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Disturbances, prescribed fire, and invasion by exotic plants in a xeric mixed-oak and oak-pine dominated area of the Ridge and Valley in eastern West Virginia

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**Disturbances, Prescribed Fire, and Invasion by Exotic Plants in a
Xeric Mixed-oak and Oak-pine Dominated Area of the Ridge and
Valley in Eastern West Virginia**

Jonathan A. Pomp

**Thesis submitted to the Davis College of Agriculture, Forestry, and
Consumer Sciences at West Virginia University
in partial fulfillment of the requirements for the degree of**

**Master of Science
in
Forestry**

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ABSTRACT

Disturbances, Prescribed Fire, and Invasion by Exotic Plants in a Xeric Mixed-oak and Oak-pine Dominated Area of the Ridge and Valley in Eastern West Virginia

Jonathan A. Pomp

This document discusses three fire and exotic invasive plant related research studies that were conducted in an area of the Ridge and Valley dominated by xeric mixed-oak and oak-pine forest communities. The first study characterizes the timing, frequency, and magnitude of canopy disturbances in the study area by investigating demographic (age distributions), anatomical (growth increment patterns), and structural (diameter distributions) evidence. In addition to the general reconstructed disturbance history, the potential link between overstory tree-of-heaven (*Ailanthus altissima* (P. Mill.) Swingle) populations and prior forest management disturbances (i.e. harvesting) is investigated. This study indicates an uneven-aged structure resulting from multiple disturbances of varying intensities throughout the area. One study area-wide disturbance event was identified around the turn of the 20th century. The majority of the areas where tree-of-heaven has ascended into the main canopy are either in harvested areas, or along roads. Native ground-layer vegetation around the main tree-of-heaven stems is in danger of being severely impacted by vegetative reproduction.

In the second study, the relationship between plot-level three-year post-fire bole scorch height, or stem-bark char, parameters (independent variables) and other fire intensity variables (litter consumption, thermocouple temperature, and sapling mortality) are investigated. This study indicates that neither overstory nor sapling bole scorch height variables are significantly related to thermocouple temperature measurements. The sum of the sapling scorch heights is significantly related to litter consumption, but only accounts for 11 % of the variation. However, all bole scorch height variables are significantly related to sapling mortality one growing season after the fire. This further illustrates the usefulness of scorch height as an estimator of relative fire intensity.

The third study examines the effects of a prescribed fire on understory exotic invasive plants. In this study, several analytical methods were employed to determine if: (1) abundance and importance increased following the fire, (2) relative fire intensity played a role, and (3) environmental variables were related to populations. The results indicated that only one species, tree-of-heaven, showed significantly higher abundance and importance following the prescribed fire. The results also implied that moister areas, where understory species richness was higher and where relative fire intensity was lower, provided a favorable microsite for the exotic invasive species. Furthermore, environmental factors played more of a role than relative fire intensity in determining the probability of presence of exotic invasive plants in the understory.

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CHAPTER 1

Introduction

The vast majority of the dry hillsides of Ridge and Valley Physiographic Province of eastern West Virginia are encompassed by fire-associated oak (*Quercus* spp.) and pine (*Pinus* spp.) woodlands. The future of these forests are at risk due to the loss of recurring fires (Abrams 1992, Waldrop and others 2006) as well as other less frequent stand initiating disturbances. The species composition, in addition to the hot, dry climate, associated with the area, makes the Ridge and Valley especially suited for the re-introduction of fire to achieve management goals (Lafon and Grissino-Mayer In Press, Thomas-Van Gundy and others 2007). The use of prescribed fire to promote historical structure and composition in oak and oak-pine forests in this region is now steadily increasing, and has become a critically important management issue to the general public, as well as to foresters, biologists, ecologists, and other land and prescribed fire managers. Chapter 2 provides a complete literature review that details the uniqueness of the forested communities in this area and how fire historically played a role in their development.

Troublesome exotic invasive plants may also pose a significant threat to these communities (Mlot 2003, Williams 1998). In forested ecosystems, highly invasive exotic plant species such as tree-of-heaven (*Ailanthus altissima* (P. Mill.) Swingle), garlic mustard (*Alliaria petiolata* (Bieb.) Cavara & Grande), and Japanese stilt grass (*Microstegium vimineum* (Trin.) A. Camus) may drastically inhibit successful establishment and development of native plants and lead to a decrease in regional biodiversity (D'Antonio and Vitousek 1992) and altered ecosystem processes (Vitousek and Walker 1989). Unfortunately, little is known about the relationship between prescribed fire and invasion by exotic invasive plants in these areas. The general

literature review (Chapter 2) also details what exotic invasive plant species are, problems associated with them, and highlights species that are especially problematic in deciduous forests of West Virginia and the eastern United States (US).

Because invasive species have a tendency to invade disturbed sites, it is important to better understand the effects of prescribed fire (an anthropogenic disturbance) on this threat to forests of West Virginia. Increases in establishment and abundance of exotic invasives may have considerable negative impacts on native plant communities. If this is the case, many areas scheduled to be burned in the future may require exotic invasive monitoring or management prior to the application of prescribed fire. For example, exotic invasive plant pests could be controlled with herbicides prior to application of the fire treatment. Pre-fire control of invasive species would aim to make the forest more resilient to invasions, reduce the likelihood of post-fire establishment, and help protect native understories and regeneration of eastern oak and pine forest communities.

Following 81 years of fire exclusion, prescribed fire treatments were applied on March 29, 2004 to Dunkle Knob, a 314 ha area composed of xeric mixed-oak and oak-pine forest communities located on the North River Ranger District (formerly the Dry River Ranger District) of the George Washington National Forest (GWNF) in the Ridge and Valley (38° 37' N, 79° 14' W). Thirty-six permanent circular 0.05 ha fixed area plots were established on the mountain in 2003 prior to the prescribed fire (Marsh 2005). Vegetation measurements were recorded in 2003 and 2004 on Dunkle Knob, as well as two adjacent areas, Brushy Knob and Heavener Mountain, to describe overstory,

midstory, and understory strata, and to assess vegetation dynamics following prescribed fire using a “prescribed fire chronosequence” (Marsh 2005). Dunkle Knob was the only mountain where pre-fire data had been recorded. Initial vegetation measurements indicated the presence of some notable exotic invaders, including; tree-of-heaven, garlic mustard, Bell’s honeysuckle (*Lonicera X bella* Zabel), and Canada bluegrass (*Poa compressa* L.). One of the conclusions of this early study was that neither percent cover or importance value of these exotic invasives changed significantly anywhere on Dunkle Knob immediately (three to four months) following prescribed fire (Marsh 2005). However, understory communities are very dynamic and composition can change within a growing season, and from growing season to growing season following fire (Bowles and others 2007, Hutchinson 2006), as well as other disturbances (Small and McCarthy 2002).

Thanks to collaborative support from several people at West Virginia University (WVU) and the United States Department of Agriculture (USDA) Forest Service, the opportunity arose to sample the plots again during the 2007 growing season. The overall objective of the new project was to determine if post-fire levels of exotic invasive plant populations were greater than pre-fire levels. This question is answered in Chapter 5 by examining pre- and post-fire data using various proven statistical techniques, including: Multi-response Permutation Procedures (MRPP), Nonmetric Multidimensional Scaling (NMS) ordination, and a single factor Analysis of Variance (ANOVA). Chapter 5 also examines exotic invasive plant population-environmental relationships.

A secondary goal of this project was to characterize the timing, frequency, and magnitude of disturbances in the study area by investigating demographic (age distributions), anatomical (growth increment patterns), and structural (diameter distributions) evidence. In general, disturbance regime descriptions allow for an evaluation of recent theories about forest development and provide a better perception of community properties. The information presented in Chapter 3 should help foresters and other land managers to better understand how disturbance frequency, intensity, and severity might affect species composition and stand dynamics in xeric mixed-oak and oak-pine forests. In addition to the general reconstructed disturbance history, the potential link between overstory tree-of-heaven populations and prior forest management disturbances (i.e. harvesting) is investigated. This analysis should show if harvesting played a role in tree-of-heaven establishment and distribution. Additionally, this analysis will provide GWNF managers with locations of tree-of-heaven infestation that may influence prescribed fire or other types of management and planning.

Another secondary goal of the project was to determine if fire intensity played a role in the distribution exotic invasive plants. Unfortunately, direct fire intensity measurements were restricted to five plots constrained in one area of Dunkle Knob. Therefore, in Chapter 4, I investigate the relationship between plot-level three-year post-fire bole scorch height, or stem-bark char, parameters (independent variables) and other fire intensity variables. Three relationships/predictions are evaluated: (1) litter consumption as a function of scorch height, (2) thermocouple estimated fire temperature as a function of scorch height, and (3) sapling mortality as a function of scorch height. The intent of

this analysis was to evaluate whether these quick, simple, and inexpensive field measurements can help characterize prescribed fire behavior and its short-term effects on forest structure and composition. Significant surrogate fire intensity variables identified in Chapter 4 are related to exotic invasive plant populations in Chapter 5.

The information from this document should provide a better understanding of the interactions of disturbance, prescribed fire, and exotic invasive plant species over time, and across the landscape, in an area of the Ridge and Valley dominated by xeric mixed oak and oak-pine forest communities. These data will also provide insight to the future potential of troublesome exotic invasive plant species in the study area.

Chapters 3, 4, and 5 are formatted so that they can be easily adjusted for submittal for publication in a journal or as conference proceedings. Chapter 4 is already published as part of the proceedings of the 16th Central Hardwood Forest Conference.

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CHAPTER 2

General Literature Review

Exotic Invasive Species: Defined

Based on Executive Order 13112 of February 3, 1999, the definition of the term exotic invasive species can be broken into two parts, exotic and invasive. The term exotic, used synonymously with alien and non-native, refers to, with respect to a particular ecosystem, any species (plant, mammal, bird, fish, insect, aquatic invertebrate, or fungi) that is not native to that ecosystem. The term invasive refers to a species whose introduction does, or is likely to, cause economic or environmental harm, or harm to human health (Clinton 1999). Therefore an exotic invasive species is a species that is not native to a particular ecosystem, it is found in that particular ecosystem, and has, or is likely to impair economic and environmental benefits associated with that particular ecosystem, or cause human harm.

Problems Imposed by Exotic Invasive Plant Species

Invasions by alien plants are a growing challenge worldwide to the management of native biodiversity and ecosystem functioning (Brooks and others 2004, Gordon 1998, Mooney and Hobbs 2000, Steele and others 2006, Vitousek and others 1996). Exotic invasive plant species have been, and are currently, causing a wide variety of problems throughout the US. The environmental problems caused by exotic invasive plant species can include, but are not limited to: diminished land values (Higgins and others 1999), loss and destruction of forage and habitat for wildlife (Allan and others 1997, Steenkamp and Chown 1996), loss of forest understory productivity (McCarthy 1997, Stinson and others 2006), loss of available grazing land (Bangsund and Leistritz 1991), soil degradation (Ashton and others 2005, Vitousek and Walker 1989), and, in some cases, diminished

recreational enjoyment (J. Pomp, personal observation). Exotic invasives can also hybridize with native species, resulting in negative genetic impacts (Mlot 2003). Furthermore, if exotic invasive plants become well established, they may alter forest fuel properties and impact fire behavior and fire regime characteristics (e.g. frequency, intensity, extent, type, and seasonality) (Brooks and others 2004, D'Antonio and Vitousek 1992, Mack and D'Antonio 1998). If these fire regime changes promote the dominance of the invaders, then an invasive plant-fire regime cycle can be established, making ecological restoration much more difficult (Brooks and others 2004). Given that the cumulative numbers of exotic invasive plant species in the forests of West Virginia may be increasing over time (Huebner 2003), few ecologists and conservation biologists will argue that the worst of the problems caused by these unwanted plant pests are behind us.

In West Virginia and throughout the mixed deciduous hardwood forests of the eastern US, invasive shrubs, vines, trees, and grasses are threatening native flora, as well as the habitats that they compose. Once established, highly invasive plants such as Japanese stilt grass, garlic mustard, and tree-of-heaven may be capable of crowding out important native vegetation in forests in only a few years time. Of particular importance, in West Virginia and throughout the central Appalachians, is the impact of exotic invasive plant species on forest reproduction and economically valuable herbaceous plant communities, e.g., American ginseng (*Panax quinquefolius* L.), goldenseal (*Hydrastis canadensis* L.), and black cohosh (*Cimicifuga racemosa* (L.) Nutt.). Exotic invasive plant species also pose a significant threat to threatened, endangered, or otherwise rare plant communities

and species. A study by Flather and others (1994) ranked exotic invasive plant species as being one of the most important threats to endangered species, second only to habitat degradation/loss.

Management for control of exotic invasive plants often proves to be difficult and costly. Some exotic invasive species have caused major economic losses in forestry, agriculture, as well as other segments of the US economy. Pimental and others (2000) estimate the total annual costs (costs due to control measures, losses, and damages) of exotic invasive plants species to equal just over \$34 billion per year. Unfortunately, ample data on the economic impacts of exotic invasive plant species in West Virginia are not available.

Important Exotic Invasive Plants in West Virginia and the Eastern US

Despite there being a high number of exotic plant species found in West Virginia, the USDA's invasive and noxious weeds list for West Virginia lists only nine species. The WVDNR recognizes twelve plant species as being especially threatening to the health and productivity of West Virginia's native ecosystems. Only one of these species, multiflora rose (*Rosa multiflora* Thunb. ex Murr.), made the USDA's invasive and noxious weed list. Known as the "Dirty Dozen," the 12 most problematic of the exotic invasive plants in West Virginia include: crown vetch (*Coronilla varia* L.), garlic mustard, Japanese knotweed (*Polygonum cuspidatum* Sieb. & Zucc.), Japanese stilt grass, kudzu (*Pueraria montana* (Lour.) Merr.), mile-a-minute weed (*Polygonum perfoliatum* L.), multiflora rose, purple loosestrife (*Lythrum salicaria* L.), reed canary grass (*Phalaris*

arundinacea L.), tree-of-heaven, water shield (*Brasenia schreberi* J.F. Gmel.), and yellow iris (*Iris pseudacoris* L.) (Grafton 2007).

The USDA Forest Service lists five of these species (crown vetch, garlic mustard, Japanese knotweed, Japanese stilt grass, and purple loosestrife) as highly invasive in the eastern region of the US. These plants are all non-native, highly invasive plants which invade natural habitats and may have the ability to replace native species (USDA Forest Service, Eastern Region 1998). Despite being the exotic invasive that is most likely to be encountered in a forest in West Virginia (Huebner 2003), multiflora rose is only considered by the USDA Forest Service as moderately invasive (less invasive than the five aforementioned species) in the eastern region. They suggest that, if this species is significantly affecting native species, then it is doing so only in local areas (USDA Forest Service, Eastern Region, 1998). The 1998 USDA Forest Service list failed to categorize/list kudzu, mile-a-minute-weed, reed canary grass, water shield, and yellow iris. In their summary report of exotic invasive species in West Virginia, the Union of Concerned Scientists identified two of previously mentioned exotic invasive plants, Japanese stilt grass and purple loosestrife, as being particularly devastating to the ecological integrity of West Virginia's native ecosystems, as well as the economy (Mlot 2003).

Given their physiological traits, range, and habitat, garlic mustard, Japanese stilt grass, multiflora rose, and tree-of-heaven can be considered as some of the exotic invasive

species that pose the greatest threats to forested ecosystems. Each of these species is introduced individually in the following paragraphs.

Garlic mustard, also commonly referred to as hedge-garlic, jack-by-the-hedge, poor-man's-mustard, jack-in-the-bush, and garlic root, is an overwintering herb of European descent (Nuzzo 2000). The exotic herb was first recorded in North America in 1868 on Long Island, New York, and has spread exponentially since its introduction (Nuzzo 1992, Nuzzo 1993). Garlic mustard can now be found in 34 states of the Union and four Canadian provinces (Nuzzo 2000). This herbaceous exotic invasive may have the ability to readily invade and dominate the understories of forests throughout the eastern US and adjacent Canada (Nuzzo 1993). Like the majority of exotic invasive species, disturbance favors the establishment, spread, and growth of garlic mustard. In general, the species can easily become established and invade forests that have experienced multiple disturbances that decrease stand quality (e.g. diameter limit harvests) (Byers and Quinn 1998, Nuzzo 1991). However, a study by Nuzzo (1999) showed that, even in “high quality” forests (relatively undisturbed), after invasion, garlic mustard spreads annually and becomes a permanent part of the community, increasing in presence, but fluctuating in cover and density.

Japanese stilt grass, also commonly referred to as stilt grass, annual jewgrass, wire grass, Nepalese browntop, and Chinese packing grass, is a shade tolerant C₄ annual grass of Asiatic origin. The exotic grass was introduced into the US in 1919 in Knoxville, Tennessee (Fairbrothers and Gray 1972). Placed in an environment with no natural

enemies, the aggressive grass readily reproduced and began to disseminate. Today, it is widely distributed in dense patches on river banks, flood plains, damp fields, swamps, lawns, woodland thickets, logging roads, and roadsides throughout the eastern US (Barden 1987, Cole and Weltzin 2005, Fairbrothers and Gray 1972). Some feel that Japanese stilt grass should be considered the greatest threat to native understory habitats of West Virginia's forests.

Japanese stilt grass possesses several traits that contribute to its invasive nature. The grass shows a very plastic response to varying habitat conditions (Claridge and Franklin 2002, Gibson and others 2002, Tu 2000, Winter and others 1992). For example, Japanese stilt grass, like all C₄ plants, prefers moderate to high light levels for reproduction, dispersal, and establishment, however, Japanese stilt grass can grow, reproduce, and disperse seed in only five percent full sunlight (Winter and others 1992). Japanese stilt grass has also been reported to promote positive feedback processes with the soil, further enhancing its ability to spread (Gibson and others 2002, Kourtev and others 1998, Tu 2000). The exotic invasive grass also possesses a persistent seed bank that remains viable for three to seven years (Barden 1987, Gibson et 2002, Tu 2000). Japanese stilt grass is slow to invade undisturbed areas; however, disturbance (natural or anthropogenic) favors the establishment of the species (Barden 1987, Oswalt and others 2007, Tu 2000). In West Virginia, Japanese stilt grass populations are especially abundant on logging roads and forest edges following operations that disturb the soil or forest canopy.

Multiflora rose, also known as rambler rose, is a perennial shrub of Asian descent. The exotic invasive was originally introduced to the eastern US from Japan in 1886 for use as an understock for ornamental roses (Eckardt 1986, Wyman 1949). The US government, specifically the Soil Conservation Service (SCS) (currently the Natural Resources Conservation Service (NRCS)), promoted the use of the plant. In the early 20th century the SCS conducted several tests and decided that multiflora rose was the best species to replace osage orange (*Maclura pomifera* (Raf.) Schneid.) as livestock barriers, or “fences”, between prairies and farms (Steavenson 1946). The problematic shrub would soon escape urban areas and fields where it was planted and readily invade various types of areas. Multiflora rose is now found in 31 of the 50 states in the US, primarily in the eastern region (Swearingen 2008). The shrub can successfully invade pastures and other unplowed lands, crowding out existing vegetation and creating dense, impenetrable thickets (Eckardt 1986). Problems associated with multiflora rose, however, are not restricted to these areas. The exotic invasive shrub can endure a wide range of edaphic and environmental conditions (Steavenson 1946) and can often be found in forested ecosystems in West Virginia, and throughout the eastern US, impairing native understory and shrub vegetation layers. A study by Huebner (2003) showed that multiflora rose was associated with forested counties in West Virginia. Her study was indicative of the potential threat multiflora rose can present to oak dominated forests in West Virginia.

Tree-of-heaven, also commonly referred to as ailanthus, Chinese sumac, and varnish tree, is a fast growing, exploitive tree of Asiatic origin. The highly invasive exotic tree-of-heaven is one of the most problematic exotic invasive species in the Appalachians and

throughout the US. Tree-of-heaven may have the ability to drastically inhibit successful establishment and development of native plants and lead to a decrease in regional biodiversity (D'Antonio and Vitousek 1992) and altered ecosystem processes (Vitousek and Walker 1989). The exotic tree was introduced into the US by William Hamilton, a Philadelphia gardener and plant collector, in 1784 (Hu 1979, Shah 1997). Tree-of-heaven may have also been introduced into the western US in the 1800's by Chinese miners as they settled in California to seek gold (Hoshovsky 1988). The exotic invasive tree is now found in 60 % of the states (Swearingen 2008). Tree-of-heaven can be found threatening thousands of acres of native flora in 38 of the 55 counties of West Virginia (69 %) (Harmon and others 2006). It has become well established in several of these counties primarily around urban areas and in disturbed areas of forests. Especially problematic, almost pure, stands of tree-of-heaven can be found in the eastern panhandle (D. McGill, Forest Resources Extension Specialist, WVU Division of Forestry and Natural Resources, personal communication).

Xeric Mixed-Oak and Oak-Pine Forests of the Ridge and Valley

The Ridge and Valley Physiographic Province of eastern West Virginia is characterized by its rugged and heterogeneous topography composed of dry slopes. Highly infertile sites that are inhospitable to mesophytic hardwood species are common throughout the area. The hot, dry hillsides in this area support xeric mixed-oak and oak-pine woodlands. The most common species found in these wooded communities include, chestnut oak (*Quercus prinus* L.), scarlet oak (*Quercus coccinea* Muenchh.), table mountain pine (*Pinus pungens* Lamb.), pitch pine (*Pinus rigida* P. Mill.), and Virginia pine (*Pinus*

virginiana P. Mill). These species possess certain silvical characteristics that allow for colonization and persistence on poor sites. All of the species, except chestnut oak, are classified as shade intolerant (Hicks 1998, Williams & Johnson 1990). Chestnut oak is classified as intermediate in shade tolerance (Hicks 1998). All of the species have site adaptive root strategies as well (i.e. they tailor the physiologic processes in their roots to allow for the greatest survival and growth on a given site). For example, the pines possess the ability to root in underdeveloped soil and rock crevices (Williams & Johnson 1990), while the oaks have superior root systems and a high root to shoot ratio. Other than scarlet oak, the species found in these forests are resistant to injuries from fire (Hicks 1998). Both the pines and the oaks have thick bark and the latter possess a superb ability to compartmentalize injury (Smith and Sutherland 1999). Also, table-mountain pine and pitch pine possess serotinous cones (Williams & Johnson 1990).

Fire in Xeric Mixed-Oak and Oak-Pine Forests

Historically, fire has played a key role in maintaining forest composition in pine-oak and mixed-oak forests in West Virginia and throughout the Appalachians (Abrams 1992, Brose and others 2001, Schuler and McClain 2003, Sutherland and others 1995). Before fire suppression became a policy in the eastern US between the late 1920s and 1950s, these forests were shaped by a combination of large scale disturbances (e.g. drought, stand-replacing wildfire, hurricane, and insect or disease outbreak) and low-intensity fires. Sutherland and others (1995) estimated that major fires occurred about every 10 to 15 years in table mountain pine stands in southwestern Virginia. Both Brose and Waldrop (2006b) and Yarnell (1998) estimated that prior to the fire suppression era, stand

level disturbances in the southern Appalachians occurred once every 6 to 13 years.

Rentch and Hicks (2003) found that large canopy disturbances involving multiple trees occurred every 17 years in oak dominated old-growth stands in the Central Hardwoods Region.

These types of disturbances formed stands of mixed-oak and pine, and more frequent low-intensity fires maintained this composition (Abrams 1992, Brose and others 2001). Several studies have estimated low intensity fire return intervals in these forests. These return intervals are comparable to those of stand-level disturbances. Sutherland and other (1995) estimated that low-intensity fires occurred every 9 to 11 years. Harmon (1982) found that periodic fires occurred, on average, about every 12 years in the Great Smoky Mountains. Shumway and others (2001) calculated fire-return intervals of 8 years in an oak forest in western Maryland. Schuler and McClain (2003) reported fire-return intervals of 7 to 15 years from 1895 to 2002 in an oak forest in the Ridge and Valley Physiographic Province of eastern West Virginia. Weibull median fire return intervals of 4 to 8 years occurred in rimrock oak-pine communities at the New River Gorge National River in West Virginia (Maxwell 2006).

The diminution of fire as a disturbance agent, and the associated change in structure and species composition, has led to a decline of both oak and, especially, pine communities in some areas (Abrams 1992, Brose and Waldrop 2006a). Continued absence of fire in oak-pine communities could soon lead to a decline in pine species, and eventual oak dominance (Abrams 1992, Sutherland and other 1995). Table mountain and pitch pine

dominated communities, in particular, are recognized by the Southern Appalachian Man and the Biosphere (SAMAB), a program that encourages the utilization of ecosystem and adaptive management principles, as one of 31 rare communities (SAMAB 1996).

Researchers once thought that, without stand initiating fires, successful regeneration of table mountain and pitch pine communities could not occur (Brose and others 2002, Sutherland and others 1995, Williams and Johnson 1990). Recent research, however, suggests that repeated low intensity fires can achieve successful table mountain pine regeneration (Waldrop and others 2006). Successful management of mixed-oak and oak-pine forests, without restoration of historical fire regimes, is difficult (Abrams 1992, Barden & Woods 1976, Marsh 2005, Sutherland and others 1995, Welch and others 2000, Williams & Johnson 1990). This information has spurred land managers to increase the number and extent of prescribed fires in these forest ecosystems in efforts to restore them to their historical structure and to promote regeneration of native oak and pine species. This increase in prescribed fire is carried out primarily on federal lands (Elliot and others 1999, Lafon and others 2005). In addition to the management resulting from altered disturbance and fire regimes, the future of these ecologically important forests are likely to be negatively affected by exotic invasive plants (Mlot 2003, Williams 1998).

Exotic Invasive Plant Species and Fire

In addition to encouraging historical forest structure and enhancing tree regeneration, natural and prescribed fire has also been shown to indirectly promote increases in both richness and cover of native herbaceous plants in forest understory communities (Bowles

and others 2007, Hutchinson 2006, Nuzzo and others 1996). These increases allow for greater diversity and, in general, a better habitat for local fauna. However, fire and other disturbances (i.e. thinning/harvesting and herbicide treatments) that decrease canopy cover and increase understory light, may also promote invasion by exotic plant species (Bossard 1991, D'Antonio 1993, Hobbs and Huenneke 1992, Hutchinson and others 2004, Larson 2003, Mooney and Hobbs 2000, Pyle 1995, Stapanian and others 1998).

The relationship between fire and invasion by exotic plants is not well understood. A few studies have directly examined this relationship in different forested ecosystems. The results, however, are variable and in some cases, contradictory. Some have found that fire, natural or anthropogenic, promoted exotic invasive plant populations. Others have found little to no change or negative responses by exotic invasives. For instance, Bridgewater and Backshall (1981) found that natural fire enhanced exotic invasive forbs and grasses in *Eucalyptus* woodlands of Western Australia. Hester and Hobbs's (1992) results contradicted those of Bridgewater and Backshall (1981). They reported decreases in the density of exotic invasive forbs and grasses in *Eucalyptus* woodlands in the same region. Wein and others (1992) found that exotic annuals and biennials invaded recently burned areas of Wood Buffalo National Park in Canada. In ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests of the western US, fire has been shown to increase both the richness and abundance of exotic invasive species (Crawford and others 2001, Griffis and others 2001). Both of these studies showed that exotic invasive species were far more abundant where fire severity, or intensity, was highest. Laughlin and others (2004), on the other hand, found little to no change in exotic invasive plant populations

following wildfire in old-growth ponderosa pine forests located in Grand Canyon National Park in Arizona. In other areas of the western US, like the California coast, interior valleys, and other western grassland and prairie ecosystems, fire is used as a management tool to control exotic invasives and promote native vegetation (Rice 2005).

Studies in the eastern US showed similar variability in the effects of fire on exotic invasive plant populations. Bowles and others (2007) showed that seventeen years of repeated annual low intensity burning promoted garlic mustard abundance in an Illinois oak forest. In their study, garlic mustard persisted and showed greater abundance in burned areas relative to unburned areas. Contrary to Bowles and others (2007) findings, Nuzzo and others (1996) found that repeated annual burnings facilitated a reduced abundance of garlic mustard in a sand forest in Illinois. Nuzzo 1991 reported that garlic mustard showed no decline with low intensity fire and some decline with mid-intensity fire. Glasgow and Matlack (2006) found that prescribed burning and canopy-opening management practices facilitated invasion by Japanese stilt grass and multiflora rose in a mixed deciduous forest of Ohio. Doren and Whiteaker (1990) reported little to no change in the populations of the exotic invasive Brazilian pepper tree (*Schinus terebinthifolius* Raddi) in Florida woodlands following fire. Based on the available literature, the variable population responses of exotic invasive plant to fire appear to be due to both regional and local landscape variation (biology, different climates and microsites, etc.), and to be both site and species specific. That is, it is highly likely that responses to fire will be different in dissimilar areas or between disparate species on the same site.

Direct assessments of the relationship between fire and exotic invasive plant populations in forested ecosystems are limited to the abovementioned studies. Furthermore, no documented studies have explored this relationship in West Virginia. Perhaps more importantly, it has not been examined in forests of the Ridge and Valley Physiographic Province, an area that because of its unique species composition and climate is especially suited for the reintroduction of fire to achieve management goals (Thomas-Van Gundy and others 2007, Lafon and Grissino-Mayer In Press). The use of prescribed fire in oak and oak-pine forests in this region is steadily increasing, especially on the GWNF, and has become a critically important management issue to the general public, as well as to foresters, biologists, ecologists, and other land and prescribed fire managers. GWNF lands experienced an astounding 2,240 % increase in area prescribed burned from the 1980s to the 1990s, and a 140 % increase from the 1990s to the 2000s, respectively.

The very hot, dry hillsides in this area provide ecologically unique ecosystems whose futures are at risk, not only due to altered fire regimes, but also due to invasion by exotic plants. Little is known about the relationship between prescribed fire and invasions by exotic invasive plants in these areas, hence, as the use of fire as a management tool increases, exotic invasives could be a major threat to future forest composition. Increases in establishment and abundance of exotic invasives may have severe negative impacts on native plant communities. If this is the case, many areas scheduled to be burned in the future may require exotic invasive monitoring or management prior to the application of prescribed fire. For example, exotic invasive plant pests could be controlled with herbicides, or other means, prior to application of the fire treatment. Pre-fire control of

invasive species would aim to make the forest more resilient to invasions, reduce the likelihood of post-fire establishment, and help protect native understories and the future of eastern oak and oak-pine forests.

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CHAPTER 3

Disturbance History and *Ailanthus altissima* (P. Mill.) Swingle Establishment in a Xeric Mixed-oak and Oak-pine Dominated Area in the Ridge and Valley of Eastern West Virginia

Abstract

Descriptions of disturbance regimes allow for evaluations of recent theories about forest development and provide a better perception of forest community properties. This chapter focuses on characterizing the disturbance history of a 313.6 ha mountain composed of xeric mixed-oak and oak-pine forest communities. In addition to the general reconstructed disturbance history, the potential link between overstory tree-of-heaven (*Ailanthus altissima* (P. Mill.) Swingle) populations and prior forest management disturbances (i.e. harvesting) is investigated. The results indicated an age distribution and structure that is a result of multiple disturbances of varying intensities throughout the area. The majority (81 %) of the releases identified from percent growth change data were small and moderate, which were likely a result of death or removal of one or two trees that neighbored a sample tree. One study area-wide major disturbance event was identified around the turn of the 20th century. For the most part, the only species to establish after the 1930s were chestnut oak (*Quercus prinus* L.), northern red oak (*Quercus rubra* L.), and Virginia pine (*Pinus virginiana* P. Mill.). Furthermore, most of the serotinous-coned yellow pines are not recruiting well in the understory. With respect to tree-of-heaven, most that had reached the main canopy were either in harvested areas, or along roads. All but one of the oldest tree-of-heaven post-dated the 1972 harvesting. Ground-layer vegetation around main tree-of-heaven stems is in danger of being severely impacted by vegetative reproduction (root suckers). Because the area is scheduled to be burned, tree-of-heaven should be eradicated in these localized areas, prior to burning, to help protect future forest composition in the study area.

Introduction

The Ridge and Valley Physiographic Province of eastern West Virginia is characterized by its rugged and heterogeneous topography composed of dry slopes. Highly infertile sites that are inhospitable to mesophytic hardwood species are common throughout the area. The hot, dry hillsides in this area support xeric mixed-oak and oak-pine woodlands.

The most common species found in these wooded communities include, chestnut oak (*Quercus prinus* L.), scarlet oak (*Quercus coccinea* Muenchh.), table mountain pine (*Pinus pungens* Lamb.), pitch pine (*Pinus rigida* P. Mill.), and Virginia pine (*Pinus virginiana* P. Mill). These species possess certain silvical characteristics that allow for colonization and persistence on poor sites. All of the species, except chestnut oak, are classified as shade intolerant (Hicks 1998, Williams & Johnson 1990). Chestnut oak is classified as intermediate in shade tolerance (Hicks 1998). All of the species have site adaptive root strategies as well (i.e. they tailor the physiologic processes in their roots to allow for the greatest survival and growth on a given site). For example, the pines possess the ability to root in underdeveloped soil and rock crevices (Williams & Johnson 1990), while the oaks have superior root systems and a high root to shoot ratio. Other than scarlet oak, the species found in these forests are resistant to injuries from fire (Hicks 1998). Both the pines and the oaks have thick bark and the latter possesses a superb ability to compartmentalize injury (Smith and Sutherland 1999). Also, table-mountain pine and pitch pine possess serotinous cones (Williams & Johnson 1990).

Historically, fire has played a key role in maintaining forest composition in pine-oak and mixed-oak forests in West Virginia and throughout the Appalachians (Abrams 1992,

Brose and others 2001, Schuler and McClain 2003, Sutherland and others 1995). Before fire suppression became a policy in the eastern United States (US) between the late 1920s and 1950s, these forests were shaped by a combination of large scale disturbances (e.g. drought, stand-replacing wildfire, hurricane, and insect or disease outbreak) and low-intensity fires. Sutherland and others (1995) estimated that major fires occurred about every 10 to 15 years in table mountain pine stands in southwestern Virginia. Both Brose and Waldrop (2006b) and Yarnell (1998) estimated that prior to the fire suppression era, stand level disturbances in the southern Appalachians occurred once every 6 to 13 years. Rentch and Hicks (2003) found that large canopy disturbances involving multiple trees occurred every 17 years in oak dominated old-growth stands in the Central Hardwoods Region.

These types of disturbances formed stands of mixed-oak and pine, and the more frequent low-intensity fires maintained this composition (Abrams 1992, Brose and others 2001). Several studies have estimated low intensity fire return intervals in these forests. These return intervals are comparable to those of stand-level disturbances. Sutherland and others (1995) estimated that low intensity fires occurred every 9 to 11 years. Harmon (1982) found that periodic fires occurred, on average, about every 12 years in the Great Smoky Mountains. Shumway and others (2001) calculated fire-return intervals of 8 years in an oak forest in western Maryland. Schuler and McClain (2003) reported fire-return intervals of 7 to 15 years from 1895 to 2002 in an oak forest in the Ridge and Valley Physiographic Province of eastern West Virginia. Weibull median fire return intervals of

4 to 8 years occurred in rimrock oak-pine communities at the New River Gorge National River in West Virginia (Maxwell 2006).

The diminution of fire as a disturbance agent, and the associated change in structure and species composition, has led to a decline of both oak and, especially, pine communities in some areas (Abrams 1992, Brose and Waldrop 2006a). Continued absence of fire in oak-pine communities could soon lead to a decline in pine species, and eventual oak dominance (Abrams 1992, Sutherland and others 1995). Table mountain and pitch pine dominated communities, in particular, are recognized by the Southern Appalachian Man and the Biosphere (SAMAB), a program that encourages the utilization of ecosystem and adaptive management principles, as one of 31 rare communities (SAMAB 1996).

Researchers once thought that, without stand initiating fires, successful regeneration of table mountain and pitch pine communities could not occur (Sutherland and others 1995, Williams and Johnson 1990). Recent research, however, suggests that repeated low intensity fires can achieve successful table mountain pine regeneration (Waldrop and others 2006). Successful management of mixed-oak and oak-pine forests, without restoration of historical fire regimes, is difficult (Abrams 1992, Barden & Woods 1976, Marsh 2005, Sutherland and others 1995, Welch and others 2000, Williams & Johnson 1990).

In addition to the management challenges resulting from altered disturbance and fire regimes, the futures of these ecologically important forests are likely to be negatively affected by exotic invasive plants (Mlot 2003, Williams 1998). Highly invasive exotic

plant species such as tree-of-heaven (*Ailanthus altissima* (P. Mill.) Swingle) are likely to drastically inhibit successful establishment and development of native plants and lead to a decrease in regional biodiversity (D'Antonio and Vitousek 1992) and altered ecosystem processes (Vitousek and Walker 1989). The highly invasive exotic tree-of-heaven is one of the most problematic exotic invasive species in the Appalachians and throughout the US.

Various types of evidence can be used for analyzing the timing, frequency, and magnitude of disturbances in forest communities, including: external physical evidence (fire scars, charcoal, uprooted trees), demographic evidence (age distribution), anatomical evidence (growth increment patterns), and structural evidence (canopy structure, diameter distribution) (Lorimer 1985). Furthermore, cohorts of similarly aged trees are one of the best indicators of past disturbance (Lorimer and Frelich 1989, Nowacki and Abrams 1992, Nowacki and Abrams 1997). Other than fire-scar analysis, disturbance (natural or anthropogenic) frequency and intensity in xeric mixed-oak and oak-pine forests of the Ridge and Valley has not been documented.

Since the mid- to late-1980s, the extent of prescribed fires has increased quite dramatically on federal lands in this area. In the decade of 1980, on average, 37.2 hectares/year were burned on the George Washington National Forest (GWNF), compared to 877.2 hectares/year in the 1990s and 2,097 hectares per year this decade. That equates to an astounding 2,240 % increase from the 1980s to the 1990s, and a 140 % increase from the 1990s to the 2000s, respectively. Concern for environmental quality is

heightened when considering the increase in prescribed fire programs combined with observations that many exotic invasive species establish prolifically in freshly disturbed areas. As part of a study to assess the impact of prescribed fire on population dynamics of exotic invasive species, an investigation of the long-term, landscape-scale disturbance history was conducted on a 313.6 ha mountain composed of xeric mixed-oak and oak-pine forest communities located on the GWNF in the Ridge and Valley Physiographic Province. The intent of this investigation was to explore evidence of forest canopy disturbances in the area in order to eliminate possible sources of variation in an associated study of the interaction between exotic invasives and prescribed fire.

This investigation had two primary objectives:

1. Identify forest canopy disturbances from the growth pattern of local trees using dendrochronology techniques.
2. Determine if a link exists between previous forest management disturbances (i.e. harvesting) and overstory tree-of-heaven populations.

A description of the disturbance regime will also allow for an evaluation of recent theories about forest development in these areas and provide a better perception of community properties. The information presented should help foresters and other land managers to better understand how disturbance frequency, intensity and severity might affect species composition and stand dynamics of this forest type. In addition to showing if harvesting played a role in tree-of-heaven establishment and distribution, the results of objective number two will provide GWNF managers with locations of localized tree-of-

heaven infestation that may influence prescribed fire planning. For example prescribed fire managers may want to remove tree-of-heaven in these areas before burning. The study area is scheduled to be burned again in the spring of 2008 (S. Croy, Ecologist-GWNF, personal communication).

Methods

Study Site

The study site, a mountain known as Dunkle Knob, encompasses 313.6 ha and is composed of xeric mixed-oak and oak-pine forest communities. Based on a 2007 inventory, the most abundant overstory species (trees ≥ 12.7 cm diameter at breast height (DBH)) species include chestnut oak, Virginia pine, and table mountain pine (Table 3.1). The understory mixture (trees ≤ 1 m in height) is well represented by red maple (*Acer rubrum* L.), striped maple (*Acer pensylvanicum* L.), and pignut hickory (*Carya glabra* (P. Mill.) Sweet) (Table 3.2).

The study site is located on the North River Ranger District (formerly the Dry River Ranger District) of the GWNF, Pendleton County, West Virginia (38° 37' N, 79° 14' W). The area is within the Ridge and Valley Physiographic Province and is in the rain shadow of the Allegheny Mountains, which causes a climate that is much drier than the rest of the state (Core 1966). The area is characterized by an average annual precipitation of 82 cm, an average annual temperature of 10.9° C, and a growing season of approximately 144 days (Estepp 1992). Forests of the region can be characterized as being part of the former oak-chestnut cover type (Braun 1950).

Table 3.1. Mean overstory composition values (IV = 0.5*(relative density + relative basal area)).

Species	Basal Area (m²/ha)	Density (trees/ha)	Importance Value (IV)
<i>Acer pensylvanicum</i>	0.08	5.56	0.77
<i>Acer rubrum</i>	0.41	18.89	2.90
<i>Acer saccharum</i>	0.06	1.11	0.26
<i>Ailanthus altissima</i>	0.03	0.56	0.12
<i>Amelanchier arborea</i>	0.02	1.11	0.17
<i>Betula lenta</i>	0.08	1.11	0.29
<i>Carya glabra</i>	1.30	46.11	7.71
<i>Carya ovata</i>	0.01	0.56	0.08
<i>Carya tomentosa</i>	0.09	3.89	0.62
<i>Nyssa sylvatica</i>	0.08	3.89	0.58
<i>Ostrya virginiana</i>	0.05	3.33	0.47
<i>Pinus pungens</i>	2.19	51.67	10.25
<i>Pinus rigida</i>	1.19	26.11	5.36
<i>Pinus strobus</i>	0.89	22.22	4.30
<i>Pinus virginiana</i>	2.14	65.00	11.55
<i>Prunus serotina</i>	0.03	1.11	0.18
<i>Quercus alba</i>	0.13	3.89	0.69
<i>Quercus coccinea</i>	0.32	8.33	1.57
<i>Quercus prinus</i>	10.94	127.78	37.37
<i>Quercus rubra</i>	1.27	34.44	6.42
<i>Quercus velutina</i>	0.86	13.89	3.33
<i>Robinia pseudoacacia</i>	0.41	15.56	2.54
<i>Sassafras albidum</i>	0.22	10.56	1.59
<i>Tsuga canadensis</i>	0.19	4.44	0.88
Total	22.98	471.11	100.00

The topography is highly dissected with slopes ranging from 6 to 70 %. Elevation ranges from 573 to 848 m. The predominant soil type is of the Berks-Weikert association. They are loamy-skeletal, mixed, Dystrochrepts formed from acidic shale, siltstone, or sandstone bedrock. The soils are droughty, infertile, and often contain numerous rock outcroppings (Estepp 1992).

Table 3.2. Mean understory composition values (IV = 0.5*(relative density + relative percent cover)).

Species	Cover (%)	Density (trees/ha)	Importance Value (IV)
<i>Acer pensylvanicum</i>	6.88	7361.11	11.63
<i>Acer rubrum</i>	17.71	12986.11	22.60
<i>Ailanthus altissima</i>	0.07	69.44	0.11
<i>Amelanchier arborea</i>	8.19	2083.33	5.63
<i>Carya glabra</i>	22.29	2222.22	11.06
<i>Fraxinus americana</i>	0.28	69.44	0.19
<i>Nyssa sylvatica</i>	9.44	1736.11	5.67
<i>Ostrya virginiana</i>	5.56	2569.44	5.24
<i>Pinus rigida</i>	0.28	69.44	0.19
<i>Pinus strobus</i>	0.56	416.67	0.72
<i>Pinus virginiana</i>	5.56	3402.78	6.26
<i>Prunus serotina</i>	0.69	138.89	0.43
<i>Quercus prinus</i>	18.96	2847.22	10.59
<i>Quercus rubra</i>	6.39	1250.00	3.93
<i>Quercus velutina</i>	21.25	1458.33	9.73
<i>Robinia pseudoacacia</i>	2.71	347.22	1.44
<i>Sassafras albidum</i>	7.01	1597.22	4.59
Total	133.82	40625.00	100.00

The site was purchased by the United States Department of Agriculture (USDA) Forest Service in 1923. After its purchase, no fires were documented in the Dunkle Knob area until a prescribed fire was ignited by GWNF personnel in March of 2004 (Marsh 2005). Five timber sales were documented in the study area after 1923. This harvesting occurred on and around the base of the mountain in the 1950s and 1960s, as well as in the early 1970s and 1980s (Figure 3.1)¹.

¹ Data on file with the USDA Forest Service, GWNF, North River Ranger District, Harrisonburg, VA.

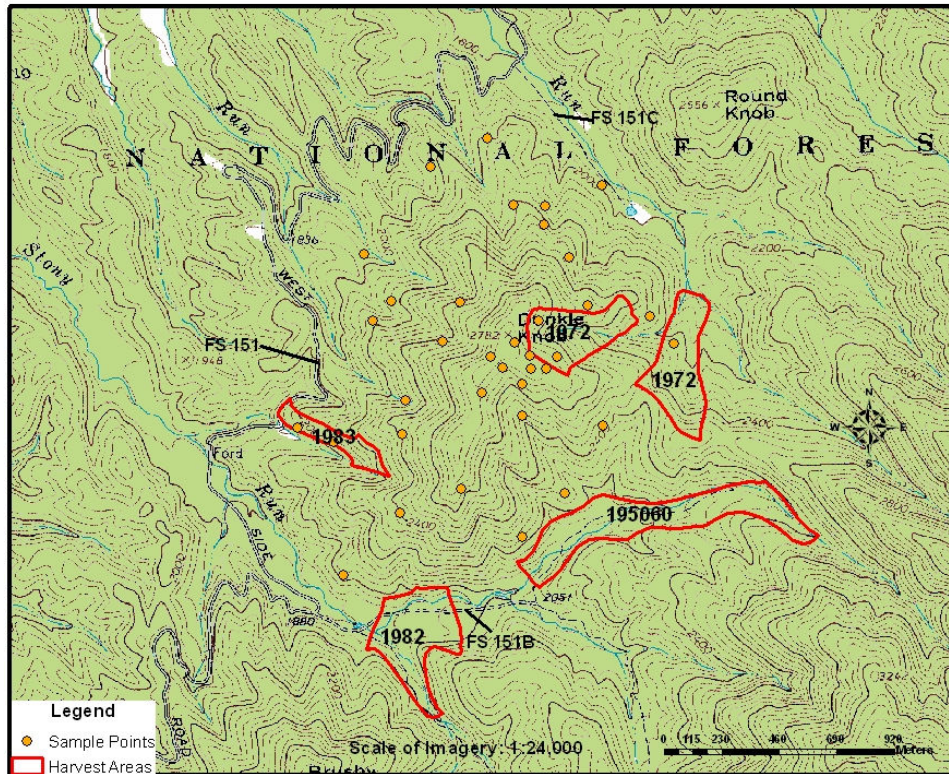


Figure 3.1. Harvest areas on, and around the base of, Dunkle Knob.

Harvesting on the mountain was restricted to lower, easily accessible, slopes and in a cove area where higher value timber would have resided (Figure 3.1). The area was first entered to extract timber in the 1950s via Forest Service Road 151B. A 14.6 ha area at the southeastern base was the target. During both the 1950s and 60s this area was selectively cut leaving a “sparse overstory of poorer material.” Timber Stand Improvement (TSI) was applied in 1978 with USFS personnel injecting undesirable trees with 2,4-D. The Dunkle Knob area was harvested again in 1972. This time, two areas, one in a cove (8.0 ha) and one on a lower slope (8.8 ha), on the northeastern side of the mountain were clearcut. These areas were accessed via extensions off of Forest Service Road 151C (Figure 3.1). Following timber removal, site preparation for natural

regeneration of hardwoods on these areas was carried out by injecting 2,4-D into all stems greater than 5.1 cm DBH or greater than 4.6 m tall. Three of the current study sample points fell into these areas that were cut in 1972 (Figure 3.1). Ten years later an 8.4 ha area was harvested at the southerly base of the mountain. The harvest type in this area was a clearcut with ten total reserve trees. The following year (1983) a similar cut (clearcut with four total reserve trees) was applied to a 4.0 ha lower slope area on the southwestern side of the mountain. Chainsaw site preparation for natural regeneration was used following the two most recent harvests. Eastern white pine (*Pinus strobus* L.) was planted in some cases after harvesting.

Field Methods

Eighteen circular 0.05 ha plots were randomly established on northeast- (315-235°) and southwest- (236-314°) facing slopes (36 total). Nine plots each were established above and below 748 m elevation, stratifying the mountain into four sampling areas: northeast aspect/upper slope (NEU), northeast aspect/lower slope (NEL), southwest aspect/upper slope (SWU), and southwest aspect/lower slope (SWL) (see Marsh (2005) for complete sampling experimental design). To aid in future relocation, sample points were marked with a Global Positioning System (GPS) and a piece of steel rebar was driven into the ground and marked with flagging at each plot center.

Each plot was inventoried in the summer of 2007 to assess stand composition. All trees having a DBH greater than 12.7 cm were sampled and identified to species. DBH was measured to the nearest 0.1 cm using a diameter tape.

A total of 165 cores were used in the disturbance reconstruction analysis (Table 3.3). At least one increment core was extracted at each plot in the summer of 2004 using an increment borer. Cores were extracted at breast height (1.3 m above the ground) and parallel to topography to avoid tension or compression wood. Larger, and presumably older, trees were preferentially selected in hopes of obtaining the longest chronological records available. No trees having a DBH < 3.5 cm were sampled.

Table 3.3. Number of cores by species.

Species	# Cores
<i>Pinus pungens</i>	25
<i>Pinus rigida</i>	14
<i>Pinus strobus</i>	14
<i>Pinus virginiana</i>	26
<i>Quercus prinus</i>	59
<i>Quercus rubra</i>	27
Total	165

To assess how harvesting related to the establishment of the exotic invasive tree-of-heaven, a GPS was used in the summer of 2007 to mark waypoints of areas where tree-of-heaven had ascended into the main canopy. The sampling method can be described as a landscape-level timed meander. Areas with single or multiple tree-of-heaven stems were identified while traversing the mountain between June 2007 and August 2007. Although it is difficult to say how close the sample came to enumerating all of the pockets of tree-of-heaven across the mountain, they were easy to spot because an abundance of 0.5 to 3.5 m tall tree-of-heaven regeneration (likely adventitious shoots, or suckers, arising from parent tree roots) was always surrounding main canopy level stems (J. Pomp, personal observation; Figure 3.2). Increment cores (n = 37) were extracted

from both large and small tree-of-heaven in these areas. The largest tree-of-heaven was always cored and no stems having a DBH < 8.0 cm were sampled.



Figure 3.2. Abundant regeneration surrounding a co-dominant tree-of-heaven on Dunkle Knob. The man in the picture is just less than two meters tall.

Laboratory Techniques

All cores were dried, mounted, and then sanded with progressively finer grit sandpaper to expose the annual rings (Stokes & Smiley 1968). Growth rings were dated for all species. Ring widths were measured for chestnut oak and all of the sampled pine species. Unfortunately, due to time constraints, the ring widths of northern red oak (*Quercus rubra* L.) and tree-of-heaven samples were not measured. Widths were measured to the nearest 0.001 mm using a Leica Stereo Zoom-5 binocular microscope, an Acu-rite measuring stage and a Quick-Check 1000 digital readout, in conjunction with Measure

J2X[®] software. Dating and measurements were validated using the program COFECHA (Grissino-Mayer and others 1997).

Data Analysis

Diameter distribution

The 2007 inventory data was summarized into four centimeter diameter classes to create a stand table (Trees per Hectare (TPH) by diameter class and species) of the three most abundant (based on relative basal area) overstory species in each sampling area. These tables were converted into a histogram for visual analysis.

Age distribution

For graphical comparison, DBH was plotted over establishment date for each sample tree by sampling area.

Growth Increment Patterns

Growth releases were identified from tree ring widths using the radial growth averaging technique proposed by Nowacki and Abrams (1997). Percent growth change (% GC) was calculated using the equation:

$$\%GC = [(M_2 - M_1) / M_1] \times 100 ,$$

where M_1 = preceding 10-year mean radial growth (inclusive of the disturbance year), and M_2 = subsequent 10-year mean (exclusive of the disturbance year) (Nowacki and

Abrams 1997). Percent GC was calculated for each sample tree (n =165) and averaged for each species, except northern red oak and tree-of-heaven, in each sampling area to produce master chronologies. Because of the low number of samples, pitch pine and table mountain pine data were combined for the master chronology calculation, with the exception of cores taken from the SWL sampling area. In addition, residuals from plots that fell in the harvested areas were analyzed separately. Two chestnut oak residuals were sampled in 1972 harvest area, while three residuals (two eastern white pines and one pitch pine) were sampled in the 1983 harvest area.

To assess how climatic trends affected radial growth responses, yearly reconstructed Palmer Drought Severity Index (PDSI) data were collected for the region (National Climactic Data Center 2006). The yearly PDSI data were converted to PDSI differences ($PDSI_{diff}$) using the formula:

$$PDSI_{diff} = P_2 - P_1,$$

where P_1 = preceding 10-year PDSI mean, and P_2 = subsequent 10-year PDSI mean (Nowacki and Abrams 1997). Simple linear regression analyses were performed examine the relationship between $PDSI_{diff}$ (independent variable) and each of % GC master chronologies. The highest recorded $PDSI_{diff}$ for each species/study area combination was then inserted into the resulting regression equations to roughly estimate the upper limits of influence on % GC master chronologies (Nowacki and Abrams 1997). If the resulting values were higher than 25 %, the release detection thresholds were adjusted upward

accordingly. Based on thinning data and polynomial regression equations relating % GC to basal area reductions, the 25 % minimum release detection threshold can be considered conservative, and is an appropriate threshold for documenting canopy disturbances in oak forests (Nowacki and Abrams 1997).

For each sample tree, releases were identified as abrupt and sustained periods of % GC that were above the 25 %, or the regression equation determined, threshold. Major releases were identified as events where % GC \geq 100 %, while moderate releases were identified by % GC values between 50 and 99 % (Lorimer and Frelich 1989). Small releases were identified as events where % GC values ranged from 25 to 49 % (Schuler and Fajvan 1999, Rentch and Hicks 2003).

The numbers of releases were tabulated by decade for each of the four sampling areas. Using this data, major disturbance decades were denoted as peaks comprised of > 25 % of the sample trees (Nowacki and Abrams 1997). Major disturbance events were not assigned unless at least two trees exhibited a release (Nowacki and Abrams 1997). Furthermore, because of the drier nature of the study site, major disturbances that occurred during positive peaks of PDSI_{diff} (decades of 1810, 1820, 1960, and 1970) (Figure 3.3) were simply considered increases in tree growth due to favorable climate shifts.

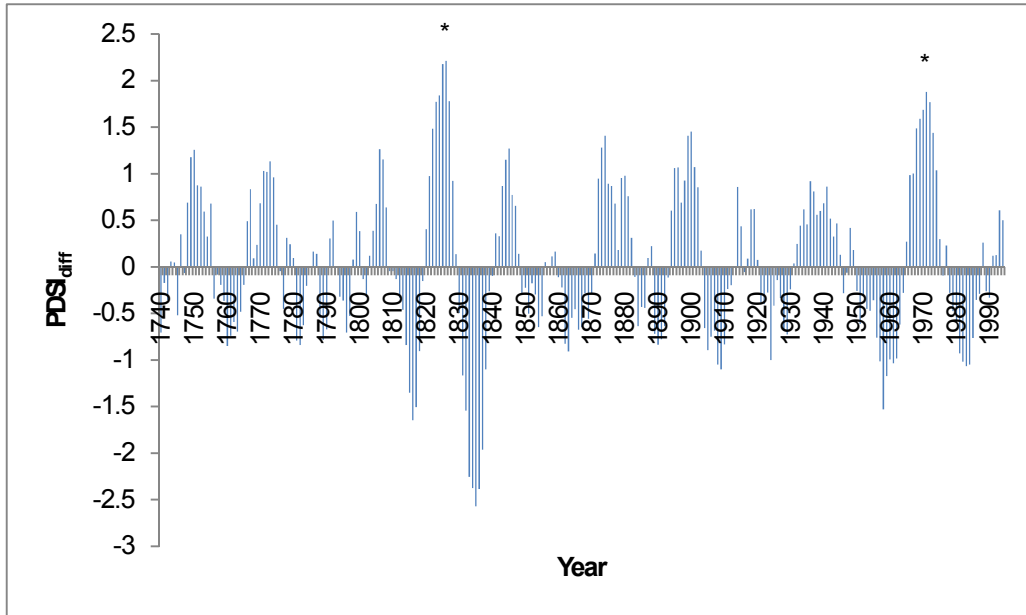


Figure 3.3. PDSI_{diff} chronology for the Dunkle Knob area. * indicates period when favorable climate would have positively affected tree growth.

Frequencies, or return intervals, of major disturbances were determined in each sampling area for both the exploitation era (prior to 1900) and the modern era (after 1900).

Exploitation era frequencies were calculated using the equation:

$$Disturbance\ Frequency_{exploitation} = \frac{1900 - old_s}{d_s},$$

where old = date of the first year % GC could be calculated in the sampling area s, and d = number of major disturbances. Modern era frequencies were calculated by dividing 94 (length of % GC chronologies in modern era) by the number of major disturbances. In both cases, evident age cohorting was also counted as a major disturbance.

In addition, average early growth rates and overall average growth patterns were analyzed to determine the canopy accession trends for each species in each sampling area. Species were considered gap origin if their growth rate during the first twenty-five years was high, and if the overall growth pattern was flat or slightly decreasing (Lorimer, 1985, Lorimer and Frelich 1989, Nowacki and Abrams 1997, Rentch and Hicks 2003). Species were considered gap origin - gap release if they displayed a high early growth rate and a growth pattern that exhibited subsequent reduced growth, followed by a major release (Rentch and Hicks 2003). Species were considered understory origin - gap release if their growth rate was low during the first twenty-five years and exhibited one or more major releases (Lorimer and Frelich 1989, Nowacki and Abrams 1997, Rentch and Hicks 2003).

Tree-of-heaven Age and Distribution

Waypoints of areas where tree-of-heaven had ascended into the main canopy were overlaid on the harvested areas shapefile using ArcMap 9.1 software (© Environmental Systems Research Institute (ESRI), Redlands, USA 2005). Establishment date was added to the attribute table of the waypoint shapefile. Maps were produced for visual examination and interpretation.

Results

Diameter Distribution

Diameter distributions were quite similar for all four sampling areas of the study site. These exhibited the classic “reverse-J” distribution shape, characteristic of an uneven-

aged structure (Figures 3.4 - 3.7). Forests with this structure are composed essentially of even-aged stands or groups of different ages (Smith 1997). For the most part, chestnut oak dominated the larger size classes in all four sampling areas. Virginia pine was one of the three most abundant species in the southwestern areas, while pignut hickory expressed high abundance on the upper slopes and table mountain pine on lower slopes. Pitch pine was present as one of the three most abundant species only in the NEL sampling area.

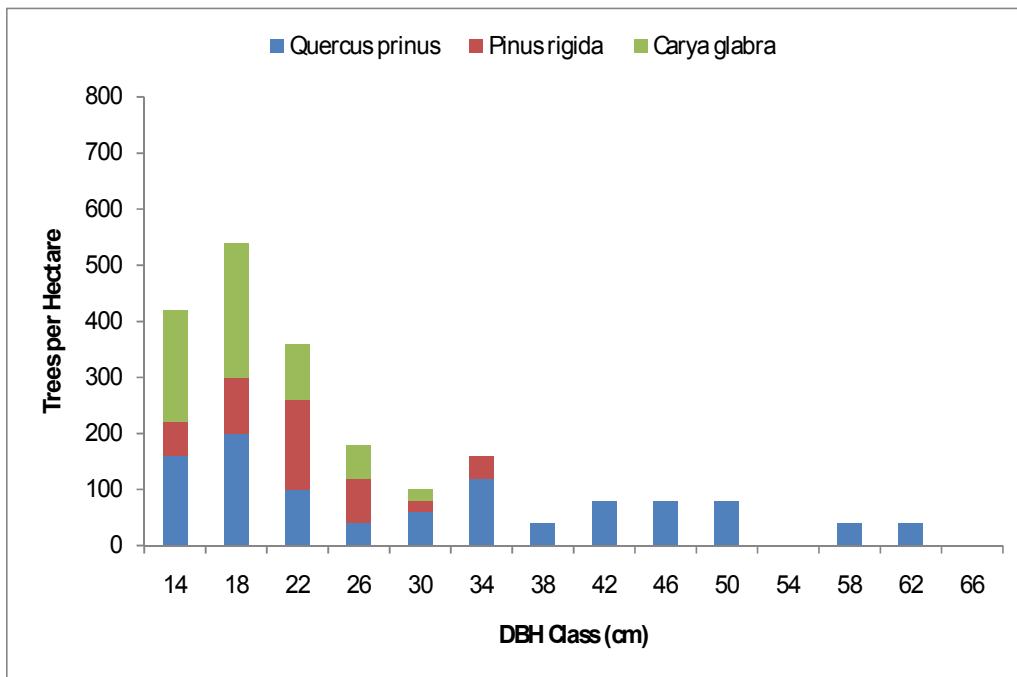


Figure 3.4. Diameter distributions for the three most abundant species: NEU.

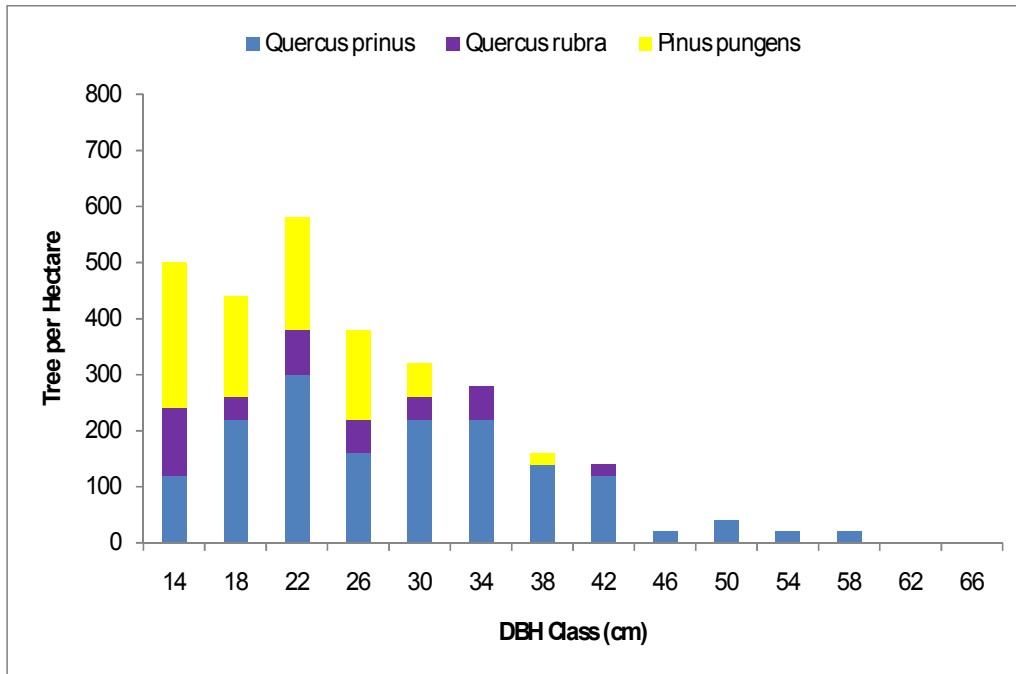


Figure 3.5. Diameter distributions for the three most abundant species: NEL

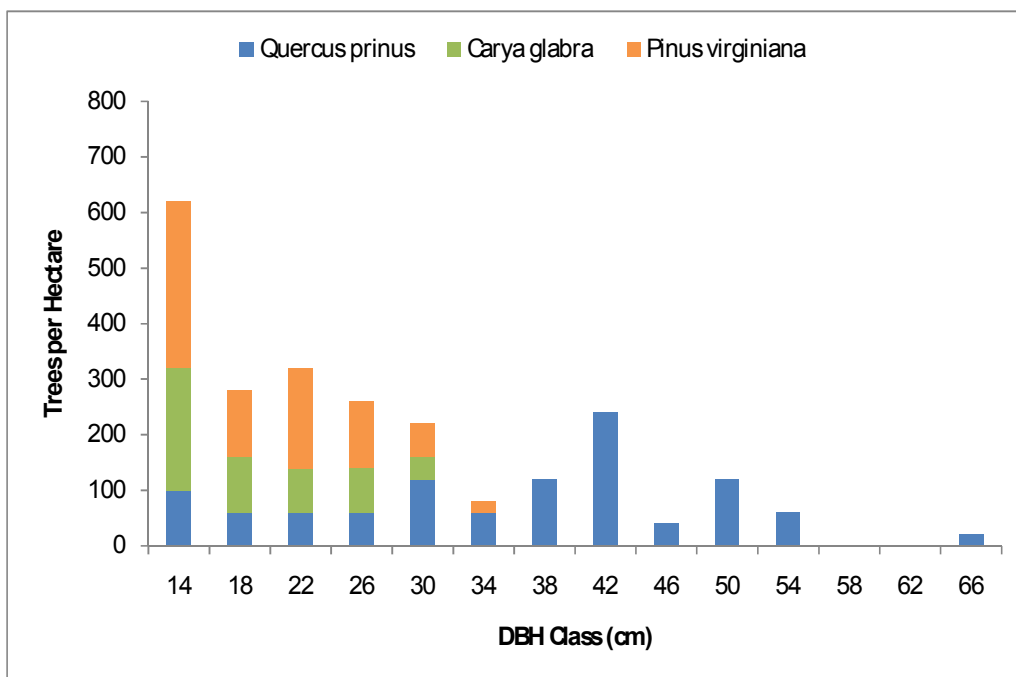


Figure 3.6. Diameter distributions for the three most abundant species: SWU

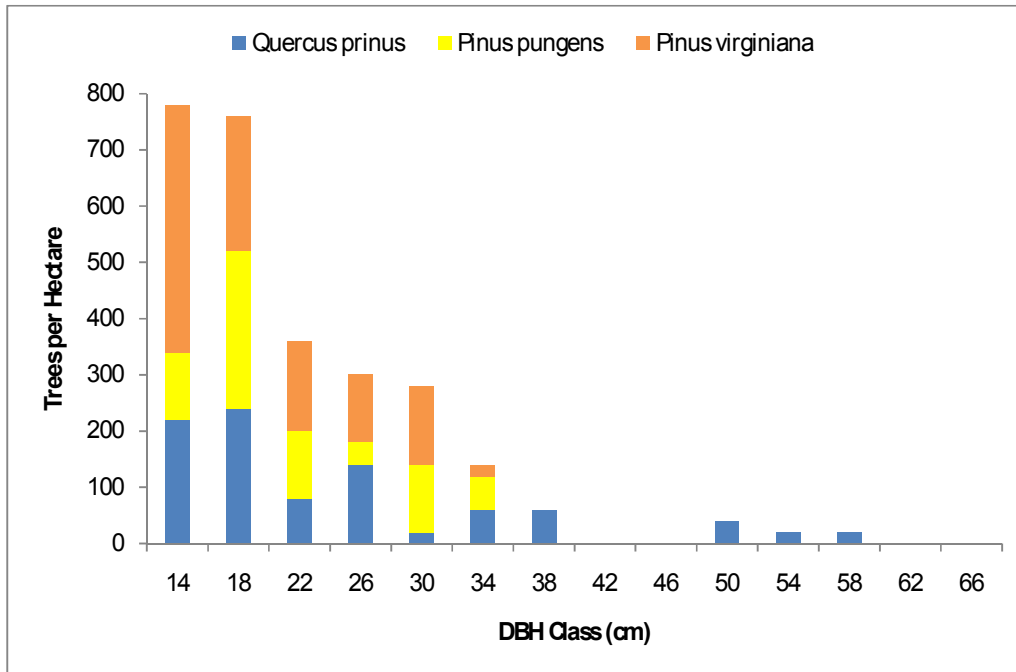


Figure 3.7. Diameter distributions for the three most abundant species: SWL

Age Distribution

Age distributions were also quite similar for all four sampling areas. They provided further evidence of uneven-aged structure, containing at least two to four recognizable cohorts (Figures 3.8 - 3.11, top). In all areas, the youngest of these cohorts arose around the turn of the twentieth century (1890 to 1910). More trees established in the decade of 1910 than any other decade examined. There has been more or less continuous establishment ever since. The oldest of the sample trees, a chestnut oak, originated in 1741 the NEL sampling area (Figure 3.9, top). All sampling areas contain at least one cohort that developed prior to 1850. These old cohorts are composed of only chestnut oak.

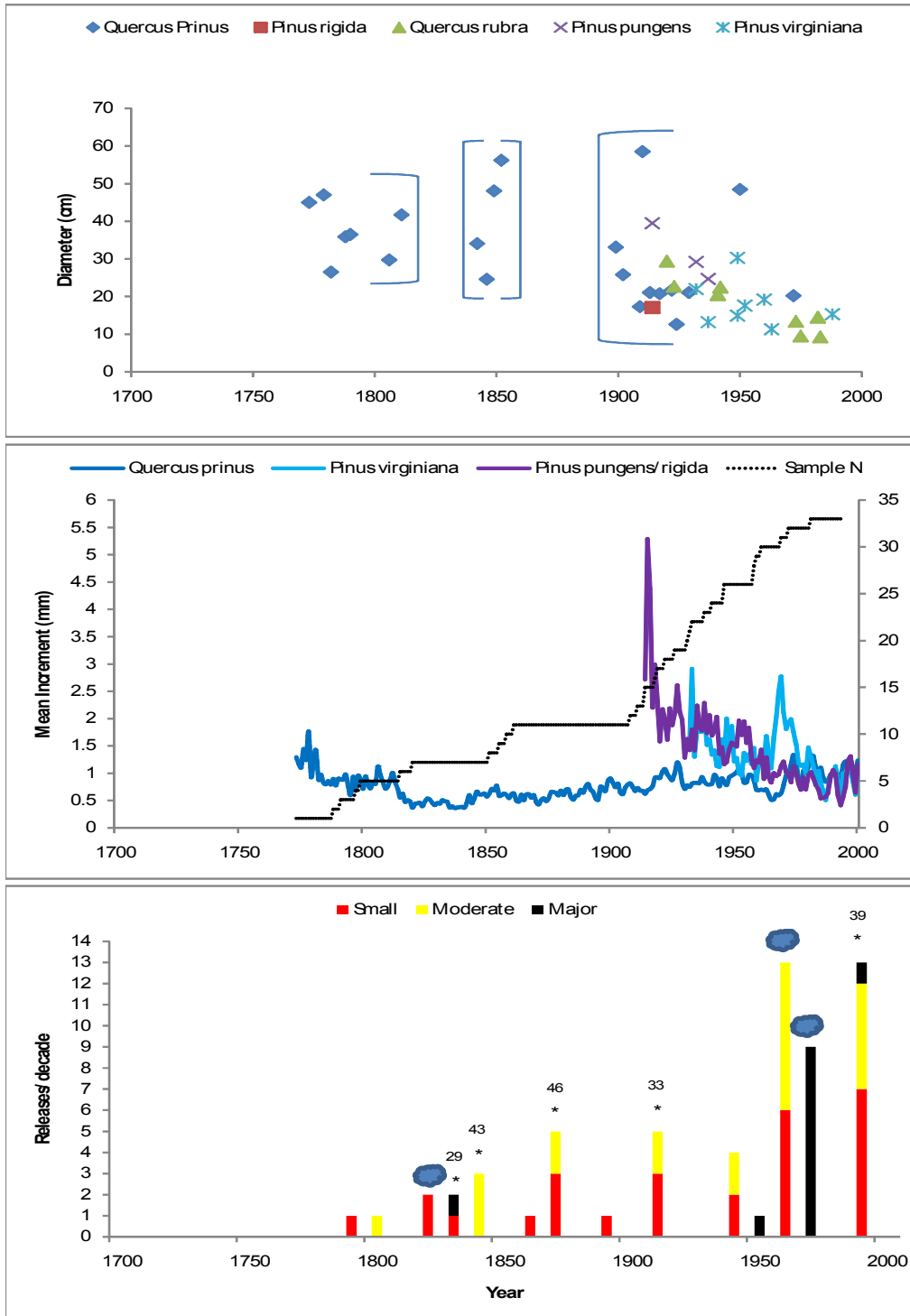


Figure 3.8. A comparative compilation of age-diameter relations (top), average growth patterns (middle), and number of releases (bottom) in the NEU sampling area. Brackets denote cohorts (top). Asterisks indicate decades in which a major disturbance occurred and clouds indicate decades when favorable climate would have positively affected tree growth (bottom). The percentage of sample trees showing releases during major disturbances are listed above the asterisks.

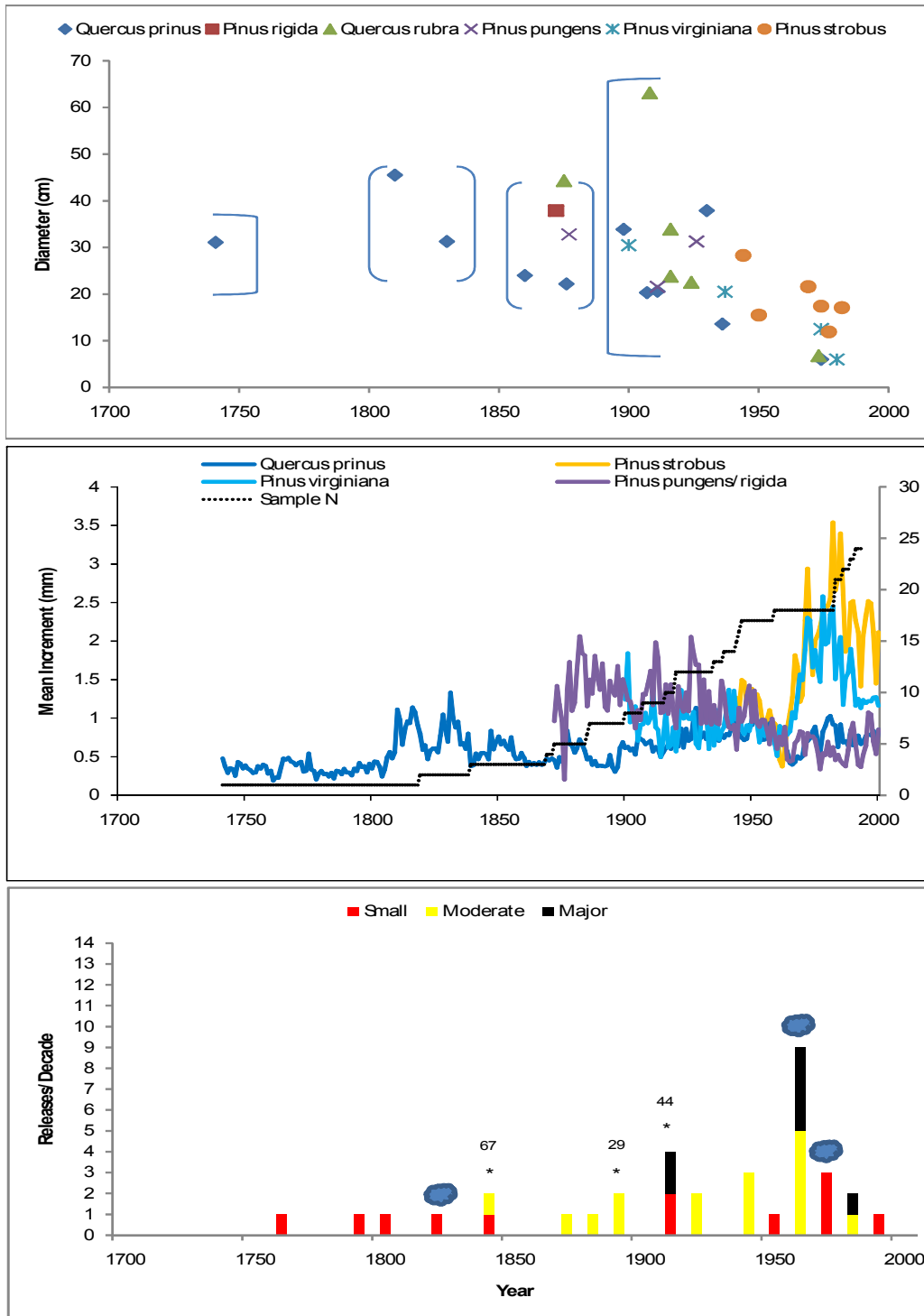


Figure 3.9. A comparative compilation of age-diameter relations (top), average growth patterns (middle), and number of releases (bottom) in the NEL sampling area. Brackets denote cohorts (top). Asterisks indicate decades in which a major disturbance occurred and clouds indicate decades when favorable climate would have positively affected tree growth (bottom). The percentage of sample trees showing releases during major disturbances are listed above the asterisks.

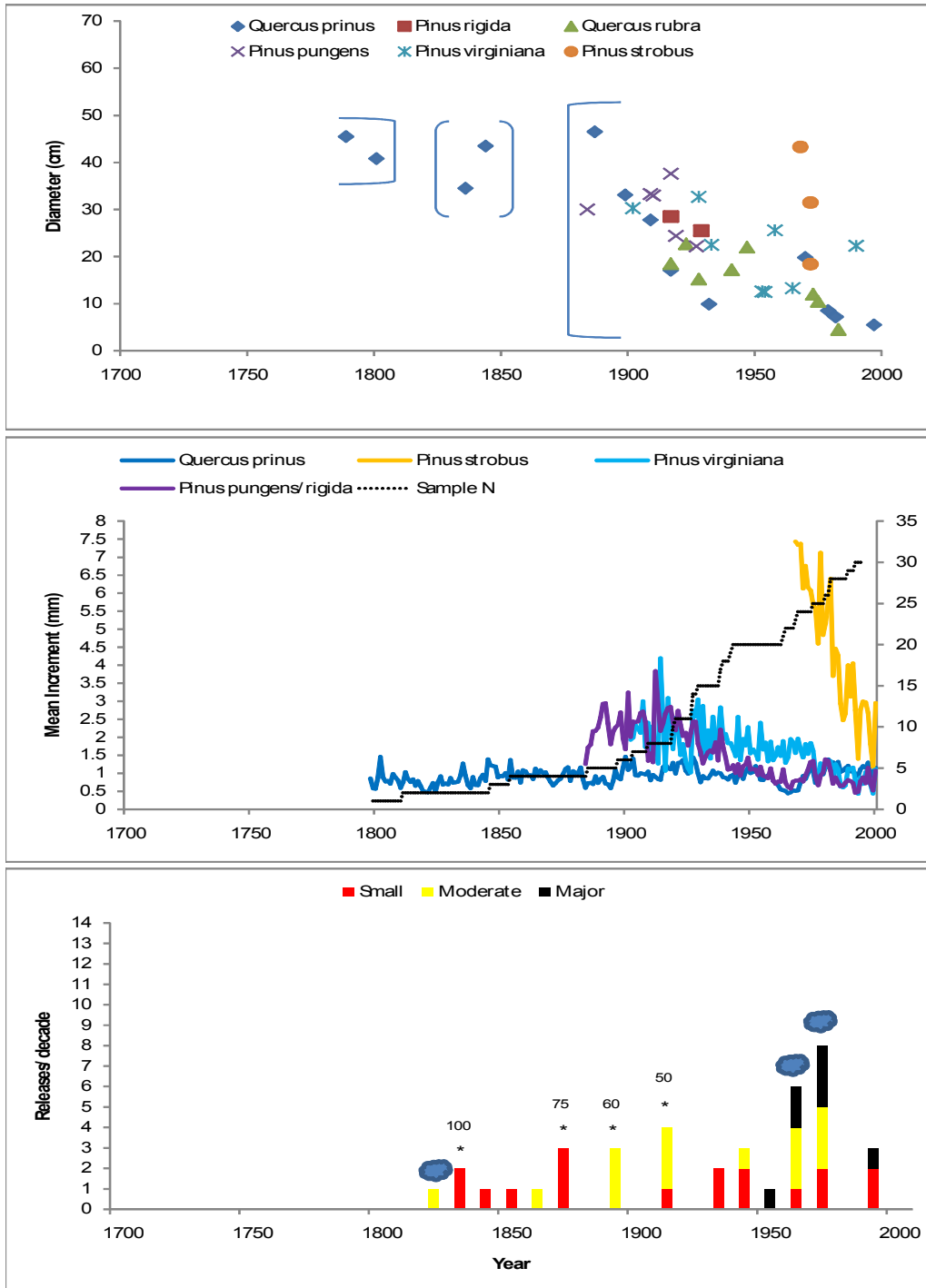


Figure 3.10. A comparative compilation of age-diameter relations (top), average growth patterns (middle), and number of releases (bottom) in the SWU sampling area. Brackets denote cohorts (top). Asterisks indicate decades in which a major disturbance occurred and clouds indicate decades when favorable climate would have positively affected tree growth (bottom). The percentage of sample trees showing releases during major disturbances are listed above the asterisks.

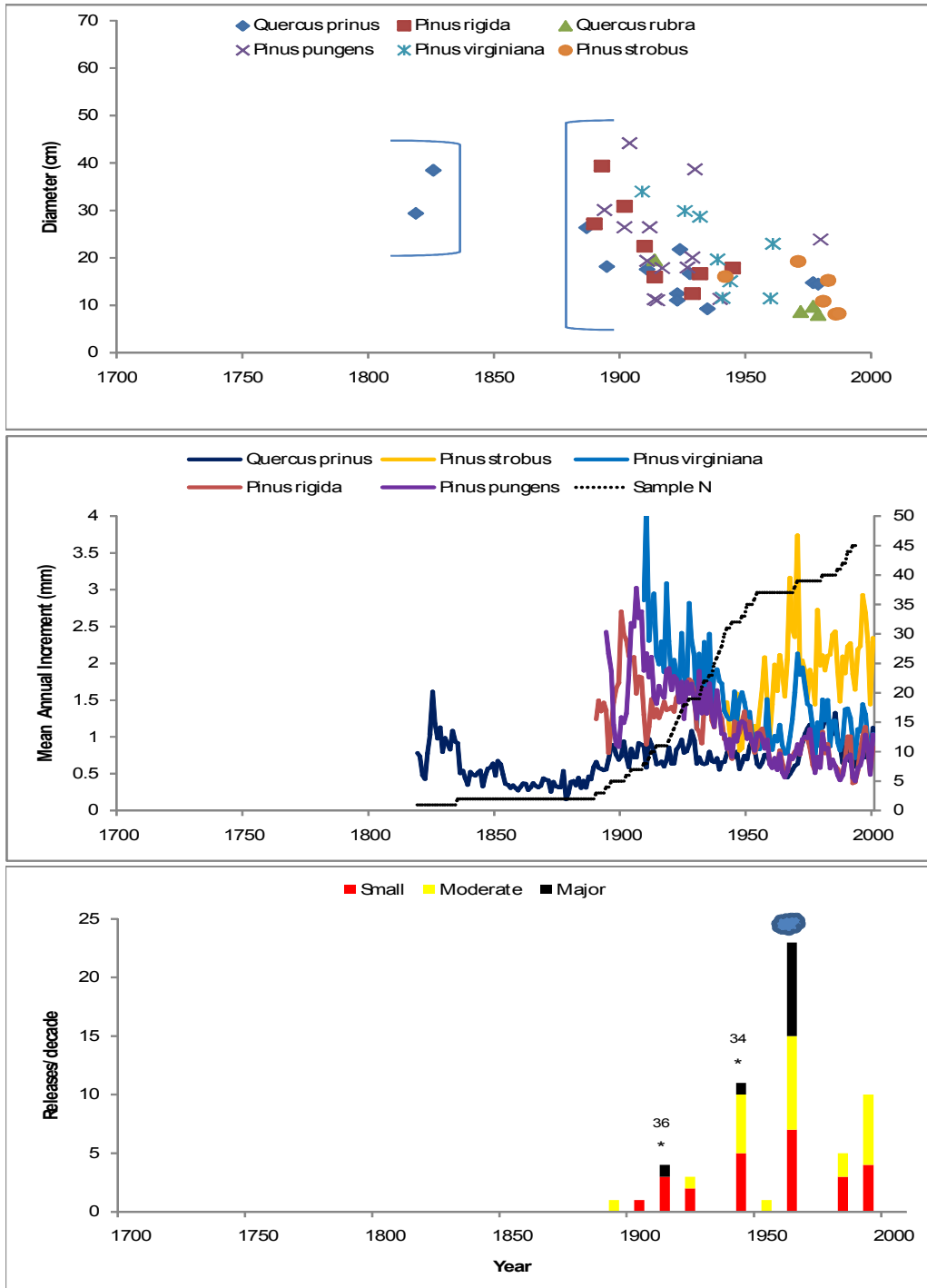


Figure 3.11. A comparative compilation of age-diameter relations (top), average growth patterns (middle), and number of releases (bottom) in the SWL sampling area. Brackets denote cohorts (top). Asterisks indicate decades in which a major disturbance occurred and clouds indicate decades when favorable climate would have positively affected tree growth (bottom). The percentage of sample trees showing releases during major disturbances are listed above the asterisks.

Growth Increment Patterns

Fifteen of the 19 ($\approx 79\%$) % GC master chronologies showed significant positive linear relationships with $PDSI_{diff}$ (Table 3.4). For these chronologies, the upper % GC responses due to climate ranged from 1.36 to 67.79 %. Eight of these values exceeded the Nowacki and Abrams (1997) 25 % release detection threshold (Table 3.4).

Table 3.4. Regression analyses relating %GC master chronologies to $PDSI_{diff}$.

Species	Location	Regression Model	n	r	R ²	F	P	Max % GC due to climate
<i>Pinus pungens</i>								
	NEU	-	0	-	-	-	-	-
	NEL	-	0	-	-	-	-	-
	SWU	-	0	-	-	-	-	-
	SWL	%GC = -7.37 + 10.77($PDSI_{diff}$)	91	0.42	0.18	19.45	<0.001*	12.88
	1972 Harvest	-	0	-	-	-	-	-
	1983 Harvest	-	0	-	-	-	-	-
<i>Pinus pungens/virginiana</i>								
	NEU	%GC = -12.72 + 7.49($PDSI_{diff}$)	71	0.37	0.14	11.73	0.001*	1.36
	NEL	%GC = -3.15 - 0.08($PDSI_{diff}$)	113	0.00	0.00	0.00	0.979	-
	SWU	%GC = -11.78 + 16.93($PDSI_{diff}$)	101	0.61	0.37	58.39	<0.001*	20.06
	SWL	-	0	-	-	-	-	-
	1972 Harvest	-	0	-	-	-	-	-
	1983 Harvest	-	0	-	-	-	-	-
<i>Pinus rigida</i>								
	NEU	-	0	-	-	-	-	-
	NEL	-	0	-	-	-	-	-
	SWU	-	0	-	-	-	-	-
	SWL	%GC = -6.02 + 13.59($PDSI_{diff}$)	104	0.62	0.39	60.94	<0.001*	19.53
	1972 Harvest	-	0	-	-	-	-	-
	1983 Harvest	%GC = 3.58 + 28.23($PDSI_{diff}$)	44	0.81	0.65	78.66	<0.001*	56.66
<i>Pinus strobus</i>								
	NEU	-	0	-	-	-	-	-
	NEL	%GC = 13.61 + 28.82($PDSI_{diff}$)	41	0.53	0.28	15.90	<0.001*	67.79
	SWU	%GC = -34.62 + 3.98($PDSI_{diff}$)	17	0.20	0.04	0.66	0.430	-
	SWL	%GC = 0.80 - 15.40($PDSI_{diff}$)	43	0.33	0.11	5.38	0.025*	-
	1972 Harvest	-	0	-	-	-	-	-
	1983 Harvest	%GC = 12.69 + 8.11($PDSI_{diff}$)	52	0.44	0.19	11.77	0.001*	27.93

Pinus virginiana

NEU	%GC = -8.50 + 4.11(PDSI _{diff})	53	0.14	0.02	0.86	0.359	-
NEL	%GC = 13.36 + 27.80(PDSI _{diff})	85	0.44	0.19	19.19	<0.001*	65.62
SWU	%GC = -15.79 + 9.28(PDSI _{diff})	92	0.67	0.45	67.22	<0.001*	1.65
SWL	%GC = -4.21 + 11.06(PDSI _{diff})	85	0.26	0.07	5.48	0.021*	16.58
1972 Harvest	-	0	-	-	-	-	-
1983 Harvest	-	0	-	-	-	-	-

Quercus prinus

NEU	%GC = 2.21 + 12.44(PDSI _{diff})	212	0.51	0.26	75.26	<0.001*	29.72
NEL	%GC = 2.50 + 9.70(PDSI _{diff})	244	0.40	0.16	46.61	<0.001*	23.95
SWU	%GC = 4.73 + 9.97(PDSI _{diff})	187	0.39	0.15	32.40	<0.001*	25.77
SWL	%GC = -1.93 + 15.37(PDSI _{diff})	166	0.58	0.34	84.16	<0.001*	26.96
1972 Harvest	%GC = 3.74 + 19.25(PDSI _{diff})	164	0.60	0.36	89.76	<0.001*	39.92
1983 Harvest	-	0	-	-	-	-	-

Major releases

The NEU area exhibited three non-climatic related major releases. These occurred in the decades of 1830, 1950, and 1990, respectively (Figure 3.8, bottom). Three major releases occurred in the NEL sampling area as well (Figure 3.9, bottom). In this area two major releases were detected in the decade of 1910, while the other occurred 1980. The SWU area showed two major releases, one in 1950 and one in 1990 (Figure 3.10, bottom).

Two major releases also occurred in the SWL sampling area (Figure 3.11, bottom).

These were identified in the decades 1910 and 1940. None of the residuals from either of the harvest areas exhibited major releases.

Moderate releases

Twelve non-climatic related moderate releases were detected in the NEU sampling area.

These releases were detected in the decades of 1800 (1), 1840 (3), 1870 (2), 1910 (2),

1940 (2), and 1990 (5) (Figure 3.8, bottom). Eleven moderate releases occurred in the NEL area: one each in the decades of 1840, 1870, 1880, and 1980, two each in the decades of 1890 and 1920, and three in the decade of 1940 (Figure 3.9, bottom). The SWU area showed nine moderate releases. These were observed in the decades of 1820 (1), 1860 (1), 1890 (3), 1910 (3), and 1940 (1), respectively (Figure 3.10, bottom). The most moderate releases were found in the SWL area (Figure 3.11, bottom). Here, 16 releases were detected: one each in the decades of 1890, 1920, and 1950, two in the decade of 1980, five in the decade of 1940; and six in the decade of 1990. Two moderate releases were identified in the 1972 harvest area. These occurred in the decades of 1870 and 1990, respectively. Neither of these releases were indicative of the harvesting. One of the eastern white pine residuals showed a moderate release in the 1983 harvest. This release occurred in the decade of 1980 and is, therefore, a result of the harvesting.

Small Releases

The NEU sampling area exhibited the highest number of small releases. Nineteen non-climactic related small releases were detected in this area during the decades of 1790 (1), 1830 (1), 1860 (1), 1870 (3), 1890 (1), 1910 (3), 1940 (2), and 1990 (7) (Figure 3.8, bottom). Only seven moderate releases occurred in the NEL area: one each in the decades of 1716, 1790, 1800, 1820, and 1840, and two in the decade of 1910 (Figure 3.9, bottom). The SWU area showed 14 major releases. These were detected during the decades of 1840 (1), 1850 (1), 1910 (1), 1830 (2), 1930 (2), 1940 (2), 1990 (2), and 1870 (3), respectively one in 1950 and one in 1990 (Figure 3.10, bottom). Eighteen small releases occurred in the SWL sampling area (Figure 3.11, bottom). These were identified

in the decades of 1900 (1), 1910 (3), 1920 (2), 1940 (5), 1980 (3), and 1990 (4). The chestnut oak residuals detected one small release in the 1972 harvest area during the decade of 1940. The eastern white pine residuals detected two small releases in the 1983 harvest area. One of these occurred in 1940, while the other was a result of the harvesting

Major Disturbance Decades

Five major disturbances, the most of any area, were identified in the NEU sampling area. Two of these major disturbances occurred during the decades of 1840 and 1910, coinciding well with cohorting (Figure 3.8). The other three major disturbances occurred in the decades of 1830, 1870, and 1990. Three major disturbances were identified in the NEL area. Again two of the major disturbances occurred during the decades of 1890 and 1910, corresponded with cohorting (Figure 3.9). The other major disturbance occurred in the decade of 1840. The SWU area also experienced two major disturbances that coincided with the cohorting around the turn of the twentieth century (Figure 3.10). Two more major disturbances were detected in this area, one each in the decades of 1830 and 1870. Only two major disturbances were identified in the SWL area. Again a major disturbance that occurred in 1910 corresponded with cohorting (Figure 3.11). The second major disturbance occurred in the decade of 1940.

Frequency of Major Disturbances

Intervals of major disturbances ranged from 26 to 94 years (Table 3.5).

Table 3.5. Return intervals (years) for major disturbances.

Era	-----Sampling Stratum-----				
	<u>NEU</u>	<u>NEL</u>	<u>SWU</u>	<u>SWL</u>	<u>Mean</u>
Exploitation (<1900)	32	30	26	81	42
Modern (>1900)	47	94	94	47	71

Canopy Accession Trends

All of the nineteen species/sampling stratum master chronologies were determined to be of gap origin.

Tree-of-heaven Establishment

Twenty-one areas where tree-of-heaven had ascended into the main canopy were identified while traversing the mountain (Figure 3.12). Approximately 71 %, or 15, of these areas were within areas that had previously undergone harvesting. Four of the remaining six areas were around the base of the mountain near roads. Two of these were found along Forest Service Road 151B, located at the southeastern base of the mountain, which provided access to the two harvests in this area. The other two, found at the northwestern base of the mountain (Figure 3.12), were 3 and 78 meters away from Forest Service Road 151, respectively. Only two areas were found on the slopes of the mountain where harvesting had not occurred. One overstory tree-of-heaven was also encountered on one plot during the inventory (Figure 3.12). Both field observations and Figure 3.12 indicate that, other than in harvested areas, canopy level tree-of-heaven were restricted to moister riparian areas.

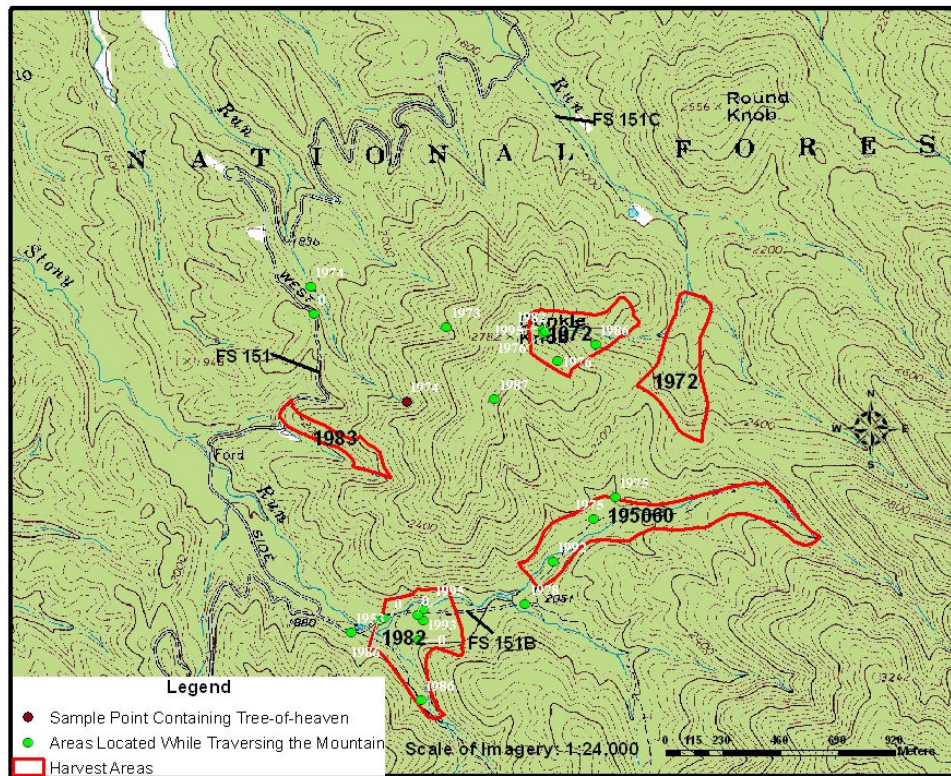


Figure 3.12. Establishment dates of the oldest tree in areas where tree-of-heaven has ascended into the main canopy. Because of their close proximity to other trees, increment cores were not extracted from areas labeled with a zero (four instances).

All of the oldest tree-of-heaven, with one exception, attained breast height after the harvesting occurred (Figure 3.12). The oldest tree of heaven is currently 55 years old (breast height attainment occurred in 1953) and has a DBH of 37 cm. This individual is located at the southern base of Dunkle Knob, next to the gate for Forest Service Road 151B (Figure 3.12).

Discussion

The results of this study represent some interesting findings and help to explain forest stand dynamics at the landscape level in the Dunkle Knob area.

General Disturbance History

The demographic and structural evidence indicate that all sampling areas on Dunkle Knob contain multiple cohorts and exhibit an uneven-aged structure. Tree ages currently range from 10 to 268 years. This age distribution and structure is an indication of multiple disturbances of varying intensities across the study area. The majority (81 %) of the releases identified from % GC data were small and moderate, which were likely a result of death or removal of one or two trees that neighbored a sample tree (Rentch and others 2002). Prior to fire exclusion in the study area, low-intensity fires could have caused some of the releases in localized areas.

Interestingly, all sampling areas experienced a major disturbance between the decades of 1890 and 1910. In all cases, these major disturbance events were consistent with age cohorting (Figures 3.8 - 3.11). The data suggested that this was the only area-wide disturbance event at the Dunkle Knob site over the 244 year period examined. Multiple species establishments occurred as a result. In the Appalachians, and throughout West Virginia, intensive harvesting practices, often followed by high-intensity fires, characterized the period from about 1880 to 1930 (Brose and others 2001). It can be assumed that, before it was purchased by the USDA Forest Service, the site was subject to exploitive logging. Some of the tremendously steep and rocky slopes of the Dunkle

Knob study site were probably inaccessible for timber harvesting (J. Pomp, personal observation), and would have allowed for scattered pockets of residuals. Furthermore, in readily accessible areas, harvesting was likely targeted at higher value oak species and, if it existed with any importance on the site, American chestnut (*Castanea dentata* (Marsh.) Borkh.). Although the work of Braun (1950) indicates that the Dunkle Knob area is situated in what was once a region dominated by the oak-chestnut forest type, no remnant carcasses or stumps were encountered there in 2007 (J. Pomp, personal observation). In any case, the harvesting would have left only low value residuals, primarily chestnut oaks and a small number of table mountain and pitch pines. American chestnut would not have been able to re-establish because of the introduction of the lethal canker causing pathogen *Cryphonectria parasitica* (Murrill) Barr.

Nearly all of the serotinous-coned table mountain and pitch pines established just after the turn of the 20th century, perhaps suggesting that a large fire followed the harvesting. Neither of these species, except for two cases on the SWL section of the mountain, established after the 1930's. For the most part, the only species that established after the decade of 1930 were chestnut oak, northern red oak, and Virginia pine. Eastern white pine was being planted by the USDA forest service on all areas, except in the NEU sampling area, from the 1940s through the 1980s.

Chestnut oak and pitch pine were the only species to exhibit releases from the combination of disturbances during the turn of the 20th century. However, these were the only species which were alive prior to the disturbance. These trees would readily accept

the extra resources provided by the harvesting, fire-induced mortality, and American chestnut decline. Average ages at release for these chestnut oaks ranged from 50 to 73 years. This was not surprising, given that other studies have documented the releases in oaks of similar ages (Rentch and Hicks 2003, Nowacki and Abrams 1997). Despite none of the chestnut oak samples exhibiting an understory origin-gap release canopy accession trend, this finding provides evidence of the ability of chestnut oak to endure lower positions in the canopy for extended time. Only one individual pitch pine showed a release around the turn of the 20th century. As expected, this individual was young when it was released (10 years old).

Interestingly, only two of the decades denoted as major disturbance decades occurred after the 1910s. One occurred in the SWL sampling area in the decade of 1940 (Figure 3.11, bottom), while the other occurred during the 1990s in the NEU sampling area (Figure 3.8, bottom). The major disturbance in the SWL sampling area may have been a result of undocumented harvesting or fire. The major disturbance in the NEU sampling may have been a result of gypsy moth (*Lymantria dispar* L.) defoliation. During the late 1980s and early 1990s several major gypsy moth outbreaks occurred in West Virginia, especially in the eastern panhandle. According to USDA Forest Service annual defoliation data (USDA Forest Service, Northern Research Station 2007), it appears that defoliation could have occurred in the study area between 1985 and 1995. The fact that most of the releases during this decade were exhibited by the pines, and approximately 60 % of the chestnut oaks that did not show releases exhibited negative % GC, reinforces this hypothesis.

Canopy Accession Trends

All of the species/sampling area master chronologies were determined to be of gap origin. This was expected because of the low tolerance to shade of the species examined. All of the yellow pines examined in this study are classified as shade intolerant (Hicks 1998, Williams & Johnson 1990). Chestnut oak is classified as intermediate in shade tolerance, but is perhaps more shade intolerant than others in the white oak group (Hicks 1998).

Disturbance Frequency

Comparisons of major disturbance frequencies to frequencies found by Schuler and Fajvan (1999) and Nowacki and Abrams (1997) are irrelevant because their return intervals are based on stand-wide disturbances. Major disturbance frequencies, during both the exploitation and modern eras at the Dunkle Knob study site were much longer than the frequencies of multiple tree events found by Rentch and Hicks 2003. However, their study was conducted at the stand level and many more cores were extracted relative to the total size of the study site, which undoubtedly provided more precise results. On Dunkle Knob only one study area, or landscape level, disturbance was noted from the data, indicating a return interval of 244 years. An assessment of disturbances in different stands across the study site would provide a more comparable measure of disturbance frequencies.

Similar to the findings of Rentch and Hicks (2003) and Nowacki and Abrams (1997), the frequency of major disturbances was, on average, higher during the exploitation era (prior to 1900) relative to the modern era. On average, disturbances occurred every 42 years in

the exploitation era, compared to 71 years during the modern era. Based on a paired t-test, these values were not significantly different ($P = 0.162$).

Releases in Harvested Areas

None of the residuals in the timber sale areas experienced major releases from the harvesting. All of the eastern white pine residuals in the 1983 harvest area however, exhibited either a small or moderate release following the harvesting. One of the goals of this clearcut with reserve trees harvest was to convert the area to eastern white pine. Harvesting records stated that “there was a good eastern white pine seed source to restock the stand.” Although species of reserve trees are not listed in the records, clearly eastern white pine would have been the species of choice. These trees would have already been a member of the overstory, and were likely co-dominant or dominant. Thus, because they were not overtopped, a major release would not be expected from these individuals. Furthermore, these trees were, on average, thirty-three years of age at the time of harvesting. Eastern white pines less than 30 years old, with at least one-third of their height in live crown, respond well to release from competition. The ability to respond, however, declines proportionately with increasing age and decreasing crown length (Burns and Honkala 1990). Interestingly, regeneration and stand development in the 1983 harvest area did not go as the USDA Forest Service planned. That is, the area was not successfully converted to eastern white pine. The area is now dominated by “scrub” stands of Virginia pine that are at an average dominant height between 20 and 25 feet with sparse understory vegetation. Because the area was dominated by mature pitch pine prior to the harvest, it is evident that a clearcut did not successfully regenerate pitch pine

either. Had a fire occurred following the clearcut, pitch pine regeneration may have been more successful.

Tree-of-heaven Establishment

Given that a sampling methodology designed to fully understand tree-of-heaven establishment was not employed in this part of the study, some areas where tree-of-heaven has reached the main canopy may have been missed. Nevertheless, the majority of the areas encountered were either in harvested areas or along roads (Figure 3.12). It should also be noted that there were other timber sales in close proximity to the study area which may have also influenced tree-of-heaven establishment. Tree-of-heaven is often associated with areas of disturbance relative to undisturbed areas (Call and Nilsen 2003, Huebner 2003). Furthermore, forest disturbances, natural or anthropogenic, generally decrease canopy cover and increase understory light, which, in turn, tends to promote invasion by exotic plant species (Bossard 1991, D'Antonio 1993, Hobbs and Huenneke 1992, Hutchinson and others 2004). Although it is well known that tree-of-heaven can tolerate a wide range of environmental conditions (Howard 2004), it appears that the extremely dry nature of the study site has limited its invasion into the overstory to disturbed portions of the mountain and, in some cases, in relatively undisturbed, moister riparian areas.

The oldest individual tree-of-heaven established before the initial timber extraction from the Dunkle Knob area in the 1950s. If this tree was a female, it could have provided seed for future cohorts following disturbance. Had the personnel that conducted the harvest

noticed this individual and eliminated it, they may have prevented several of the other areas from becoming invaded. Seeds may have also been dispersed from other seed producing trees that were not encountered on or near the study site, as tree-of-heaven seed can be dispersed as far as 200 meters from the female parent (Kota 2005).

Conclusions and Management Implications

The oak-pine forest type in the study area was presumably converted from an oak-chestnut forest type as a result of a combination of anthropogenic and natural disturbances in the early 20th century. The results indicate that the disturbance regime that followed this conversion is not sufficient for the serotinous-coned yellow pines to maintain their role as an important part of the overstory at Dunkle Knob. That is, neither table mountain pine or pitch pine are not recruiting well in the understory (Table 3.2).

The loss of fire as a disturbance agent in 1923 likely played a key role in this developmental pathway. Post-1930 species establishment indicate that oaks have been slowly gaining importance in the study area. However, considering the economic and wildlife value of oak species, as well as the current widespread oak regeneration problem throughout the Appalachians, this trend does not rectify some management objectives.

Tree-of-heaven is one of the most problematic exotic invasive species in the eastern US. In addition to its highly invasive nature, it also possesses allelopathic properties (Heisey 1996). Native understory plants, as well as the benefits they provide (e.g. biodiversity, food for wildlife), are in danger of being severely impacted by abundant tree-of-heaven vegetative reproduction (root suckers) in localized areas around the main stems that have

ascended into the main canopy. Furthermore, Huebner (2006) suggests that, if large seed producing females exist, tree-of-heaven may increase in abundance with repeated burning. Managers should attempt to eradicate these tree-of-heaven “hot spots” prior to applying the next burn. Effective chemical control methods are available for controlling this unwanted exotic invader (Eck 2005). Pre-fire control of tree-of-heaven would aim to make the forest more resilient to further invasion and reduce the likelihood of post-fire establishment.

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CHAPTER 4

Investigating the Relationship between Bole Scorch Height and Fire Intensity Variables in the Ridge and Valley Physiographic Province, West Virginia

Abstract

Prescribed fires are carried out on the George Washington National Forest (GWNF) to promote long-term resource and social values, including tree regeneration, improving wildlife habitat and aesthetics, and maintenance of low woody fuel loading. Prescribed fire programs have increased on the GWNF over the past twenty years. Although prescribed fire is widely used on the GWNF, methods and techniques for monitoring fire behavior are still in the early stages of development. As part of a larger study to assess the effects of prescribed fire on exotic invasive species in the Ridge and Valley of eastern West Virginia, I used simple linear correlation and regression analysis to investigate the relationships between plot-level three-year post-fire bole scorch height parameters, fire temperature as estimated by thermocouple probes, litter consumption, and sapling mortality to better understand and predict fire effects in mixed-oak and oak-pine forests. I collected information on these parameters as they were observed on 36 research plots in an area where a prescribed fire had been applied in the Ridge and Valley. The prescribed burn totaled 313.6 ha and was conducted in March 2004. Neither overstory or sapling bole scorch height variables were significantly related to thermocouple temperature measurements recorded at five locations on a southwest facing slope ($\alpha = 0.05$). The sum of the sapling scorch heights was significantly related to litter consumption, but only accounted for 11 % of the variation. However, all bole scorch height variables were significantly related to sapling mortality one growing season after the fire. This further illustrates the usefulness of scorch height as an estimator of relative fire intensity, and provides a useful prediction model for xeric mixed-oak and oak-pine fire managers.

Introduction

Prescribed fire is primarily used on the George Washington National Forest (GWNF) to manage areas unsuitable for timber production. These sites are often characterized by infertile soils, steep/rocky slopes, non-commercial/low-value species mixtures, or pine-oak stands in general (United States Department of Agriculture (USDA) Forest Service 1993). The fires are applied to promote long-term resource and social values like maintenance of low woody fuel loading, aesthetics, tree regeneration and increased abundance of understory plants that are valuable for wildlife forage, such as blueberries (*Vaccinium spp.*), huckleberry (*Gaylussacia baccata* (Wangenh.) K. Koch), scrub oak (*Quercus ilicifolia* Wangenh.), and various grasses. Since the mid- to late-1980s, the extent of these fires has increased quite dramatically (Figure 4.1). In the decade of 1980, GWNF managers treated, on average, 37.2 hectares/year, compared to 877.2 hectares/year in the 1990s and 2,097.0 hectares per year this decade. That equates to an astounding 2,240 % increase from the 1980s to the 1990s, and a 140 % increase from the 1990s to the 2000s, respectively.

Several studies have examined fire behavior in a number of various regions. However, fire behavior-related studies in the Appalachian region are limited (Franklin and others 1997, Clinton and others 1998, Vose and others 1999, Iverson and others 2004).

Although prescribed fire is widely used on the GWNF, methods and techniques for monitoring fire behavior are still in the early stages of development.

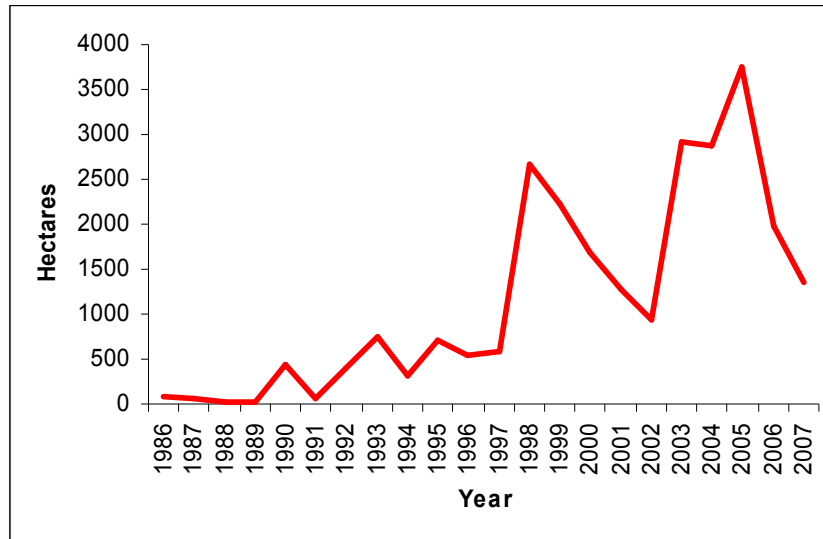


Figure 4.1. Area treated with prescribed fire by year on the GWNF.

Bole scorch height, otherwise known as stem-bark char or height of bole blackening, variables have been proven to be significant predictors of post-fire mortality in various forest types (Loomis 1973, Wyant and others 1986, Regelbrugge and Conrad 1993, Regelbrugge and Smith 1994). It has also been proven to be highly correlated with areal percent canopy scorch and top-kill of individual trees (Regelbrugge and Smith 1994). Although bole scorch height underestimates flame length in prescribed burns (Cain 1984), these findings indicate that it can be used to estimate relative fire intensity at the stand-level.

As part of a larger study to assess the effects of prescribed fire on exotic invasive species in eastern West Virginia, I investigated the relationship between plot-level three-year post-fire bole scorch height parameters (independent variables) and other fire intensity variables to better understand and predict fire effects in xeric mixed-oak and oak-pine forests. A better understanding of the role of fire in determining the structure and

composition of xeric mixed-oak and oak-pine forests of the Ridge and Valley Physiographic Province will enhance management efforts to sustain these forests.

In this chapter, I focus on the relationships between measured variables and the usefulness of these parameters for predicting changes in stand structure. Three relationships are evaluated, including; (1) litter consumption as a function of scorch height, (2) thermocouple estimated fire temperature as a function of scorch height, and (3) sapling mortality as a function of scorch height. The intent of this analysis is to evaluate whether these quick, simple, and inexpensive field measurements can help characterize prescribed fire behavior and its short-term effects on forest structure and composition. I expect increasing litter consumption, thermocouple temperature, and sapling mortality with increasing bole scorch height.

Methods

Study Site

This study was conducted at Dunkle Knob located on the North River Ranger District (formerly the Dry River Ranger District) of the GWNF, Pendleton County, West Virginia (38° 37' N, 79° 14' W). The study area encompassed 313.6 ha and is composed of xeric mixed-oak and oak-pine communities. The area is within the Ridge and Valley Physiographic Province and is in the rain-shadow of the Allegheny Mountains, which causes a climate that is much drier than the rest of the state (Core 1966). The area is characterized by an average annual precipitation of 82 cm, an average annual temperature of 10.9° C, and a growing season of approximately 144 days (Estepp 1992). Forests of

the region can be characterized as being part of the former oak-chestnut cover type (Braun 1950).

The topography is highly dissected with slopes ranging from 6 to 70 %. Elevation ranges from 573 to 848 m. The predominant soil type is of the Berks-Weikert association. They are loamy-skeletal, mixed, Dystrochrepts formed from acidic shale, siltstone, or sandstone bedrock. The soils are droughty, infertile, and often contain numerous rock outcroppings (Estepp 1992).

The Dunkle Knob area was purchased by the USDA Forest Service in 1923. There have been no documented fires in the Dunkle Knob area from its purchase date up to the time of the prescribed fire (Marsh 2005). Dunkle Knob was treated with a prescribed fire on March 29, 2004. Air temperatures ranged from 11°-21° C and relative humidity ranged from 29 to 76 %. Winds were primarily from the northwest and southeast at a speed of 2-10 Km/H. The interior of the mountain was ignited in a northeast-southwest pattern from top to bottom by a helicopter dropping delayed aerial ignition devices. The prescribed fire perimeter was established using existing roads and limited new fire line construction as fire breaks. This perimeter area was ignited by ground personnel using drip torches (Marsh 2005).

Pre-fire (2003) total basal area for the overstory (trees \geq 12.7 cm diameter at breast height (DBH)) averaged 20.8 m²/ha, and increased to 23.0 m²/ha by 2007. Chestnut oak

(*Quercus prinus* L.), table mountain pine (*Pinus pungens* Lamb.), and Virginia pine (*Pinus virginiana* P. Mill.) made up 66.5 % of the overstory basal area in 2007.

Field Methods

Thirty-six circular 0.05 ha overstory plots were randomly established at the Dunkle Knob study site during the summer of 2003 (pre-fire). Eighteen plots were randomly established on northeast- (315-235°) and southwest- (236-314°) facing slopes. Within each aspect class, nine plots each were established above and below 748 m elevation (see Marsh 2005 for complete sampling experimental design). To aid in future relocation, sample points were located with a Global Positioning System (GPS) and a piece of steel rebar was driven into the ground at plot center. Also, plot boundaries were flagged.

Each plot was inventoried in the summers of 2003 (pre-fire), 2004 (three to four months post-fire) and 2007, three full growing seasons after the fire. All overstory trees (DBH > 12.7 cm), both living and dead, were tallied. Each tree was identified to species (although a small number of dead trees could not be positively identified), and crown class was recorded. Stem diameter was measured to the nearest 0.1 cm using a diameter tape. During the 2007 inventory, bole scorch height, defined as height of the highest point of bole blackening on the uphill face of the tree (Regelbrugge and Smith 1994), was measured to the nearest 15 cm using a height pole. The sapling layer (all trees $2.54 \text{ cm} \leq \text{DBH} \leq 12.6 \text{ cm}$) was sampled using a circular 0.01 ha plot located at the center of each overstory plot (n = 36). All saplings were identified to species and DBH. Bole scorch height on saplings was also recorded as described above.

Also within each overstory plot, 1 m² circular plots were established at 12.06 m from the overstory plot center in each of the four cardinal directions (n = 144). These were primarily for understory vegetation measurements; however, litter layer (Oi) depth was measured to the nearest 0.1 cm at the west end of each plot using a ruler in 2003 and 2004.

On five south-facing overstory plots, 9.14 m² square sampling plots were established. On these plots, the prescribed fire itself was monitored using a network of thermocouple probes. These sample points were selected due to logistical considerations and the expectation that the prescribed fire would be the most intense on southerly aspects (Marsh 2005). At the corners and at the center of these square plots, 25 cm thermocouple probes were installed and HOBO[®] data loggers (Onset Computer Corporation, Bourne, MA) were buried just below the surface in a manner similar to that of Iverson and others (2004) and set to record thermocouple temperature at four second intervals the day of the prescribed fire. All data loggers were collected immediately following the treatment. It is well known that Type K thermocouple probes cannot record flame or air temperature around the probe during a prescribed fire because of the lag time of the probes and the relatively short duration of the most intense heating (Iverson and others 2004). However, it was my desire to use a small sample of thermocouple probe temperature data to characterize fire behavior on what I believed would be the most intense area of the fire, and to assess its usefulness for predicting fire effects.

Analytical Methods

Pre- and post-fire litter depth (LD) measurements were averaged for each overstory plot (n = 4), and litter consumption was quantified by the average percentage change in these values (n= 36). Litter consumption was calculated using the equation:

$$\text{Litter Consumption} = \left(\frac{(LD_{post} - LD_{pre})}{LD_{pre}} \right) * 100$$

Scatter plots were used to graphically examine all relationships for linearity, as well as any potential non-linear relationships. Following graphical examination, simple linear correlation and regression analysis were performed to examine the relationship between litter consumption (dependent) and both overstory and sapling bole scorch height measures (mean and sum) (independent) for all plots. The sum of hardwood tree heights is commonly used as a measure of hardwood competition in young southern pine plantations. Similarly, I tested the sum of the scorch heights as an independent variable that combines the number of stems scorched and degree of charring to determine its usefulness as a surrogate for fire behavior.

The analysis was also performed separately for mostly scorched plots. In this study, mostly scorched plots are defined as plots in which $\geq 50\%$ of the overstory trees were scorched. This separate analysis was performed to reduce within-plot scorch height variability. This method reduced this variability considerably in a pilot study of plot level bole scorch height against temperature sensitive paint used to estimate fire

temperature from another prescribed fire in a mixed-oak stand in the Ridge and Valley Province².

Linear correlation and regression were also used to examine the relationships between recorded thermocouple temperature (dependent) and bole scorch height parameters.

To better understand how post-fire stand structure attributes were influenced by surrogates of fire intensity, a third correlation/regression analysis was performed to observe how the change in the number of understory living saplings (percent mortality) varied with plot-level bole scorch height variables.

All statistical analyses were performed using SAS 9.1 software (SAS Institute, 2003), with significance determined at an $\alpha = 0.05$. Effects of independent variables were assessed using the General Linear Model (GLM) procedure. Mean comparisons were evaluated with paired t-tests. The DFFITS procedure was used to examine plot data and identify influential observations (outliers). The SAS DFFITS procedure identified one plot as a statistical outlier. Therefore this plot was removed from the analyses. Interestingly, this was the only plot in the study area in which the overstory was dominated by pitch pine (> 50 % of the overstory basal area). Both average and sum of the scorch heights on this plot were much greater than on others.

² Data on file with the USDA Forest Service, Northern Research Station, Parsons, WV.

Results and Discussion

Litter Consumption

Immediately following the prescribed fire, litter depth differed significantly from pre-fire conditions. The fire, on average, resulted in a 49.5 % decrease in litter depth (2.14 cm in 2003 compared to 1.08 cm in 2004). Average three year post-fire bole scorch height variables were significantly different between the overstory and understory strata based on paired t-tests. Loucks and others (2004) found that mean scorch heights increased with larger diameter classes in an Appalachian hardwood forest in Kentucky. Similarly, our study showed means that were significantly higher for the overstory stratum (Table 4.1). These scorch height differences suggest that researchers or managers interested in measuring scorch height should also stratify their sample by these forest layers.

Table 4.1. Average scorch height parameters (cm) by stratum.

<u>Stratum</u>	<u>Mean</u>	<u>Sum</u>
Overstory	56.01	1450.41
Sapling	38.37	286.08

Both means were significantly different between strata at $\alpha = 0.05$.

Neither of the overstory stratum independent variables (mean and sum of the scorch heights) could effectively predict litter consumption (Table 4.2). The sum of the sapling scorch heights, however, proved to be a significant litter consumption predictor.

Unfortunately, the model can only account for 11 % of the variation in litter consumption. More intricate fuel measurements (e.g. fuel transects and moisture content) may show stronger relationships to the measured variables than simple percentage change

in litter depth. Mass changes in the litter layer may alter ecosystem functions, but can be time consuming to measure. Developing efficient methods for estimating such changes would be helpful to forest managers. The fire effects on the litter layer were similar to the work of Brown and others (1991) and Elliot and others (2002).

Table 4.2. Relationships between bole scorch height variables (cm) and litter consumption (%) for all plots.

Stratum	Variable	r	R ²	P
Overstory	Mean	0.20	0.04	0.253
	Sum	0.14	0.02	0.407
Sapling	Mean	0.29	0.09	0.128
	Sum	0.33	0.11	0.049*

*Indicates significance at $\alpha = 0.05$

Contrary to my expectations, the results were similar when the analysis was restricted to plots in which greater than 50 % of the trees were scorched (Table 4.3). Even though I found a positive linear correlation, there was little evidence that any measure of scorch height was a useful predictor of litter consumption, due to the large amount of unexplained variation in the model. The most promising explanatory variable in this restricted analysis was the sum of the sapling scorch heights (P = 0.060). I suspect that the weak correlation between these variables is due to the litter layer itself. That is, given the thin pre-fire litter layer (2.14 cm on average), plots experiencing even the modest intensity would likely lose nearly their entire litter layer. Future efforts to develop easy to measure predictor variables to estimate litter consumption should narrow the ecological amplitude of the explanatory variables to potentially improve model performance.

Table 4.3. Relationships between bole scorch height variables (cm) and litter consumption (%) for plots with ≥ 50 % of the trees scorched (n = 18).

Stratum	Variable	r	R ²	P
Overstory	Mean	0.22	0.05	0.393
	Sum	0.10	0.01	0.634
Sapling	Mean	0.35	0.12	0.168
	Sum	0.45	0.20	0.060

Thermocouple Temperature

Although I found a general positive relationship between scorch height and thermocouple temperature, there was little evidence of a useful predictive relationship (Table 4.4).

When compared to litter consumption predictions, R² values were, in general, higher for overstory scorch height based thermocouple temperature predictions. However, this larger value is likely due to the fact that only a small number of points could be used to examine the relationships between scorch height and thermocouple temperature.

Table 4.4.--Relationships between bole scorch height variables (cm) and fire temperature (°C) for five plots fitted with thermocouples.

Stratum	Variable	r	R ²	P
Overstory	Mean	0.62	0.39	0.264
	Sum	0.70	0.50	0.184
Sapling	Mean	0.04	0.002	0.940
	Sum	0.14	0.02	0.823

It should be emphasized that the thermocouple temperatures used in the analysis were the average of five thermocouple probes on each plot. Due to the limitation of such instrumentation, averaging probe temperature may mask significant spatial variability among fire intensity throughout the sample area, ultimately weakening the relationship.

Despite the results, I speculate that useful predictive relationships between thermocouple probe temperatures and fire effects could be developed if more spatially limited procedures were utilized. Employing more intensive sampling would likely limit bias and produce more indicative and statistically significant results as well. However, the time consuming nature of using thermocouple probes combined with their potentially limited spatial extrapolations may ultimately restrict their usefulness by managers.

Sapling Mortality

Total pre-fire number of living saplings averaged 1000 trees per hectare (TPH), compared to 725 TPH in 2004. These values were significantly different ($P < 0.001$). Chestnut oak, pignut hickory (*Carya glabra* (P. Mill.) Sweet), Virginia pine, and striped maple (*Acer pensylvanicum* L.) made up 53.7 % of the number of saplings in 2007.

There was ample evidence of a relationship between scorch height and sapling mortality induced by the prescribed fire. All overstory and sapling layer scorch height variables were significant predictors of sapling mortality (Table 4.5). Future analyses should assess species specific predictive relationships. Average overstory bole scorch height provided the strongest relationship and accounted for 63 % of the variation in sapling mortality (Figure 4.2). By contrast, Regelbrugge and Smith (1994) found that the average scorch height accounted for 96 % of the variation in the percentage of the number of fire-killed trees. However, their study was conducted in the aftermath of a 1900 ha wildfire which exhibited greater amplitude in scorch height and fire severity than the Dunkle Knob prescribed fire I monitored. They also chose stands with both low and

high fire severity to provide a wide range of fire intensity and subsequent effects. My results further illustrate the usefulness of this relationship even when conditions are much more homogenous.

Table 4.5. Sapling mortality (%) models and associated statistics.

Model	r	R ²	SEE	P
%Mort.= 4.03 + 0.39(mean overstory scorch height (cm))	0.79	0.63	4.37	<0.001*
%Mort.= 9.75 + 0.01(sum of overstory scorch heights (cm))	0.69	0.47	4.89	<0.001*
%Mort.=10.96 + 0.39(mean sapling scorch height (cm))	0.42	0.18	7.28	0.010*
%Mort.= 16.41 + 0.03(sum of sapling scorch heights (cm))	0.44	0.19	5.87	0.008*

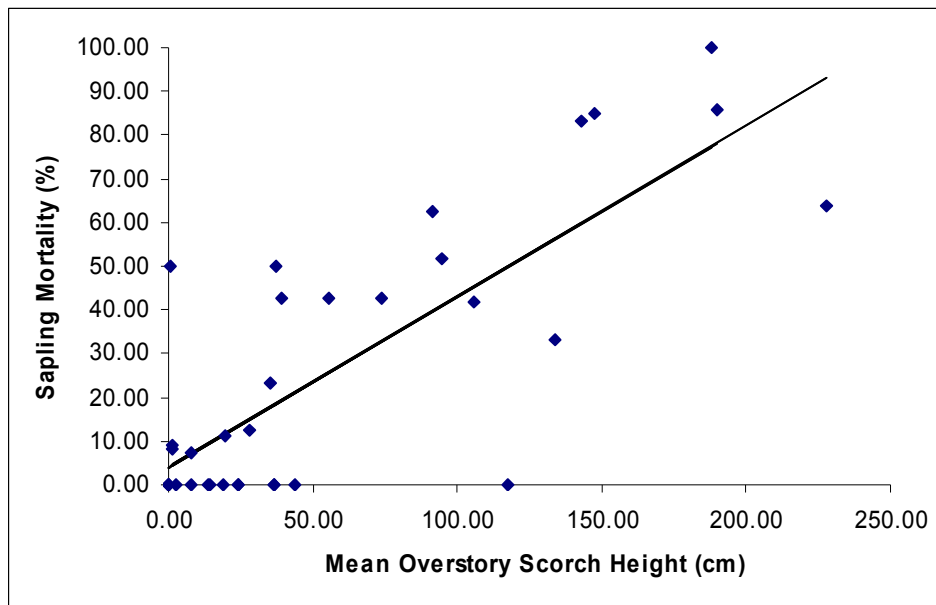


Figure 4.2. The relationship between mean overstory scorch height (cm) and sapling mortality (%).

When the sapling mortality analysis was restricted to mostly scorched plots, relationships/models were not nearly as sound. Therefore, these results were not reported.

Conclusions

The results indicate that, at the plot level, the measured three-year post-fire overstory and sapling bole scorch height variables (mean and sum) are not significantly correlated to litter consumption or thermocouple temperature. The results do, however, indicate that sapling mortality (one growing season post-fire) is a function of the tested bole scorch height parameters, with the strongest independent variable being average overstory scorch height. This further illustrates the usefulness of scorch height as an estimator of relative fire intensity and fire effects, and will allow for an examination of how relative fire intensity relates to changes in exotic invasive plant populations (Chapter 5).

The relationship between average scorch height and sapling mortality also has the potential to be very useful for xeric mixed-oak and oak-pine fire managers. The mean overstory scorch height model indicates that 50 % mortality can be expected at an average overstory scorch height of just over one meter. An average overstory scorch height just less than two meters would result in about 75 % sapling mortality. An average overstory scorch height of about two and one-half meters would likely result in 95 % mortality. This model can be used to make a quick and inexpensive assessment of fire-induced mortality in areas where pre- and post-fire saplings were not sampled. Managers could also use the information to adjust ignition patterns to alter the potential sapling mortality in similar xeric mixed-oak and oak-pine prescribed fires.

However, the model does have its limitations. It has not been tested against independent data, and should therefore be considered preliminary. However, I do feel that the model

will provide a sensible estimate of plot-level sapling mortality (one year post-fire). Use of the model should be limited to xeric mixed-oak and oak-pine forests of the Ridge and Valley physiographic province. Also, due to the statistical outlier and its removal, model predictions should not be applied when pitch pine makes up a high percentage (> 50 %) of the basal area in the area being burned.

Further analysis of existing information about both dependent and independent variables would help to better understand the relationships examined. For example, measures of scorch height models of mortality may be different for disparate species such as red maple (*Acer rubrum* L.) and chestnut oak. Also, because topography is related to scorch height (Loomis 1973), the effect slope percent, aspect, elevation, slope position, landform shape, etc. should be examined. Loucks and others (2004) found higher scorch heights in more xeric locations. Logistic or multiple regression models may provide improved predictions by incorporating these and other influential variables.

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CHAPTER 5

Prescribed Fire and Invasion by Exotics in the Understory of a Xeric Appalachian Mixed-oak and Oak-pine Dominated Forest

Abstract

The number and extent of prescribed fires in xeric mixed-oak and oak-pine ecosystems of the Appalachians has been steadily increasing. In general, these fires are set to promote restoration of historical forest structure and enhance tree regeneration of desirable species. These fires may, however, promote invasion by non-native invasive plants. Unfortunately, the relationship between fire and invasion by exotic plants in these forests is not well understood. This chapter focuses on the effects of a prescribed fire on understory exotic invasive plants in a xeric Appalachian mixed-oak and oak-pine dominated area of the Ridge and Valley. Several analytical methods were employed to determine if: (1) abundance and importance increased following the fire, (2) relative fire intensity played a role, and (3) environmental variables were related to populations. Indicator Species Analysis (ISA) and Nonmetric Multidimensional Scaling (NMS) ordination suggested that the prescribed fire had little effect on the abundance of understory exotic invasive plants. An Analysis of Variance (ANOVA) test ($n = 36$) indicated that tree-of-heaven (*Ailanthus altissima* (P. Mill.) Swingle) showed significantly higher abundance and importance immediately following the prescribed fire, but was back to pre-fire levels after three full growing seasons. NMS ordination also implied that moister areas, where understory species richness is higher and where relative fire intensity is lower, provides a favorable microsite for the exotic invasive species. Abundance was positively correlated to these areas. Multiple logistic regression models indicated that environmental factors play more of a role than relative fire intensity in determining the probability of presence of exotic invasives in the understory. More research is needed on this topic for it to be fully understood.

Introduction

Historically, fire has played a key role in maintaining forest composition in pine-oak and mixed-oak forests in West Virginia and throughout the Appalachians (Abrams 1992, Brose and others 2001, Schuler and McClain 2003, Sutherland and others 1995). Before fire suppression became a policy in the eastern United States (US) between the late 1920s and 1950s, these forests were shaped by a combination of large scale disturbances (e.g. drought, stand-replacing wildfire, hurricane, and insect or disease outbreak) and low-intensity fires. Sutherland and others (1995) estimated that major fires occurred about every 10 to 15 years in table mountain pine (*Pinus pungens* Lamb.) stands in southwestern Virginia. Both Brose and Waldrop (2006b) and Yarnell (1998) estimated that prior to the fire suppression era, stand level disturbances in the southern Appalachians occurred once every 6 to 13 years. Rentch and Hicks (2003) found that large canopy disturbances involving multiple trees occurred every 17 years in oak (*Quercus* spp.) dominated old-growth stands in the Central Hardwoods Region.

These types of disturbances formed stands of mixed-oak and pine, and more frequent low-intensity fires maintained this composition (Abrams 1992, Brose and others 2001). Several studies have estimated low intensity fire return intervals in these forests. These return intervals are comparable to those of stand-level disturbances. Sutherland and others (1995) estimated that low intensity fires occurred every 9 to 11 years. Harmon (1982) found that periodic fires occurred, on average, about every 12 years in the Great Smoky Mountains. Shumway and others (2001) calculated fire-return intervals of 8 years in an oak forest in western Maryland. Schuler and McClain (2003) reported fire-return

intervals of 7 to 15 years in an oak forest in the Ridge and Valley Physiographic Province of eastern West Virginia. Weibull median fire return intervals of four to eight years occurred in rimrock oak-pine communities at the New River Gorge National River in West Virginia (Maxwell 2006).

The diminution of fire as a disturbance agent, and the associated change in structure and species composition, has led to a decline of both oak and, especially, pine communities in some areas (Abrams 1992, Brose and Waldrop 2006a). Continued absence of fire in oak-pine communities could soon lead to a decline in pine species, and eventual oak dominance (Abrams 1992; Sutherland and others 1995). Table mountain and pitch pine (*Pinus rigida* P. Mill.) dominated communities, in particular, are recognized by the Southern Appalachian Man and the Biosphere (SAMAB), a program that encourages the utilization of ecosystem and adaptive management principles, as one of 31 rare communities (SAMAB 1996). Researchers once thought that, without stand initiating fires, successful regeneration of table mountain and pitch pine communities could not occur (Sutherland and others 1995, Williams and Johnson 1990). Recent research, however, suggests that repeated low intensity fires can achieve successful table mountain pine regeneration (Waldrop and others 2006). Successful management of mixed-oak and oak-pine forests, without restoration of historical fire regimes, is difficult (Abrams 1992, Barden & Woods 1976, Marsh 2005, Sutherland and others 1995, Welch and others 2000, Williams & Johnson 1990). This information has spurred land managers to increase the number and extent of prescribed fires in these forest ecosystems in efforts to restore them to their historical structure and to promote regeneration of native oak and

pine species. This increase in prescribed fire is carried out primarily on federal lands (Elliot and others 1999, Lafon and others 2005).

In addition to the management challenges resulting from altered disturbance and fire regimes, the futures of these ecologically important forests are likely to be negatively affected by exotic invasive plants (Mlot 2003, Williams 1998). Highly invasive exotic plant species such as tree-of-heaven (*Ailanthus altissima* (P. Mill) Swingle), garlic mustard (*Alliaria petiolata* (Bieb.) Cavara & Grande), and Japanese stilt grass (*Microstegium vimineum* (Trin.) A. Camus) may drastically inhibit successful establishment and development of native plants and lead to a decrease in regional biodiversity (D'Antonio and Vitousek 1992) and altered ecosystem processes (Vitousek and Walker 1989). Furthermore, if exotic invasive plants become well established, they may alter forest fuel properties and impact fire behavior and fire regime characteristics (e.g. frequency, intensity, extent, type, and seasonality) (Brooks and others 2004, D'Antonio and Vitousek 1992, Mack and D'Antonio 1998). If these fire regime changes promote the dominance of the invaders, then an invasive plant-fire regime cycle can be established, making ecological restoration much more difficult (Brooks and others 2004). Given that the cumulative numbers of exotic invasive plant species in the forests of West Virginia may be increasing over time (Huebner 2003), few ecologists and conservation biologists will argue that worst of the problems caused by these unwanted plant pests are behind us.

In addition to encouraging historical forest structure and enhancing tree regeneration, natural and prescribed fire has been shown to indirectly promote increases in both richness and cover of native herbaceous plants in forest understory communities (Bowles and others 2007, Hutchinson 2006, Nuzzo and others 1996). These increases allow for greater diversity and, in general, a better habitat for local fauna. However, fire and other disturbances (i.e. thinning/harvesting and herbicide treatments) that decrease canopy cover and increase understory light, also may promote invasion by exotic plant species (Bossard 1991, D'Antonio 1993, Hobbs and Huenneke 1992, Hutchinson and others 2004, Larson 2003, Mooney and Hobbs 2000, Pyle 1995, Stapanian and others 1998).

The relationship between fire and invasion by exotic plants is not well understood. A few studies have directly examined this relationship in different forested ecosystems. The results, however, are variable and in some cases, contradictory. Some have found that fire, natural or anthropogenic, promoted the establishment and abundance of exotic invasive plants. Others have found little to no change or negative responses by exotic invasives. For instance, Bridgewater and Backshall (1991) found that natural fire enhanced exotic invasive forbs and grasses in *Eucalyptus* woodlands of Western Australia. Hester and Hobbs's (1992) results contradicted those of Bridgewater and Backshall (1991). They reported decreases in the density of exotic invasive forbs and grasses in *Eucalyptus* woodlands of the same region. Wein and others (1992) found that exotic annuals and biennials invaded recently burned areas of Wood Buffalo National Park in Canada. In ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests of the western US, fire has been shown to increase both the richness and abundance of exotic

invasive species (Crawford and others 2001, Griffis and others 2001). Both of these studies showed that exotic invasive species were far more abundant where fire severity, or intensity, was highest. Laughlin and others (2004), on the other hand found little to no change in exotic invasive plant populations following wildfire in ponderosa pine forests located in Grand Canyon National Park in Arizona. In other areas of the western US, like the California coast, interior valleys, and other western grassland and prairie ecosystems, fire is used as a management to control exotic invasives and promote native vegetation (Rice 2005).

Studies in the eastern US showed similar variability in the effects of fire on exotic invasive plants. Bowles and others (2007) showed that seventeen years of repeated annual low intensity burning promoted garlic mustard abundance in an Illinois oak forest. In their study, garlic mustard persisted and showed greater abundance in burned areas relative to unburned areas. Contrary to Bowles and other's (2007) findings, Nuzzo and others (1996) found that repeated annual burnings facilitated a reduced abundance of garlic mustard in a sand forest in Illinois. Nuzzo (1991) reported that garlic mustard showed no decline with low intensity fire and some decline with mid-intensity fire. Glasgow and Matlack (2006) found that prescribed burning and canopy-opening management practices facilitated invasion by Japanese stilt grass and multiflora rose (*Rosa multiflora* Thunb. ex Murr.) in a mixed deciduous forest of Ohio. Doren and Whiteaker (1990) reported little to no change in the populations of the exotic invasive Brazilian pepper tree (*Schinus terebinthifolius* Raddi) in Florida woodlands following fire. Based on the available literature, the variable population responses of exotic

invasive plants to fire appear to be due to both regional and local landscape variation (biology, different climates and microsites, etc.), and to be both site and species specific. That is, it is highly likely that responses to fire will be different in dissimilar areas or between disparate species on the same site.

Direct assessments of the relationship between fire and exotic invasive plant populations in forested ecosystems are limited to the abovementioned studies. Furthermore, no documented studies have explored this relationship in West Virginia. Perhaps more importantly, it has not been examined in forests of the Ridge and Valley Physiographic Province, an area that because of its unique species composition and climate is especially suited for the reintroduction of fire to achieve management goals (Thomas-Van Gundy and others 2007, Lafon and Grissino-Mayer In Press). The use of prescribed fire in oak and oak-pine forests in this region is steadily increasing, especially on the George Washington National Forest (GWNF), and has become a critically important management issue to the general public, as well as to foresters, biologists, ecologists, and other land and prescribed fire managers. GWNF lands experienced a 2,240 % increase in area prescribed burned from the 1980s to the 1990s, and a 140 % increase from the 1990s to the 2000s.

The hot, dry hillsides in this area provide ecologically unique ecosystems whose futures are at risk, not only due to altered fire regimes, but also due to invasion by exotic plants. Little is known about the relationship between prescribed fire and invasions by exotic invasive plants in these areas, hence, as the use of fire as a management tool increases,

exotic invasives could be a major threat to future forest composition. Increases in establishment and abundance of exotic invasives may have considerable negative impacts on native plant communities. If this is the case, many areas scheduled to be burned in the future may require exotic invasive monitoring or management prior to the application of prescribed fire.

This study focuses on the effects of a prescribed fire on exotic invasive plants in a xeric Appalachian mixed-oak and oak-pine dominated forest of the Ridge and Valley. The analysis focuses on the understory stratum where all early establishing species, including exotic invasives, will be found (Huebner 2007). Three research questions were investigated: (1) how does prescribed fire affect the abundance and importance of exotic invasive plants?, (2) does relative fire intensity play a role in abundance changes?, and (3) are there exotic invasive plant-environmental relationships? The intent of this study was to help foresters and prescribed fire managers to better understand the interaction of prescribed fire and exotic invasive plant species over time and across the landscape in xeric mixed-oak and oak-pine forest communities.

Methods

Study Site

This study was conducted at Dunkle Knob located on the North River Ranger District (formerly the Dry River Ranger District) of the GWNF, Pendleton County, West Virginia (38° 37' N, 79° 14' W). The study area encompassed 313.6 ha and is composed of xeric mixed-oak and oak-pine communities. The area is within the Ridge and Valley

Physiographic Province and is uniquely situated in the rain-shadow of the Allegheny Mountains, which causes a climate that is much drier than the rest of the state (Core 1966). The area is characterized by an average annual precipitation of 82 cm, an average annual temperature of 10.9° C, and a growing season of approximately 144 days (Estep 1992). Forests of the region can be characterized as being part of the former oak-chestnut cover type (Braun 1950).

The topography is highly dissected with slopes ranging from 6 to 70 %. Elevation ranges from 573 to 848 m. The predominant soil type is of the Berks-Weikert association. They are loamy-skeletal, mixed, Dystrochrepts formed from acidic shale, siltstone, or sandstone bedrock. The soils are droughty, infertile, and often contain numerous rock outcroppings (Estep 1992).

The Dunkle Knob area was purchased by the USDA Forest Service in 1923. After its purchase, no fires were documented in the Dunkle Knob area until a prescribed fire was ignited by GWNF personnel on March 29 of 2004 (Marsh 2005). At the time of the fire, air temperatures ranged from 11°-21° C and relative humidity ranged from 29 to 76 %. Winds were primarily from the northwest and southeast at a speed of 2-10 Km/H. The interior of the mountain was ignited in a northeast-southwest pattern from top to bottom by a helicopter dropping delayed aerial ignition devices. The prescribed fire perimeter was established using existing roads and limited new fire line construction as fire breaks. This perimeter area was ignited by ground personnel using drip torches (Marsh 2005). Based on data from five thermocouple probes, the fire burned at an average temperature

of 148.3° C with an average rate of spread of 2.3 m/min. Timber harvesting occurred on, and around the base of, the mountain in the 1950s and 1960s, as well as in the early 1970s and 1980s (Figure 5.1). The harvests are described in detail in Chapter 3. Eastern white pine (*Pinus strobus* L.) was planted in some cases after harvesting.

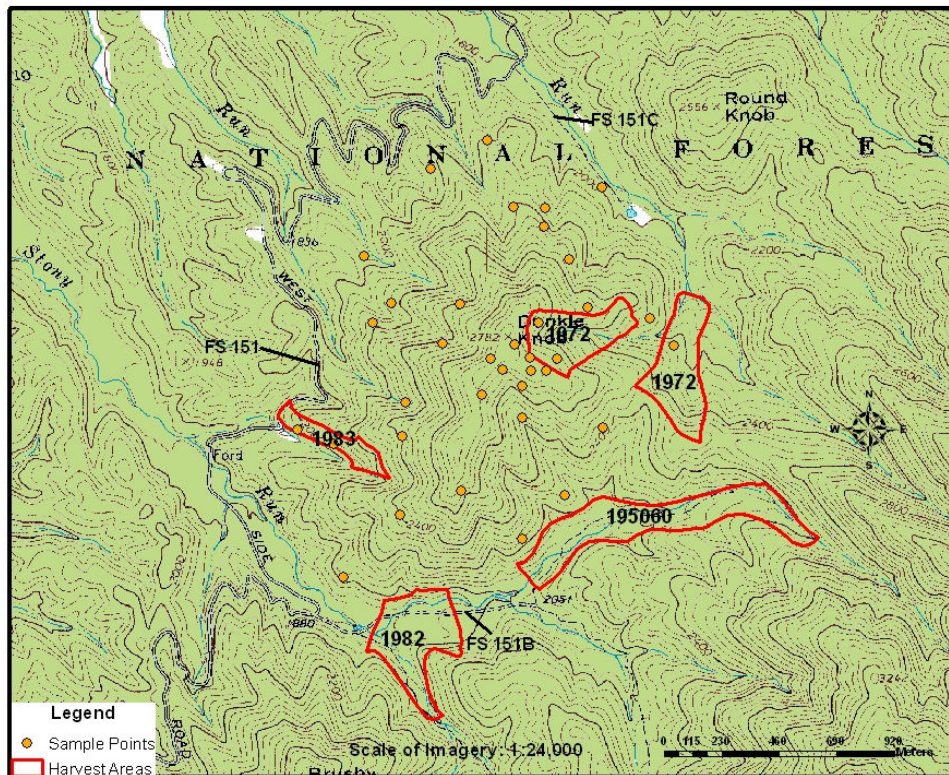


Figure 5.1. Harvest areas on, and around the base of, Dunkle Knob.

Pre-fire (2003) total basal area for the overstory (trees ≥ 12.7 cm diameter at breast height (DBH)) averaged 20.8 m²/ha, and increased to 23.0 m²/ha by 2007 (three full growing seasons after the fire). In 2007 the most abundant overstory species included chestnut oak (*Quercus prinus* L.), Virginia pine (*Pinus virginiana* P. Mill.), and table mountain pine (see Chapter 3).

Field Methods

Thirty-six circular 0.05 ha overstory plots were randomly established at the Dunkle Knob study site during the summer of 2003 (pre-fire). Eighteen plots were randomly established on northeast- (315-235°) and southwest- (236-314°) facing slopes. Within each aspect class, nine plots each were established above and below 748 m elevation (see Marsh (2005) for complete sampling experimental design). To aid in future relocation, sample points were located with a Global Positioning System (GPS) and a piece of steel rebar was driven into the ground at plot center. Also, plot boundaries were flagged.

Each plot was inventoried in the summers of 2003 (pre-fire), 2004 (three to four months post-fire) and 2007. All overstory trees, both living and dead, were tallied. Each tree was identified to species (although a small number of dead trees could not be positively identified), and crown class was recorded. Stem diameter was measured to the nearest 0.1 cm using a diameter tape. During the 2007 inventory, bole scorch height, defined as height of the highest point of bole blackening on the uphill face of the tree (Regelbrugge and Smith 1994), was measured to the nearest 15 cm using a height pole. The sapling layer (all trees $2.54 \text{ cm} \leq \text{DBH} \leq 12.6 \text{ cm}$) was sampled using a circular 0.01 ha plot located at the center of each overstory plot ($n = 36$) (Figure 5.2). All saplings were identified to species and DBH. Bole scorch height on saplings was also recorded as described above.

Bole scorch height is also described as stem-bark char or height of bole blackening. This variable has been proven to be a significant predictor of post-fire mortality in various

forest types (Loomis 1973, Regelbrugge and Conrad 1993, Regelbrugge and Smith 1994, Wyant and others 1986). It has also been proven to be highly correlated with areal percent canopy scorch and top-kill of individual trees (Regelbrugge and Smith 1994). Although bole scorch height underestimates flame length in prescribed burns (Cain 1984), these findings indicate that it can be used to estimate relative fire intensity at the stand-level. Chapter 4 illustrates the usefulness of scorch height as an estimator of relative fire intensity and fire effects at the plot level on the Dunkle Knob site.

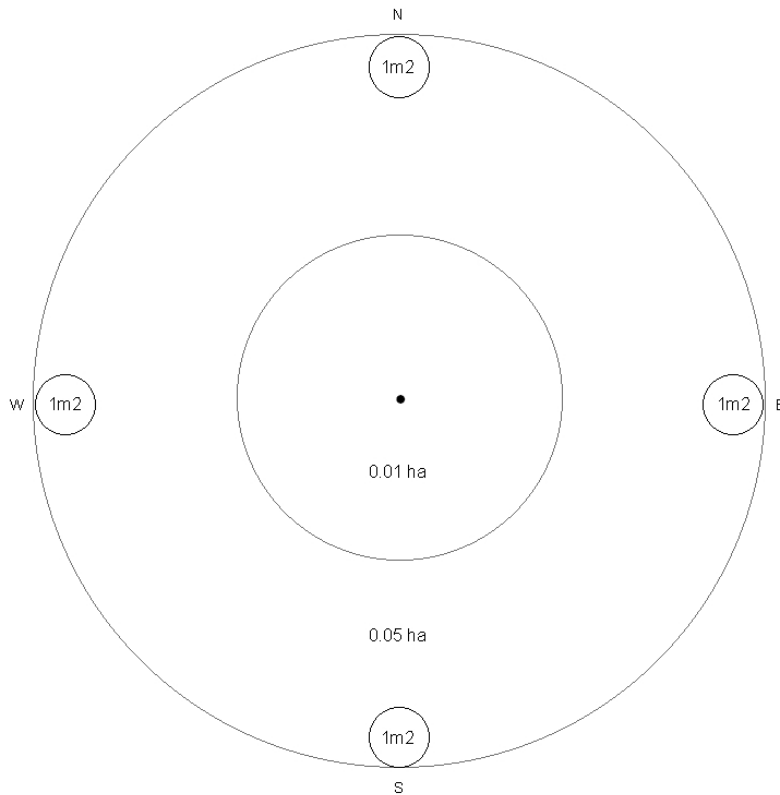


Figure 5.2. Measurement plots at a sample point.

Within each 0.05 ha plot, 1 m² circular plots were established at 12.06 m from the plot center in each of the four cardinal directions for understory vegetation measurements (Figure 5.2). In 2003, 2004, and 2007 (three full growing seasons after the fire), a 50 cm² Mylar® circle representing 0.5 % was used to estimate cover (to the 0.5% level) of each herb, shrub, vine, and tree seedling ≤ 1 m in height rooted in these plots. There was a <1%-cover category, which was given a value of 0.5% for data analysis purposes. Estimates of percent ground cover of moss/lichen, rock, bare ground, dead wood, living wood, and litter were also recorded.

A vegetation census using the timed-meander technique was performed on each of the 0.05 ha plots (n=36) in 2007. This method has been proven effective for documenting species richness, defining flora, and for detecting both exotic invasive (Huebner 2007) and threatened and endangered species (Goff and others 1982). Each plot was thoroughly walked, starting and ending at North from plot center, and time was noted every ten minutes as new species were tallied. All species having a height of ≤ 1 m were tallied. Unknown plants were collected and pressed for later identification by botanical expert Dr. Cynthia Huebner, Botanist, United States Department of Agriculture (USDA) Forest Service, Northeastern Research Station. The largest portion of the samples were identified by Dr. Huebner, however, in a few cases, herbaceous plant expert William Grafton, Wildlife Extension Specialist, West Virginia University (WVU) Division of Forestry and Natural Resources, participated in identification.

Spherical crown densiometer readings were taken to estimate percent canopy cover on Dunkle Knob in 2003, 2004, and 2007. Five densiometer measurements were taken at each sample point; one at the center of the overstory and sapling plots and one at each of the four 1m² sub-plots. The measurement at the center of the overstory plot was taken facing downhill, while the remaining were made facing plot center.

Analytical Methods

Exotic invasive species were defined using published state and regional flora (Core 1966, Gleason and Cronquist 1991, Harmon and others 2006) and available species lists (The Bugwood Network 2006, USDA Forest Service, Eastern Region 1998). Definitions were crosschecked by Dr. Cynthia Huebner to ensure correctness. Nomenclature follows Harmon and others (2006) and the Integrated Taxonomic Information System (ITIS) (2001). For some of the pressed samples, the distinguishing characteristics between certain species were not present (e.g., reproductive structures) or the plant was too underdeveloped for accurate identification. In these instances, taxonomic classification to only the family or genus level was possible. In some cases, positive identification of a particular specimen could not be made. These specimens were classified to the closest taxonomic level that could be achieved with confidence, and cf. was added to the nomenclature.

Pre-treatment vs. Post-treatment

Pre-treatment (2003) and post-treatment (2007) understory species composition was evaluated with Multi-response Permutation Procedures (MRPP) using PC-ORD 5

software (McCune and Mefford 1999). Although MRPP, and other multivariate analysis methods, avoid distributional assumptions, one should adjust the data matrix if it is extremely variable. McCune and others (2002) suggest that row and column variability, measured by the Coefficient of Variation (CV), between 50 and 100 % will have a moderate effect on the outcome of multivariate analyses.

Input data were pre-fire and post-fire species plot percent cover values (average of 1m² (n=144) for each 0.05 ha plot (n=36). 2004 data were limited, and therefore were not included in the analysis. Initial data screening indicated a high CV of column (species) totals (448 %). Therefore data were adjusted to reduce this variability. First, rare species (those occurring in less than five percent of the sample units) were deleted. This reduced the CV of column totals to 297 %. Data were then subjected to arcsine square root transformation, further reducing the CV of column totals to 161 %. Because the goal of < 100 % CV had not been met, a general relativization was used. The data were also examined for species (columns) that fell more than two standard deviations above or below the mean. Using this method, no potential outliers were identified. The final matrix (data set of plot x species) exhibited row and column totals with CVs of 56 and 0 %, respectively.

MRPP is a nonparametric procedure for testing the hypothesis of no difference between two or more groups of entities (McCune and others 2002). This method has been proven useful in various types of ecological studies (Hartman and McCarthy 2007, Rentch and others 2005, Zimmerman and others 1985). The procedure yields three statistics: a

weighted within-group average distance or δ (using Sorensen , or city-block, distance), the T statistic, and the A statistic. The T statistic describes the separation between groups and is calculated using the formula:

$$T = \frac{OBSERVED \delta - EXPECTED \delta}{S.DEV. OF EXPECTED \delta},$$

where δ = the weighted within-group distance (McCune and others 2002). The T statistic describes the separation between groups; stronger separations between groups are represented by more negative T values. PC-ORD 5 (McCune and Mefford 1999) provides a probability value for the likelihood of observing the difference, T, by chance. The A statistic is an estimate of within-group homogeneity, compared with random expectation and is calculated using the formula:

$$A = 1 - \frac{OBSERVED \delta}{EXPECTED \delta},$$

again where δ = the weighted within-group distance (McCune and others 2002). This statistic provides a description of the “effect size” that is independent of the sample size with values ranging from 0 (heterogeneity within groups equals expectation by chance) to 1 (sample units are identical within groups) (McCune and others 2002).

Indicator Species Analysis (ISA) (Dufrene and Legendre 1997) was used in conjunction with MRPP to determine if any understory species, particularly exotic invasives, showed

a greater than expected fidelity to pre-fire or post-fire conditions based on abundance and occurrence. Again, PC-ORD 5 software (McCune and Mefford 1999) was used for the analysis. PC-ORD 5 first calculates an indicator value for each species. The indicator values (IV_{kj}) can range from 0 (no indication) to 100 (perfect indication) and are calculated using the equation:

$$IV_{kj} = 100(RA_{kj} \times RF_{kj}),$$

where j is each species in each group k , and RA = relative abundance and RF = relative frequency (McCune and others 2002). The statistical significance of the highest IV_{kj} for each species is then tested using a Monte Carlo method that randomly assigns each sample plot to groups 1000 times. The null hypothesis (H_0) is that the maximum IV_{kj} for each species is no larger than would be expected by chance (i.e., the species has no indicator value) (McCune and others 2002). The significance level for hypothesis tests was set at $\alpha = 0.05$.

In addition, Analysis of Variance (ANOVA) was used to compare exotic invasive species percent cover and Importance Value (IV) means for each sample period (pre-fire, three to four months post-fire, and three full growing seasons post-fire). IVs were calculated from the 1 m² data (averaged for each 0.05 ha plot) using the formula:

$$IV = \frac{RELATIVE PERCENT COVER + RELATIVE FREQUENCY}{2}$$

The AVOVA tests were performed using SAS 9.1 software (SAS Institute, 2003) using a General Linear Model (GLM) procedure. The significance level for these tests was set at $\alpha = 0.05$. If the ANOVA tests showed significant differences in means, the means were evaluated using Fisher's protected Least Significant Difference (LSD) test (t-test).

The Nonmetric Multidimensional Scaling (NMS) ordination technique was also used to compare pre-fire and post-fire understory species composition. Input data for the NMS ordination were the same as the MRPP input data (percent cover or abundance) and is described above. NMS is a non-parametric iterative ordination technique that uses ranked distances to arrange sample plots along a number of axes determined by a minimal stress configuration, and is considered by McCune and others (2002) as the most generally effective ordination method for ecological community data. The method has been proven useful in various types of forestry studies (De Grandpré and others 2005, Hutchinson and others 2005, Schuler 2004, Small and McCarthy 2002).

The NMS ordination was performed using PC-ORD 5 software (McCune and Mefford 1999) with the Sorensen, or city-block, distance measure. The following settings were used to determine appropriate dimensionality and to avoid local minima: a six dimensional solution (six axes), 50 runs of real data and 50 runs of randomized data (maximum of 500 iterations for each run) for a Monte Carlo test of significance, an initial step length of .20, an instability criterion of 0.0005, and random starting coordinates.

Following a finding that the results were not due to chance ($P = 0.020$), a three dimensional solution provided the lowest stress and was selected as the final solution for the NMS ordination. After finding the optimal number of axes for the solution, the best ending point in the preliminary analysis was used as the starting point in the final run. Ordination axes were rotated to align the horizontal axis with the strongest explanatory (relative fire intensity and environmental) variables. The explanatory variables were then correlated with the ordination axes. Understory species were also correlated to the ordination axes. Relative fire intensity variables used in the correlation included average overstory scorch height and sapling mortality. Environmental variables were: percent slope, Topographic Relative Moisture Index (TRMI) (Parker 1982), understory species richness, percent change in canopy cover (2003 to 2007), percent canopy cover (2007), elevation (m), distance (m) from areas where tree-of-heaven has ascended into the main canopy (see Chapter 3), distance (m) from harvesting, distance from roads (m), and percent ground cover of moss, rock, bare ground, living wood, dead wood, and litter. Initial data screening indicated that percent ground cover of rock, bare ground, and both living and dead wood had a skewness of greater than one; therefore, as recommended by McCune and others (2002), a Log 10 transformation was applied to these variables.

Exotic Invasive Plant Species and Relative Fire Intensity/Environmental Variables

The NMS ordination technique was also used to examine post-fire data for relationships between understory exotic invasive plant species and both relative fire intensity and environmental variables. This NMS ordination was performed using the post-fire (2007) timed meander data (presence/absence) for each 0.05 ha plot ($n=36$). Initial data

screening indicated a high CV of column (species) totals (116 %). Therefore data were adjusted to reduce this variability. First, rare species (those occurring in less than five percent of the sample units) were deleted. This reduced the CV of column totals to 83 %. CV of rows was found to equal 22 %. Beals smoothing (Beals 1984) was then applied to the data. Beals smoothing is an extremely powerful transformation that replaces presence/absence data with quantitative values that represent the “favorability” of each sample for each species, regardless of whether the species was present in the sample (McCune and others 2002). The data were also examined for species (columns) that fell more than two standard deviations above or below the mean. Using this method, no potential outliers were identified. The final “smoothed” matrix (data set of plot x species) exhibited row and column totals with CVs of 4 and 82 %, respectively.

The NMS ordination methods, and explanatory variables, were the same as those used for the NMS ordination used to examine pre-fire versus post-fire data and are described in the previous section. Following a finding that the results were not due to chance ($P = 0.020$), a two dimensional solution provided the lowest stress and was selected as the final solution for the NMS ordination of “smoothed” timed meander data.

In addition to the NMS ordinations, relationships between understory exotic invasive plants and relative fire intensity and environmental variables were examined by developing multiple logistic regression models to predict the post-fire (2007) probability of presence of tree-of-heaven and garlic mustard in the understory. Each of these species was found on seven of the 0.05 ha plots. Logistic regression models differ from

regression models in that they are not linear and explanatory variables can be either quantitative or qualitative. These models have become the standard method of analysis for describing the relationship between a discrete, or dichotomous, response variable and one or more explanatory variables across several different disciplines (Hosmer and Lemeshow 2000). A third multiple logistic regression model was developed to predict the post-fire (2007) likelihood of the presence of all species listed by the USDA Forest Service as highly invasive in the eastern region of the US. These plants are known as “Category 1” invasives and are all non-native, highly invasive plants which invade natural habitats and may have the ability to replace native species (USDA Forest Service, Eastern Region 1998). For this study, Category 1 exotic invasives included: tree-of-heaven, garlic mustard, Japanese stilt grass, and autumn olive (*Elaeagnus umbellata* Thunb.). As a group these species were found on twelve of the 0.05 ha plots. A fourth multiple logistic regression model was developed to predict the probability of presence of tree-of-heaven immediately post-fire (2004). During this time tree-of-heaven was found on eleven of the 0.05 ha plots. All 2007 post-fire response variable data came from the timed meander inventory. In all models the explanatory, or predictor, variables were the same as those used in the NMS ordinations.

For exploratory data analysis purposes, each independent variable was first evaluated individually using the method of Maximum Likelihood Estimates (MLE) to estimate the coefficients of the logistic regression function. The MLEs were calculated using SAS 9.1 software (SAS Institute, 2003) using a Logistic procedure. Significance was determined at $\alpha = 0.05$ using the Wald statistic. Those independent variables with a statistically

significant MLE were then evaluated together using multiple logistic regression with backward selection (P to enter and P to stay = 0.10). In these multiple logistic regression models, coefficients were derived from MLEs and expressed as odds ratios. In this study, odds ratios represent how a unit change in the value of an independent variable is associated with the probability of presence of a particular exotic invasive plant species in the understory stratum. The Wald statistic was again used to test the significance of explanatory variables in the full models. A Hosmer-Lemeshow test (Hosmer and Lemeshow 1980) was used to determine the goodness-of-fit between observed and predicted values for each multiple logistic regression model.

Results

A total of 152 vascular plants were identified in the understory from the 1m² plots (n=144) prior to the prescribed fire in 2003 (Appendix Table 1). Of these, only four, or approximately 3 % were defined as exotic invasives. These included: tree-of-heaven, garlic mustard, Canada bluegrass (*Poa compressa* L.), and Bell's honeysuckle (*Lonicera X bella* Zabel). Forty-five percent of the identified plants were ferns and forbs, while 22, 16, and 16 % were graminoids, shrubs and vines, and trees, respectively. In 2007 (three full growing seasons post-fire), a total of 139 vascular plants were identified in the understory from the 1m² plots (Appendix Table 2). Only three, or approximately 2 %, of these were defined as exotic invasives: tree-of-heaven, garlic mustard, and Canada bluegrass. Again the majority of the identified plants were ferns and forbs (55%). Graminoids, shrubs and vines, and trees made up 16, 16, and 14 % of the identified plants, respectively. Based on paired t-tests, mean species richness was, on average,

significantly higher on the 36 0.05 ha plots prior to the fire (17.53 pre-fire compared to 16.06 post-fire) ($P = 0.022$). Evenness and diversity averaged 0.66 and 1.85 pre-fire, compared to 0.67 and 1.81 post-fire, respectively. Neither of these measures were significantly different ($P = 0.361$ and 0.254).

The post-fire timed meander inventory identified a total of 245 vascular plant species (Appendix Table 3). Of these, only six, or approximately 2 % were defined as exotic invasives. Exotic invasives included: tree-of-heaven, garlic mustard, common sheep sorrel (*Rumex acetosella* L.), Japanese stilt grass, Canada bluegrass, and autumn olive. Percentages of habit types were similar to those calculated from the 2007 1m² data. Ferns and forbs, graminoids, shrubs and vines, and trees made up 58, 12, 15, and 15 % of the identified plants, respectively. Based on a paired t-test, the timed meander inventory yielded a significantly higher average species richness of 48.56, compared to 16.06 calculated from the 1m² ($P < 0.001$). Evenness and diversity were not compared for the two sampling methods due to the fact that the timed meander only provided presence/absence data.

Pre-treatment vs. Post-treatment

MRPP and ISA

The MRPP provided evidence that the pre-fire (2003) and post-fire (2007) understory layers can be characterized by somewhat different species composition. The T statistic, which represents the separation between pre-fire and post fire composition, was found to equal -10.81 and was highly significant ($P < 0.001$). The A statistic, an estimate of

within group homogeneity, was found to equal 0.021, indicating low homogeneity between plots within groups. A Mantel randomization test (Mantel 1967), which tests the hypothesis of no difference between two matrices (in this case pre-fire plot x species and post-fire plot x species), also indicated that compositional differences exist between pre-fire and post-fire understory layers ($r = 0.46$, $P = 0.001$).

The ISA suggested that some species were more likely to occur in the understory post-fire than pre-fire and vice versa (Table 5.1). Although the exotic invasives garlic mustard and Canada bluegrass each exhibited a stronger IV_{kj} in the post-fire understory, their observed IV_{kj} did not exceed their expected IV_{kj} and were, therefore, not significant indicators. Tree-of-heaven and Bell's honeysuckle were not considered in the analysis because they occurred in less than five percent of the sample plots. Fireweed (*Erechtites hieraciifolia* (L.) Raf. ex DC.), Unknown (UK) *Poaceae*, and grape (*Vitis* spp.) were most strongly associated with the pre-fire understory ($P < 0.002$). Pignut hickory (*Carya glabra* (P. Mill) Sweet) and Carolina saxifrage (*Saxifraga caroliniana* Gray) showed the strongest associations with the post-fire understory ($P < 0.004$).

ANOVA

Both percent cover and IVs were quite low for all exotic invasive species during all measurement periods (Figures 5.3 and 5.4). None of the species yielded an average percent cover that was greater than 0.5 %. All IVs were lower than 1.0, except for garlic

mustard pre-fire, and varied by species (Figure 5.4). Bell's honeysuckle was not considered in the analysis because it only occurred on one plot pre-fire and was absent during both post-fire measurement periods.

Table 5.1. Pre-fire and post-fire species Indicator Values (IVkj). Only species whose observed IVkj exceeds expected IVkj at P-values < 0.05 are shown.

Species	Observed IVkj		Maximum Expected IVkj	SD	P
	Pre-fire	Post-fire			
<i>Acer rubrum</i>	0	44.3	34.8	4.54	0.042
<i>Carex cephalophora</i>	0	25.2	13.4	3.66	0.009
<i>Carex cf pensylvanica/lucorum</i>	22.3	0	13.4	3.7	0.032
<i>Carya glabra</i>	0	47.3	28	4.69	0.002
<i>Carya ovata</i>	16.7	0	7.4	2.82	0.024
<i>Dichanthelium commutatum</i>	0	17.5	10.1	3.21	0.031
<i>Erechtites hieraciifolia</i>	26.9	0	13.5	3.77	0.001
<i>Houstonia longifolia</i>	0	31.9	22.5	4.2	0.034
<i>Pinus spp.</i>	19.4	0	8.4	2.83	0.012
<i>Potentilla simplex</i>	0	37.1	25.9	4.68	0.028
<i>Quercus prinus</i>	0	39.6	30.2	4.71	0.047
<i>Saxifraga caroliniana</i>	0	25	10.1	3.22	0.003
UK Poaceae	36.1	0	13.4	3.6	0.001
<i>Vaccinium pallidum</i>	0	54.5	45	3.69	0.017
<i>Vitis spp.</i>	38.9	0	14.3	3.67	0.001

For all exotic invasive species, except tree-of-heaven, the null hypothesis (H_0) of $\mu_1 = \mu_2 = \mu_3$ was accepted ($P > 0.05$) for both mean percent cover and IVs. Therefore, only tree-of-heaven percent cover and IV means were evaluated using Fisher's Least Significant Difference (LSD) test (t-test). Immediately following the fire, tree-of-heaven had an average percent cover that was significantly higher than both pre-fire and three full growing seasons following the fire (0.030 in 2004 compared to 0.002 and 0.003 in 2003 and 2007, respectively) (Figure 5.3). The same pattern occurred for tree-of-heaven IVs

(Figure 5.4). Even at its highest point in 2004, tree-of-heaven still only had a mean IV of 0.83.

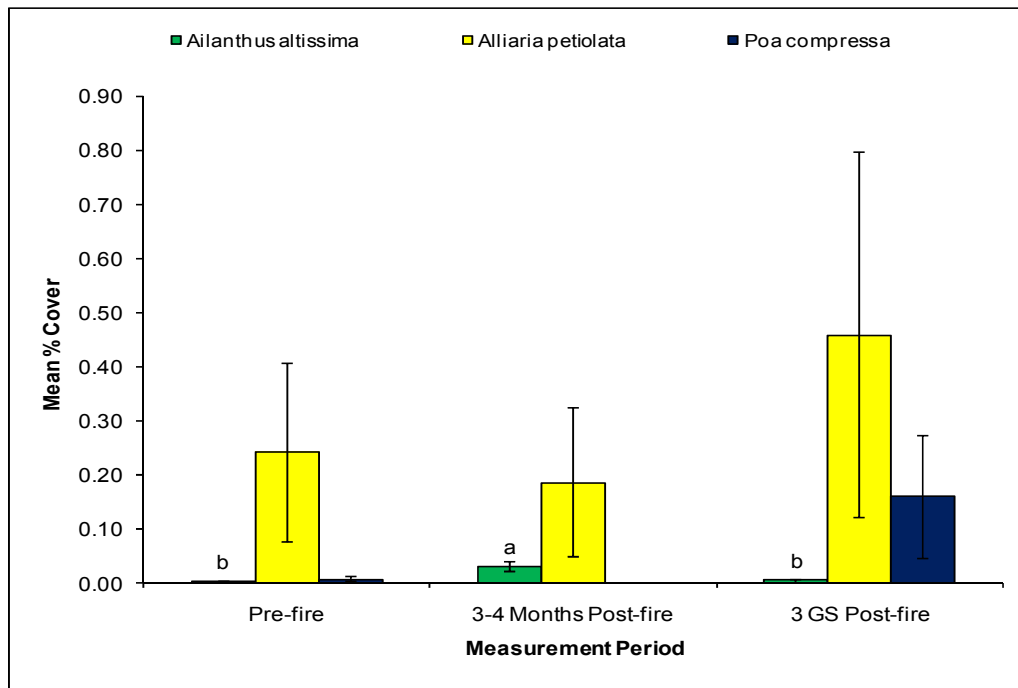


Figure 5.3. Average percent cover by species and measurement period. Means with the same letter are not significantly different at $\alpha = 0.05$. Error bars represent +/- one standard error. GS = growing seasons.

NMS Ordination

The NMS ordination (a three dimensional solution) of combined pre- and post-fire abundance data yielded a final stress and instability of 21.79 and 0.032, respectively. Stress and instability values this high generally do not indicate a reliable solution (McCune and others 2002), therefore the results must be used with caution.

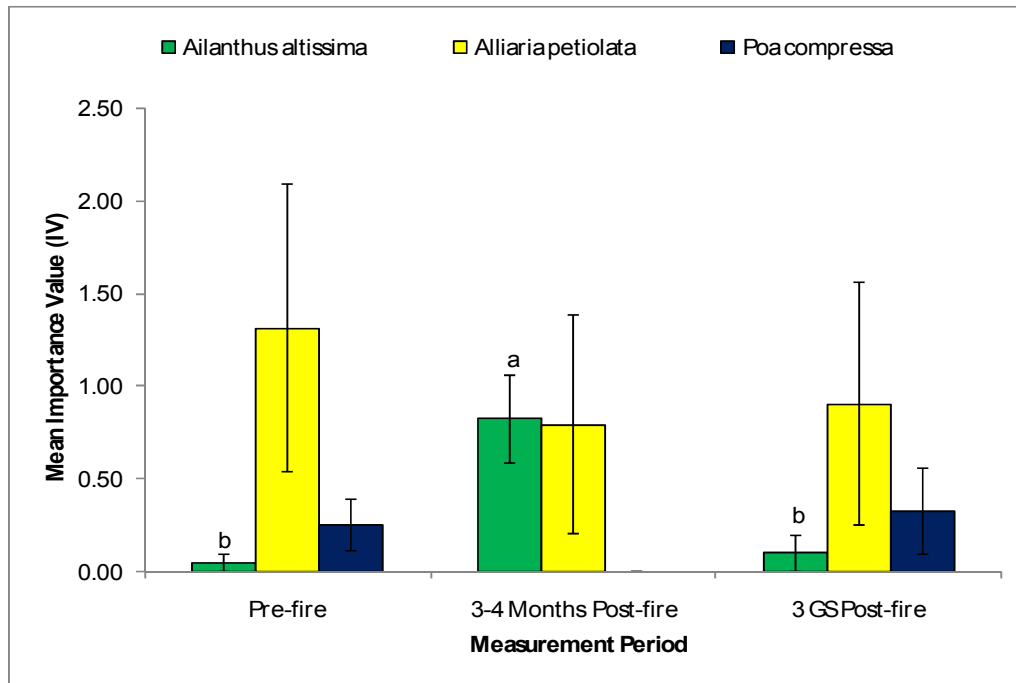


Figure 5.4. Average IV by species and measurement period. Means with the same letter are not significantly different at $\alpha = 0.05$. Error bars represent \pm one standard error. GS = growing seasons.

Axis 1, 2, and 3 represented 10.5, 18.4, and 17.8 % of the variance in species composition (cumulative $R^2 = 0.47$). Axes 2 and 3 are presented and discussed as they reflected most of the variability in species composition. The NMS results provided further evidence that pre-fire and post-fire understory layers can be characterized by somewhat different species compositions. Graphical inspection identified axis 3 as most influential for separating sampling plots based on the effects of the prescribed fire (Figure 5.5). Interestingly, this axis showed a rather strong negative correlation with percent change in canopy cover ($r = -0.41$), indicating that canopy cover decreased from pre- to post-fire.

Axis 3 scores were most positively correlated with striped maple (*Acer pensylvanicum* L.) ($r = 0.39$), grape ($r = 0.38$), and witch-hazel (*Hamamelis virginiana* L.) ($r = 0.30$) (Appendix Table 4). Although the Pearson correlation coefficients were rather low, this indicates that these species were greater in abundance prior to the prescribed fire. To some extent this agrees with the ISA results. Grape was a significant indicator of pre-fire understory composition. Although striped maple and witch hazel exhibited a stronger IV_{kj} in the pre-fire understory in the ISA, their observed IV_{kj} , however, did not exceed their expected IV_{kj} and were, therefore, not significant indicators. Axis 3 scores were most negatively correlated with common cinquefoil (*Potentilla simplex* Michx.) ($r = -0.61$), summer bluets (*Houstonia longifolia* Gaertn.) ($r = -0.50$), and upland low blueberry (*Vaccinium pallidum* Ait.) (-0.45). These correlations indicate that these species were greater in abundance following the prescribed fire. All three of these species were significant indicators of post-fire composition in the ISA.

The exotic invasive garlic mustard showed only weak positive correlations with axis 3 scores ($r = 0.14$) (Appendix Table 4). Canada bluegrass exhibited a negative correlation ($r = -0.11$). These exotic invasives showed stronger correlations to axis 2 ($r = 0.45$ for garlic mustard and $r = 0.22$ for Canada bluegrass), suggesting that additional factors are related to their abundance. Species richness was positively correlated with axis 2 scores (Figure 5.5), suggesting that the exotic invasives are in higher abundance in areas where species richness is higher. Sapling mortality was negatively associated with axis 2, implying that the exotic invasives are in lower abundance in areas where relative fire

intensity was lower. This also indicates that understory species richness is higher in areas where relative fire intensity was lower.

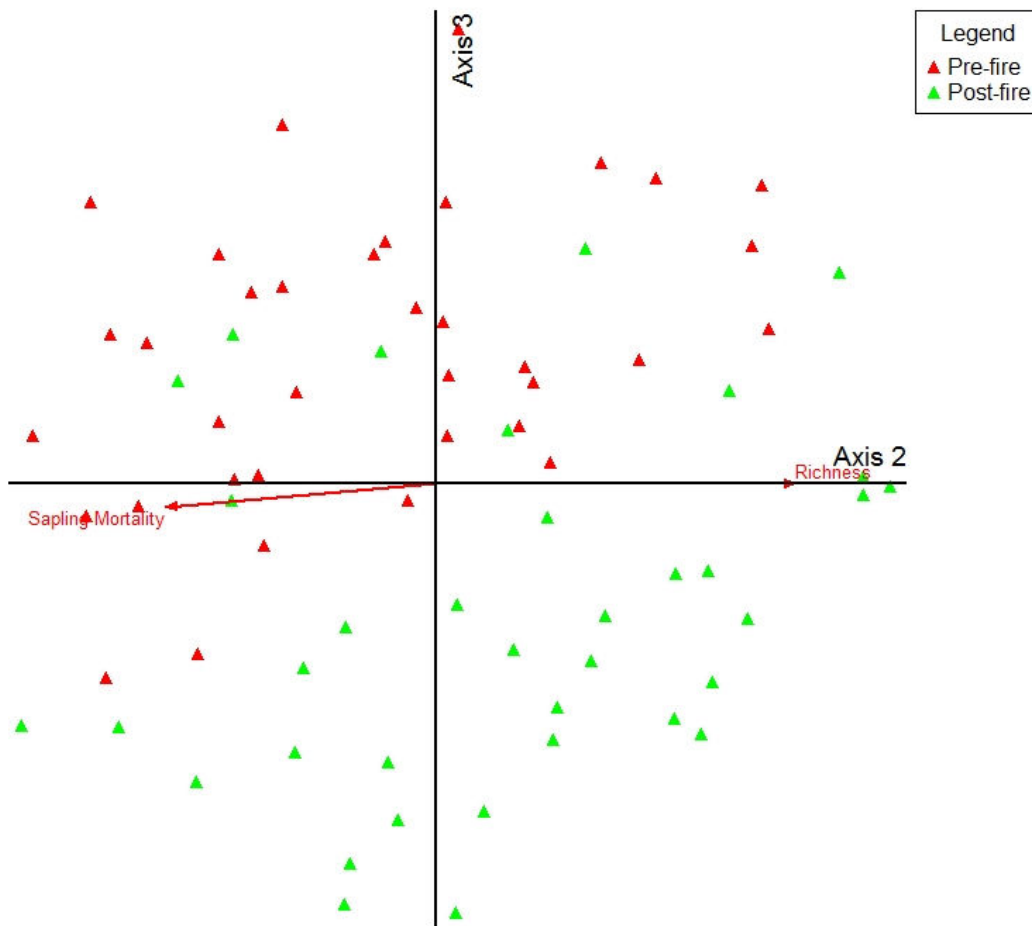


Figure 5.5. NMS ordination (axes 2 and 3) of sample plots (data are understory abundance (percent cover)) before and after the prescribed fire. Vectors showing influential explanatory variables are also shown (R^2 cutoff = 0.20).

Exotic Invasive Plant Species and Relative Fire Intensity/Environmental Variables

NMS Ordination

The NMS ordination (a two dimensional solution) of post-fire “smoothed” timed meander data yielded a final stress and instability of 9.189 and 0.038, respectively.

Axis 1 and 2 represented 79.0 and 16.0 % of the variance in species composition (cumulative $R^2 = 0.95$). Similar to the results of the NMS ordination of pre- and post-fire abundance data, these NMS results also suggested that sapling mortality and species richness can help explain the variability in species composition across the study site (Figure 5.6). In addition, this ordination also identified mean bole scorch height and TRMI as strong explanatory variables. Average bole scorch height and sapling mortality showed rather strong negative correlations with axis 1 ($r = -0.51$ and $r = -0.42$). Species richness, on the other hand, exhibited a strong positive correlation with axis 1 ($r = 0.54$). This implies that understory species richness is higher in areas where relative fire intensity (expressed by average bole scorch height) was lower and vice versa. Because they are highly correlated (see Chapter 4), it also makes sense that bole scorch height and sapling mortality show similar correlations to axis 1. TRMI showed a quite strong negative correlation ($r = -0.57$) with axis 2 scores, while sapling mortality exhibited a moderate positive correlation ($r = 0.41$). This implies that relative fire intensity (expressed by sapling mortality) was higher in dryer areas and lower in moister areas.

The exotic invasives considered in the analysis showed positive correlations with axis 1, suggesting a higher “favorability” for these species in areas of higher understory species

richness and where relative fire intensity was lower (Appendix Table 6). Common sheep sorrel and autumn olive were not considered in the analysis because they occurred in less than five percent of the sample plots. Garlic mustard, tree-of-heaven, Japanese stilt grass, and Canada bluegrass showed correlations of $r = 0.86$, $r = 0.75$, $r = 0.48$, and $r = 0.42$, respectively, with axis 1 scores. With the exception of Canada bluegrass, negative correlations of these exotic invasives to axis 2 also indicate higher “favorability” in moister areas (Appendix Table 6). Japanese stilt grass “favorability” showed the strongest association with these moister areas ($r = -0.58$), followed by garlic mustard ($r = -0.56$) and tree-of-heaven ($r = -0.35$), respectively.

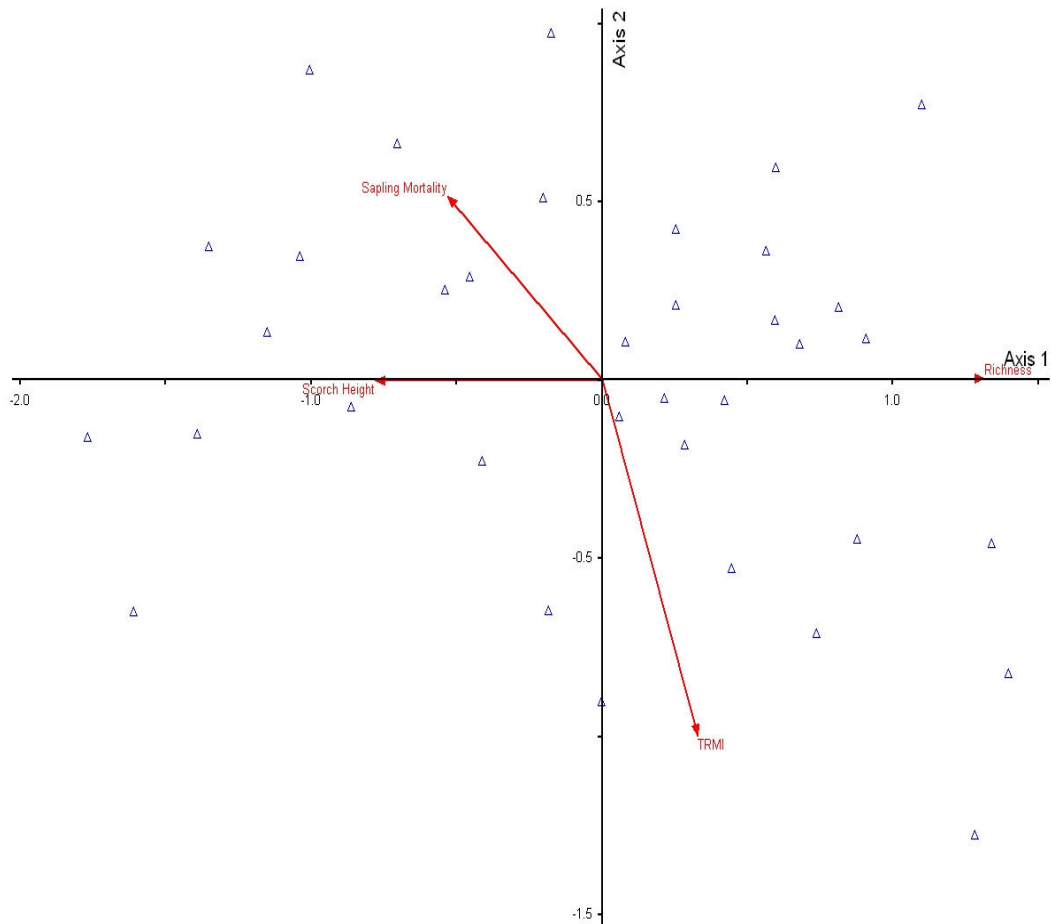


Figure 5.6. NMS ordination (axes 1 and 2) of sample plots (data are “smoothed” understory presence/absence) after the prescribed fire (2007). Vectors showing influential explanatory variables are also shown (R^2 cutoff = 0.20).

Logistic Regression

Seven of the environmental variables were statistically significant when evaluated individually (Table 5.2). Five of these seven variables were statistically significant in the multiple logistic regression models (Table 5.3). Neither of the relative fire intensity variables were significant in either the individual or full models. The Hosmer-Lemeshow goodness of fit tests indicated that the estimates for all of the multiple logistic regression models fit the data at an acceptable level (Table 5.3). However, it should be noted that, because of the small sample size in this study, there is an extremely high probability that the multiple logistic regression models are overfit. Hosmer and Lemeshow (2000) suggest that logistic regression models should use no more than $p + 1 \leq \min(n_1, n_0)/10$ parameters, where p = the number of parameters (variables) and $\min(n_1, n_0)$ is, in this study, the minimum number of plots with either presence or absence of a give exotic invasive species. The results, therefore, should be used with caution.

Elevation was a significant independent variable for all exotic invasive models except garlic mustard. In addition to elevation, distance from harvesting was included in the 2004 (3 - 4 months post-fire) tree-of-heaven multiple logistic regression model. Percent change in canopy cover was a second independent variable included in both the 2007 (three full growing seasons post-fire) tree-of-heaven and Category 1 exotic invasives multiple logistic regression models. Distance from areas where tree-of-heaven has reached the main canopy was the only variable included in the garlic mustard logistic regression model.

Table 5.2. Results of individual logistic runs for each statistically significant variable.

Model	Independent Variable	MLE	SE	p	Odds ratio	
<i>Ailanthus altissima</i> (2004)	Distance from AA in main canopy (m)	-	0.009	0.004	0.015	0.991
	Distance from harvesting (m)	0.120	0.004	0.009	0.988	
	Distance from roads (m)	0.007	0.002	0.008	1.007	
	Elevation (m)	0.020	0.007	0.003	1.020	
<i>Ailanthus altissima</i> (2007)	Distance from roads (m)	0.006	0.003	0.038	1.006	
	Elevation (m)	0.017	0.007	0.012	1.017	
	Percent canopy cover	-	0.105	0.050	0.038	0.900
	Shading (%)	-	0.119	0.050	0.001	0.888
<i>Alliaria petiolata</i> (2007)	Distance from AA in main canopy (m)	-	0.012	0.006	0.035	0.988
	Distance from roads (m)	0.007	0.003	0.026	1.007	
	Elevation (m)	0.016	0.007	0.019	1.016	
Category 1 Exotic Invasives (2007)	Distance from AA in main canopy (m)	-	0.007	0.003	0.019	0.993
	Distance from roads (m)	0.007	0.003	0.005	1.007	
	Elevation (m)	0.020	0.007	0.003	1.020	
	Shading (%)	-	0.076	0.038	0.044	0.927
	Species Richness	0.079	0.04	0.047	1.038	

For ease of interpretation, the effects of the independent variables (based on their odds ratios) on the probability of presence for the exotic invasive species are listed in Table 5.4. Every one unit increase in any given independent variable in Table 5.4 is associated with an increase or decrease in the chance of a given exotic invasive species being present in the understory. For example, for the 2004 tree-of-heaven model: every one meter increase in distance from harvesting is associated with a one percent decrease in the likelihood that tree-of-heaven will be present in the understory (Table 5.4).

Table 5.3. Results of multiple logistic regression models.

Model	Independent Variable	MLE	SE	p	Odds ratio	R ²	χ ²	p
<i>Ailanthus altissima</i> (2004)	Distance from harvesting (m)	-						
		0.012	0.006	0.039	0.988	0.44	7.94	0.338
	Elevation (m)	0.017	0.007	0.013	1.017			
<i>Ailanthus altissima</i> (2007)	Elevation (m)	0.022	0.010	0.032	1.022	0.37	3.11	0.874
	% Δ in Canopy Cover	-						
		0.130	0.056	0.021	0.878			
<i>Alliaria petiolata</i> (2007)	Distance from AA in main canopy (m)	-						
		0.012	0.006	0.035	0.988	0.23	12.69	0.080
Category 1 Exotic Invasives (2007)	Elevation (m)	0.021	0.007	0.005	1.021	0.40	5.29	0.624
	% Δ in Canopy Cover	-						
		0.074	0.042	0.081	0.929			

Table 5.4. Interpretation of odds ratios. Every one unit increase in any given independent variable is associated with an increase or decrease in the odds (effect) of a given exotic invasive species being present in the understory.

Model	Independent Variable	Effect
<i>Ailanthus altissima</i> (2004)	Distance from harvesting (m)	1 % ↓
	Elevation (m)	2 % ↑
<i>Ailanthus altissima</i> (2007)	Elevation (m)	2 % ↑
	% Δ in Canopy Cover	12 % ↓
<i>Alliaria petiolata</i> (2007)	Distance from AA in main canopy (m)	1 % ↓
Category 1 Exotic Invasives (2007)	Elevation (m)	2 % ↑
	% Δ in Canopy Cover	7 % ↓

Discussion

Contrary to the findings of other studies (Arthur and others 1998, Bowles and others 2007, Hutchinson and others 2005, Welch and others 2002), species richness was found to be significantly lower ($P < 0.05$) in the post-fire understory. Marsh (2005) reported a significant increase in overall understory species richness at the same site three to four months post fire. However, understory communities are very dynamic and composition can change within a growing season, and from growing season to growing season following fire (Bowles and others 2007, Hutchinson 2006), as well as other disturbances (Small and McCarthy 2002).

Prior to the application of the prescribed fire, the percentage of exotic invasive plants found on the 1m² understory plots was very small when compared to the total number of plants. Four exotic invasive species were recorded prior to the fire in 2003. However, tree-of-heaven and Bell's honeysuckle were found on only one of the 0.05 ha plots.

Garlic mustard and tree-of-heaven occurred on the same plot, a plot that was located in an area that was harvested in 1972. Garlic mustard was also found on two other plots prior to the fire, one of which was in a different 1972 harvest area. This is an indication that the harvesting may have played a role in initial establishment of the exotic invasive species. Unfortunately, it is unknown whether or not these species occurred on the study site prior to the harvesting in 1972.

Three full growing seasons after the fire, the percentage of exotic invasive plants found on the 1m² understory plots was also very low. Garlic mustard and Canada bluegrass

were found on the same 0.05 ha plots they were found on prior to the fire, with the exception that Canada bluegrass was found on one less plot.

The timed meander inventory identified the same percentage of post-fire exotic invasives as the 1 m² plots. However, the results do indicate that, at the Dunkle Knob study site, the timed meander inventory method is superior to the 1m² understory plots for documenting true understory species richness, defining true understory flora, and for detecting the true number exotic invasive plants found in the understory. If the understory at Dunkle Knob is to be monitored for exotic invasive species, or simply for compositional changes, following future burns, it is suggested that the timed meander inventory method be employed on the sample plots. However, for this method to be effective over multiple measurement periods, those who conduct the inventory should be skilled in flora identification at the site, which may limit its applicability both in this and in other cases (Huebner 2007). Also, because the timed meander method does not provide a true measure of abundance, the 1 m² understory plots should still be employed to monitor changes in this measure.

Pre-treatment vs. Post Treatment

When comparing pre-treatment and post-treatment abundances, neither ISA nor the NMS ordination indicated significance with respect to the exotic invasives. Instead the NMS ordination indicated that abundance of exotic invasives was correlated with areas of higher species richness and lower relative fire intensity. Similarly, other studies have shown that exotic plant invasions are positively associated with higher species richness

(Bridgewater and Backshall 1981, DeFerrari and Naiman 1994, Huebner and Tobin 2006, Robinson and others 1995). Interestingly, also at the 1m² scale, Stohlgren and others (1999) found that higher exotic invasive plants species richness was associated with areas of higher native plant species richness in forests of the Rocky Mountains. The findings of higher exotic invasive abundances in areas of lower fire intensity disagree with those of Glasgow and Matlack (2006). However, exotic invasive plant seeds were added in their study, it took place in a more mesophytic hardwood forest, and they examined two species that did not occur on the 1m² plots at the Dunkle Knob study site.

The results of the MRPP, Mantel test, and NMS ordination did, however, suggest that the pre-fire and post-fire understory layers were characterized by somewhat different species composition. However, some of the differences are likely due to inconsistencies in sampling from period to period. For example, the ISA results suggest that pignut hickory is an indicator of post-fire composition, while shagbark hickory (*Carya ovata* (P. Mill.) K. Koch) is an indicator of pre-fire composition. This is likely a result of misidentification afield during one of the two sampling periods. Changes in investigators, or those who sample the permanent plots, can produce spurious apparent changes (Ketchledge and Leonard 1984, McCune and Menges 1986), especially in the understory where high plant identification skill is required. Based on extensive work with the data, I also feel that although *Pinus* spp., UK Poaceae, and *Vitis* spp., in particular, were statistically significant indicators of pre-fire composition, they are not ecologically significant. That is, these species were deemed as indicators due to a grouping or “lumping” effect. *Vitis* spp. also showed higher abundance pre-fire in the

NMS ordination. These three species groups were classified to species in many more cases during the post-fire inventory. If the post-fire data were “lumped” into groups that matched the pre-fire groups, or if the pre-fire groups could be broken down to the species level, these results would likely be more reliable.

Nevertheless, the ISA results suggest that the fire resulted in an improvement in regeneration for the chestnut oak and red maple (*Acer rubrum* L.). The NMS ordination results also suggest a deterioration in striped maple and witch-hazel regeneration. An increase in wildlife forage is also indicated from both the NMS ordination and ISA results due to increases in upland low blueberry. An improvement in oak regeneration following a single low-intensity fire has been previously documented (Kuddes-Fischer and Arthur 2002, Barnes and Van Lear 1998, Adams and Reiske 2001). The study by Kuddes-Fischer and Arthur (2002) also reported an increase in red maple seedling density. Blankenship and Arthur (2006) also observed abundant red maple regeneration following fire in oak stands in Kentucky. Because upland low blueberry is well adapted to fire (Tirmenstein 1991), it is not surprising to see a greater than expected fidelity to the post-fire understory for this species. Information of the effects of fire on striped maple is lacking, however, its physiological traits would suggest abundant sprouting from dead stems following fire. This, however, was not apparent from the data or from field observations. Witch-hazel is low in resistance and readily killed by fire (Coladonato 1993).

The ANOVA indicated that tree-of-heaven increased in both abundance and importance immediately following the fire, but was back to pre-fire levels after three full growing seasons. Interestingly, tree of heaven seedlings were recorded on eleven of the 0.05 ha plots (data from the 1m² plots) in 2004, compared to only one in 2003 and 2007.

Immediate post-fire increases in seedlings likely came from large seed producing female trees located on or near the study site. Tree-of-heaven employs minimal seed banking (Kowarik 1995) and, if large seed producing females exist, most post-fire establishment would likely be by seed (Huebner 2006). Furthermore, tree-of-heaven seed can be dispersed as far as 200 meters from the female parent (Kota 2005). The seedlings would have benefited from decreased competition for light or mineral resources because the fire had killed or “knocked back” pre-existing vegetation (Adams and others 1994, Boerner and others 2000). Given the xeric nature of the Dunkle Knob study site, native vegetation, in particular those species that showed higher abundances post-fire from the ISA and NMS ordination, would soon re-sprout and out compete the tree-of-heaven seedlings for required resources. Studies that directly assess the effect of fire on tree-of-heaven do not exist. However, Huebner (2006) suggests that, if large seed producers exist, tree-of-heaven may increase in abundance with repeated burning. The results from the Dunkle Knob study site cannot confirm this hypothesis because only a single fire was applied to the area.

Exotic Invasive Plant Species and Relative Fire Intensity/Environmental Variables

Similar to the results of the NMS ordination of pre-fire and post-fire data, the NMS ordination of “smoothed” presence/absence data indicated that areas where relative fire

intensity was lower and understory species richness was higher were more “favorable” for the exotic invasive species. Garlic mustard and tree-of-heaven showed the strongest relationships with this pattern ($r > 0.74$) (Appendix Table 6). TRMI was also an influential independent variable in this NMS ordination. Higher soil moisture and fertility often provide higher understory species richness (Small and McCarthy 2002) and, furthermore, relative fire intensity would tend to be lower in areas of higher soil and litter layer moisture. This suggests that, higher moisture (expressed by TRMI), not higher species richness, is likely the direct cause of higher abundances of exotic invasive plants. Small and McCarthy (2002) noted that the frequency of non-native understory species was greater in moister areas relative to drier areas in an oak forest of the central Appalachians.

Despite the possibility of overfitting, the results of the multiple logistic regression models provided some interesting findings. The relative fire intensity variables were not important in determining the probability of presence of the exotic invasive species in the understory of the study site. Instead, the presence of understory exotic invasives was found to be closely related to environmental variables. Elevation was a significant explanatory variable for all exotic invasive species, except garlic mustard, with a higher probability of presence as elevation increases. This relationship was contrary to results the post-fire NMS ordination, which indicated that moister areas provided favorable microsites for the exotic invasive species. It is well known that on a mountain such as Dunkle Knob, lower slopes have more moisture available for plant use relative to upper slopes. Furthermore, no published studies have documented increases in understory

exotic invasives with increasing elevation. This anomaly may be a result of the combination of increased sampling density near the top of Dunkle Knob (Figure 5.1) and the possible overfitting of the logistic regression models due to a low number of presence observations.

In addition to elevation, both the 2007 tree-of-heaven and Category 1 exotic invasive models also indicated a higher probability of presence where canopy cover has decreased over the last four years. The 2004 tree-of-heaven model also included distance from harvesting as a significant explanatory variable (negative relationship). Tree-of-heaven can tolerate a wide range of environmental conditions (Howard 2004). Several list the species as being shade intolerant (Bourdeau and Laverick 1958, Forgione 1993, Kowarik 1995) and it is often associated with areas of disturbance relative to undisturbed areas (Call and Nilsen 2003, Huebner 2003). Forest disturbances, natural or anthropogenic, generally decrease canopy cover and increase understory light, which, in turn, tends to promote invasion by exotic plant species (Bossard 1991, D'Antonio 1993, Hobbs and Huenneke 1992, Hutchinson and others 2004).

Interestingly, the only significant explanatory variable in the garlic mustard model was distance from areas where tree-of-heaven has reached the main canopy. No published studies have documented this phenomenon, but it is likely that is a result of these areas being associated with disturbance (see Chapter 3). Given the highly invasive nature of these species, coupled with the fact that both possess allelopathic properties (Heisey 1996, McCarthy and Hanson 1998), this suggests that these areas have the potential to be

very troublesome to native understory vegetation in the future. That is, the understory in areas where tree-of-heaven has reached the main canopy may soon be dominated by tree-of-heaven suckers and garlic mustard. Native understory plants, as well as the benefits they provide (e.g. biodiversity, food for wildlife), could be severely impacted in these localized areas.

Sampling Limitations

The location of sampling and sampling methodology employed in this study likely biased the results. A more systematic inventory, employing cruise line perpendicular to topography and more sample plots, would likely produce less conservative estimates of compositional parameters, limit bias, and generate more indicative results. In an area as large as Dunkle Knob (313.6 ha), a much greater number of plots would be required for high confidence in estimates (Avery and Burkhart 2002, Shiver and Borders 1996).

Furthermore, it should be emphasized that the understory abundance measures used in the analyses were the average of four 1m² plots on each 0.05 ha plot. Averaging percent cover measurements likely masked significant variability among understory vegetation within the 0.05 ha plots. Also, delineating and sampling within stand types would also allow for improved results and conclusions because exotic invasives appeared to be associated with certain stand types over others (e.g. cove communities versus table mountain/pitch pine communities (J. Pomp, personal observation). Moreover, fire intensity groupings or the use of a control site (burn versus no burn) would certainly have benefited this study.

Conclusions and Management Implications

The results of this study indicate that the prescribed fire had little effect on the populations of understory exotic invasive plants. Both exotic invasive abundance and importance was, in general, rather low both before and after the prescribed fire treatment. Tree-of-heaven was the only exotic invasive that showed significantly higher abundance and importance following the prescribed fire. This increase, however, was short-lived. Nevertheless, given the limitations of this study, prescribed fire managers in the Ridge and Valley area should attempt to locate exotic invasive plant “hot spots,” especially tree-of-heaven, and, if economically feasible, eradicate them prior to applying a burn. Effective chemical control methods are available for controlling a majority of the unwanted exotic invaders found in this study (tree-of-heaven (Eck 2005), Japanese stilt grass (Pomp and others, In Press), garlic mustard (Nuzzo 1991), Canada bluegrass (Waller and Schmidt 1983), and autumn olive (Kuhns 1986). Pre-fire control of exotic invasive species would aim to make the forest more resilient to further invasion, reduce the likelihood of post-fire establishment, and help protect native understories and the future of eastern oak and pine forest communities, as well as the ecological benefits they provide.

Although the prescribed fire induced changes in understory vegetation, the results suggest that environmental factors play more of a role in determining the location and abundance of exotic invasives at the Dunkle Knob study site. Furthermore, prior harvesting, and road construction associated with it, undoubtedly played a role in the initial establishment of the exotic invasives. It is highly likely that responses exhibited by exotic invasive

plant populations, with respect to fire, will be different in dissimilar areas, or even on similar sites within the Ridge and Valley Physiographic Province. That is, due to differences in prior disturbances and the number and abundance of exotic invasive species present, the effects of prescribed fire on exotic invasive plants will be highly site and species specific. Clearly, however, more research is needed on this topic for it to be fully understood. Investigation of multiple burns, fire intensity groupings, and burning during different seasons would undoubtedly better our understanding of the effect of prescribed fire on understory exotic invasive plant species in xeric mixed-oak and oak-pine forests of the Appalachians.

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**Table 1. Pre-fire (2003) understory layer vascular plants encountered on 1m² plots (n = 144).
UK = unknown.**

Scientific Name	Habit	Functional Type	Duration
<i>Acer pensylvanicum</i>	Tree	Native Weed	Perennial
<i>Acer rubrum</i>	Tree	Native Weed	Perennial
<i>Ageratina altissima</i> var. <i>altissima</i>	Ferns and Forbs	Native Weed	Perennial
<i>Ailanthus altissima</i>	Tree	Exotic Invasive	Perennial
<i>Alliaria petiolata</i>	Ferns and Forbs	Exotic Invasive	Annual
<i>Allium cernuum</i>	Ferns and Forbs	Native	Perennial
<i>Allium</i> cf. <i>cernuum</i>	Ferns and Forbs	Native?	Perennial
<i>Amelanchier arborea</i> var. <i>arborea</i>	Tree	Native	Perennial
<i>Amphicarpaea bracteata</i>	Ferns and Forbs	Native	Annual
<i>Antennaria neglecta</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria plantaginifolia</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria virginica</i>	Ferns and Forbs	Native	Perennial
<i>Asclepias quadrifolia</i>	Ferns and Forbs	Native	Perennial
<i>Asplenium platyneuron</i>	Ferns and Forbs	Native	Perennial
<i>Aster</i> spp.	Ferns and Forbs	Native?	Perennial
<i>Aureolaria laevigata</i>	Ferns and Forbs	Native	Perennial
<i>Aureolaria virginica</i>	Ferns and Forbs	Native	Perennial
<i>Bromus</i> cf. <i>latiglumis</i>	Graminoid	Native	Perennial
<i>Bromus latiglumis</i>	Graminoid	Native	Perennial
<i>Carex cephalophora</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum/tonsa</i>	Graminoid	Native	Perennial
<i>Carex digitalis</i>	Graminoid	Native	Perennial
<i>Carex laxiflora</i>	Graminoid	Native	Perennial
<i>Carex pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> spp.	Graminoid	Native	Perennial
<i>Carex willdenowii</i>	Graminoid	Native	Perennial
<i>Carya glabra</i>	Tree	Native	Perennial
<i>Carya ovata</i>	Tree	Native	Perennial
<i>Carya</i> spp.	Tree	Native	Perennial
Cf. <i>Bromus ciliatus/Calamagrostis porteri</i>	Graminoid	Native	Perennial
Cf. <i>Bromus pubescens</i>	Graminoid	Native	Perennial
Cf. <i>Saxifraga caroliniana</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Bromus</i> spp.	Graminoid	Native	Perennial
Cf. <i>Ceanothus americanus</i>	Shrubs and Vines	Native	Perennial
Cf. <i>Cerastium arvense</i>	Ferns and Forbs	Native Invasive	Perennial
Cf. <i>Festuca subverticillata</i>	Graminoid	Native	Perennial
Cf. <i>Helianthus</i> sp.	Ferns and Forbs	Native?	Annual
Cf. <i>Lonicera x bella</i>	Shrubs and Vines	Exotic Invasive	Perennial
Cf. <i>Muhlenbergia schreberi/frondosa</i>	Graminoid	Native	Perennial
Cf. <i>Packera aurea</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Packera obovata</i>	Ferns and Forbs	Native	Perennial

<i>Cf. Parnassia spp.</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Poa sylvestris</i>	Graminoid	Native	Perennial
<i>Cf. Solidago arguta/speciosa</i>	Ferns and Forbs	Native	Perennial
<i>Cf. UK Asteraceae</i>	Ferns and Forbs	Native?	Perennial
<i>Cf. Verbascum blattaria</i>	Ferns and Forbs	Exotic	Biennial
<i>Chimaphila maculata</i>	Shrubs and Vines	Native	Perennial
<i>Convolvulus spp.</i>	Ferns and Forbs	Native	Perennial
<i>Corydalis flavula</i>	Ferns and Forbs	Native	Annual
<i>Cratageous spp.</i>	Shrubs and Vines	Native	Perennial
<i>Cunila origanoides</i>	Ferns and Forbs	Native	Perennial
<i>Danthonia compressa</i>	Graminoid	Native	Perennial
<i>Danthonia spicata</i>	Graminoid	Native	Perennial
<i>Deschampsia flexuosa</i>	Graminoid	Native	Perennial
<i>Dichanthelium boscii</i>	Graminoid	Native	Perennial
<i>Dichanthelium commutatum</i>	Graminoid	Native	Perennial
<i>Dichanthelium depauperatum</i>	Graminoid	Native	Perennial
<i>Dichanthelium latifolium</i>	Graminoid	Native	Perennial
<i>Dichanthelium linearifolium</i>	Graminoid	Native	Perennial
<i>Dichanthelium spp.</i>	Graminoid	Native	Perennial
<i>Dioscorea villosa</i>	Ferns and Forbs	Native	Perennial
<i>Draba ramosissima</i>	Ferns and Forbs	Native	Perennial
<i>Dryopteris marginalis</i>	Ferns and Forbs	Native	Perennial
<i>Elymus hystrix</i>	Graminoid	Native	Perennial
<i>Epigaea repens</i>	Shrubs and Vines	Native	Perennial
<i>Erechtites hieracifolia</i>	Ferns and Forbs	Native Weed	Annual
<i>Festuca subverticillata</i>	Graminoid	Native	Perennial
<i>Galium circaezans var. circaezans</i>	Ferns and Forbs	Native	Perennial
<i>Galium lanceolatum</i>	Ferns and Forbs	Native	Perennial
<i>Galium triflorum</i>	Ferns and Forbs	Native	Perennial
<i>Gaultheria procumbens</i>	Shrubs and Vines	Native	Perennial
<i>Gaylussacia baccata</i>	Shrubs and Vines	Native	Perennial
<i>Hamamelis virginiana</i>	Vines	Native	Perennial
<i>Hedeoma pulegioides</i>	Ferns and Forbs	Native Weed	Annual
<i>Helianthus sp.</i>	Ferns and Forbs	Native	Annual
<i>Hepatica nobilis</i>	Ferns and Forbs	Native	Perennial
<i>Heuchera americana</i>	Ferns and Forbs	Native	Perennial
<i>Hieracium pilosella var. pilosella</i>	Ferns and Forbs	Exotic	Perennial
<i>Hieracium venosum</i>	Ferns and Forbs	Native	Perennial
<i>Houstonia cf. caerulea</i>	Ferns and Forbs	Native	Perennial
<i>Houstonia longifolia</i>	Ferns and Forbs	Native	Perennial
<i>Juncus tenuis</i>	Graminoid	Native	Perennial
<i>Kalmia latifolia</i>	Shrubs and Vines	Native	Perennial
<i>Lespedeza hirta ssp. hirta</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza procumbens</i>	Ferns and Forbs	Native	Perennial
<i>Lolium arundinaceum</i>	Graminoid	Exotic	Perennial

<i>Maianthemum racemosum ssp. racemosum</i>	Ferns and Forbs	Native	Perennial
<i>Nyssa sylvatica</i>	Tree	Native	Perennial
<i>Ostrya virginiana</i>	Tree	Native	Perennial
<i>Parietaria pensylvanica</i>	Ferns and Forbs	Native	Annual
<i>Paronychia canadensis</i>	Ferns and Forbs	Native	Annual
<i>Parthenocissus quinquefolia</i>	Shrubs and Vines	Native	Perennial
<i>Phlox subulata</i>	Shrubs and Vines	Native	Perennial
<i>Pinus pungens</i>	Tree	Native	Perennial
<i>Pinus rigida</i>	Tree	Native	Perennial
<i>Pinus spp.</i>	Tree	Native	Perennial
<i>Pinus strobus</i>	Tree	Native	Perennial
<i>Pinus virginiana</i>	Tree	Native	Perennial
<i>Pinus virginiana/pungens</i>	Tree	Native	Perennial
<i>Poa cf. trivialis</i>	Graminoid	Exotic	Perennial
<i>Poa compressa</i>	Graminoid	Exotic Invasive	Perennial
<i>Polygonatum biflorum</i>	Ferns and Forbs	Native	Perennial
<i>Polygonum cf. convolvulus</i>	Shrubs and Vines	Exotic	Annual
<i>Polygonum scandens</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Polypodium vulgare</i>	Ferns and Forbs	Native	Perennial
<i>Potentilla simplex</i>	Ferns and Forbs	Native	Perennial
<i>Prenanthes alba</i>	Ferns and Forbs	Native	Biennial
<i>Prunus serotina</i>	Tree	Native	Perennial
<i>Pseudognaphalium obtusifolium ssp. obtusifolium</i>	Ferns and Forbs	Native	Annual
<i>Pteridium aquilinum</i>	Ferns and Forbs	Native Invasive	Perennial
<i>Quercus coccinea</i>	Tree	Native	Perennial
<i>Quercus ilicifolia</i>	Shrubs and Vines	Native	Perennial
<i>Quercus prinus</i>	Tree	Native	Perennial
<i>Quercus rubra</i>	Tree	Native	Perennial
<i>Quercus spp.</i>	Tree	Native	Perennial
<i>Quercus velutina</i>	Tree	Native	Perennial
<i>Rhus aromatica</i>	Shrubs and Vines	Native	Perennial
<i>Robinia pseudoacacia</i>	Tree	Native Invasive	Perennial
<i>Rosa carolina</i>	Shrubs and Vines	Native	Perennial
<i>Rubus spp.</i>	Shrubs and Vines	Native Weed	Perennial
<i>Sassifraga albidum</i>	Tree	Native	Perennial
<i>Sedum ternatum/glaucophyllum</i>	Ferns and Forbs	Native	Perennial
<i>Smilax rotundifolia</i>	Shrubs and Vines	Native	Perennial
<i>Solidago cf. curtisii</i>	Ferns and Forbs	Native	Perennial
<i>Solidago sp.</i>	Ferns and Forbs	Native	Perennial
<i>Sphenopholis nitida</i>	Graminoid	Native	Perennial
<i>Spiraea betulifolia var. corymbosa</i>	Shrubs and Vines	Native	Perennial
<i>Symphyotrichum cf. divaricatum</i>	Ferns and Forbs	Native	Annual

<i>Symphytotrichum divaricatum</i>	Ferns and Forbs	Native	Annual
<i>Thalictrum thalictroides</i>	Ferns and Forbs	Native	Perennial
<i>Triodanis perfoliata</i>	Ferns and Forbs	Native	Annual
UK Asteraceae	Ferns and Forbs	Native?	Perennial
UK Dicot	Ferns and Forbs?	Native?	Perennial?
UK Fern	Ferns and Forbs	Native	Perennial
UK Poaceae	Graminoid	Native?	Perennial
<i>Uvularia perfoliata</i>	Ferns and Forbs	Native	Perennial
<i>Uvularia puberula</i>	Ferns and Forbs	Native	Perennial
<i>Uvularia sessilifolia</i>	Ferns and Forbs	Native	Perennial
<i>Vaccinium pallidum</i>	Shrubs and Vines	Native	Perennial
<i>Vaccinium stamineum</i>	Shrubs and Vines	Native	Perennial
<i>Verbascum sp.</i>	Ferns and Forbs	Native?	Perennial?
<i>Viburnum acerifolium</i>	Shrubs and Vines	Native	Perennial
<i>Viburnum prunifolium</i>	Tree	Native	Perennial
<i>Vicia caroliniana</i>	Shrubs and Vines	Native	Perennial
<i>Viola cf. palmata</i>	Ferns and Forbs	Native	Perennial
<i>Viola cf. sororia</i>	Ferns and Forbs	Native	Annual
<i>Viola palmata</i>	Ferns and Forbs	Native	Perennial
<i>Viola pedata</i>	Ferns and Forbs	Native	Perennial
<i>Vitis aestivalis var. bicolor</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Vitis sp.</i>	Shrubs and Vines	Native Invasive	Perennial

Table 2. Post-fire (2007) understory layer vascular plants encountered on 1m² plots (n = 144). UK = unknown.

Species	Habit	Functional Type	Duration
<i>Acer pensylvanicum</i>	Tree	Native Weed	Perennial
<i>Acer rubrum</i>	Tree	Native Weed	Perennial
<i>Ageratina altissima</i> var. <i>altissima</i>	Ferns and Forbs	Native Weed	Perennial
<i>Ailanthus altissima</i>	Tree	Exotic Invasive	Perennial
<i>Alliaria petiolata</i>	Ferns and Forbs	Exotic Invasive	Annual
<i>Allium cernuum</i>	Ferns and Forbs	Native	Perennial
<i>Ambrosia artemisiifolia</i>	Ferns and Forbs	Native Weed	Annual
<i>Amelanchier arborea</i>	Tree	Native	Perennial
<i>Amphicarpaea bracteata</i>	Ferns and Forbs	Native	Annual
<i>Antennaria neglecta</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria plantaginifolia</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria virginica</i>	Ferns and Forbs	Native	Perennial
<i>Asclepias quadrifolia</i>	Ferns and Forbs	Native	Perennial
<i>Asplenium platyneuron</i>	Ferns and Forbs	Native	Perennial
<i>Aureolaria flava</i> var. <i>flava</i>	Ferns and Forbs	Native	Perennial
<i>Botrychium virginianum</i>	Ferns and Forbs	Native	Perennial
<i>Brachyletrum erectum</i>	Graminoid	Native	Perennial
<i>Campanula divaricata</i>	Ferns and Forbs	Native	Perennial
<i>Carex cephalophora</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum/tonsa</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>tonsa</i>	Graminoid	Native	Perennial
<i>Carex digitalis</i>	Graminoid	Native	Perennial
<i>Carex laxiflora</i>	Graminoid	Native	Perennial
<i>Carex pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> spp.	Graminoid	Native	Perennial
<i>Carya glabra</i>	Tree	Native	Perennial
<i>Ceanothus americanus</i>	Shrubs and Vines	Native	Perennial
Cf. <i>Aster</i> spp.	Ferns and Forbs	Native?	Perennial
Cf. <i>Aureolaria laevigata</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Bromus ciliatus/Calamagrostis porteri</i>	Graminoid	Native	Perennial
Cf. <i>Festuca subverticillata</i>	Graminoid	Native	Perennial
Cf. <i>Hieracium</i> spp.	Ferns and Forbs	Native?	Perennial
Cf. <i>Ilex montana</i>	Tree	Native	Perennial
Cf. <i>Lysimachia quadrifolia</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Phlox buckleyi</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Sassafras albidum</i>	Tree	Native	Perennial
Cf. <i>Scrophularia lanceolata</i>	Ferns and Forbs	Native	Perennial
Cf. <i>Sericocarpus asteroides</i>	Ferns and Forbs	Native	Perennial
Cf. UK <i>Asteraceae</i>	Ferns and Forbs	Native?	Perennial
<i>Chimaphila maculata</i>	Shrubs and Vines	Native	Perennial
<i>Comandra</i> cf. <i>umbellata</i>	Ferns and Forbs	Native	Perennial

<i>Corydalis flavula</i>	Ferns and Forbs	Native	Annual
<i>Corylus americana</i>	Shrubs and Vines	Native	Perennial
<i>Cratageous spp.</i>	Shrubs and Vines	Native	Perennial
<i>Cunila origanoides</i>	Ferns and Forbs	Native	Perennial
<i>Danthonia spicata</i>	Graminoid	Native	Perennial
<i>Danthonia spp.</i>	Graminoid	Native	Perennial
<i>Dennstaedtia punctilobula</i>	Ferns and Forbs	Native Invasive	Perennial
<i>Deschampsia flexuosa</i>	Graminoid	Native	Perennial
<i>Dichanthelium boscii</i>	Graminoid	Native	Perennial
<i>Dichanthelium commutatum</i>	Graminoid	Native	Perennial
<i>Dichanthelium depauperatum</i>	Graminoid	Native	Perennial
<i>Dichanthelium latifolium</i>	Graminoid	Native	Perennial
<i>Dioscorea villosa</i>	Ferns and Forbs	Native	Perennial
<i>Draba ramosissima</i>	Ferns and Forbs	Native	Perennial
<i>Dryopteris marginalis</i>	Ferns and Forbs	Native	Perennial
<i>Erechtites hieraciifolia</i>	Ferns and Forbs	Native Weed	Annual
<i>Festuca subverticillata</i>	Graminoid	Native	Perennial
<i>Fraxinus americana</i>	Tree	Native	Perennial
<i>Galium circaezans</i> var. <i>circaezans</i>	Ferns and Forbs	Native	Perennial
<i>Galium lanceolatum</i>	Ferns and Forbs	Native	Perennial
<i>Galium triflorum</i>	Ferns and Forbs	Native	Perennial
<i>Gaylussacia baccata</i>	Shrubs and Vines	Native	Perennial
<i>Goodyera spp.</i>	Ferns and Forbs	Native	Perennial
<i>GRubus spp.</i>	Shrubs and Vines	Native	Perennial
<i>Hamamelis virginiana</i>	Shrubs and Vines	Native	Perennial
<i>Hedeoma pulegioides</i>	Ferns and Forbs	Native Weed	Annual
<i>Helianthus cf. divaricatus</i>	Ferns and Forbs	Native	Perennial
<i>Helianthus divaricatus</i>	Ferns and Forbs	Native	Perennial
<i>Hepatica nobilis</i> var. <i>obtusata</i>	Ferns and Forbs	Native	Perennial
<i>Heuchera americana</i>	Ferns and Forbs	Native	Perennial
<i>Hieracium pilosella</i> var. <i>pilosella</i>	Ferns and Forbs	Exotic	Perennial
<i>Hieracium venosum</i>	Ferns and Forbs	Native	Perennial
<i>Houstonia longifolia</i>	Ferns and Forbs	Native	Perennial
<i>Iris verna</i>	Ferns and Forbs	Native	Perennial
<i>Kalmia latifolia</i>	Shrubs and Vines	Native	Perennial
<i>Lespedeza cf. violacea</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza hirta</i> ssp. <i>hirta</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza repens</i>	Ferns and Forbs	Native	Perennial
<i>Maianthemum racemosum</i> ssp. <i>racemosum</i>	Ferns and Forbs	Native	Perennial
<i>Melampyrum lineare</i>	Ferns and Forbs	Native	Annual
<i>Mentha suaveolens</i>	Ferns and Forbs	Exotic	Perennial
<i>Muhlenbergia tenuiflora</i>	Graminoid	Native	Perennial
<i>Nyssa Sylvatica</i>	Tree	Native	Perennial
<i>Ostrya virginiana</i>	Tree	Native	Perennial
<i>Oxalis cf. stricta</i>	Ferns and Forbs	Native	Perennial
<i>Parietaria pensylvanica</i>	Ferns and Forbs	Native	Annual

<i>Paronychia canadensis</i>	Ferns and Forbs	Native	Annual
<i>Paronychia fastigiata/montana</i>	Ferns and Forbs	Native	Annual
<i>Parthenocissus quinquefolia</i>	Shrubs and Vines	Native	Perennial
<i>Phytolacca americana var. americana</i>	Ferns and Forbs	Native Weed	Perennial
<i>Pinus rigida</i>	Tree	Native	Perennial
<i>Pinus strobus</i>	Tree	Native	Perennial
<i>Pinus virginiana</i>	Tree	Native	Perennial
<i>Poa compressa</i>	Graminoid	Exotic Invasive	Perennial
<i>Polygala senega</i>	Ferns and Forbs	Native	Perennial
<i>Polygonatum biflorum</i>	Ferns and Forbs	Native	Perennial
<i>Polygonum cf. convolvulus</i>	Shrubs and Vines	Exotic	Annual
<i>Potentilla simplex</i>	Ferns and Forbs	Native	Perennial
<i>Prenanthes alba</i>	Ferns and Forbs	Native	Biennial
<i>Prunus serotina</i>	Tree	Native	Perennial
<i>Pteridium aquilinum</i>	Ferns and Forbs	Native Invasive	Perennial
<i>Quercus ilicifolia</i>	Shrubs and Vines	Native	Perennial
<i>Quercus prinus</i>	Tree	Native	Perennial
<i>Quercus rubra</i>	Tree	Native	Perennial
<i>Quercus velutina</i>	Tree	Native	Perennial
<i>Robinia pseudoacacia</i>	Tree	Native Invasive	Perennial
<i>Rosa carolina</i>	Shrubs and Vines	Native	Perennial
<i>Rubus spp.</i>	Shrubs and Vines	Native Weed	Perennial
<i>Rumex cf. obtusifolius</i>	Ferns and Forbs	Exotic	Perennial
<i>Sassafras albidum</i>	Tree	Native	Perennial
<i>Saxifraga caroliniana</i>	Ferns and Forbs	Native	Perennial
<i>Sedum ternatum/glaucophyllum</i>	Ferns and Forbs	Native	Perennial
<i>Smilax rotundifolia</i>	Shrubs and Vines	Native	Perennial
<i>Solidago caesia</i>	Ferns and Forbs	Native	Perennial
<i>Solidago cf. curtisii</i>	Ferns and Forbs	Native	Perennial
<i>Solidago cf. nemoralis</i>	Ferns and Forbs	Native	Perennial
<i>Spiraea betulifolia var. corymbosa</i>	Shrubs and Vines	Native	Perennial
<i>Symphyotrichum divaricatum</i>	Ferns and Forbs	Native	Annual
<i>Taenidia integerrima</i>	Ferns and Forbs	Native	Perennial
<i>Tephrosia virginiana</i>	Ferns and Forbs	Native	Perennial
<i>Thalictrum thalictroides</i>	Ferns and Forbs	Native	Perennial
<i>Toxicodendron radicans</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Triodanis perfoliata var. perfoliata</i>	Ferns and Forbs	Native	Annual
UK Asteraceae	Ferns and Forbs	Native?	Perennial
UK Dicot	Ferns and Forbs?	Native?	Perennial?
<i>Uvularia perfoliata</i>	Ferns and Forbs	Native	Perennial
<i>Uvularia puberula</i>	Ferns and Forbs	Native	Perennial
<i>Vaccinium pallidum</i>	Shrubs and Vines	Native	Perennial

<i>Vaccinium stamineum</i>	Shrubs and Vines	Native	Perennial
<i>Veronica officinalis</i>	Ferns and Forbs	Exotic	Perennial
<i>Viburnum acerifolium</i>	Shrubs and Vines	Native	Perennial
<i>Vicia caroliniana</i>	Shrubs and Vines	Native	Perennial
<i>Viola cf. palmata</i>	Ferns and Forbs	Native	Perennial
<i>Viola palmata</i>	Ferns and Forbs	Native	Perennial
<i>Vitis aestivalis var. bicolor</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Vitis vulpina</i>	Shrubs and Vines	Native Invasive	Perennial

Table 3. Post-fire (2007) understory layer vascular plants encountered during the timed meander inventory (n = 36). UK = unknown.

Species	Habit	Functional Type	Duration
<i>Acer pensylvanicum</i>	Tree	Native Weed	Perennial
<i>Acer rubrum</i> var. <i>rubrum</i>	Tree	Native Weed	Perennial
<i>Acer saccharum</i> var. <i>saccharum</i>	Tree	Native	Annual
<i>Actaea racemosa</i> var. <i>racemosa</i>	Ferns and Forbs	Native	Perennial
<i>Adiantum pedatum</i>	Ferns and Forbs	Native	Biennial
<i>Ageratina altissima</i> var. <i>altissima</i>	Ferns and Forbs	Native Weed	Perennial
<i>Ailanthus altissima</i>	Tree	Exotic Invasive	Perennial
<i>Alliaria petiolata</i>	Ferns and Forbs	Exotic Invasive	Perennial
<i>Allium cernuum</i>	Ferns and Forbs	Native	Perennial
<i>Allium</i> cf. <i>cernuum</i>	Ferns and Forbs	Native?	Perennial
<i>Ambrosia artemisiifolia</i>	Ferns and Forbs	Native Weed	Perennial
<i>Amelanchier arborea</i> var. <i>arborea</i>	Tree	Native	Perennial
<i>Amphicarpaea bracteata</i>	Ferns and Forbs	Native	Perennial
<i>Anaphalis margaritacea</i>	Ferns and Forbs	Native Weed	Perennial
<i>Antennaria neglecta</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria plantaginifolia</i>	Ferns and Forbs	Native	Perennial
<i>Antennaria virginica</i>	Ferns and Forbs	Native	Perennial
<i>Apocynum</i> spp.	Ferns and Forbs	Native Weed	Perennial
<i>Arabis canadensis</i>	Ferns and Forbs	Native	Perennial
<i>Aralia spinosa</i>	Tree	Native Weed	Perennial
<i>Aristolochia serpentaria</i>	Ferns and Forbs	Native	Perennial
<i>Asclepias quadrifolia</i>	Ferns and Forbs	Native	Perennial
<i>Asplenium platyneuron</i>	Ferns and Forbs	Native	Perennial
<i>Aster</i> spp.	Ferns and Forbs	Native?	Perennial
<i>Aureolaria flava</i> var. <i>flava</i>	Ferns and Forbs	Native	Perennial
<i>Aureolaria laevigata</i>	Ferns and Forbs	Native	Biennial
<i>Botrychium virginianum</i>	Ferns and Forbs	Native	Perennial
<i>Brachyletrum erectum</i>	Graminoid	Native	Perennial
<i>Campanula divaricata</i>	Ferns and Forbs	Native	Perennial
<i>Carex cephalophora</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>pensylvanica/lucorum/tonsa</i>	Graminoid	Native	Perennial
<i>Carex</i> cf. <i>tonsa</i>	Graminoid	Native	Perennial
<i>Carex digitalis</i>	Graminoid	Native	Perennial
<i>Carex laxiflora</i>	Graminoid	Native	Perennial
<i>Carex pensylvanica</i>	Graminoid	Native	Perennial
<i>Carex</i> spp.	Graminoid	Native	Annual
<i>Carex</i> spp. 1	Graminoid	Native	Perennial
<i>Carya alba</i>	Tree	Native	Perennial
<i>Carya glabra</i>	Tree	Native	Annual
<i>Carya ovata</i>	Tree	Native	Perennial
<i>Castanea dentata</i>	Tree	Native	Perennial
<i>Caulophyllum thalictroides</i>	Ferns and Forbs	Native	Perennial
<i>Ceanothus americanus</i>	Shrubs and	Native	Perennial

	Vines		
<i>Cercis canadensis</i> var. <i>canadensis</i>	Tree	Native	Perennial
<i>Cf. Agrostis perennans</i>	Graminoid	Native?	Perennial
<i>Cf. Aster</i> spp.	Ferns and Forbs	Native?	Perennial
<i>Cf. Aureolaria laevigata</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Bromus ciliatus/Calamagrostis porteri</i>	Graminoid	Native	Perennial
<i>Cf. Bromus pubescens</i>	Graminoid	Native	Perennial?
<i>Cf. Calystegia spithamea</i> ssp. <i>purshiana</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Doellingeria infirma</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Euthamia</i> spp.	Ferns and Forbs	Native Weed	Perennial
<i>Cf. Festuca subverticillata</i>	Graminoid	Native	Perennial
<i>Cf. Hackelia virginiana</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Heuchera americana</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Hieracium</i> spp.	Ferns and Forbs	Native?	Perennial
<i>Cf. Ilex montana</i>	Tree	Native	Perennial
<i>Cf. Ionactis linariifolius</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Lactuca</i> spp.	Ferns and Forbs	Native?	Perennial
	Shrubs and		
<i>Cf. Lathyrus venosus</i>	Vines	Native	Perennial
	Shrubs and		
<i>Cf. Lindera benzoin</i>	Vines	Native	Perennial
	Shrubs and		
<i>Cf. Lyonia mariana</i>	Vines	Native	Perennial
<i>Cf. Lysimachia quadrifolia</i>	Ferns and Forbs	Native	Perennial
	Shrubs and		
<i>Cf. Menziesia pilosa</i>	Vines	Native	Perennial
<i>Cf. Phlox buckleyi</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Phytolacca americana</i> var. <i>americana</i>	Ferns and Forbs	Native Weed	Perennial
<i>Cf. Prenanthes</i> spp.	Ferns and Forbs	Native	Perennial
<i>Cf. Pycnanthemum</i> spp.	Ferns and Forbs	Native	Perennial
<i>Cf. Sassafras albidum</i>	Tree	Native	Perennial
<i>Cf. Scrophularia lanceolata</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Sericocarpus asteroides</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Sericocarpus linifolius</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Solidago bicolor</i>	Ferns and Forbs	Native	Annual
<i>Cf. Solidago curtisii</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Thaspium barbinode</i>	Ferns and Forbs	Native	Perennial
<i>Cf. Thaspium/Zizia</i> spp.	Ferns and Forbs	Native	Perennial
<i>Cf. UK Asteraceae</i>	Ferns and Forbs	Native?	Perennial
	Shrubs and		
<i>Chimaphila maculata</i>	Vines	Native	Perennial
<i>Cirsium palustre</i>	Ferns and Forbs	Exotic	Perennial
<i>Cirsium vulgare</i>	Ferns and Forbs	Exotic	Biennial
<i>Comandra</i> cf. <i>umbellata</i>	Ferns and Forbs	Native	Perennial
<i>Conopholis americana</i>	Ferns and Forbs	Native	Perennial
<i>Cornus florida</i>	Tree	Native	Perennial
<i>Corydalis flavula</i>	Ferns and Forbs	Native	Perennial
	Shrubs and		
<i>Corylus americana</i>	Vines	Native	Perennial
	Shrubs and		
<i>Crataegus</i> spp.	Vines	Native	Perennial
<i>Cunila origanoides</i>	Ferns and Forbs	Native	Perennial

<i>Cynoglossum virginianum</i>	Ferns and Forbs	Native Weed	Perennial
<i>Danthonia cf. spicata</i>	Graminoid	Native	Perennial
<i>Danthonia spicata</i>	Graminoid	Native	Annual
<i>Danthonia spp.</i>	Graminoid	Native	Annual
<i>Daucus carota</i>	Ferns and Forbs	Exotic	Biennial
<i>Dennstaedtia punctilobula</i>	Ferns and Forbs	Native Invasive	Perennial
<i>Deschampsia flexuosa</i>	Graminoid	Native	Annual
<i>Desmodium nudiflorum</i>	Ferns and Forbs	Native	Perennial
<i>Desmodium rotundifolium</i>	Ferns and Forbs	Native	Perennial
<i>Desmodium spp.</i>	Ferns and Forbs	Native	Perennial
<i>Dichanthelium boscii</i>	Graminoid	Native	Perennial
<i>Dichanthelium clandestinum</i>	Graminoid	Native Weed	Perennial
<i>Dichanthelium commutatum</i>	Graminoid	Native	Perennial?
<i>Dichanthelium depauperatum</i>	Graminoid	Native	Perennial?
<i>Dichanthelium latifolium</i>	Graminoid	Native	Annual
<i>Dioscorea villosa</i>	Ferns and Forbs	Native	Perennial
<i>Doellingeria infirma</i>	Ferns and Forbs	Native	Perennial
<i>Draba ramosissima</i>	Ferns and Forbs	Native	Perennial
<i>Dryopteris marginalis</i>	Ferns and Forbs	Native	Perennial
<i>Elaeagnus umbellata var. parviflora</i>	Shrubs and Vines	Exotic Invasive	Annual
<i>Elymus hystrix var. hystrix</i>	Graminoid Shrubs and	Native	Perennial
<i>Epigaea repens</i>	Vines	Native	Perennial
<i>Erechtites hieraciifolia</i>	Ferns and Forbs	Native Weed	Perennial
<i>Erigeron strigosus</i>	Ferns and Forbs	Native Weed	Perennial
<i>Festuca subverticillata</i>	Graminoid	Native	Perennial
<i>Fraxinus americana</i>	Tree	Native	Perennial
<i>Galium circaezans var. circaezans</i>	Ferns and Forbs	Native	Annual
<i>Galium lanceolatum</i>	Ferns and Forbs	Native	Perennial
<i>Galium triflorum</i>	Ferns and Forbs	Native	Perennial
<i>Gaultheria procumbens</i>	Shrubs and Vines	Native	Perennial
<i>Gaylussacia baccata</i>	Shrubs and Vines	Native	Perennial
<i>Goodyera spp.</i>	Ferns and Forbs	Native	Perennial
<i>GRubus spp.</i>	Shrubs and Vines	Native	Perennial
<i>Hamamelis virginiana</i>	Shrubs and Vines	Native	Perennial
<i>Hedeoma pulegioides</i>	Ferns and Forbs	Native Weed	Perennial
<i>Helianthus cf. divaricatus</i>	Ferns and Forbs	Native	Perennial
<i>Helianthus divaricatus</i>	Ferns and Forbs	Native	Perennial
<i>Helianthus spp.</i>	Ferns and Forbs	Native	Perennial
<i>Hepatica nobilis var. obtusa</i>	Ferns and Forbs	Native	Perennial
<i>Heuchera americana</i>	Ferns and Forbs	Native	Perennial
<i>Hieracium caespitosum</i>	Ferns and Forbs	Exotic	Perennial
<i>Hieracium pilosella var. pilosella</i>	Ferns and Forbs	Exotic	Perennial
<i>Hieracium venosum</i>	Ferns and Forbs	Native	Perennial
<i>Houstonia longifolia</i>	Ferns and Forbs	Native	Perennial
<i>Iris verna</i>	Ferns and Forbs	Native	Perennial
<i>Juniperus virginiana var. virginiana</i>	Tree	Native Weed	Perennial

<i>Kalmia latifolia</i>	Shrubs and Vines	Native	Perennial
<i>Laportea canadensis</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza cf. violacea</i>	Ferns and Forbs	Native	Annual
<i>Lespedeza hirta ssp. hirta</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza procumbens</i>	Ferns and Forbs	Native	Perennial
<i>Lespedeza repens</i>	Ferns and Forbs	Native	Annual
<i>Lespedeza violacea</i>	Ferns and Forbs	Native	Annual
	Shrubs and		
<i>Lindera benzoin</i>	Vines	Native	Perennial
<i>Lobelia spp.</i>	Ferns and Forbs	Native?	Perennial
<i>Maianthemum racemosum ssp. racemosum</i>	Ferns and Forbs	Native	Annual
<i>Melampyrum lineare</i>	Ferns and Forbs	Native	Perennial
<i>Mentha suaveolens</i>	Ferns and Forbs	Exotic	Perennial
	Shrubs and		
<i>Menziesia pilosa</i>	Vines	Native	Perennial
<i>Microstegium vimineum</i>	Graminoid	Exotic Invasive	Perennial
<i>Monotropa uniflora</i>	Ferns and Forbs	Native	Perennial
<i>Muhlenbergia tenuiflora</i>	Graminoid	Native	Perennial
<i>Nyssa sylvatica</i>	Tree	Native	Perennial
<i>Ostrya virginiana var. virginiana</i>	Tree	Native	Perennial
<i>Oxalis cf. stricta</i>	Ferns and Forbs	Native	Annual
<i>Parietaria pensylvanica</i>	Ferns and Forbs	Native	Perennial
<i>Paronychia canadensis</i>	Ferns and Forbs	Native	Perennial
<i>Paronychia fastigiata/montana</i>	Ferns and Forbs	Native	Perennial
	Shrubs and		
<i>Parthenocissus quinquefolia</i>	Vines	Native	Perennial
<i>Penstemon canescens</i>	Ferns and Forbs	Native	Perennial
<i>Penstemon cf. canescens</i>	Ferns and Forbs	Native	Biennial
	Shrubs and		
<i>Phlox subulata</i>	Vines	Native	Perennial
<i>Phytolacca americana var. americana</i>	Ferns and Forbs	Native Weed	Perennial
<i>Pilea pumila</i>	Ferns and Forbs	Native	Perennial
<i>Pinus pungens</i>	Tree	Native	Perennial
<i>Pinus rigida</i>	Tree	Native	Perennial
<i>Pinus strobus</i>	Tree	Native	Perennial
<i>Pinus virginiana</i>	Tree	Native	Perennial
<i>Pinus virginiana/pungens</i>	Tree	Native	Perennial
<i>Poa compressa</i>	Graminoid	Exotic Invasive	Perennial
<i>Polygala senega</i>	Ferns and Forbs	Native	Perennial
<i>Polygonatum biflorum</i>	Ferns and Forbs	Native	Perennial
	Shrubs and		
<i>Polygonum cf. convolvulus</i>	Vines	Exotic	Perennial
<i>Polystichum acrostichoides</i>	Ferns and Forbs	Native	Perennial
<i>Potentilla simplex</i>	Ferns and Forbs	Native	Perennial
<i>Prenanthes alba</i>	Ferns and Forbs	Native	Perennial
<i>Prenanthes serpentaria</i>	Ferns and Forbs	Native	Perennial
<i>Prenanthes spp.</i>	Ferns and Forbs	Native	Perennial
<i>Prunella vulgaris</i>	Ferns and Forbs	Native	Perennial
<i>Prunus americana</i>	Tree	Native	Perennial
<i>Prunus pensylvanica var. pensylvanica</i>	Tree	Native	Perennial
<i>Prunus serotina</i>	Tree	Native	Perennial

<i>Pteridium aquilinum</i>	Ferns and Forbs	Native Invasive	Perennial
<i>Pycnanthemum incanum</i>	Ferns and Forbs	Native	Perennial
<i>Quercus alba</i>	Tree	Native	Perennial
<i>Quercus coccinea</i> var. <i>coccinea</i>	Tree	Native	Perennial
<i>Quercus ilicifolia</i>	Shrubs and Vines	Native	Perennial
<i>Quercus marilandica</i> var. <i>marilandica</i>	Shrubs and Vines	Native	Perennial
<i>Quercus prinus</i>	Tree	Native	Perennial
<i>Quercus rubra</i>	Tree	Native	Perennial
<i>Quercus velutina</i>	Tree	Native	Perennial
<i>Rhus glabra</i>	Tree	Native Weed	Perennial
<i>Rhus typhina</i>	Tree	Native Weed	Perennial
<i>Robinia pseudoacacia</i>	Tree	Native Invasive	Perennial
<i>Rosa carolina</i> var. <i>carolina</i>	Shrubs and Vines	Native	Perennial
<i>Rubus occidentalis</i>	Shrubs and Vines	Native Weed	Perennial
<i>Rubus</i> spp.	Shrubs and Vines	Native Weed	Perennial
<i>Rumex acetosella</i>	Ferns and Forbs	Exotic Invasive	Perennial
<i>Rumex</i> cf. <i>obtusifolius</i>	Ferns and Forbs	Exotic	Perennial
<i>Sassafras albidum</i>	Tree	Native	Perennial
<i>Saxifraga caroliniana</i>	Ferns and Forbs	Native	Perennial
<i>Scutellaria ovata</i>	Ferns and Forbs	Native	Perennial
<i>Sedum ternatum/glaucophyllum</i>	Ferns and Forbs	Native	Perennial
<i>Silene stellata</i>	Ferns and Forbs	Native	Annual
<i>Smilax glauca</i>	Shrubs and Vines	Native	Perennial
<i>Smilax rotundifolia</i>	Shrubs and Vines	Native	Perennial
<i>Solidago bicolor</i>	Ferns and Forbs	Native	Annual
<i>Solidago caesia</i>	Ferns and Forbs	Native	Perennial
<i>Solidago</i> cf. <i>curtisii</i>	Ferns and Forbs	Native	Perennial
<i>Solidago</i> cf. <i>nemoralis</i>	Ferns and Forbs	Native	Perennial
<i>Spiraea betulifolia</i> var. <i>corymbosa</i>	Shrubs and Vines	Native	Perennial
<i>Symphytotrichum</i> cf. <i>cordifolium</i> / <i>Eurybia</i> cf. <i>schreberi</i>	Ferns and Forbs	Native	Perennial
<i>Symphytotrichum</i> cf. <i>divaricatum</i>	Ferns and Forbs	Native	Perennial
<i>Symphytotrichum divaricatum</i>	Ferns and Forbs	Native	Perennial
<i>Symphytotrichum praealtum</i>	Ferns and Forbs	Native	Annual
<i>Symphytotrichum undulatum</i>	Ferns and Forbs	Native	Perennial
<i>Taenidia integerrima</i>	Ferns and Forbs	Native	Perennial
<i>Tephrosia virginiana</i>	Ferns and Forbs	Native	Perennial
<i>Thalictrum thalictroides</i>	Ferns and Forbs	Native	Annual
<i>Thelypteris noveboracensis</i>	Ferns and Forbs	Native	Perennial
<i>Toxicodendron radicans</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Triodanis perfoliata</i> var. <i>perfoliata</i>	Ferns and Forbs	Native	Perennial
UK Asteraceae	Ferns and Forbs	Native?	Perennial
UK Asteraceae1	Ferns and Forbs	Native?	Perennial
UK Asteraceae2	Ferns and Forbs	Native?	Perennial

<i>UK Asteraceae</i> ³	Ferns and Forbs	Native?	Perennial
<i>UK Dicot</i>	Ferns and Forbs?	Native?	Perennial
<i>UK Dicot1</i>	Ferns and Forbs?	Native?	Perennial
<i>UK Poaceae</i>	Graminoid	Native?	Perennial
<i>Ulmus rubra</i>	Tree	Native	Perennial
<i>Uvularia perfoliata</i>	Ferns and Forbs	Native	Perennial
<i>Uvularia puberula</i>	Ferns and Forbs	Native	Perennial
<i>Vaccinium pallidum</i>	Shrubs and Vines	Native	Perennial
<i>Vaccinium stamineum</i>	Shrubs and Vines	Native	Perennial
<i>Verbascum thapsus</i>	Ferns and Forbs	Exotic	Biennial
<i>Veronica officinalis</i>	Ferns and Forbs	Exotic	Perennial
<i>Viburnum acerifolium</i>	Shrubs and Vines	Native	Perennial
<i>Viburnum prunifolium</i>	Tree	Native	Perennial
<i>Vicia caroliniana</i>	Shrubs and Vines	Native	Perennial
<i>Vicia cf. villosa</i>	Shrubs and Vines	Exotic	Annual
<i>Viola cf. palmata</i>	Ferns and Forbs	Native	Perennial
<i>Viola cf. sororia</i>	Ferns and Forbs	Native	Perennial
<i>Viola palmata</i>	Ferns and Forbs	Native	Perennial
<i>Viola pedata</i>	Ferns and Forbs	Native	Perennial
<i>Vitis aestivalis var. bicolor</i>	Shrubs and Vines	Native Invasive	Perennial
<i>Vitis vulpina</i>	Shrubs and Vines	Native Invasive	Perennial

Table 4. Pearson correlations for each understory species with axes 2 and 3 of the NMS ordination of combined pre- and post-fire abundance data. UK = unknown. Exotic invasives are shown in bold.

Species	R ² = 0.184	R ² = 0.178
	Axis 2	Axis 3
<i>Acer pensylvanicum</i>	0.535	0.39
<i>Acer rubrum</i> var. <i>rubrum</i>	0.153	-0.028
<i>Ageratina altissima</i> var. <i>altissima</i>	0.627	-0.048
<i>Alliaria petiolata</i>	0.446	0.136
<i>Amelanchier arborea</i> var. <i>arborea</i>	-0.006	-0.093
<i>Amphicarpaea bracteata</i>	0.311	-0.059
<i>Antennaria neglecta</i>	-0.092	-0.409
<i>Antennaria plantaginifolia</i>	0.028	-0.073
<i>Antennaria virginica</i>	0.067	-0.092
<i>Asplenium platyneuron</i>	0.442	-0.122
<i>Carex cephalophora</i>	0.459	-0.204
<i>Carex</i> cf. <i>pensylvanica/lucorum</i>	-0.029	0.207
<i>Carex</i> cf. <i>pensylvanica/lucorum/tonsa</i>	-0.108	-0.247
<i>Carex digitalis</i>	0.091	0.068
<i>Carya glabra</i>	0.235	-0.352
<i>Carex laxiflora</i>	0.418	0.03
<i>Carya ovata</i>	-0.077	0.292
<i>Carex pensylvanica</i>	0.144	-0.388
<i>Carex</i> spp.	0.241	0.025
Cf. UK Asteraceae	0.219	0.204
<i>Chimaphila maculata</i>	-0.069	-0.059
<i>Cratageous</i> spp.	0.242	-0.182
<i>Danthonia spicata</i>	-0.218	-0.378
<i>Deschampsia flexuosa</i>	0.012	-0.285
<i>Dennstaedtia punctilobula</i>	0.365	-0.025
<i>Dichanthelium boscii</i>	0.123	-0.102
<i>Dichanthelium commutatum</i>	0.095	-0.364
<i>Dichanthelium depauperatum</i>	-0.059	-0.311
<i>Dichanthelium latifolium</i>	0.15	-0.131
<i>Dioscorea villosa</i>	-0.016	-0.009
<i>Dryopteris marginalis</i>	0.382	0.109
<i>Erechtites hieraciifolia</i>	0.214	0.246
<i>Festuca subverticillata</i>	0.187	-0.069
<i>Gaylussacia baccata</i>	-0.239	0.225
<i>Galium circaezans</i> var. <i>circaezans</i>	0.516	-0.06
<i>Hamamelis virginiana</i>	0.112	0.301

<i>Heuchera americana</i>	0.031	-0.118
<i>Hepatica nobilis var. obtusa</i>	0.152	-0.088
<i>Hedeoma pulegioides</i>	0.062	-0.054
<i>Houstonia longifolia</i>	0.229	-0.498
<i>Kalmia latifolia</i>	-0.313	0.226
<i>Nyssa sylvatica</i>	-0.025	0.122
<i>Ostrya virginiana var. virginiana</i>	0.501	0.083
<i>Paronychia canadensis</i>	0.355	0.095
<i>Parietaria pensylvanica</i>	0.542	0.262
<i>Parthenocissus quinquefolia</i>	0.189	0.146
<i>Pinus spp.</i>	-0.033	0.201
<i>Pinus strobus</i>	0.17	0.035
<i>Pinus virginiana</i>	-0.255	-0.335
<i>Polygonatum biflorum</i>	-0.209	-0.155
<i>Polygonum cf. convolvulus</i>	0.523	0.103
<i>Poa compressa</i>	0.216	-0.105
<i>Potentilla simplex</i>	-0.013	-0.61
<i>Prunus serotina</i>	0.233	0.073
<i>Quercus ilicifolia</i>	-0.456	-0.056
<i>Quercus prinus</i>	-0.057	-0.27
<i>Quercus rubra</i>	0.001	-0.268
<i>Quercus velutina</i>	-0.153	-0.224
<i>Rosa carolina var. carolina</i>	-0.202	-0.167
<i>Robinia pseudoacacia</i>	0.083	-0.245
<i>Sassafras albidum</i>	0.332	0.183
<i>Saxifraga caroliniana</i>	0.454	-0.115
<i>Sedum ternatum/glaucophyllum</i>	0.145	-0.272
<i>Smilax rotundifolia</i>	-0.09	-0.176
<i>Solidago caesia</i>	0.276	-0.217
<i>Symphyotrichum divaricatum</i>	0.241	-0.018
<i>Thalictrum thalictroides</i>	0.286	-0.083
UK Asteraceae	0.201	-0.065
UK Dicot	0.131	-0.05
UK Poaceae	-0.049	0.238
<i>Uvularia perfoliata</i>	0.204	0.042
<i>Uvularia puberula</i>	-0.063	0.027
<i>Vaccinium pallidum</i>	-0.347	-0.451
<i>Vaccinium stamineum</i>	-0.042	-0.196
<i>Viburnum acerifolium</i>	-0.216	0.079
<i>Vitis aestivalis var. bicolor</i>	0.208	0.007
<i>Vicia caroliniana</i>	0.375	-0.068
<i>Viola cf. palmata</i>	0.044	-0.245
<i>Viola palmata</i>	0.462	-0.045

Vitis spp.

-0.083

0.378

Table 5. Pearson correlations for each understory species with axes 1 and 2 of the NMS ordination of "smoothed" post-fire presence/absence data. UK = unknown. Exotic invasives are shown in bold.

Species	R ² =	R ² =
	Axis 1	Axis 2
<i>Acer pensylvanicum</i>	0.880	-0.507
<i>Acer rubrum</i> var. <i>rubrum</i>	-0.590	0.017
<i>Acer saccharum</i> var. <i>saccharum</i>	0.584	-0.629
<i>Adiantum pedatum</i>	0.362	-0.719
<i>Ageratina altissima</i> var. <i>altissima</i>	0.922	-0.411
<i>Ailanthus altissima</i>	0.745	-0.348
<i>Allium cernuum</i>	0.266	-0.065
<i>Alliaria petiolata</i>	0.855	-0.555
<i>Amelanchier arborea</i> var. <i>arborea</i>	-0.479	0.822
<i>Ambrosia artemisiifolia</i>	0.821	-0.164
<i>Amphicarpaea bracteata</i>	0.945	-0.189
<i>Antennaria neglecta</i>	-0.059	0.902
<i>Antennaria plantaginifolia</i>	0.169	0.631
<i>Antennaria virginica</i>	0.377	0.476
<i>Aristolochia serpentaria</i>	0.620	-0.495
<i>Asplenium platyneuron</i>	0.889	-0.423
<i>Asclepias quadrifolia</i>	0.824	0.156
<i>Aster</i> spp.	0.326	0.081
<i>Botrychium virginianum</i>	0.434	-0.534
<i>Brachyletrum erectum</i>	0.828	-0.325
<i>Carya alba</i>	0.373	0.637
<i>Carex cephalophora</i>	0.835	0.010
<i>Carex</i> cf. <i>pensylvanica</i>	0.499	0.479
<i>Carex</i> cf. <i>pensylvanica/lucorum</i>	0.598	-0.259
<i>Carex</i> cf. <i>pensylvanica/lucorum/tonsa</i>	-0.345	-0.213
<i>Campanula divaricata</i>	0.634	-0.198
<i>Carex digitalis</i>	0.056	-0.026
<i>Carya glabra</i>	-0.710	0.544
<i>Carex laxiflora</i>	0.923	-0.383
<i>Carex pensylvanica</i>	0.238	0.024
<i>Carex</i> spp.	0.407	-0.602
<i>Ceanothus americanus</i>	0.478	0.316
Cf. <i>Aureolaria laevigata</i>	0.511	-0.229
Cf. <i>Bromus ciliatus/Calamagrostis porteri</i>	0.028	-0.408
Cf. <i>Heuchera americana</i>	0.230	-0.125

<i>Cf. Hieracium spp.</i>	0.130	0.592
<i>Cf. Lactuca spp.</i>	0.158	-0.480
<i>Cf. Lysimachia quadrifolia</i>	-0.053	-0.558
<i>Cf. Phlox buckleyi</i>	0.226	0.520
<i>Cf. Prenanthes spp.</i>	0.408	-0.320
<i>Cf. Scrophularia lanceolata</i>	0.841	0.080
<i>Cf. Sericocarpus linifolius</i>	0.259	0.444
<i>Cf. UK Asteraceae</i>	-0.056	0.249
<i>Chimaphila maculata</i>	0.060	0.421
<i>Conopholis americana</i>	-0.145	-0.093
<i>Cornus florida</i>	-0.218	-0.456
<i>Corydalis flavula</i>	0.616	-0.671
<i>Crataegus spp.</i>	0.440	0.516
<i>Cunila origanoides</i>	0.253	-0.276
<i>Daucus carota</i>	0.299	0.013
<i>Danthonia spp.</i>	0.428	-0.002
<i>Danthonia spicata</i>	-0.214	0.675
<i>Deschampsia flexuosa</i>	-0.160	0.592
<i>Desmodium nudiflorum</i>	-0.221	-0.084
<i>Dennstaedtia punctilobula</i>	0.682	-0.739
<i>Desmodium spp.</i>	0.385	-0.625
<i>Dichanthelium boscii</i>	0.525	-0.105
<i>Dichanthelium commutatum</i>	-0.444	-0.135
<i>Dichanthelium depauperatum</i>	-0.809	0.152
<i>Dichanthelium latifolium</i>	0.278	-0.391
<i>Dioscorea villosa</i>	0.741	-0.621
<i>Dryopteris marginalis</i>	0.856	-0.468
<i>Draba ramosissima</i>	0.507	0.469
<i>Elymus hystrix var. hystrix</i>	0.545	0.302
<i>Epigaea repens</i>	-0.840	0.151
<i>Erigeron strigosus</i>	0.220	0.310
<i>Festuca subverticillata</i>	0.729	-0.416
<i>Fraxinus americana</i>	0.470	-0.175
<i>Gaylussacia baccata</i>	-0.878	-0.069
<i>Galium circaezans var. circaezans</i>	0.980	-0.342
<i>Galium lanceolatum</i>	0.857	-0.295
<i>Gaultheria procumbens</i>	-0.759	-0.188
<i>Galium triflorum</i>	0.549	-0.128
<i>Goodyera spp.</i>	-0.060	-0.554
<i>GRubus spp.</i>	-0.481	0.291
<i>Hamamelis virginiana</i>	-0.582	-0.505
<i>Helianthus divaricatus</i>	0.657	0.293
<i>Heuchera americana</i>	0.578	-0.230

<i>Hepatica nobilis</i> var. <i>obtusata</i>	0.526	-0.412
<i>Hedeoma pulegioides</i>	0.726	0.138
<i>Hieracium caespitosum</i>	-0.238	0.363
<i>Hieracium pilosella</i> var. <i>pilosella</i>	-0.548	0.820
<i>Hieracium venosum</i>	-0.496	0.313
<i>Houstonia</i> cf. <i>longifolia</i>	0.438	0.610
<i>Iris verna</i>	-0.699	0.543
<i>Kalmia latifolia</i>	-0.877	-0.032
<i>Lespedeza</i> cf. <i>violacea</i>	0.294	0.348
<i>Lespedeza hirta</i> ssp. <i>hirta</i>	0.489	0.372
<i>Lespedeza procumbens</i>	0.605	0.365
<i>Lespedeza repens</i>	-0.481	0.291
<i>Maianthemum racemosum</i> ssp. <i>racemosum</i>	0.101	0.098
<i>Melampyrum lineare</i>	-0.747	0.346
<i>Microstegium vimineum</i>	0.478	-0.584
<i>Monotropa uniflora</i>	-0.004	-0.130
<i>Nyssa sylvatica</i>	-0.794	-0.224
<i>Ostrya virginiana</i> var. <i>virginiana</i>	0.546	-0.619
<i>Paronychia canadensis</i>	0.854	-0.363
<i>Paronychia fastigiata/montana</i>	0.320	0.564
<i>Parietaria pennsylvanica</i>	0.740	-0.659
<i>Parthenocissus quinquefolia</i>	0.689	-0.710
<i>Penstemon canescens</i>	-0.334	0.653
<i>Pinus pungens</i>	-0.697	0.508
<i>Pinus rigida</i>	-0.831	0.106
<i>Pinus strobus</i>	-0.366	0.154
<i>Pinus virginiana</i>	-0.640	0.802
<i>Pinus virginiana/pungens</i>	-0.319	-0.184
<i>Polystichum acrostichoides</i>	-0.080	-0.615
<i>Polygonatum biflorum</i>	0.390	0.665
<i>Polygonum</i> cf. <i>convolvulus</i>	0.942	-0.191
<i>Poa compressa</i>	0.420	0.390
<i>Polygala senega</i>	0.762	-0.474
<i>Potentilla simplex</i>	-0.153	0.882
<i>Prunus serotina</i>	-0.011	0.523
<i>Pteridium aquilinum</i>	-0.749	-0.226
<i>Pycnanthemum incanum</i>	0.644	-0.358
<i>Quercus alba</i>	-0.779	0.423
<i>Quercus coccinea</i> var. <i>coccinea</i>	-0.894	0.290
<i>Quercus ilicifolia</i>	-0.907	0.184
<i>Quercus marilandica</i> var. <i>marilandica</i>	-0.045	0.601
<i>Quercus prinus</i>	-0.138	0.400
<i>Quercus velutina</i>	-0.866	0.594

<i>Rhus typhina</i>	-0.472	0.287
<i>Rosa carolina</i> var. <i>carolina</i>	0.101	0.659
<i>Robinia pseudoacacia</i>	0.729	0.110
<i>Rubus occidentalis</i>	0.318	-0.774
<i>Rubus</i> spp.	-0.119	-0.675
<i>Sassafras albidum</i>	-0.388	-0.490
<i>Saxifraga caroliniana</i>	0.947	-0.171
<i>Sedum ternatum/glaucophyllum</i>	0.458	0.553
<i>Silene stellata</i>	0.743	-0.107
<i>Smilax rotundifolia</i>	-0.823	0.169
<i>Solidago caesia</i>	0.848	-0.337
<i>Solidago</i> cf. <i>nemoralis</i>	0.014	0.187
<i>Spiraea betulifolia</i> var. <i>corymbosa</i>	0.126	0.026
<i>Symphyotrichum</i> cf. <i>divaricatum</i>	0.454	-0.376
<i>Symphyotrichum divaricatum</i>	0.499	0.328
<i>Symphyotrichum undulatum</i>	0.657	0.306
<i>Taenidia integerrima</i>	0.648	-0.048
<i>Tephrosia virginiana</i>	-0.466	0.487
<i>Thalictrum thalictroides</i>	0.773	-0.184
<i>Toxicodendron radicans</i>	0.338	0.010
<i>Triodanis perfoliata</i> var. <i>perfoliata</i>	0.502	0.463
UK Asteraceae	-0.089	0.807
UK Asteraceae1	0.005	0.515
UK Dicot	0.464	0.207
UK Poaceae	-0.720	0.103
<i>Ulmus rubra</i>	0.434	-0.534
<i>Uvularia perfoliata</i>	0.798	-0.700
<i>Uvularia puberula</i>	-0.700	-0.358
<i>Vaccinium pallidum</i>	-0.583	0.782
<i>Vaccinium stamineum</i>	-0.684	0.766
<i>Verbascum thapsus</i>	0.438	0.431
<i>Viburnum acerifolium</i>	-0.497	-0.420
<i>Vitis aestivalis</i> var. <i>bicolor</i>	0.382	0.091
<i>Vicia caroliniana</i>	0.811	-0.236
<i>Viola</i> cf. <i>palmata</i>	0.032	0.527
<i>Viola palmata</i>	0.963	-0.310
<i>Viola pedata</i>	-0.842	0.404
<i>Vitis vulpina</i>	-0.270	-0.108