsley 1999, Royo et al. 2010a). We suggest that moderate browse pressure, rather than its complete absence, may be critical to sustain plant diversity following a harvest or a disturbance (Faison et

al. 2016, Murray et al. 2016). Moderating deer browse impact could be accomplished by coordinated actions at larger scales, including controlling population through hunting pressure and attenuating impact through active silvicultural management (Redick and Jacobs 2020).

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Repeated Harvesting of Narrow Forest Strips for Energy Biomass Benefits the Herbaceous Layer

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INTRODUCTION

Repeatedly harvesting wood as a source of energy biomass is a potential option for reducing fossil fuel use. Despite the interest in increasing forest bioenergy use, gaps remain in the understanding of how much, how often, and in what spatial pattern forest biomass can be removed from a site before biomass removal begins to adversely affect forest ecosystem resources. Among these ecosystem benefits are soil fertility and biomass yield, water quality, and wildlife habitat (Abbas et al. 2011). In addition, little research has been done on the effects of repeated cutting for woody biomass on the forest herbaceous layer. The herbaceous layer is an important part of forest ecosystems, and it performs multiple vital functions and services such as providing nutrient storage (Gerken Golay et al. 2016), accelerating nutrient cycling (De Groote et al. 2018), and serving as a biodiversity reservoir (Gilliam 2007). Light is a limiting factor to the growth of the herbaceous layer in the closed-canopy forest. Thus, harvesting can provide both woody biomass and light, potentially benefiting abundance and diversity of the herbaceous layer abundance and diversity. After an initial harvest, densely regrowing stands are commonly managed through crop tree release cuts. Although this practice maintains an open environment for the herbaceous layer, the cut (but scattered) biomass is usually unused due to cost. An alternative method may be to cut biomass in strips. This alternative practice would achieve crop tree release in residual strips (Martin et al. 2017) while allowing the cut biomass to be collected using automated biomass harvesters (Dykstra 2010, Visser and Obi 2021).

In this study, herbaceous layer cover, species richness, and Shannon-Wiener diversity were compared between (1) cut strips (harvested areas), residual strips left between the cut strips, and an uncut control, and (2) between different cut strip widths and the uncut control. We predicted that the herbaceous layer would benefit the most in the cut strips and to a lesser degree in residual strips relative to the control and that it would benefit more in the wider cut strips relative to narrower cut strips.

METHODS

The study was conducted in three stands in the West Virginia University Research Forest (Martin 2015), located about 16 km east of Morgantown, WV (39.669189, -79.812176) (Fig. 1). In each stand, the first strip cut occurred in 2014 in 11-year-old stands; the second strip cut occurred in 2020 in 6-year-old stands. Each stand contained three treatment plots and one uncut control plot (each about 30 m × 30 m). Each treatment plot had five pairs of alternating cut and residual strips. Residual strips were 2.4 m wide. Cut strip widths were 2.4 m (narrow), 3.7 m (intermediate), and 4.9 m (wide) (based on one or two passes with a harvester) in their respective treatment plots. The uncut control had three adjacent strips that were 3.5 m wide (Fig. 1). All strips were oriented in a north-south direction.

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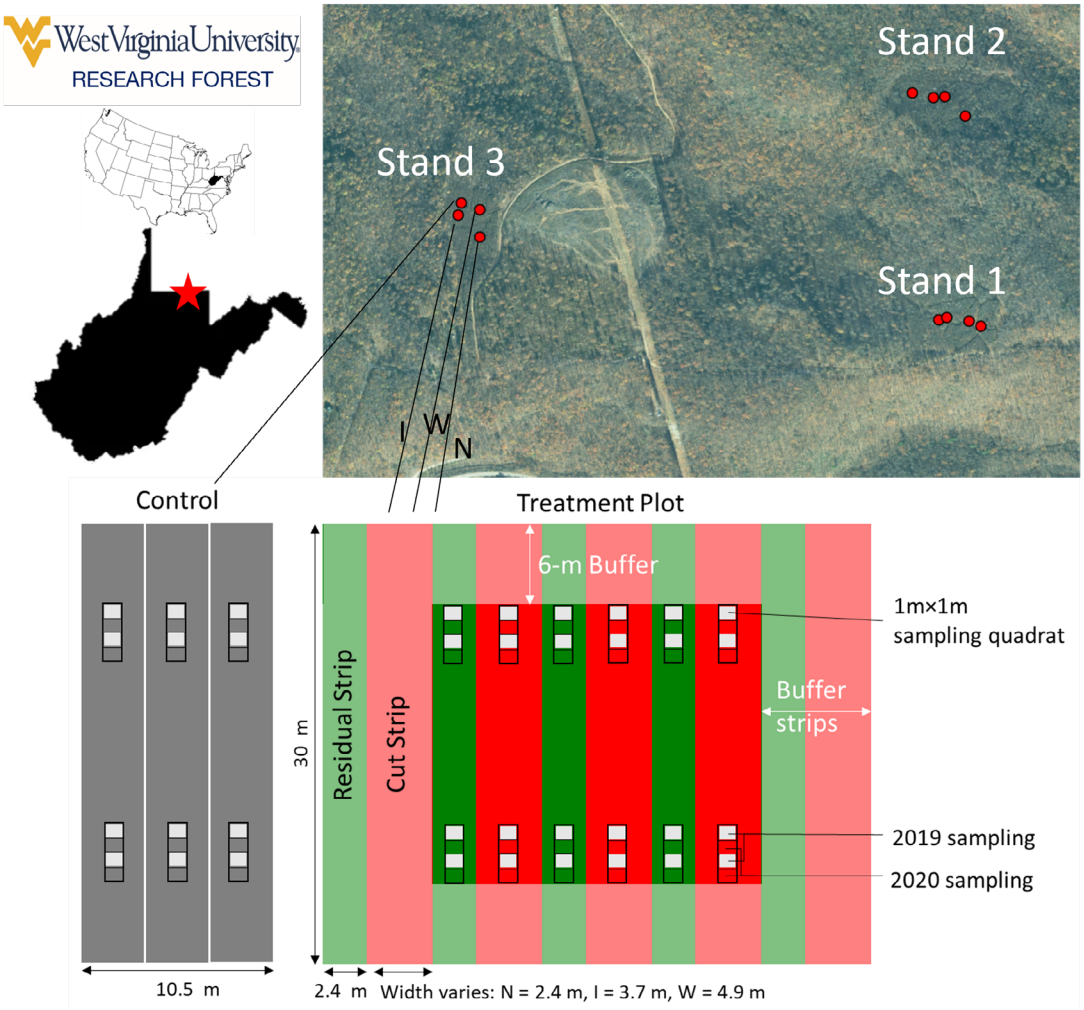

Figure 1.—Maps showing location of the strip cut harvesting experiment (West Virginia University Forest, Bruceton Mills, WV) and diagram of the study design. Each stand contains a control (gray) and three treatment plots consisting of three cut strips (red) and residual cut strips between cut strips (green). N: narrow; I: intermediate; W: wide. All strips are oriented north-south.

Herbaceous layer cover was recorded in four 1 sq. m quadrats in the centermost three cut and residual strip pairs (Fig. 1) and in the three control strips in summer 2019 (pre-second cut) and 2020 (post-second cut). Cover was recorded as leaf area using the hand-area method (Walter et al. 2015) at species level, if possible. Per strip, we calculated total cover (C, sq. dm of leaf area of all species present in the four 1 sq. m sampled quadrats) as a measure of abundance, species richness (S, number of species present in the 4 sq. m), and Shannon-Wiener diversity (H', in the 4 sq. m). H' is an index used to characterize species diversity by accounting for both species richness and relative abundance of each species (Shannon and Weaver 1949). H' increases with increasing species richness and more equitable relative abundances among species.

Data from this split plot randomized complete block design were analyzed in SAS (SAS Institute 2013) with repeated-measures analysis of variance (ANOVA; accounting for measurements in 2019 and 2020) for (1) an effect of treatment (cut, residual, control), and (2) strip cut width (wide, intermediate, narrow, control) on herbaceous layer characteristics; multiple comparisons of means were Tukey-Kramer adjusted.

RESULTS AND DISCUSSION

In the herbaceous layer (defined as vegetation less than 1 m tall), 78 plant species were recorded (37 forbs, 8 ferns, 13 shrubs/vines, 20 tree seedlings/resprouts; graminoids were not resolved beyond the family level). Shrubs/vines and ferns contributed most to total leaf area (32 percent and 32 percent, respectively), followed by tree seedlings/resprouts (17 percent), forbs (16 percent) and graminoids (3 percent). Two nonnative species, Japanese barberry (*Berberis thunbergii*) and multiflora rose (*Rosa multiflora*), were found among the shrub/vine growth form; however, their contribution to total shrub leaf area was very small (0.4 percent).

Mean values of C, S, and H' were generally greater in cut strips (regardless of width) than in the control (Fig. 2A,B,C). Only the difference in cover between cut strips and the control (when averaged across both years) was significant (p = 0.03).

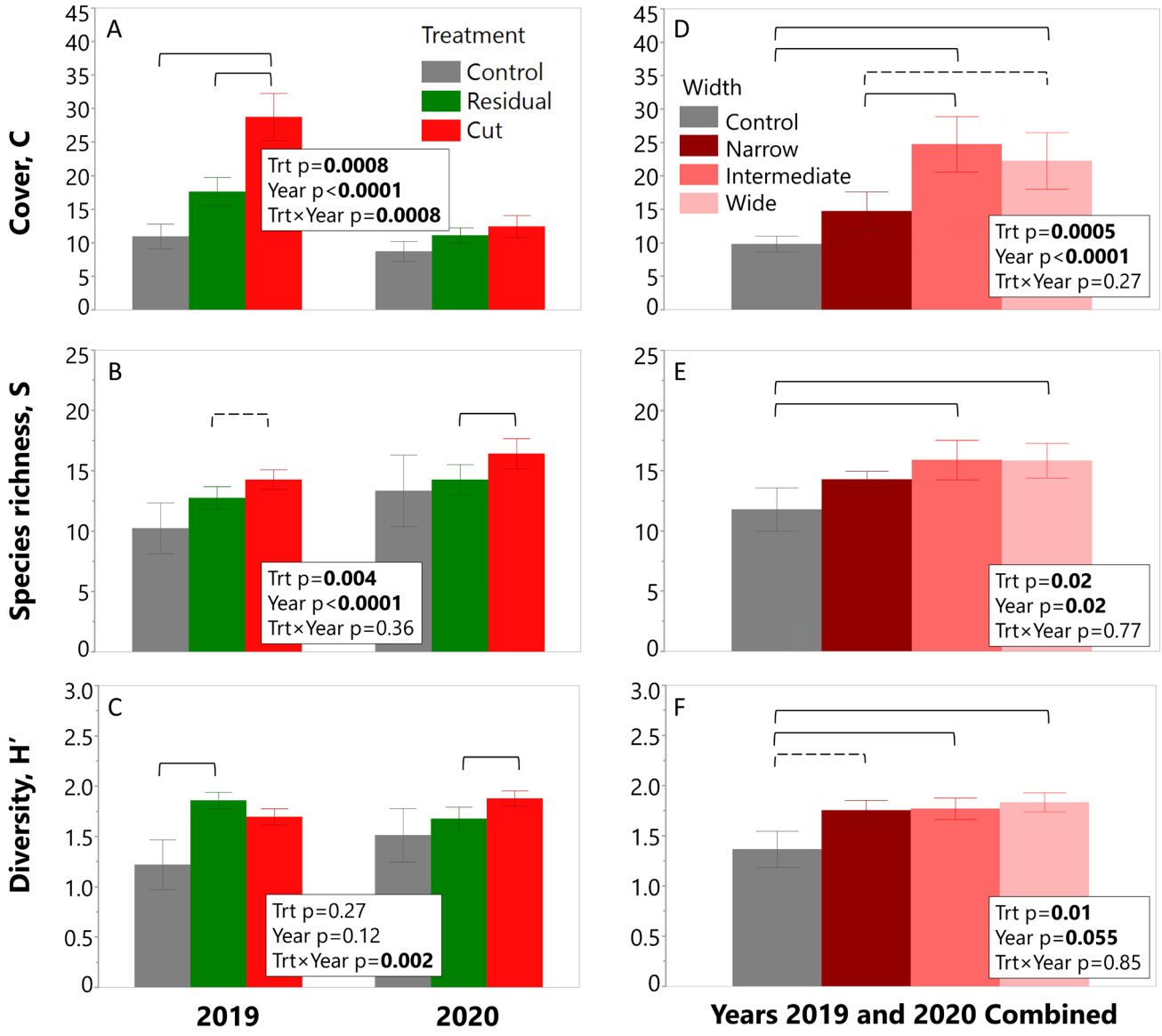

Figure 2.—Effects of strip cuts for energy biomass on herbaceous layer characteristics (West Virginia University Forest, Bruceton Mills, WV), by treatment type (trt; control versus residual strips versus cut strips) and cut strip width, respectively, on (A, D) cover (sq. dm in four 1 sq. m quadrats; %), (B, E) species richness (per 4 sq. m), and (C, F) Shannon diversity index (per 4 sq. m). ANOVA results are based on 2 years of data collection. Brackets show significant differences between treatments using Tukey-Kramer multiple comparisons; solid lines p ≤ 0.05, dashed lines p ≤ 0.1.

Positive effects on herbaceous layer cover are expected due to greater light levels and increased moisture from altered canopy cover (Burkhart et al. 2013, Márialigeti et al. 2016, Vander Yacht et al. 2020). The statistical power of this analysis may not have been sufficient to detect a treatment effect on species richness and diversity (see the following discussion). However, effects of forest cutting on measures of herbaceous layer diversity may not always be straightforward. In a compilation of several studies on the effects of clearcutting on herbaceous layer richness and diversity in forests in the northeastern United States, effects varied between studies. Roberts and Gilliam (2014) reported that species richness and diversity in 2- to 26-year-old clearcuts were the same as or higher than in mature second-growth forest (55 to more than 150 years old).

Comparing the cut strips of different widths (narrow, intermediate, and wide), no statistically significant differences in C, S, or H' were detected. This result is possibly due to the small differences between studied cut width or small sample size. Across both years, intermediate and wide strips had significantly greater C, S, and H' than the control (Fig. 2D,E,F). Thus, wider cut strips may be more effective in promoting the herbaceous layer than narrow strips.

In the residual strips, herbaceous layer characteristics were intermediate between the control and cut strips (Fig. 2A,B,C). Replicating the analyses described earlier but comparing the herbaceous layer in residual strips to the control, C and H' (but not S) were greater in residual strips than in the control ($p \leq 0.05$; data not shown). This likely indicates an edge effect from the cut strips on the residual strips (Remy et al. 2018) and may represent an additional benefit of strip cutting to the herbaceous layer community. However, whether cut or residual strips can maintain elevated species richness and diversity of native plants into the future depends on the ability of invasive species to colonize these areas (Mortensen et al. 2009) and on the level of deer browse (Waller 2014).

CONCLUSIONS

Strip cutting for woody biomass as an energy source appears to be sustainable and likely beneficial to the abundance, species richness, and diversity of the herbaceous layer, at least after the initial two cuts. Further studies should evaluate long-term cumulative effects, including effects on soil properties and on compaction and compositional changes due to vehicles used for harvesting (Closset-Kopp et al. 2019). Although cut strips of 3.7 m and 4.9 m widths benefited the herbaceous layer the most in this study, future studies are needed to evaluate wider cut strips and their effects on overall site management goals of integrating biomass yield, timber from residual strips, and ecological benefits from early successional habitat (Swanson et al. 2011) and edges.

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Restoration

Creating Woodlands from Scratch: Intensive Management Used to Craft a Rare Forest Ecosystem in Virginia

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Abstract

The Fairfax County Park Authority in Fairfax, VA is using herculean efforts to move an unmanaged Virginia pine (*Pinus virginiana*)-eastern redcedar (*Juniperus virginiana*)-hardwood forest toward a rare northern hardpan basic oak (*Quercus*)-hickory (*Carya*) forest type. Initial treatments included mechanical removal of Virginia pine and eastern redcedar using a forestry masticator that retained approximately 60 to 80 square feet of Virginia pine basal area to create understory light conditions conducive to oak establishment and growth. This treatment also removed undesirable mesophytic species such as red maple (*Acer rubrum*), elm (*Ulmus*), and ash (*Fraxinus*). Oak, hickory, redbud (*Cercis*), and dogwood (*Cornus*) were retained in the understory where present. Nonnative invasive species such as Japanese stiltgrass (*Microstegium vimineum*), autumn olive (*Elaeagnus umbellata*), and Japanese honeysuckle (*Lonicera japonica*) were treated with herbicide. Because scant advance oak reproduction existed, the site was artificially planted with bare-root hardwood seedlings at 800 seedlings per acre; white oak (*Q. alba*), northern red oak (*Q. rubra*), and pignut hickory (*Carya glabra*) were planted in upland areas and swamp white oak (*Q. bicolor*), and pin oak (*Q. palustris*) were planted in hydric areas. The entire project area was fenced to exclude deer using a solar-powered electric fence system. Four years into project implementation, the site has been successfully transformed into a pine savanna with a diverse understory of native grasses and herbs. However, the project goal was to establish a cohort of oak and hickory reproduction to build the foundation of the rare forest type. The status of the oak reproduction remains tenuous, as seedling quality and planting procedures were substandard and competing vegetation remains an ongoing challenge. While project goals have been partially achieved, many lessons were learned that may guide future forest restoration projects. Today's contemporary forests will require these intensive management efforts to sustain desirable oak-dominated stands.

INTRODUCTION

Degraded stands are common throughout eastern forests, often the result of misguided or no management, coupled with severe impacts from nonnative and invasive species (Clatterbuck 2006, Oswalt and Oswalt 2011). Restoration treatments are recommended according to site characteristics, owner objectives, and available resources for management activities. Improving stands from degraded to desirable conditions by changing species composition or stand structure, or both, can be an intensive undertaking.

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Depending on the degree of degradation and objectives, the restoration process can entail multiple activities over an extended time. Applying the best available science should be a key starting point.

The Fairfax County Park Authority (FCPA) in Virginia was presented with a unique opportunity to restore a rare forest community with dedicated resources to meet restoration objectives. The FCPA stewards approximately 24,000 acres of public park land, of which 17,000 acres are undeveloped. As part of its programmatic restoration goals, the FCPA planned an oak-hickory (*Quercus-Carya*) forest restoration project adjacent to a Virginia state natural area preserve under its management. The desired forest type is a northern hardpan basic oak-hickory forest, a rare edaphic community that exists in the Triassic Basin area west of Washington, DC and is associated with mixed mesic hardwoods and oak-heath (*Erica*) forests (Fleming and Patterson 2017). Selection of this project was based on the rare status of this ecological community, ranked as "imperiled" (G2/S2) by NatureServe and existing only in Virginia and Maryland. The project presented an opportunity to rebuild and expand a rare forest type, protect the adjacent high-quality stands from degradation, and conserve rare and vulnerable native plant species (NatureServe 2022).

The current forest type is a Virginia pine (*Pinus virginiana*)-eastern redcedar (*Juniperus virginiana*) forest, with a heavy infestation of invasive species. Because of the severe degradation, restoration was aimed at changing species composition by removing undesirable trees and treating understory invasive plants. The goal was to restore the native tree species of the northern hardpan basic oak-hickory forest type, with oak and hickory being preferred. Although much research in upland hardwood forests, including those described as oak-hickory forest types, has focused on regeneration methods for the oak species, few studies have examined the ecology and silviculture of hickory (Hannah 1987, Johnson et al. 2002, Loftis 1983, Roach and Gingrich 1968, Sander 1977). Little information exists on regenerating hickory, and natural regeneration response to disturbance shows that hickory does not respond favorably to some disturbances (Pile Knapp et al. 2021, Schweitzer 2010). Artificial regeneration of hickory under active management has received only limited attention (Muller et al. 2019).

Research shows that oak stocking in a newly regenerated stand is proportional to the amount of oak advance reproduction present prior to treatment and those oaks must be of sufficient size and densities to be competitive sources of regeneration (Dey et al. 2012). Major issues arise when oak reproduction is either small or absent, as the development of new oak seedlings or growth of existing small seedlings after disturbance is not reliable (Gottschalk 1994, Parker and Dey 2008). Promoting oak advance reproduction requires active management to ensure adequate stocking of oak. The shelterwood method of regeneration has flexible applications and is widely recommended. Shelterwood treatments create understory light and moisture conditions conducive to oak seedling growth, while these conditions aid in controlling competition from shade-intolerant species. However, shelterwood prescriptions are predicated on growing advance (existing) oak reproduction (Schweitzer and Dey 2011). If no oak advance reproduction exists, or if it is not in a competitive position, planting oaks as part of a shelterwood prescription mimics the process of positioning naturally regenerating oak seedlings to recruit and respond to the final harvest release (Dey et al. 2009). However, high-quality seedlings are rarely available (Clark et al. 2022). Tree planters who have had experience with the best planting

techniques are also scarce. Despite these challenges, a shelterwood prescription with underplanting was identified as the most applicable for restoring the FCPA site.

METHODS

The FCPA has 420 parks on more than 24,000 acres of land throughout Fairfax County, VA, where the FCPA is a primary authority on conservation and stewardship of public lands. Its ecosystem restoration program, called Helping Our Land Heal, was begun in 2012. The FCPA has been charged with using capital improvement funding to address the protection and restoration of natural capital assets, such as undeveloped forests, grasslands, and wetlands, through active management activities, including prescribed fire, treatment of invasive species, deer management, and restoration of rare species and communities. The project area detailed here is adjacent to a nature preserve that contains a rare forest community—northern hardpan basic oak-hickory forest—found primarily in the Culpeper basin of Virginia and Maryland, with small patches in the Piedmont foothills and lower Blue Ridge Mountains (NatureServe 2022). This forest type is characterized by the dominance of white oak (*Q. alba*), white ash (*Fraxinus americana*), and pignut hickory (*Carya glabra*) with eastern redcedar, eastern redbud (*Cercis canadensis*), and flowering dogwood (*Cornus florida*) as common understory trees. Other dominant species include northern red oak (*Q. rubra*), black oak (*Q. velutina*), post oak (*Q. stellata*), mockernut hickory (*Carya tomentosa*), and slippery elm (*Ulmus rubra*). Associated herbaceous community species include forbs such as nightshade (*Circaea canadensis*) and shining bedstraw (*Galium concinnum*), grasslike plants including rock muhly (*Muhlenbergia sobolifera*), poverty oat grass (*Danthonia spicata*), and bottlebrush grass (*Elymus hystrix*), and vines such as hog-peanut (*Amphicarpaea bracteata*).

The project area was released from agriculture approximately 50 years ago, abandoned soon afterwards, and allowed to naturally succeed into a stand of low-quality Virginia pine and eastern redcedar. The understory is dominated by invasive shrubs, vines, and grasses, including autumn olive (*Elaeagnus umbellata*), Japanese honeysuckle (*Lonicera japonica*), Oriental bittersweet (*Celastrus orbiculatus*), wineberry (*Rubus phoenicolasius*), and Japanese stiltgrass (*Microstegium vimineum*). Desired oak and hickory seedlings and saplings are rare due to heavy deer pressure, heavy shade, and competition from invasive plant species.

The desired forest community, northern hardpan basic oak-hickory forest, is characterized by unique shrink-swell soils and a mixture of oaks, hickories, eastern redbud, and forest grasses and forbs. Soils are generally about 8 inches of loam underlaid with heavy clay and hardpan development. The dominant soil series in the project area are Hapludalfs of the Jackland and Haymarket series, which contain an upper layer of silt underlaid by a sticky, plastic clay layer that supports a perched water table during the wet season and a droughty hardpan layer during the dry season. These soils are derived from igneous bedrock commonly known as diabase. Limited areas of Rowland silt loam occur at the edges of the project area near the floodplain of a nearby stream and Elbert silt loam occurs in a small depression that has very poor drainage. These areas are alternately wet and dry, with erosion; leaching on higher areas, combined with greater water limitations, influences species composition and growth. Since the stand is not naturally succeeding to oak-hickory, and there is no significant seed source for desirable species, restoration plans included removing undesirable canopy and midstory species followed by an underplanting

of oak and hickory. A shelterwood prescription, in which canopy reduction is done in two phases, was recommended to control understory conditions (Table 1); this treatment would increase light for regeneration establishment and growth while the shade from residual trees would also help by mitigating drought and heat stresses and deterring more-open conditions conducive to invasive species establishment and growth.

Restoration Treatment Descriptions

The project goal is to restore the forest to conditions resembling those of the adjacent natural area preserve, specifically, to create a young forest that would follow a trajectory toward a northern hardpan basic oak-hickory forest as classified by the Virginia Division of Natural Heritage. The vegetation would be dominated by mature white and red oak with an understory consisting of eastern redbud and flowering dogwood. The forest floor would contain a rich diversity of herbaceous species including characteristic grasses and flowering species. To achieve this state, a new cohort of oak seedlings would be chaperoned to maturity through active management.

Stand conditions at the start of the restoration process were 200 eastern redcedar trees per acre with an 8-inch diameter at breast height (d.b.h.) interspersed with 150 Virginia pine trees per acre that averaged 10 inches d.b.h. Canopy Virginia pine trees were approximately 50 to 60 feet tall, while most of the smaller redcedar trees were considerably shorter. Average basal area was 140 square feet per acre and canopy cover was 100 percent.

Table 1.—Restoration treatment challenges and methods used to move a degraded Virginia pine-eastern redcedar stand toward northern hardpan basic oak-hickory forest in Fairfax County, VA

Challenge	Proposed treatment	Justification	Drawbacks
Increase light availability in the understory	Reduce stand density with forestry mulcher	No markets for timber harvest	Deep mulch layer created; subjected residuals to windthrow
Remove undesirable trees	Forestry mulcher	No markets for removed species	More mulch; no income generated
Control woody plants	Herbicide	Control sprouting of undesirable species	Expensive and laborious
Control invasive species	Herbicide	Site highly infested with multiple invasive species	Expensive; not always effective; nontarget species concerns
Reduce deer herbivory	Tree tubes and fencing	Tree growth above browse line desired	Expensive; fence requires maintenance
Deploy artificial regeneration	Hand planting	No desired reproduction present	Expensive; poor quality seedlings are common; planting techniques are laborious; planting must be monitored
Balance function and biodiversity	Herbaceous community conservation	Broad restoration goals	Disturbances not targeted; open conditions favor some invasive species over desired species

Stand density was reduced using forestry mulching attachments (FAE, Trento, Italy) on rubber-tracked loaders in March 2018 on 21 acres. The scope of the original project was 30 acres of treatment, but 6 acres were removed from treatment because they could not tolerate the disturbance required for mechanical treatments and tree planting, and 3 acres were left undisturbed along drainages to protect water quality. While some eastern redcedar trees were exchanged with labor offsets to a local boutique industry buyer as per FCPA authorization, there were no markets for the small-diameter and low-quality pine and redcedar stems. Trees to be retained were marked prior to mulching. The resulting stand had 60 square feet of basal area comprising primarily 10-inch d.b.h. Virginia pines. Canopy cover was reduced to 65 percent, ranging from 58 to 87 percent across the stand. Shortly after the density-reduction treatment, the 2018 Mid-Atlantic Derecho storm event caused major blowdown and after 1 year windthrow had further reduced basal area to 55 square feet per acre (Fig. 1a,b).

Figure 1.—Stand conditions immediately after stem density reduction with a forestry mulcher (upper photo) and understory response 1 year later (lower photo), Fairfax County, VA. Courtesy photos by Owen Williams, Fairfax County Park Authority, used with permission.

The forestry mulcher left a deep chip layer on the forest floor that made seedling planting and tree tube installation difficult. In addition, the mulch layer did not deter invasive species and there was concern over the potential increase in fuel loading. In summer 2018, the site was chemically treated to reduce invasive species. Restoration of desired species required careful herbicide application and skilled contractors who could distinguish native species to be left untreated. Treatment included both foliar and bark treatments, using backpack sprayers and a spray unit mounted on an all-terrain vehicle. Broadleaf plants (*Acer, Ulmus, Fraxinus*) and vines (*Lonicera, Vitis, Smilax, Toxicodendron, Celastris, Parthenocissus quinquefolia,* and *Persicaria perfoliata*) were treated with triclopyr and grass (*Microstegium*) was treated with clethodim; glyphosate was used as a backup for spot treatments.

In January 2019, a two-layer three-strand solar-powered electric fence was installed around the project site to protect any naturally regenerating stems and stump sprouts. The fence was 85 percent successful based on a deer-browse survey. The cost of this installation was $1 per linear foot, and it required 7 hours per 10 acres each month to repair damage and remove limbs from fence lines, tighten lines, remove interfering vegetation, and check and troubleshoot the power source.

The first planting was done in March 2019. This smaller-scale test planting took place on 3 acres, with a planned 435 trees per acre to be planted. Species included white oak, northern red oak, mockernut hickory, and pignut hickory. Trees were sourced from a local nursery (unknown seed source and genetic diversity) and were 1-0 bare-root seedlings averaging 1 foot in height. No specifications were provided for seedling quality. Planting was done with a hoedad with a blade 4 inches wide and 6 inches long. Seedlings were fitted with a 5-foot-tall Tubex® tree shelter (Berry Global, Aberdare, South Wales, UK) that was staked and fitted with a bird netting. Tree tubes were difficult to install due to the deep layer of mulch and shelters were prone to toppling. After one growing season, only 40 percent of the seedlings had survived.

Several errors occurred during the planning and execution of this planting effort.

- Minimal supervision of the planting activities was conducted. A consulting forester had been hired for the project and was responsible for supervising the planting crew, but quality control was incomplete.
- Characterization of site hydrological conditions was partially inaccurate and as a result, a hydric area was planted with upland species, leading to high mortality.
- Bitternut hickory (*Carya cordiformis*) had been substituted by the nursery for pignut and mockernut hickory without approval from project managers. This error was discovered by project managers as the planting began and the substitution was allowed so that the project would not be delayed.
- The planting contractor planted only 270 trees per acre instead of the contracted rate of 435 trees per acre. This error was not discovered until the end of the growing season during survival surveys.

In December 2019 the second planting took place on 20 acres. We planted white oak, northern red oak, pin oak (*Q. palustris*), and swamp white oak (*Q. bicolor*), along with mockernut and pignut hickories, matching species to microtopographic and moisture

regimes. As part of the contract, we specified seedling characteristics to include a 24-inch shoot length, 3/8-inch minimum root collar diameter, five first-order lateral roots, and root pruning to 8-inch root length (Clark et al. 2000, Dey et al. 2008, Kormanik et al. 1994). The goal was to plant 800 trees per acre (Stringer et al. 2010) and tree tubes were not installed because the site was fenced. However, the planting crew arrived on-site with seedlings that did not meet the specifications. The trees appeared to be 1-0 seedlings with a shoot length approximately 12 inches long and a root collar diameter smaller than the minimum requested. A decision was made to commence with the planting because of budget constraints and timelines. Trees were planted with hoedads (4-inch by 8-inch blades), as no specifications were contracted for the planting tool. Each planted tree was marked with a pin flag to facilitate future survival surveys. Survival surveys took place in fall 2020 and found a 37 percent survival rate of plantings. An average of 565 seedlings per acre were recorded, of which 53 percent were naturally regenerating volunteers.

We conducted one additional planting for December 2021 using more advanced nursery protocols. For this planting, we worked with the contractor to meet seedling specifications identified for the second planting to improve success, as has been recommended for hardwood species (Clark et al. 2000, Kormanik et al. 1994). We planted the same oak and hickory species using a mix of KBC bars (Jim-Gem® KBC bar, Forestry Suppliers, Inc., Jackson, MS) with a blade 6 inches wide and 10 inches long and hoedads of similar dimensions. The contractor was able to meet the 24-inch shoot length specification for all seedlings; however, only 60 percent of the white oak (7,000 out of 12,000 seedlings) had a shoot length of 24 inches, as is typical for this species and stock type (Clark et al. 2022, Kormanik et al. 1994). Remaining white oak seedlings had shoot lengths between 12 and 18 inches. FCPA personnel were on-site for the entire planting operation, and ensured stock specifications were consistently met, roots were not overpruned, seedlings were planted at the proper depth, and the entire area was planted.

MANAGEMENT IMPLICATIONS

While there were many lessons learned in applying the best science available to this restoration effort, we did succeed in moving this stand toward more desirable species composition and structure. Over 30,000 trees were planted from 2019 through 2021. Survival was 37 to 40 percent (not including the 2021 planting) and 50 percent of the trees per acre were volunteers. Seedlings planted in 2021 have not yet been evaluated, but we hope for improved success from using larger seedlings. Treating nonnative invasive species was a colossal undertaking, and without multiple herbicide treatments, the site would not be progressing toward our desired condition but would be dominated by those species. Herbicide treatment was complicated by a need to selectively spray and to recognize (and not treat) native herbaceous community development. Forty percent of the project cost was spent on multiple invasive plant treatments over 5 years. Over that same time, 27 percent of the project cost was for staff, and 20 percent went to planting practices (Table 2). We have succeeded in transforming a dense and heavily invaded stand of Virginia pine into a pine savanna with an incredible diversity and abundance of native forbs and grasses, but much work remains.

Table 2.—Restoration treatment costs used to move a degraded Virginia pine-eastern redcedar stand toward northern hardpan basic oak-hickory forest in Fairfax County, VA

Activity	Cost per acre	Total cost	Percentage of total cost
Archaeological survey	$900	$28,500	6
Consulting forester	$175	$5,000	1
Shelterwood cut (forestry mulching)	$1,300	$30,000	6
Invasive and woody species control	$8,800 over 5 years	$185,000	40
Deer fences	$500	$10,000	2
Artificial regeneration	$2,235	$91,700	20
Staff time	$6,000 over 5 years	$125,000	27

We decided that moving this stand toward an oak-hickory forest type using a uniform shelterwood prescription was an excellent starting point. Eastern redcedar and other low-quality hardwoods were targeted for removal. Due to a lack of local markets for smaller diameter and lower quality trees, higher costs were incurred to conduct this phase of our restoration prescription. The use of the mulcher to remove the midstory and understory did have multiple benefits, including changing understory light levels and setting back competition to a more manageable state. As part of the prescription, we assessed the spacing and density of the pine and removed a few pine trees by thinning from below with the mulcher within existing dense clumps; our goal was to increase light to the understory and create a uniform spacing of codominant and dominant pines (Dey et al. 2012). A residual overstory of Virginia pine created understory light levels conducive to growing oak and hickory, but the heavy infestation of undesirable species was also stimulated.

Enriching the understory with additional oak and hickory seedlings accelerated the process of moving this stand toward an oak-hickory forest type. It was unnecessary to plant redbud as this species will naturally regenerate. Choosing oak and hickory species best suited to these site conditions (soils) and altering our planting regime to match species to microsite conditions was of paramount importance. One difficult specification to control was seedling seed source; while seedlings planted should be from local seed sources, these species remain largely undomesticated as tree improvement programs have dwindled in the last few decades (Wheeler et al. 2015). Additionally, some nurseries and contractors may not be well acquainted with procedures for developing and planting high-quality hardwood seedlings and often go outside their state or region to obtain seed (Clark et al. 2022). We used these enrichment plantings to encompass a range of 50 to 100 seedlings per acre, as the goal was to supplement any natural reproduction that occurs; 50 to 75 trees per acre will result in an oak-dominated stand at maturity. In addition, if the resultant stand is overstocked with oaks and hickories, density reduction can be done later.

Because of the high threat of browsing, the desired reproduction, regardless of natural or supplemental origin, had to be protected. Tree shelters have been proven to protect seedlings from browse and increase early height growth without being detrimental to the seedlings themselves (Ponder 2003, Schuler et al. 2005, Tuley 1985, Ward et al. 2000). Both fencing and tree shelters require additional time and resources for installation and maintenance, but research from areas of similar deer browsing pressure

in Pennsylvania and West Virginia has shown that without some type of protection to the desired reproduction the stand will not regenerate as preferred (Royo and Stout 2019), and many more trees would need to be planted (Ward et al. 2000). Thinning or caging individual oaks can be done by concentrating on releasing and protecting individual crop trees selected to best ascend to dominance at maturity. Thus, not all oak or hickory seedlings need protecting or thinning release during regeneration and recruitment into the overstory. When considering tubing or caging seedlings to protect them from deer browsing, a tree height of 5 to 6 feet is a common threshold to increase the likelihood that deer will not get to the terminal leader. Planting higher quality (i.e., larger) seedlings is the most practical method to get seedlings above browse line sooner and achieve recruitment objectives (Clark and Schlarbaum 2019, Ward et al. 2000). One unexpected problem we encountered was that planting and tree shelter installation in this deep mulch layer were difficult.

One key to success under the project conditions described was to control the competition. We erroneously thought that the layer of pine-cedar mulch that was created with the initial treatment would serve as an excellent barrier to other vegetation that may compete with planted seedlings. The initial shelterwood treatment and mulching did not accomplish this. The invasive species were treated chemically (herbicide) several times during the 5-year project as dense and diverse populations of invasive plants were difficult to eradicate with a single treatment. In addition to treating the invasive species, benefit was gained from also treating with herbicide any undesirable hardwood or pine species that invaded the site post-mulching and that were in direct competition with the desired species. These repeated treatments of competition control were imperative to any success of this prescription. If herbicides are not an option, then prescribed burning done periodically may be a suitable alternative for controlling competing vegetation and invasives (Dey et al. 2012); however, fire would preclude any planting as young newly planted seedlings will not respond positively to fire until root systems are well established. Depending on the species, invasive control may require a combination of mechanical cutting, herbicide application, and prescribed burning.

As previously mentioned, planted seedlings were indirectly impacted by the deep chip layer because it made planting and shelter installation difficult. Damage to reproduction with harvest activities was not a concern, as well-established seedlings and saplings will sprout if their aboveground structures are disturbed, and the establishment time under the initial shelterwood treatment will enhance root development, thus improving the sprouting ability. It will be important to assess the removal of the residual overwood at some time in the future, depending on how the site responds to the initial disturbance, seedling survival and growth, and stand composition and structure. The "shelterwood period" is the best opportunity to manage or direct the future composition of the replacement stand, so care should be taken to monitor the site and conduct the final release harvest when the next stand is in place. Further, natural mortality of the pine over time mimics a natural disturbance regime, which may be better for growth and recruitment of hickory (Pile Knapp et al. 2021). Because Virginia pine has a propensity to die back and have somewhat sparse crowns, retaining the overstory of pine indefinitely and not conducting a second treatment/harvest may also be an option. Unless the Virginia pine market drives a different decision, maintaining a two-aged forest on this site seems biologically, economically, and socially acceptable.

CONCLUSIONS

This project is a unique collaboration between management and scientists that was forged during the implementation stage of the restoration project and is not meant to be a research project. However, it serves as part of a larger learning network as implementation of large-scale field-based research is difficult and takes decades to yield meaningful results. Additionally, forest management science is largely place based. Results can vary dramatically from site to site and year to year, and depend on nuanced differences in prescriptions such as planting dates, seedling or stock types, herbicide type, and harvesting equipment. We hope this project can serve as an example for others working on hardwood restoration projects in the Central Hardwood region.

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A Decade of Growth at Two Hybrid American Chestnut Plantings in West Virginia

Melissa Thomas-Van Gundy[1]

Abstract

In 2011, American, Chinese, and two generations of hybrid chestnut seedlings were planted in east-central West Virginia. The original objectives of these plantings were to determine whether any BC_3F_3 hybrid families were phenotypical of the American chestnut and blight resistant. Summarized here are the 10-year results for growth and mortality on two sites. A composite ranking combining height and diameter growth was constructed to explore differences between BC_3F_3 families. Mortality for all seedling types in 2021 was 60 percent on one site and 69 percent on the other site. Mean diameter at breast height after 10 years for all BC_3F_3 hybrid families was 4.3 cm on the first site and 3.3 cm on the second site. Differences in diameter growth between the BC_3F_3 hybrids were not significant at either site. The large differences in family success between the two sites, as seen in the composite ranking scores for families planted on both sites, underscore the importance of integrating native genotypic diversity into American chestnut breeding and restoration efforts.

INTRODUCTION

The forests of the eastern United States have been impacted by nonnative species since the first settlers arrived from Europe, with some species experiencing substantial negative changes. The history of the American chestnut (*Castanea dentata*) in the forests of the eastern United States and its functionally extinct status due to chestnut blight (*Cryphonectria parasitica*) and root rot disease (*Phytophthora cinnamomi*) are well documented (Crandall et al. 1945, Smith 2000, Youngs 2000). American chestnut now persists in the eastern forest as understory sprouts that may reach reproductive age (Paillet 2002, Woods and Shanks 1959) but are neither contributing to mast production nor dominating the canopy as in the past.

The loss of overstory American chestnut affected species composition across the range with northern red oak (*Quercus rubra*), chestnut oak (*Q. prinus*), and white oak (*Q. alba*) increasing in importance (Keever 1953). On mesic sites, hemlock (*Tsuga canadensis*) and yellow-poplar (*Liriodendron tulipifera*) replaced American chestnut (Elliott and Swank 2008). If diseased trees were not removed, individual trees could take 2 to 10 or more years to die from the blight, allowing adjacent trees to gradually fill in the resultant gaps (Woods and Shanks 1959).

Efforts to create blight-resistant or -tolerant American chestnut hybrids started not long after the disease effects were realized (Anagnostakis 2012). The USDA Bureau of Plant Industry began crossing American chestnut with Asian species to produce trees with American chestnut form and Asian resistance (Clapper 1954a, 1954b; Graves 1942).

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These efforts later became part of the Connecticut Agricultural Experiment Station's work on restoring American chestnut (Anagnostakis 2012). The backcross method of breeding was introduced into these efforts in 1986 (Burnham et al. 1986). The American Chestnut Foundation™ (TACF), founded in 1983, began with the mission to backcross blight resistance from the Chinese chestnut (*C. mollissima*) into the American chestnut (Ellingboe 1994, Hebard 2001). TACF has produced a third generation of the third backcross (BC_3F_3) hybrid predicted to have blight resistance similar to the Asian species and the "timber" form of the American chestnut (Hebard 2001).

West Virginia lies in the center of the former range for American chestnut (Little 1977), where the species was estimated to compose about 12 percent of the overstory (Brooks 1910). Based on trees in land surveys, American chestnut was a relatively minor component of European-contact forests in north-central West Virginia with importance values of 1.5, 1.7, and 2.4 in three counties (Rentch and Hicks 2005). Witness trees for the area that later became the Monongahela National Forest (MNF), in east-central West Virginia, show that about 6 percent of the trees listed in deeds were American chestnut. The species was not evenly distributed across the MNF, ranging from 4 to 18 percent of witness trees depending on ecological subsection (Thomas-Van Gundy and Strager 2012).

In 2011, about 1,750 seedlings of American chestnut, Chinese chestnut, and two generations of American chestnut hybrids (BC_3F_2 and BC_3F_3) were planted at 2 sites on the MNF in West Virginia. There were 23 families of BC_3F_3 hybrid American chestnuts represented on the 2 planting sites. The original objectives of the plantings were to determine: (1) to what degree the BC_3F_3 hybrid seedlings resemble American chestnuts in form and growth habit, (2) to what degree the BC_3F_3 hybrid seedlings are resistant to naturally occurring blight, and (3) how long resistance persists in these hybrids. However, since 2011, blight resistance in the BC_3F_3 hybrids has been shown to be lower than expected (Steiner at al. 2017).

These hybrid seedlings were from the TACF traditional breeding program. Under TACF's 3-Bur initiative—breeding, biotechnology, and biocontrol—traditional breeding is expected to continue with stringent selection of the most blight- and phytophthora root rot-resistant parents, additional generations of intercrossing the selections, and integration of the transgenic blight-tolerant American chestnut containing the oxalate oxidase gene into the breeding program (Westbrook et al. 2019). Selected families of hybrid chestnuts could also be crossed with the transgenic chestnut under the biotechnology initiative. Progeny test planting, like the two sites in West Virginia, represent the near real-world restoration planting conditions to which the hybrid seedlings would be subject. As such, findings from these plantings, along with orchard studies of blight resistance, will aid in the selection of American chestnut hybrid growing stock for restoration of this iconic species.

The analysis presented here is part of the long-term monitoring and measurement of these progeny tests and as such are interim results with initial study objectives listed earlier. In this report I present the mean diameter at breast height (d.b.h.) growth of the seedlings in 2016, 2017, and 2021 by site and seedling type (American chestnut, Chinese chestnut, BC_3F_2, and BC_3F_3) and report mortality by site and seedling type. To incorporate an expanded measure of growth and tree form, a composite growth ranking was created for the BC_3F_3 hybrid families, combining previously reported height growth (Thomas-

Van Gundy et al. 2017) and the 2021 d.b.h. growth reported here. Based on previous work and genetic studies, I expect that the mean traits of the hybrid seedling types will be intermediate between the American and Chinese types. Where possible, statistical differences will be determined between seedling type and between families of BC_3F_3 hybrids.

METHODS

Study Areas

This study was originally described in detail when an analysis of height data was conducted (Thomas-Van Gundy et al. 2017), but the site and planting methods will be summarized here to address the objectives for this analysis. The planting on the Cheat Ranger District of the MNF (Cheat site) included 7 acres of a commercial clearcut harvest, fenced to reduce deer (*Odocoileus virginianus*) browsing in 2008, and located in Tucker County near the town of Saint George, WV (39°08′56.7″N 79°41′50.1″W; Fig. 1). The planting on the Gauley Ranger District of the MNF (Gauley site) is a 2-acre portion of a larger clearcut located in Webster County above Sawyer Run, a tributary to the Williams River near the town of Cowen, WV (38°23′57.8″N 80°26′28.4″W; Fig. 1). At the Cheat site, herbicide (2 percent Razor®Pro [Nufarm Americas, Alsip, IL] applied to cut stumps) was used to control red maple (*Acer rubrum*) stump sprouts before planting, and on the Gauley site VisPore® mats were used to control competing vegetation as trees were planted. On both sites, planted seedlings were released from overtopping vegetation, once with herbicide in fall 2011 on the Cheat site, and once through hand clipping in spring 2013 on the Gauley site. Naturally occurring American chestnut saplings were noted on both sites before harvest and planting, and soil tests found no *Phytophthora* on either site before planting.

The Gauley site is higher in elevation (about 1,000 m) than the Cheat site (760 m) and both sites have moderate site indices (index species of northern red oak): 72 for the Gauley site and 75 for the Cheat site. The Gauley site faces mainly southwest, and the Cheat site faces northwest. Soils are mapped on the Gauley site as the Clifftop-Laidig association and on the Cheat site as Gilpin. Surface soil on the Gauley planting site appears higher in sand content than the Cheat site and there are scattered sandstone boulders on the forest floor of the Gauley site.

Seedlings were planted in March and April 2011 in rows about 2.4 m apart with about 2.4 m between seedlings as site conditions allowed. The seedlings were 1-year-old bare-root stock that had been grown at the Virginia state nursery near Waynesboro, VA. The BC3F3 seedlings were grown from nuts produced in seed orchards at TAFC's farm near Meadowview, VA. Most of the American chestnut ancestors of the trees in the seed orchard were in the vicinity of Mount Rogers National Recreation Area, VA, but some were farther south in the Appalachian Mountains. Mount Rogers National Recreation Area is approximately 225 km from the Gauley planting site, about 300 km from the Cheat planting site, and 2 to 3 degrees of latitude south of the two planting sites.

A resolvable incomplete block design (Patterson and Williams 1976) was chosen for this study to address the concerns of expected mortality, the number of families to test, and differences in planting site conditions. Approximately 875 trees were planted at each site from 35 families with 25 replications/seedlings of each family.

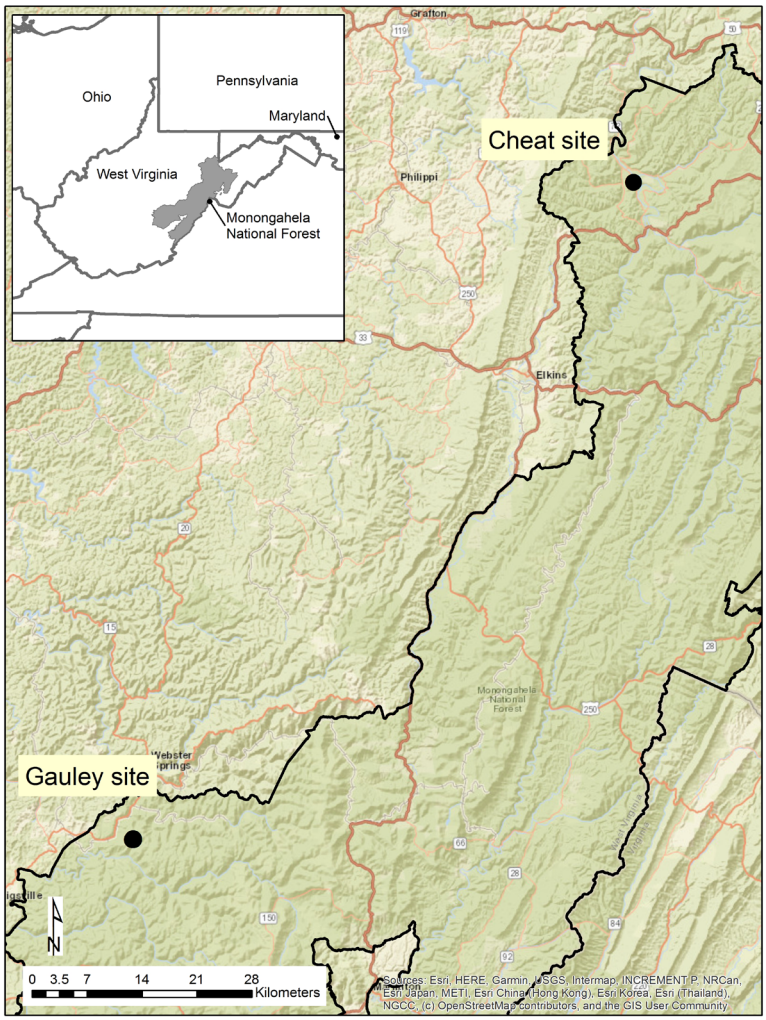

Figure 1.—Map of study areas and vicinity, Monongahela National Forest.

The seedlings were arranged in seven 5-tree incomplete blocks with this blocking replicated 25 times across the site. We arbitrarily chose a 5-tree block to minimize environmental variation among experimental units. Seedling types represented in these plantings are pure American chestnut, pure Chinese chestnut, BC_3F_2 hybrids, and BC_3F_3 hybrids. For more details on the planting design see Thomas-Van Gundy et al. (2017).

Data Collection and Analysis

In 2016, 2017, and 2021, live trees at least 1.4 m tall were measured for d.b.h. to the nearest 0.25 cm. Survival, damage, and form were noted for each seedling using qualitative descriptions when diameters were taken. Mean d.b.h. by year and type (hybrid generation, American, and Chinese) were compared through logistic regression (PROC GLIMMIX, SAS 9.4; SAS Institute 2016) as a resolvable incomplete block design. Fixed effects in the model were time, type, site (Cheat or Gauley), and the interactions between type and time, and type, time, and site. The gamma distribution was used to model the data with log used as link function and degrees of freedom calculated by the Kenward-Roger approximation. Complete block number and incomplete block number were included as random variables. Unequal variance was accounted for by the inclusion of year as a random variable with variance components modeled as autoregressive structure. Significant differences (α =

0.05) were determined though least square means comparisons with p-value adjustments for pairwise comparisons made through the Tukey-Kramer procedure. The sites were analyzed separately due to large differences in initial mortality; the sites were also analyzed separately for height growth (Thomas-Van Gundy et al. 2017).

Because determinations of good or poor form in young seedlings can be subjective, another method for gauging how similar these families are to the desired growth habits was developed. The BC_3F_3 families were ranked by d.b.h. growth (mean d.b.h. in 2021) and total height growth (data from Thomas-Van Gundy et al. 2017). The families ranked from 1 to 23 (largest/tallest to smallest/shortest) on the Cheat site and 1 to 17 on the Gauley site. The two rankings were averaged to create a mean score.

Descriptive statistics were used for percentage of mortality (number of dead compared to total trees each year) by seedling type and site. Mortality was calculated by seedling type and includes stems that resprouted. Given the extensive mortality on the Gauley site, several families included only a few individual stems by 2021. Stems in the following families were dropped from d.b.h. growth analysis: D2-26-72 (1 stem), CH297 (3 stems), D2-28-52 (3 stems), D2-50-15 (3 stems), D4-18-30 (3 stems), W1-31-63 (3 stems), and D2-28-42 (4 stems). A total of 20 trees from 7 families (1 BC_3F_2, 6 BC_3F_3) were removed from analyses of d.b.h. change; all stems were included in the calculation of mortality.

RESULTS

Mortality

Current mortality, the number of seedlings dead in 2021, on the Cheat site is at 60 percent across all types; seedlings of the Chinese type had the lowest mortality, with 51 percent dead in 2021. BC_3F_2 and American seedlings showed the highest mortality at about 60 percent for each type (Fig. 2a). Current mortality is higher on the Gauley site at about 69 percent across all seedling types (Fig. 2b). Chinese chestnut seedlings have the lowest mortality at 28 percent and BC_3F_2 the highest at 80 percent.

Differences Between Types

As with the height data (Thomas-Van Gundy et al. 2017), a significant difference in mean d.b.h. was found between sites ($p < 0.001$); therefore, mean d.b.h. results are reported by site. On the Cheat site the largest mean d.b.h. in 2021 was found in the BC_3F_3 type with an average of 6.1 cm and the lowest in the Chinese chestnut type at 3.3 cm (Fig. 3; Table 1). In all time periods, mean d.b.h. of BC_3F_3 seedlings was significantly larger than the Chinese type (Fig. 3).

On the Cheat site, mean change in d.b.h. in 2021 was statistically different between types ($p = 0.0253$); however, when the errors for multiple comparisons are controlled, only the difference between BC_3F_3 and Chinese seedling types was statistically significant ($p = 0.0319$). The largest mean change in d.b.h. from 2016 to 2021 was in the BC_3F_3 type at 2.7 cm. The greatest mean change in d.b.h. on the Cheat site was in trees of the D2-50-15 family at 4.1 cm.

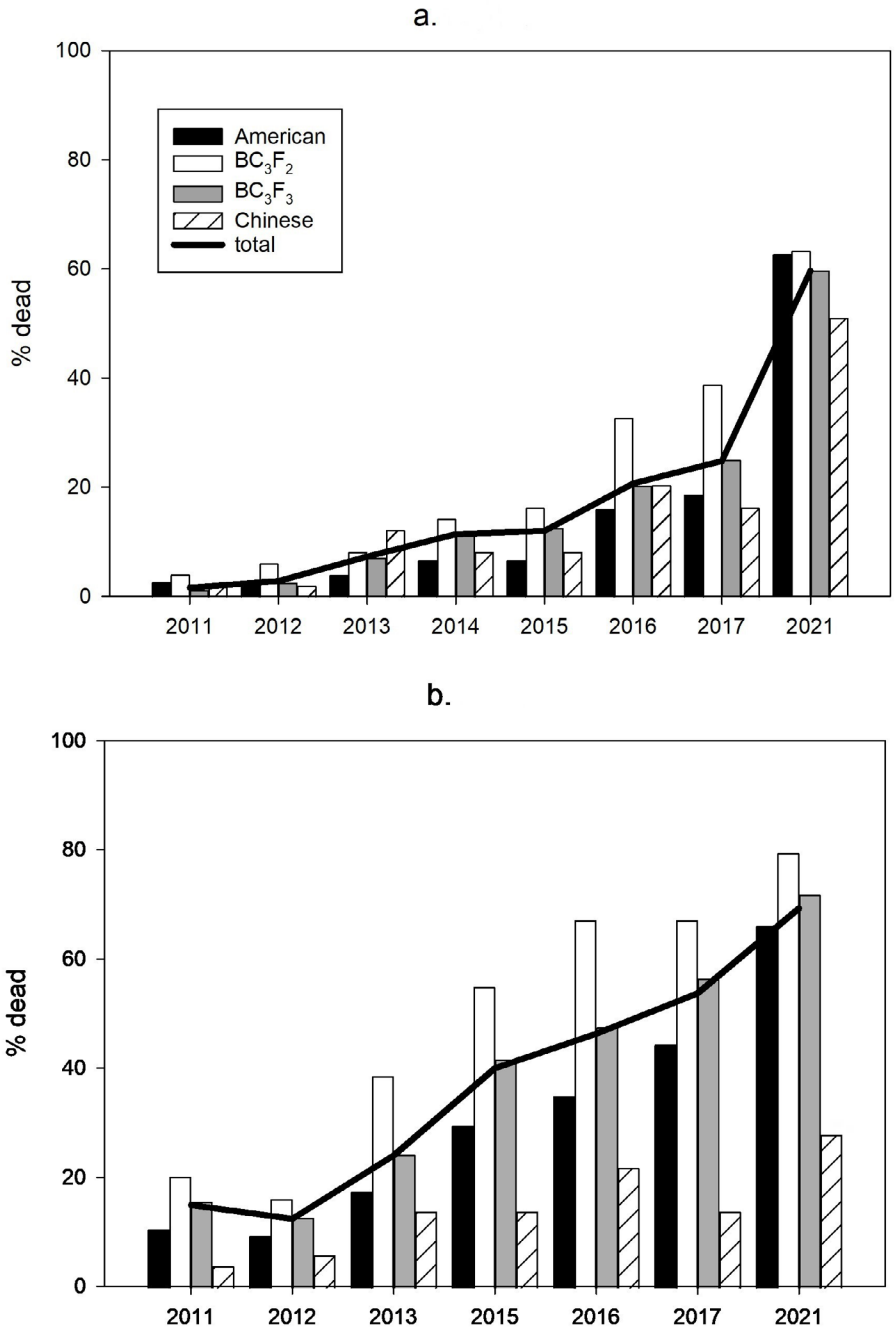

Figure 2.—Chestnut seedling mortality from 2011 to 2021 by site, Cheat (a) and Gauley (b), and type, Monongahela National Forest.

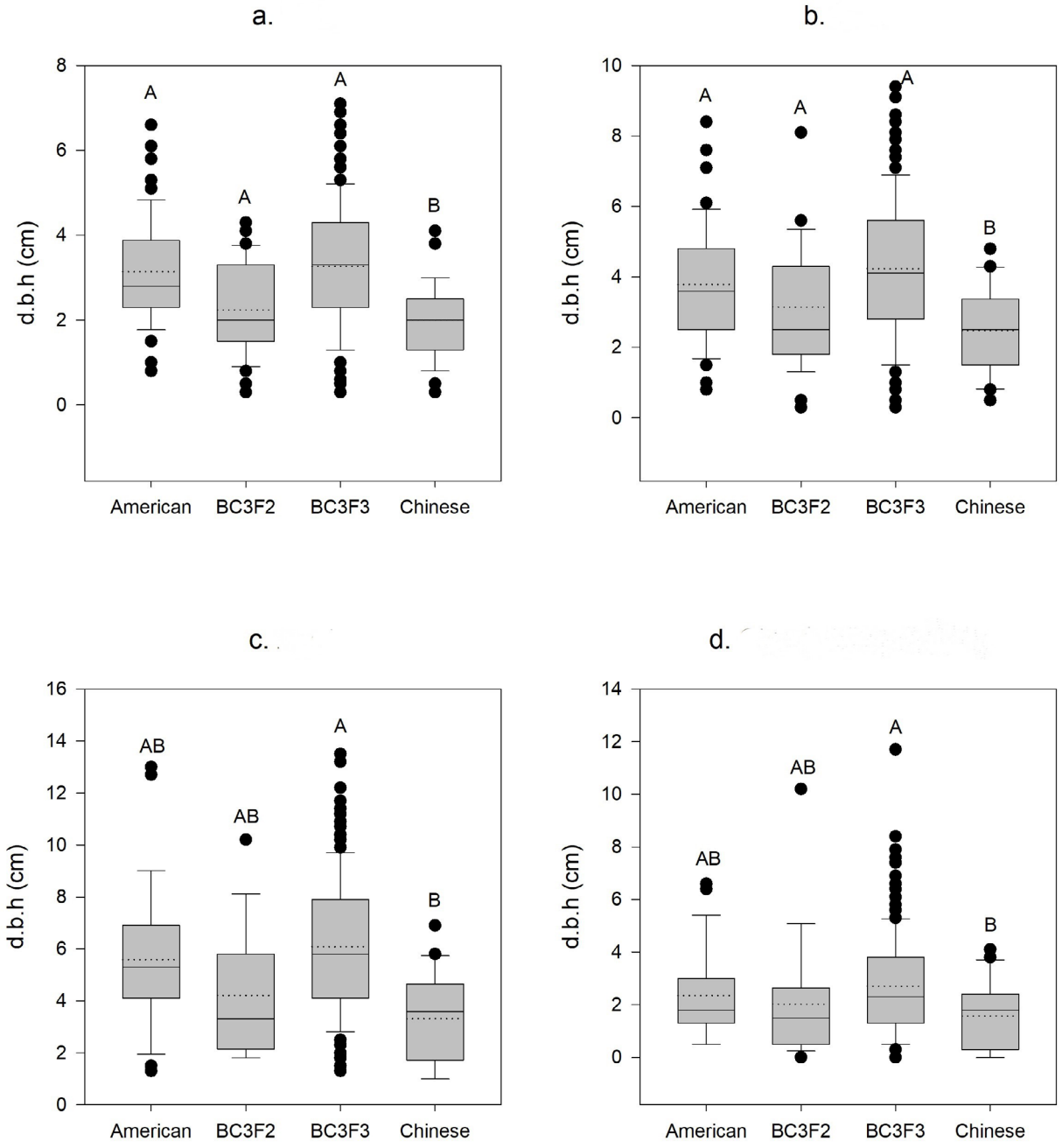

Figure 3.— Mean diameter at breast height (d.b.h.) in 2016 (a), 2017 (b), and 2021 (c) and mean change in d.b.h., 2016 to 2021 (d) for the Cheat site by chestnut seedling type, Monongahela National Forest. Dashed line is the mean, solid line is the median, box outlines the 5th and 95th percentiles, and all outliers are graphed. Means with different letters are significantly different (α = 0.05).

American chestnut seedlings on the Gauley site averaged 6.1 cm d.b.h. in 2021 and were the largest type on that site. Seedlings of the Chinese type had the lowest mean d.b.h. on the Gauley site at 3.0 cm (Table 2). Mean change in d.b.h. was statistically significant between seedling types on the Gauley site ($p = 0.0078$), with American chestnuts significantly larger than Chinese (Fig. 4). No statistically significant differences were found between mean change in diameter of American chestnuts and either hybrid type or between the two hybrid types. The greatest change in d.b.h. was found in the American type with a mean of 2.4 cm.

Table 1.—Mean diameter at breast height (d.b.h.) and standard error (SE) for live stems only for 2016, 2017, and 2021, and change in mean d.b.h. by chestnut seedling type on the Cheat site, Monongahela National Forest

Type	Mean d.b.h. (SE), cm, 2016	Count, 2016	Mean d.b.h. (SE), cm, 2017	Count, 2016	Mean d.b.h. (SE), cm, 2021	Count, 2021	Change in mean d.b.h. (SE), cm, 2016–2021	Count, 2021
American	3.1 (0.17)	58	3.8 (0.23)	55	5.6 (0.53)	28	2.4 (0.33)	28
BC_3F_2	2.2 (0.19)	31	3.0 (0.33)	27	4.3 (0.58)	17	2.0 (0.58)	17
BC_3F_3	3.3 (0.07)	483	4.3 (0.10)	458	6.1 (0.15)	275	2.7 (0.11)	271
Chinese	2.0 (0.16)	31	2.5 (0.20)	30	3.3 (0.38)	22	1.6 (0.28)	21
All types	3.1 (0.06)	604	4.1 (0.08)	570	5.8 (0.15)	342	2.6 (0.10)	337

Table 2.—Mean diameter at breast height (d.b.h.) and standard error (SE) for live stems only for 2016, 2017, and 2021, and change in mean d.b.h. by chestnut seedling type on the Gauley site, Monongahela National Forest

Type	Mean d.b.h. (SE), cm, 2016	Count, 2016	Mean d.b.h. (SE), cm, 2017	Count, 2016	Mean d.b.h. (SE), cm, 2021	Count, 2021	Change in mean d.b.h. (SE), cm, 2016–2021	Count, 2021
American	2.3 (0.19)	40	3.3 (0.28)	39	6.1 (0.53)	24	2.4 (0.33)	28
BC_3F_2	1.7 (0.23)	14	2.5 (0.41)	14	4.6 (0.81)	6	2.0 (0.58)	17
BC_3F_3	2.3 (0.08)	292	3.0 (0.13)	260	5.3 (0.25)	164	2.7 (0.11)	270
Chinese	1.3 (0.10)	33	1.5 (0.10)	38	3.0 (0.20)	34	1.6 (0.28)	21
All types	2.2 (0.07)	379	3.0 (0.10)	351	5.1 (0.20)	228	2.6 (0.10)	336

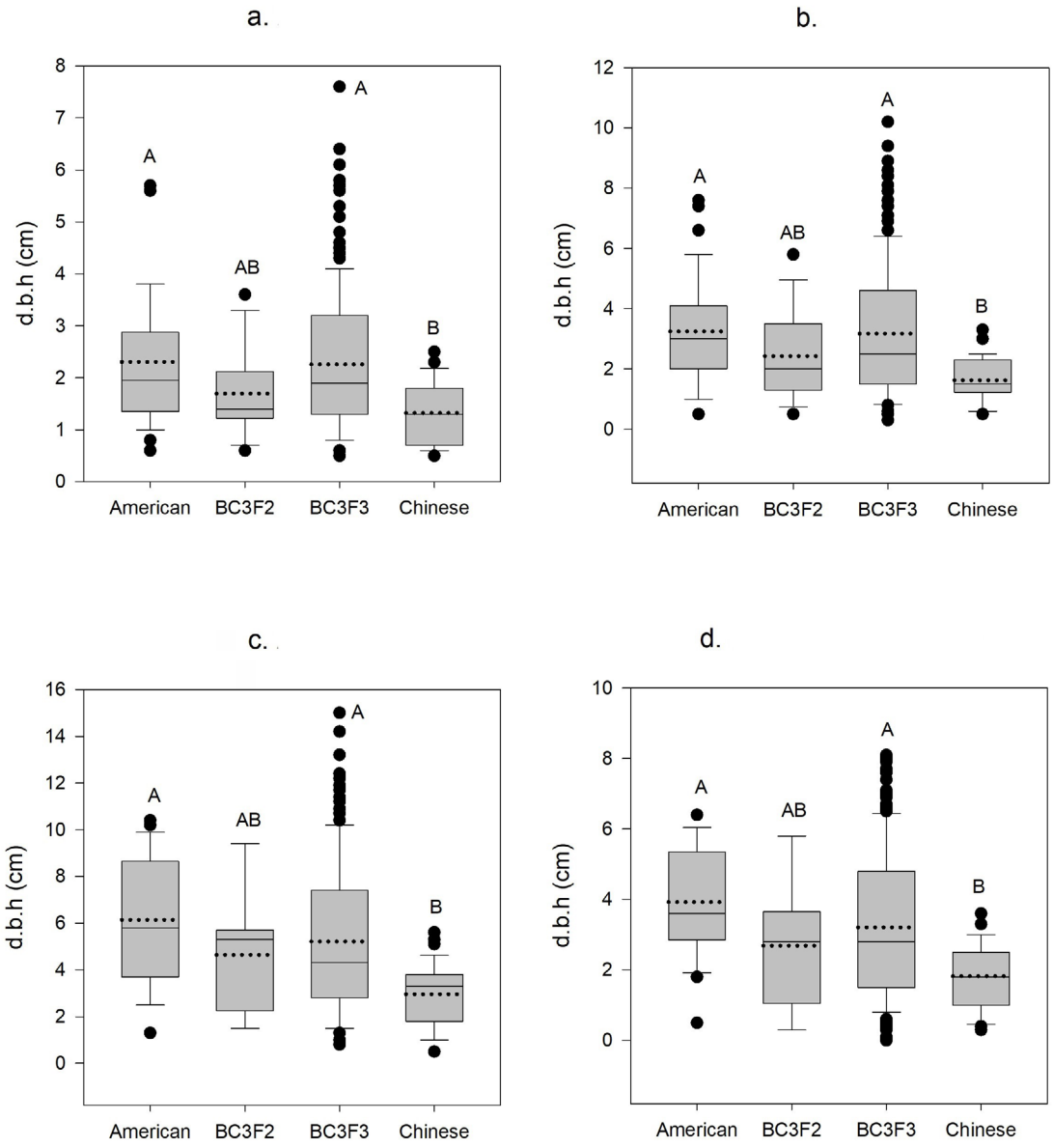

Figure 4.— Mean diameter at breast height (d.b.h.) in 2016 (a), 2017 (b), and 2021 (c) and mean change in d.b.h., 2016 to 2021 (d) for the Gauley site by chestnut seedling type, Monongahela National Forest. Dashed line is the mean, solid line is the median, box outlines the 5th and 95th percentiles, and all outliers are graphed. Means with different letters are significantly different (α = 0.05).

Differences Between Families

Site and family were found to be significant in explaining differences in the change in d.b.h. for the BC_3F_3 families. Significant differences between BC_3F_3 families were found on the Cheat site (Fig. 5), with seedlings in the D5-17-130 family significantly smaller than seedlings in either the D2-50-15, D4-20-65, D5-26-54, D5-27-101, or W1-31-65 family. No statistically significant differences in mean change in d.b.h. between BC_3F_3 families were found on the Gauley site when p-values were adjusted for multiple comparisons (Fig. 5).

The composite rankings created by combining d.b.h. growth from this analysis with past data on height growth shows some interesting differences between families at the two study sites (Table 3). D5-26-54 and D4-20-65 hybrid families displayed good growth on both sites. In 2021, there were 16 individuals on each site for D5-26-54 and 10 (Cheat) and

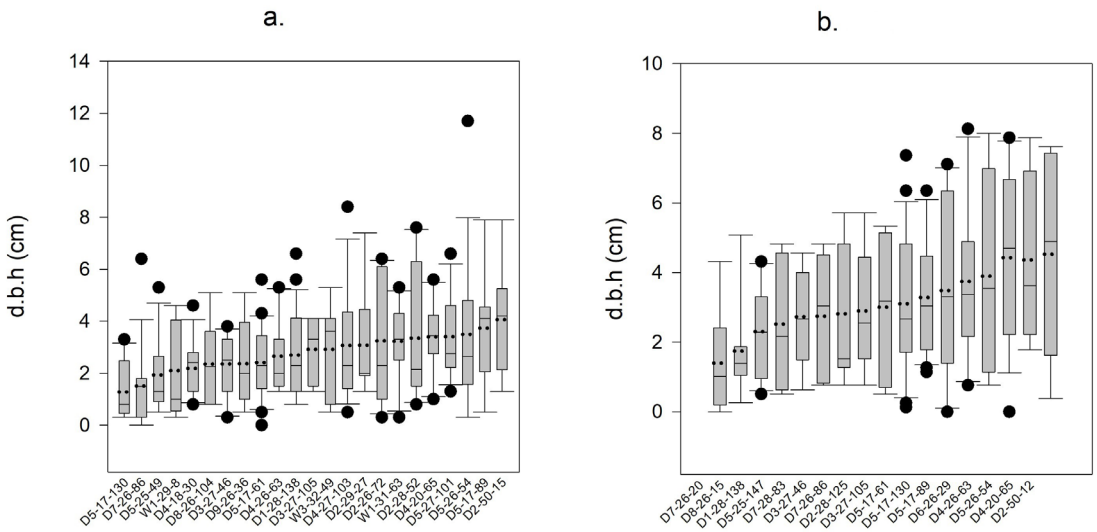

Figure 5.— Mean change in diameter at breast height (d.b.h.) from 2016 to 2021 for BC3F3 families by site, Monongahela National Forest. Dashed line is the mean, solid line is the median, box outlines the 5th and 95th percentiles, and all outliers are graphed.

Table 3.—Composite score results for BC_3F_3 chestnut families based on combining rankings by total height growth and mean growth in diameter at breast height (d.b.h.). Families were ranked from 1 to 23 on the Cheat site and 1 to 17 on the Gauley site with 1 assigned to the family with the greatest mean d.b.h. or tallest mean height. Outlined families are those planted on both sites on the Monongahela National Forest.

Family	Composite score for Cheat site planting	Composite rank for Cheat site planting	Family	Composite score for Gauley site planting	Composite rank for Gauley site planting
D2-26-72	1.5	1	D2-50-12	2	1
D5-26-54	4	2	D5-17-89	4	2
W3-32-49	5	3	D4-26-63	4	2
D4-20-65	5	3	D5-26-54	5	3
D5-27-101	5	3	D4-20-65	5	3
W1-31-63	5	3	D2-28-125	6.5	4
D2-50-15	8	4	D7-28-83	7.5	5
D5-17-89	9.5	5	D3-27-105	7.5	5
D2-29-27	9.5	5	D5-17-61	9	6
D3-27-46	10	6	D6-26-29	9	6
D4-18-30	10.5	7	D5-25-147	10	7
D2-28-52	10.5	7	D1-28-138	12	8
D3-27-105	12	8	D5-17-130	12.5	9
D4-26-63	14	9	D7-26-86	14	10
D1-28-138	15.5	10	D8-26-15	14.5	11
D5-17-61	15.5	10	D7-26-20	15	12
D4-27-103	15.5	10	D3-27-46	15.5	13

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Table 3 (continued).

Family	Composite score for Cheat site planting	Composite rank for Cheat site planting	Family	Composite score for Gauley site planting	Composite rank for Gauley site planting
D9-26-36	16	11	[No additional data]		
D8-26-104	19.5	12			
D5-25-49	20	13			
W1-29-8	21	14			
D7-26-86	21.5	15			
D5-17-130	23	16			

6 (Gauley) alive for the D4-20-65 family. Family D2-26-72 stems had a high composite ranking on the Cheat site based on only 11 trees and family D4-26-63 had a high composite ranking on the Gauley site but based on only 7 trees.

DISCUSSION AND CONCLUSIONS

Differences in mortality between the two sites have been apparent since the first year after planting (Thomas-Van Gundy et al. 2017). While site environmental factors and some differences in site preparation (weed mats or herbicide) cannot be completely ruled out, early mortality on the Gauley site appeared to be related to the use of mechanized equipment to remove treetops before planting. As reported previously (Thomas-Van Gundy et al. 2017), first-year mortality on the Gauley site appeared as a linear pattern that tracked the lightly used skid trails created during this site preparation. Yearly observations have found that few of these stems have resprouted.

As found for the height growth of these third-generation hybrids, mean d.b.h. growth was intermediate between American chestnut and Chinese chestnut types, supporting part of the first hypothesis investigated. Statistical differences between the sites for d.b.h. growth were found, contrary to the second part of the first hypothesis. Site differences were also found for height growth (Thomas-Van Gundy et al. 2017). For the second hypothesis, the rankings for third-generation hybrid families on the Cheat site did not meaningfully change over time, however. Given the significance of the interaction of time and family on the Gauley site, there are meaningful differences in rankings on this site. On this higher elevation site, planted seedlings have been noted as experiencing early leaf-out and frost or freeze damage. Perhaps these harsher site conditions are impacting the stability of the hybrid family rankings on this site. Tracking of the families by this composite ranking system will continue.

Desired characteristics in these hybrids, which may be used to restore chestnut to eastern forests, include rapid height and diameter growth, and tall, single stem form. While descriptions of stem form can be subjective and reflect site conditions as well as genetic potential, d.b.h. and height growth can be used as a proxy for the desired American chestnut growth habit. Although no statistical differences between families could be documented for either site, family D5-17-130 is beginning to stand out here in West Virginia and in plantings in Virginia. This family has shown relatively high blight resistance in field and orchard settings (Clark et al. 2019). However, on average,

seedlings of this family do not appear to have the growth habit of American chestnut when compared to other BC_3F_3 hybrids. The mean ranking for family D5-17-130 on these West Virginia sites is last on the Cheat site and fifth to last on the Gauley site. However, these results are based on relatively few seedlings: 25 individuals on the Gauley site and 18 on the Cheat site.

There is a need to determine whether the hybrid seedlings will function within the ecosystem similarly to American chestnuts. A pilot study of soil fauna on the Cheat site found no functional difference in non-Acarina fauna between hybrids and pure American chestnuts, suggesting that hybrid chestnuts will not negatively affect soil and litter fauna (Naylor et al. 2018). Based on leaf and twig morphology, 96 percent of the third backcross seedlings produced by the TACF breeding program were found to differ from Chinese chestnuts and resemble American chestnuts (Diskin et al. 2006). This finding gives support to the idea that the third-generation hybrids will match American chestnuts in ecological attributes and functions.

In a description of the breeding program under which the hybrid seedlings studied here were developed, the difficulties of the large numbers of trees needed for the effort and the extended timeframes involved are noted as limiting progress (Steiner et al. 2017). In their summary of progress to date, Steiner et al. note the differences between hypothesized blight resistance of the hybrids and realized resistance. This difference, they note, is most likely because resistance is controlled by more than three gene loci; however, the large number of hybrid lines (or families) developed should allow for some capture of resistant alleles from the parent trees (Steiner et al. 2017). In the 10 years since the establishment of this study, new methods for genotyping and assessment have been developed by TACF (Westbrook et al. 2019). Based on current methods of assessing ancestry and a blight index developed by TACF, hybrid families D5-26-54, D4-20-65, and D4-26-63 are considered to have 100 percent *C. dentata* ancestry and poor blight resistance.[2] TACF had no data on hybrid families D2-26-72 and D5-17-130, suggesting that the representative trees of this hybrid were removed from the Meadowview Research Farm. Though this may suggest these hybrid families are no longer important for future restoration, individual trees from these hybrid families may still be of importance to future breeding, despite recent findings suggesting that some alleles for blight tolerance were lost in some backcross lines (Westbrook et al. 2019). Individuals from high ranking third-generation hybrids on these two study sites could be used in new backcross efforts to diversify restoration efforts to include the blight-tolerant Darling 58 American chestnut created by insertion of an oxalate oxidase gene in the genome of an American chestnut from New York (Westbrook et al. 2021).

Wild native American chestnut saplings were present on both sites before timber harvest and experimental planting. The third-generation backcross hybrids used in this study are crosses of the Clapper and Graves lines of chestnuts originating from trees found near Glenn Dale, MD crossed with native American chestnuts from higher elevation mountain sites in Virginia. Initial results of sowing nuts of the American chestnut outside its native range (New Hampshire) to evaluate the possibility of assisted migration determined that survival of the seedling was controlled by competition with immediate neighbors and cumulative winter injury due to cold intolerance (Clark et al. 2022).

[2] Personal communication from Director of Science J. Westbrook, TACF, April 18, 2022.

The nuts in the New Hampshire assisted migration study and the study reported on here are from the same set of backcross trees. The large differences in family success between the two sites, as seen in the composite ranking scores of families planted on both sites, underscore the importance of integrating genotypic diversity into American chestnut breeding and restoration efforts. Though American chestnuts from the uplands of West Virginia will not have resistance to the blight, they would be better adapted to northern latitudes.

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Conifer Component

Development of a Norway Spruce Stand on a Gauged Watershed in the Fernow Experimental Forest, West Virginia

Mary Beth Adams, Benjamin Rau, Jacob Kemner, and Charlene Kelly[1]

Abstract

In 1973, a 55-acre watershed at the Fernow Experimental Forest (FEF) in West Virginia was planted with 2-0 Norway spruce (*Picea abies*) seedlings to evaluate the effects of conifers on streamflow and water quality. This conversion required considerable effort to ensure survival of the Norway spruce "island" in the sea of hardwoods. After nearly 50 years of tree growth and stand development, we report the impacts of the forest type conversion on stand and biomass development, water yield, water quality, soils, and nutrient cycling within the watershed. Conversion from hardwoods to Norway spruce resulted in significant decreases in annual water yield, relative to the reference watershed. Stream water quality was also significantly altered, with virtually no nitrate being detected in the stream water and a significantly lower stream pH, in contrast to the other FEF watersheds. The stream channel has also been altered. The forest floor in the spruce watershed exhibits an accumulated mass and increased carbon and nitrogen storage. However, mineral soil carbon and nitrogen storage are significantly lower relative to the hardwood reference. We discuss the implications of this unusual research project.

INTRODUCTION

Long-term watershed research studies allow scientists to evaluate the effects of manipulations over the lifetime of trees and forest ecosystems. Such research also improves understanding of ecosystem processes and their variability. In this paper we report on one such study conducted on the Fernow Experimental Forest (FEF) in West Virginia, where a gauged watershed was planted to Norway spruce (*Picea abies*) in 1973. We report on nearly 50 years of stand development, water yield, water chemistry, soil nutrients, and soil carbon (C) and compare the conifer watershed with a nearby native hardwood watershed of the same stand age. Conversion of native forests to planted conifer stands is a common anthropogenic landscape alteration. Understanding ecosystem biogeochemistry in these altered systems is necessary for management of ecosystem services and forecasting future ecosystem processes. Alterations to canopy architecture, litter chemistry and mass, and root exudation processes affect the chemical and biological processes in forest soils, which may influence ecosystem C and nitrogen (N) retention or loss.

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METHODS

Site Description

Watershed 6 (WS6) is a 22.34-ha, south-facing catchment located on the FEF. Prior to clearcutting in 1964 through 1969, the stand was mature second-growth mixed hardwood, and the dominant tree species were white oaks (*Quercus alba*) and chestnut oaks (*Q. montana*), with a basal area of 33.06 sq. m/ha (Patric and Reinhart 1971). Elevation ranges from 730 to 830 m. Soils in the watershed are mapped as Calvin channery silt loam (loamy-skeletal, mixed, active mesic Typic Dystrudepts), and as Cateache channery silt loam (fine loamy-mixed active, mesic Ultic Hapludalfs) (Adams et al. 2020).

Conversion from Hardwood to Conifer Species

A 120° V-notch weir was installed in November 1956 (weir coordinates 39.06612N, 79.67698W) and the initial calibration period occurred between November 1956 and February 1964. Between March and October 1964, the lower half of the watershed was clearcut, which removed 51 percent of the original basal area, followed by the removal of pulpwood and all other standing vegetation greater than 1 inch in diameter. Between May 1965 and October 1969, the lower half was maintained barren with herbicides (Patric and Reinhart 1971). Between October 1967 and February 1968, the remaining upper half of the watershed was clearcut, removing 49 percent of the original stand basal area. From May 1968 to October 1969, the upper half was also maintained barren with herbicides.

Natural regeneration occurred from 1970 to 1973, mostly from seed sources from the surrounding forest. During March and April 1973, 2-0 Norway spruce seedlings were planted at a 3.05 m × 3.05 m spacing, resulting in 1,077 trees/ha; however, notes in the study file suggest the trees were planted more closely than the original design.[2] Follow-up herbicide treatments in August 1975 and September 1980 were performed to release the Norway spruce trees from competing hardwood tree species, hay-scented fern (*Dennstaedtia*), and blackberry (*Rubus*) (Wendel and Kochenderfer 1984).

Stand growth and development were measured periodically during the next 50 years (Adams et al. 2020). Allometric equations for calculating aboveground and belowground biomass are described in Kelly (2010). Streamflow and stream chemistry were monitored routinely (Rau et al. 2022), and several studies of N and C biogeochemistry have been added (Kelly 2010, Kemner 2020). The nearby hardwood watersheds used for reference purposes (WS7 and WS4) are described in Adams et al. (2012).

RESULTS

Stand Development

The initial stocking survey in December 1975 found 1,371 live trees/ha (Adams et al. 2020). Repeated herbicide treatments allowed the Norway spruce trees to survive and develop. As measured in 2018, total spruce basal area was 39.2 sq. m/ha, and density was around 824 spruce trees/ha. By that point, a hardwood tree component had developed. Hardwood basal area was 4.1 sq. m/ha (9.5 percent of total stand basal area) and the hardwood tree density was 270 stems/ha (about 25 percent of spruce density; Fig. 1). The most common hardwoods in WS6 include yellow poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*).

[2] All study files and reports referenced are on file at the USDA Forest Service Timber and Watershed Laboratory, 459 Nursery Bottom Road, Parsons, WV 26287.

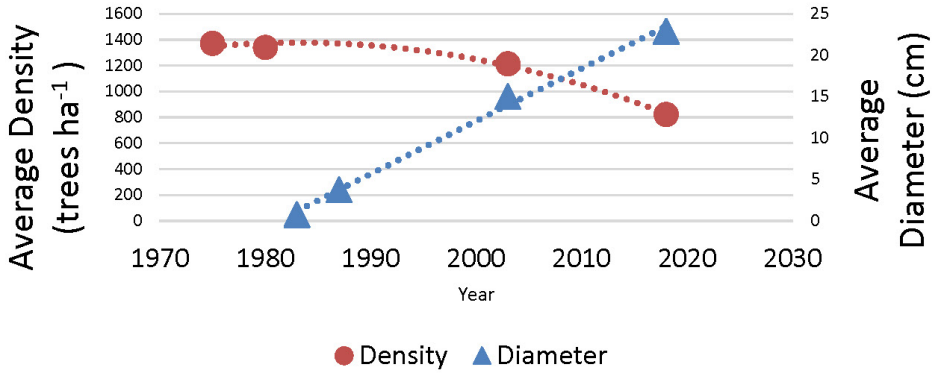

Figure 1.—Average density and diameter of Norway spruce on watershed 6, Fernow Experimental Forest, measured over time. Spruce trees were planted in 1973.

As measured in 2008, total aboveground biomass estimates were approximately 30 percent less in WS6 (116,800 kg/ha) relative to WS7 (166,000 kg/ha). Estimates of belowground biomass were approximately 45 percent lower in WS6 (21,000 kg/ha) relative to WS7 (38,000 kg/ha; Kemner 2020). Total biomass of both watersheds increased since 2008, though live tree biomass in WS6 increased more (136 percent) compared to WS7 (49 percent) as measured in 2020. However, total biomass values did not differ significantly between the watersheds (WS6: 260,471 kg/ha, WS7: 242,660 kg/ha; Kemner 2020).

Streamflow and Stream Chemistry

During the pretreatment period (1957–1964) stream discharge from WS6 (and WS7) closely followed the discharge of WS4, the long-term reference watershed. Discharge increased in WS6 and WS7 after harvesting and herbicide treatments relative to WS4 until the late 1980s (Fig. 2) (Rau et al. 2022). Beginning after 1994, annual discharge from WS6 decreased significantly relative to both WS4 and WS7. Analysis of seasonal stream discharge (growing season versus dormant season) revealed that dormant season discharge in WS6 decreased dramatically beginning in the late 1980s relative to both WS4 and WS7, and that growing season discharge declined faster in WS7 relative to WS6. Overall, the spruce vegetation has dramatically decreased stream discharge relative to the hardwood reference watersheds, especially in the dormant season.

One of the most striking trends is the absolute decline in N exports in stream water from WS6 (Fig. 3) (Kemner et al. 2020, Rau et al. 2022). This is a result not only of decreased streamflow volume—mostly due to changes in dormant season flows—but also decreased concentrations of nitrate (NO_3-N) in stream water. WS7, a 50-year-old hardwood stand, exports approximately 8 kg/ha/ year NO_3-N, while NO_3-N exports from WS6 have been nearly zero for almost 40 years (mean of about 0.6 kg/ha/year; Adams and Kochenderfer 2007, Kelly 2010, Kemner et al. 2020). Ammonium export is minimal, as it is from most FEF streams (Adams and Kochenderfer 2007, Adams et al. 1994, Kelly et al. 2011a) and dissolved organic N (DON) concentrations are also very low from WS6 relative to WS7 (Kelly 2010). Annual DON export from WS6 was also very low (mean of 0.48 kg ha/year).

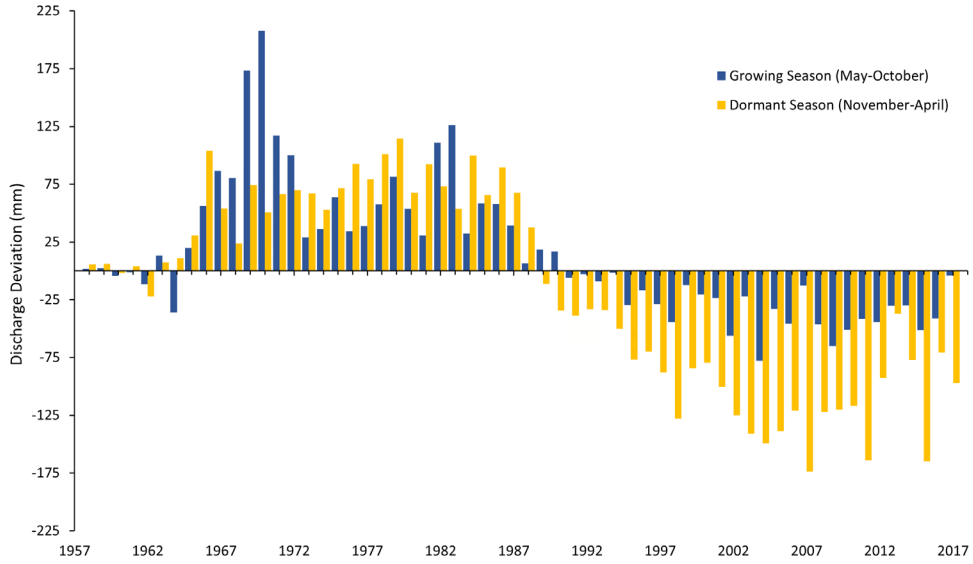

Figure 2.—Growing season and dormant season stream discharge from watershed 6, Fernow Experimental Forest, measured as the departure from the predicted discharge, 1960–2018. Watershed 4 was the reference watershed used to calculate the prediction equation.

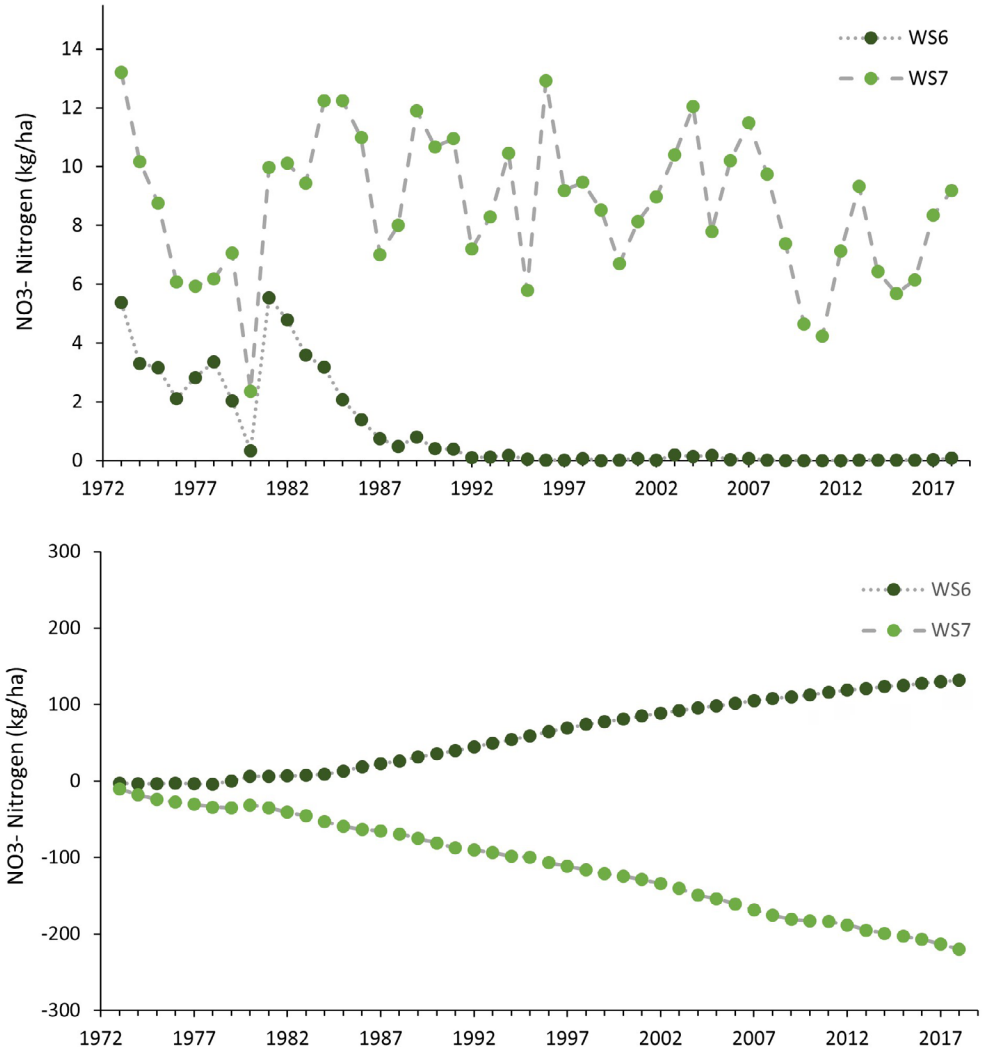

Figure 3.—Export of nitrate (NO_3-) nitrogen over time from watershed 6 (WS6; Norway spruce) and watershed 7 (WS7; hardwood reference), Fernow Experimental Forest, 1973–2018. Upper panel is annual export, and bottom panel represents a cumulative flux.

Soil Properties

The deforestation experiment had a nearly immediate impact on the soil, with most of the litter layer (L layer or Oi horizon, estimated at 2,300 kg/ha) removed by the end of the first year of devegetation (Troendle et al. 1974). The F+H (Oe+Oa) layer also appeared to decrease in depth, but the net weight of organic matter content remained consistent throughout the first 4 years. Roots in the upper 15 cm of soil decreased linearly during the 4 years, as did soil bulk density (rock-free). Carbon content (measured as loss on ignition) decreased in the A horizon, as did the infiltration rate (from 250 cm/hour to 50 cm/hour; Troendle et al. 1974).

However, Kelly et al. (2011a) described the forest floor (O horizon) of WS6 approximately 45 years after planting (2008) as a relatively thick horizon (2 to 8 cm) of nondecomposed needles (Oi) above further decomposed organic material of spruce origin (Oe+Oa). The forest floor C content in WS6 was 3.814 kg C/ha (Kelly 2010) and was significantly greater than that of the hardwood watershed (WS7; 2,343 kg C/ha). In 2020, forest floor biomass had increased in both watersheds, although to a much greater extent in WS6, and C content for WS 6 was 24,034 kg C/ha (Kemner 2020), almost 3 times the 8,322 kg C/ha in WS7.

In 2008, mineral soil from WS6 contained significantly less C and N than soil from WS7 in both A- and B-horizons (Fig. 4). For example, in 2008, A-horizon soil WS6 contained approximately 17 percent less C (kg/ha) than spruce soil from WS7, and also a smaller pool of N than WS6. Mean annual net nitrification rates were significantly less under spruce WS6: 4.2 kg NO_3-N/ha/year compared with 144.8 kg NO_3-N/ha/year under hardwood WS7 (Kelly 2010, Kelly et al. 2011b).

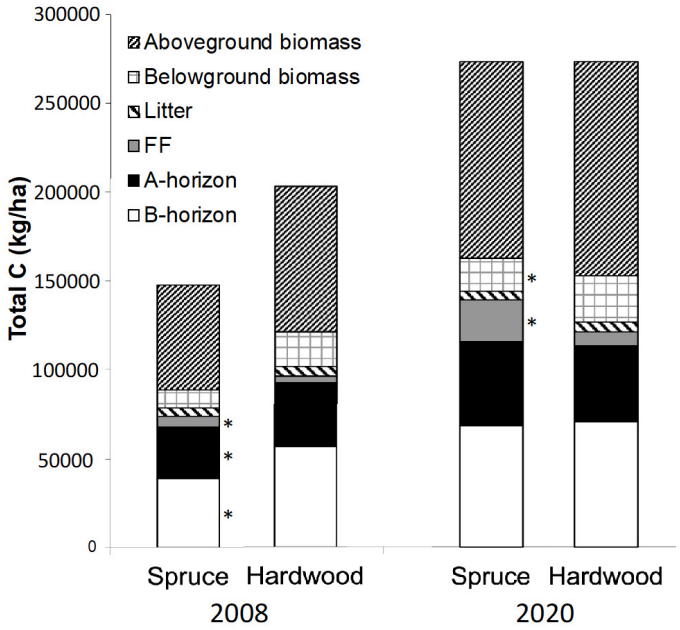

Figure 4.—Watershed budget depicting carbon (C) content contained within each soil and biomass compartment in the spruce (watershed 6) and hardwood (watershed 7) watersheds, Fernow Experimental Forest. Asterisks denote significant differences between watersheds in the associated pool in the respective year (α = 0.05) (Kemner 2020). FF = forest floor.

Rates of C and N accumulation changed, however, and by the 2020 assessment, no significant differences were detected in soil C or N content between watersheds in either soil horizon (Kemner 2020). Moreover, in the 2020 assessment, inorganic N content in WS6 as extracted from field-fresh soil is less than one-fifth the content in WS7 soil (WS6 = 0.97; WS7 = 5.3 mg inorganic N/kg soil; $p < 0.001$). When total inorganic N is separated into NO_3-N and NH_4-N pools, soil from WS6 has significantly less NO_3-N than soil from WS7 (WS6 = 0.25; WS7 = 4.3 mg/kg soil; $p < 0.001$), though there is no significant difference in extractable NH4-N between watersheds.

Soil aggregate distribution in WS6 exhibited a strong shift to dominance by micro-aggregates (53 to 250 μm and less than 53 μm) relative to soil from WS7), which was observed in both A- and B-horizons (Kemner 2020, Kemner et al. 2020). This may be a function of macro-aggregate dispersion via acidification of soil (A-horizon soil pH WS6 = 3.85, WS7 = 3.97; B-horizon soil pH WS6 = 4.03, WS7 = 4.16) and lower microbial activity and byproducts. Changes in aggregation are likely related to more recalcitrant litter, and acidic conditions in WS6 would result in decreased bacterial activity and therefore a greater proportion of organic matter occurring in larger aggregate size classes or as particulate organic matter unassociated with soil particles.

DISCUSSION

The Norway spruce seedlings planted on WS6 in 1973 have matured into a stand that approximates the level of productivity of the native hardwood forest above-ground and below-ground. We estimate crown closure occurred in the late 1990s or early 2000s, based on observations during our 2003 measurements. At that time, the stand was very dense, with little self-pruning having taken place little light hitting the forest floor, resulting in large populations and diversity of mushrooms (see Figure 5 in Adams et al. 2020). As the trees continued to grow and the stand developed, however, self-pruning occurred and more light reached the floor of the watershed; hardwoods began to grow as well. Currently, the stand on WS6 contains a little less than 25 percent hardwood stems (density). There is some evidence of spruce regeneration occurring, albeit limited (M.B. Adams, personal observation, July 2018).

The planted spruce also resulted in changes in the soil on the watershed, particularly related to C storage. The deforestation experiment prior to planting eliminated the forest floor (at least the Oi layer) from both WS6 and WS7, yet within a short period of time two distinctive forest floors had developed. The forest floor (O horizon) eventually accumulated significantly more C in WS6, relative to that in the hardwood WS7. The mineral soil in WS6 became more acidic, as a result of the uptake and leaching from the conifer litter. There were differences in other chemical and physical soil properties, including soil pH and aggregation. There is also evidence of a spodic E-horizon in some locations in the watershed (Kemner et al. 2020), which can be an important property linked to the presence of spruce (Nauman et al. 2015). All these changes point to significant alterations in C cycling and storage due to afforestation with spruce. Such changes were observed only after decades of tree growth and development, however. The results of other studies support these observations. Other research has reported that soils under conifer forests generally store more organic C throughout the soil profile compared to hardwood stands (Chiti et al. 2012, Nauman et al. 2015, Ohno et al. 2017). In addition, Diochon et al. (2009) showed that for red spruce, soil organic C stocks shrank during the

first 45 years of stand development, but grew quickly after that point, similar to what has been observed on WS6.

Changes in C cycling will also alter the cycling of other nutrients. Our data show that the rate of N accumulation in the spruce watershed is accelerating in the most recent sampling relative to that of the hardwood system, and is likely linked with the rate of accumulation of C. There likely occurred a significant loss of both C and N immediately after conversion from hardwoods to Norway spruce, and as the stand on WS6 ages, this spruce watershed may store more biomass, C, and N relative to the hardwood forest of the same age. We are still unable to account for the large long-term discrepancy of N export from the two watersheds, however.

Finally, changes above-ground and below-ground had important implications for water use and water quality. Decreased annual stream discharge in WS6 relative to the hardwood watersheds is attributed to greater annual transpiration by the spruce and potentially less throughfall in the spruce vegetation. Although not measured, there was likely greater interception by the higher leaf area on the spruce watershed, with the possibility of greater evaporation back into the atmosphere, resulting in less water reaching the soil. A thicker forest floor on WS6, of more recalcitrant litter, also likely contributed to more water being held in the rooting zone of the spruce relative to the hardwood forest floor.

CONCLUSIONS

Despite the large investment required up-front to establish the Norway spruce stand on WS6, the current stand is productive and reasonably stable at 50 years of age, although native hardwood trees are again becoming more common. The species conversion, from hardwood to conifer, significantly altered the soil properties, water use, and biogeochemistry of the stand. More carbon is being stored in the conifer stand, which is encouraging in a world where elevated carbon dioxide threatens to lead to drastic changes in climate. Elevated carbon storage and evidence of spodic horizon podzolization have significance for land managers interested in restoring native red spruce to the landscape of the central Appalachians after 100 years of dominance of northern hardwoods. Considerations for water availability, carbon cycling, nutrient cycling, and costs of restoration should be included in all plans for species conversion, as the effects are likely to be slow to develop and long-lasting.

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Structure and Productivity of a 65-Year-Old Eastern White Pine Plantation 20 Years After Three Levels of Commercial Thinning in the Southern Appalachian Mountains

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Abstract

Eastern white pine (*Pinus strobus;* EWP) is a commercially valuable conifer of the Appalachian Mountain region that is highly productive in monoculture plantings and is well suited as a timber crop for private landowners. Little information is available on growth responses to commercial thinning in EWP plantations. The purpose of this unreplicated case study was to evaluate the response of a 45-year-old plantation of EWP on a low-quality site to three levels of partial harvest treatments (thinning), which consisted of reducing stand basal area by either 0 percent (no harvest, control), 33 percent, or 66 percent. After 20 years of growth following thinning, residual stand basal area increased 27, 52, and 72 percent, respectively, with similar responses of mean tree diameter, and cubic and board foot volumes. Study results provide resource managers with information for evaluating potential economic returns from mid-rotation commercial thinning.

INTRODUCTION

Eastern white pine (*Pinus strobus;* EWP) occupies the Appalachian Mountain region of eastern North America from northern Georgia to Labrador, Canada and west to the Great Lakes region, where it is a minor component in mixed hardwood stands (Cope 1932, Kimberly 1933). EWP is a valuable commercial timber species, particularly in the New England states, which can be successfully regenerated naturally and by planting, is highly productive, and responds to management (Wendel and Smith 1990). It has long been a favored species for reforestation plantings in the southern Appalachian Mountain region, as demonstrated by many trial plantings on the Biltmore Estate (Asheville, NC) between 1890 and 1900 (Haasis 1930). Planted EWP has also been demonstrated as superior to other species for increasing productivity of low-quality sites (Barrett 1933). EWP can be intensively managed in plantations for production of high-value products (Torbert et al. 2000, Yawney and Trimble 1958). More recently, EWP has been evaluated for reclamation of surface mined lands (Rodrigue and Burger 2004) and carbon sequestration (Leverett et al. 2021).

Stands of EWP respond well to management, such as pruning (Wright et al. 1980) and thinning (Casselman et al. 2007) for high-quality sawlog production. Following extensive exploitative logging in the southern Appalachians during the early 1900s, EWP was often planted to control soil erosion and increase forest productivity (Jemison 1946, Minckler 1941, Wahlenberg and Doolittle 1950). In the northern part of its range, stocking charts

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are available to maintain fully stocked EWP stands following thinning (Leak 1982, Leak et al. 1995). Response to thinning has been reported from only a few studies in Canada (Reátegui et al. 2021), New England (Seymour 2007), and North Carolina (Della-Bianca 1981). Except for a small case study on a deeply eroded site at the Biltmore Estate (Della-Bianca 1981), little information is available on the long-term stand response to commercial thinning at mid-rotation age of around 50 years in the southern Appalachians, where productivity tends to be higher compared to New England (Kimberly 1933).

Vimmerstedt (1962) provides growth and yield information of unthinned planted EWP stands in the southern Appalachians. However, intermediate stand management of planted stands of EWP has received little attention. In early 1997, a long-term demonstration of the response of EWP to thinning was established in a 45-year-old plantation in the Bent Creek Experimental Forest. The main objective of the investigation reported here was to describe the stand structure and productivity resulting from three levels of thinning in that plantation after 20 years. A secondary objective was to compare measured stand data from the plantation with the predicted values from other studies.

METHODS

Study Site

The study was installed in the Bent Creek Experimental Forest, in the Pisgah District of the Pisgah National Forest, south of Asheville, NC (35.4975N, -82.6131W). Nesbitt (1941) indicated the 3-acre study site was under private ownership from 1838 until 1914 and unlike nearby areas was apparently never cultivated. However, the tract likely served as a woodlot and provided pasture for livestock owned by settlers practicing subsistence farming. Following purchase by the USDA Forest Service in 1914 for establishment of the Pisgah National Forest, the tract was undisturbed until 1951, when the planting of EWP for the present study was established. Establishment records are not available for the plantation, but a 1958 map shows the tract as a demonstration for increasing productivity of low-quality sites, which was based on previous research on EWP in the experimental forest (Wahlenberg and Doolittle 1950).

Elevation of the study site is 2,200 feet; topography is nearly level (3 percent gradient). Soil series is primarily Tate, which is a fine-loamy, mixed, semiactive, mesic Typic Hapludult. Tate soils are deep (more than 40 inches) to felsic high-grade metamorphic rock parent material. Vegetation on these soils consists of a shade-intolerant overstory of mixed oaks (*Quercus*) and hickories (*Carya*). Oaks include southern red (*Q. falcata*), scarlet (*Q. coccinea*), and white (*Q. alba*); hickories include mockernut (*C. tomentosa*) and pignut (*C. glabra*). A shade-tolerant midstory consists of red maple (*Acer rubrum*), sourwood (*Oxydendrum arboreum*), and blackgum (*Nyssa sylvatica*). Oak site index averages 72 feet at 50 years. A sign on the site stated the area was planted in 1952, indicating the site was probably cleared of natural vegetation for the demonstration planting in 1951. Two-year-old EWP seedlings were probably obtained from a state nursery in February 1952 and then manually planted. Seedling spacing is unknown, but based on other EWP plantings of that time it was likely to have been 6-foot spacing in rows and 6 feet between rows, resulting in an initial density of approximately 1,200 seedlings per acre. The first growing season of the newly established EWP stand was during summer 1952.

Thinning Treatments

The 3-acre tract was subdivided into three nearly equal-size treatment plots. Each plot was assigned one of three basal area reduction treatments: 0, 33, or 66 percent, referenced as unthinned (or control), light, or heavy thinning treatments, respectively, in the following text. Widely spaced potential crop trees were selected in the heavy treatment, which resulted in free (not touching) crowns throughout. In the light thinning treatment, all intermediate and selected codominant trees were marked for harvest; no trees were marked for harvest in the control plot.

Marked trees were felled and moved to a loading area outside of the study area with a wheeled skidder at age 45. Although EWP is susceptible to the annosus root rot disease (*Heterobasidion annosum*) (Boyce 1962), the cut surface of stumps was not treated with borax as a prevention for inoculation by airborne spores of the disease because the hazard rating for infection was low (Kuhlman et al. 1976). All hardwoods greater than 4.5 feet tall were marked for cutting in the plots in the light and heavy thinning treatments. The volume and value of timber harvested from the two thinned plots are not available.

Tree Inventory and Data Summary

All residual trees were inventoried by diameter at breast height (d.b.h.) to the nearest 0.1 inch, one growing season after harvest, at 46 years of age. The d.b.h. of each residual tree immediately after thinning at age 45 was estimated using the relationship:

$$\text{d.b.h. (age 45)} = \text{d.b.h. (age 46)} - [\text{d.b.h. (age 46)}/46]$$

The d.b.h. and total height were measured on 10 randomly selected dominant and codominant trees in each treatment plot at age 49 and 10 years later at age 59. At age 65, the d.b.h. of all surviving trees on each plot was recorded.

Stand structure on each treatment plot was quantified by density, mean d.b.h., basal area, cubic feet, board feet, and predicted total height. Mean d.b.h. of residual trees on the treated plots was calculated as quadratic mean diameter (QMD) (Curtis and Marshall 2000). Regression was used to derive a model for predicting tree height from d.b.h. using data at age 49. That model was then used to predict tree heights of each tree in the 1997 and 2017 inventories, which were required for calculation of cubic volume to a 3-inch outside bark top diameter and sawtimber board feet volume (International ¼-inch) reported by Vimmerstedt (1962). All trees on each plot were included in this summary, including trees adjacent to plot boundaries and several hardwoods 4 inches d.b.h. or greater.

RESULTS AND DISCUSSION

Mean Diameter and Diameter Distributions

QMD at age 45 was 13.8, 15.4, and 16.9 inches for the 0, 33, and 66 percent thinning treatments, respectively (Table 1). During the two decades following thinning, QMD increased 26, 29, and 41 percent for the three treatments. Immediately following thinning, the d.b.h. range of residual trees was about the same across the three treatments, averaging

around 15 inches. After 20 years, the range of d.b.h. increased to about 20 inches on the 0 and 33 percent thinnings, and 15 inches on the 66 percent treatment. The diameter distribution of residual trees resembled normal distributions for the three plots at stand age 45, which is typical of even-aged stands of mid-tolerant species, such as EWP (Lorimer and Krug 1983) (Fig. 1). The d.b.h. distributions were skewed slightly toward larger sizes at age 65. The number of sawlog-size trees decreased about 9 percent on average between stand ages 45 and 65 years, resulting from natural mortality.

Basal Area

Before the harvest treatments, basal area of the plantation was approximately 226 square feet per acre based on the control plot (Table 1). During the 20 years after the harvest treatments, basal area increment ranged from 27 percent on the control treatment to 72 percent on the heavy thinning. Measured basal area for the control plot at age 65 was 287 square feet per acre, which was greater than the predicted amount of 198 square feet per acre based on a model developed from unthinned plantations sampled across a range of ages and sites in western North Carolina (Hepp et al. 2015). In another comparison, measured basal area of the control plot was greater than predicted (218 square feet per acre) from a growth and yield study of EWP in Ohio (Dale et al. 1989)

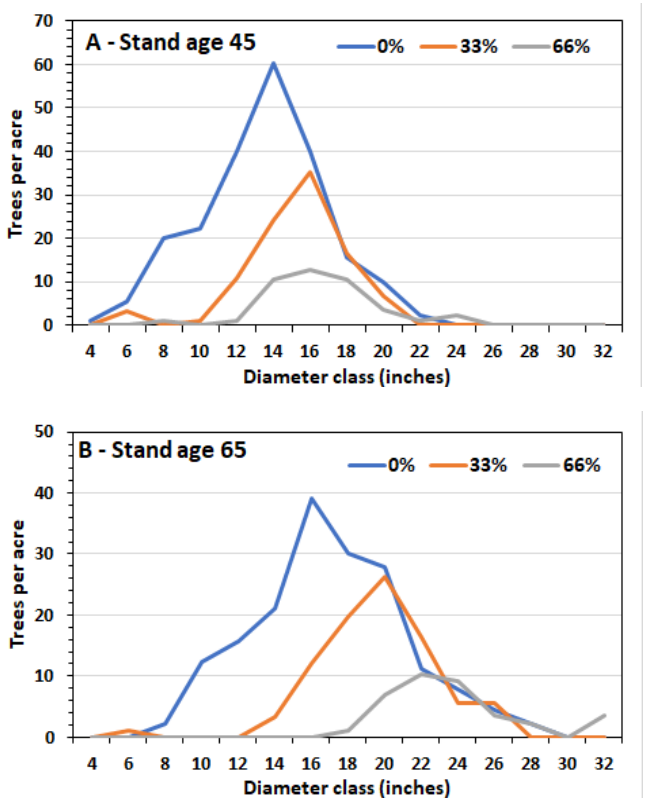

Figure 1.—Diameter distributions of an eastern white pine plantation in the Bent Creek Experimental Forest immediately following three partial harvest basal area treatments (A) and 20 years later (B).

Table 1.—Change in stand parameters between 1997 and 2017 resulting from three basal area reduction treatments in a plantation of 45-year-old eastern white pines in the Bent Creek Experimental Forest

Stand parameter	0 percent basal area reduction treatment; 1997 harvest	0 percent basal area reduction treatment; 2017 harvest	0 percent basal area reduction treatment; %	33 percent basal area reduction treatment; 1997 harvest	33 percent basal area reduction treatment; 2017 harvest	33 percent basal area reduction treatment; %	66 percent basal area reduction treatment; 1997 harvest	66 percent basal area reduction treatment; 2017 harvest	66 percent basal area reduction treatment; %
Density (n per acre)	217.4	173.9	-20.0	97.7	90.0	-7.9	42.8	37.0	-13.6
Quadratic diameter at breast height (d.b.h.; inches)	13.8	17.4	26.1	15.4	19.9	29.2	16.9	23.8	40.8
Minimum d.b.h.	5.0	7.8	56.0	5.1	6.8	33.3	8.8	17.5	98.9
Maximum d.b.h. (inches)	21.3	28.3	32.9	20.8	26.7	28.4	23.3	32.6	39.9
Sawlog trees (n per acre) [a]	171	154	-9.9	86	81	-5.8	36	32	11.1
Basal area (square feet per acre)	226.0	287.4	27.2	127.0	193.7	52.5	66.4	114.3	72.1
Volume (MCF per acre) [b]	10.8	14.0	29.6	6.1	9.5	55.7	3.0	5.4	80.0
Volume (MBF per acre) [c]	51.7	70.7	36.8	29.9	48.5	62.2	15.1	28.5	88.7
Production (thousand board feet per acre) [d]	--	70.7	--	--	70.3	--	--	65.1	--

[a] Sawlog-size trees: ≥10-inch class.

[b] MCF = thousand cubic feet.

[c] MBF = thousand board feet (International ¼-inch).

[d] Production = 2017 MBF/acre + 1997 harvest; 1997 harvest in 0 percent thinning = 0 MBF/acre; 1997 harvest in 33 percent thinning = 21.8 MBF/acre; 1997 harvest in 66 percent thinning = 36.6 MBF/acre.

Height, Live Crown Ratio, and Site Index

Mean heights of dominant and codominant trees at age 49 (4 years after thinning) across all treatments were relatively consistent (Table 2), averaging approximately 99 feet. The live crown ratio soon after the thinning treatments (49 years) was smallest for the control plot (35 percent) and nearly the same (40 and 39 percent) for the light and heavy thinning treatments, respectively (Table 2). Crown ratios after 10 years (59 years) differed only slightly from the earlier estimates. Mean site index of EWP (mean height of dominant and codominant trees at 50 years) for the study area is approximately 100 feet, determined from mean measured height at 49 years (99 feet) plus 1 foot for the estimated annual increment during the growing season at age 50.

Cubic Foot and Board Foot Volumes

Mean cubic foot volumes of EWP on the study tract before the thinning treatments were estimated from the control plot as 10.8 thousand cubic feet (MCF) per acre at stand age of 45 years and 14.0 MCF per acre at 65 years, resulting in an increase of 29.6 percent (Table 1). The 20-year increment of cubic foot volumes for the light thinning treatment was about 25 percent greater than for the control plot. Similarly, the increment of the heavy harvest treatment was about 25 percent greater than for the light harvest treatment. Board foot volume increments followed the trends observed for cubic volumes, but percent increases were slightly greater. For the control plot, board foot volume increased about 37 percent compared to 62 and 89 percent for the light and heavy thinning treatments, respectively (Table 1).

SUMMARY AND RECOMMENDATIONS

The objective of this study was to quantify long-term changes in the structure of an EWP stand following three levels of silvicultural thinning, which resulted in the sale of commercial products. One purpose of thinning is to increase economic returns from a forestry investment by harvesting trees that would otherwise die from intraspecific competition. Thinning therefore maintains optimum stand density for maximum productivity and value of products (Zeide 2006). The lack of data on product volumes and harvest values does not allow a direct economic analysis of thinning.

Table 2.—Height to top and base of crown for dominant and codominant eastern white pines in the three basal area reduction treatments at plantation ages 49 and 59 years, Bent Creek Experimental Forest

Stand parameter	0 percent basal area reduction treatment; 49 years	0 percent basal area reduction treatment; 59 years	33 percent basal area reduction treatment; 49 years	33 percent basal area reduction treatment; 59 years	66 percent basal area reduction treatment; 49 years	66 percent basal area reduction treatment; 59 years
Top of crown (feet)	102.3	112.2	101.3	108.7	95.5	104.2
Base of crown (feet)	66.4	77.8	60.4	68.4	58.2	62.0
Crown ratio (%)	35.1	30.6	40.4	37.1	39.0	40.5

However, an indirect evaluation of thinning can be estimated from the total production of sawtimber, which is the amount harvested by thinning and the subsequent periodic growth of the residual stand (Baldwin et al. 1998). In our study, assuming uniform prethinning stand conditions throughout the plantation, harvested sawtimber volumes can be estimated as the difference between the post-thinning residual volume on the control plot and the residual volume on each thinned plot. Using the plot in the light thinning as an example, total sawtimber production was 48.5 MBF/acre + (51.7 MBF/acre - 29.9 MBF/acre) = 70.3 MBF/acre (Table 1). Like results of the conifer thinning studies reported by Zeide (2006) and Baldwin et al. (1998), neither the 33 nor 66 percent thinning treatments increased total EWP production during the study interval of 20 years. The results of an economic analysis are likely to be different, however, because proceeds from sale of the harvested sawtimber could have been invested for additional financial gains.

The findings reported here are from an unreplicated long-term study established for demonstration purposes. In the absence of designed studies, these results provide interim information on the response of an EWP plantation to three intensities of commercial thinning at a mid-rotation age of 50 to 60 years. The high level of productivity of this unthinned EWP stand on a dry site is demonstrated by greater than 25 percent increases in d.b.h., basal area, and volumes. Comparison of stand basal areas for the Bent Creek plantation with several other studies revealed large differences and suggests the need for additional refinement of the growth and yield model developed by Hepp et al. (2015). Equally or more important to evaluation of tree responses to thinning is assessment of economic values resulting from different thinning treatments, as demonstrated by Clatterbuck and Ganus (2000). EWP is a highly productive species on dry, low elevation sites in the southern Appalachians (Barrett 1933); similarly, shortleaf pine (*P. echinata*) is a native conifer that can be grown in monoculture plantings, but it is more often a component of mixed pine-hardwood stands (Pile and Waldrop 2016).

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