Forest dynamics of pine- and oak-dominated communities on southeastern-facing slopes of Warm Springs Mountain, Virginia

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ABSTRACT

Warm Springs Mountain (WSM), a priority conservation area for The Nature Conservancy in Bath County, Virginia, is home to a rare montane pine barren and large tracts of uninterrupted mixed pine and deciduous forest extending east into the George Washington National Forest. Limited documentation of past disturbances and their influence on WSM forests presents challenges for land managers desiring to understand historic conditions for these ecosystems. The only formal study of vegetation dynamics on WSM noted an absence of pitch pine (*Pinus rigida* Mill.) regeneration and an increase in fire-intolerant species during recent decades in the pine barren community that is probably linked to fire suppression. Dendrochronological studies of disturbance history in the central and southern Appalachians have mostly focused on ridgetop and southwestern-facing slopes. This study examines long-term forest dynamics in the pine- and oak-dominated forests on southeastern slopes of Warm Springs Mountain and downslope from the higher elevation pine barren using dendrochronology and vegetation analysis. We studied trees in six 20 x 50 m plots to develop a tree ring chronology and document changes in stand composition and structure through time. We found an increase in fire-intolerant species and decline in fire-dependent pines and oaks through time. Pitch pines have not recruited since 1954 in our sites due to a lack of burning, while *Acer rubrum* L. has produced high numbers of seedlings in recent years. This study of vegetation dynamics over space and time will provide insights for land managers and inform fire restoration practices.

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Chapter 1: Introduction

1.1. Problem Statement and Objective

Natural and anthropogenic disturbances are major drivers of vegetation dynamics in the southern Appalachians (Brose et al. 2001). Climate, pathogens, land use patterns, and altered fire regimes are among the most important in changing or maintaining the forests of this region (Williams 1998). These disturbances can drastically alter species composition, especially when multiple cycles are disrupted simultaneously.

Fire, for example, has been an important disturbance in eastern deciduous forests for millennia (Lorimer 1985, Abrams and Nowacki 1992, Fesenmeyer and Christensen 2010). Although lightning has been known to cause fire in the southern Appalachians (Barden and Woods 1976), anthropogenic fires are much more common (Lafon et al. 2005). Fires were ignited by Native Americans, who burned forests for increased mast production, game trails, and agricultural land (Pyne 1983, Rufner and Abrams 2002). Later, European colonization brought about higher populations that continued burning for similar reasons. Fire became more common when commercial logging and charcoal production became a frequent practice in the eastern United States during the Industrial Revolution (Rufner and Abrams 2002).

The southern Appalachians have undergone major changes since European settlers first began using the forests (Brose et al. 2001). Not only has the long dominant American chestnut (*Castanea dentata* L.) been removed by the chestnut blight (*Cryphonectricia parasitica*), but extensive fire suppression has also had major impacts on forest ecosystems of the eastern United States (Elliot and Swank 2008). Some southern Appalachian forest canopies were estimated to contain 40–45% *C. dentata* before the blight (Reed 1905, Keever 1953). Another human disturbance, heavy commercial logging, coincided with the chestnut decline to dramatically change the forests of the southern Appalachians (Burke 2011, Aldrich 2010). Some areas have extensive records regarding logging activity and other land use changes (Guyette and Spetich 2003, Signell et al. 2005), while others, especially remote areas, are less well known.

Later, the U.S. government began actively suppressing fire because of proximity to settlements, towns, and factories. Management practices and land fragmentation effectively removed fire from the landscape by the early 20th century; prior to fire suppression by the U.S. Forest Service, many forests would have looked dramatically different than today's forests (Brose et al. 2001). Many parts of the southern Appalachians have never been heavily populated but a mountainous landscape, land fragmentation, and fire suppression since the 1930s underlie a complex change in vegetation that has not yet been well documented.

Altered fire regimes and climatic changes influence vegetation and are further complicated by increased fuel loads, land use changes, and topography (Lafon et al. 2005, Flatley et al. 2010). Other regions have experienced changes in vegetation composition and structure, such as the longleaf pine savannas of the southeastern coastal plains, which have been converted to closed canopy forests with increased deciduous components and understory density under the current fire suppression regime (Frost 1993). In the southern Appalachians, complex topography is a prominent factor affecting both climatic and fire patterns and needs more study.

Our study area on Warm Springs Mountain (WSM), located in Bath County, Virginia, is among the largest and most biologically significant forest blocks in the southern Appalachians (Thorne and Anderson 2003). Over 3600 hectares of the mountain has been owned by The Nature Conservancy (TNC) since 2002 and managed as the non-profit conservation organization's largest nature preserve in Virginia. The Warm Springs Mountain Preserve shares an eastern border 21 km in length with the George Washington National Forest and effectively provides an 8000 ha conservation area collaboratively managed as a public-private partnership. Only scant information exists about vegetation change for the higher elevations at WSM. The pine barrens on the ridgetop flats are the only expression of this vegetation type in Virginia and one of only a few in the world. Warm Springs Mountain is located in the southern Appalachian region and the delicate pine barrens on its flattened peaks have been described as strongly influenced by fire, in terms of stand structure and composition (Powers 2010). Forests downslope from the pine barrens of this mountainous region consist of a mosaic of community types, with pines and oaks as important canopy components. Fires have been suppressed in the WSM pine barrens for at least 80 years (Marek Smith, personal communication) and the pre-suppression fire regime on the lower slopes, especially southeastern aspects, is unknown. Aspect may alter the return interval due to changes in vegetation dynamics, sun exposure, and other environmental variables on different facing slopes. Most work on fire history in the region has been carried out on southwestern slopes, which are driest and considered most likely to burn. We were interested in slightly more mesic southeastern slopes and the long-term vegetation change associated with this aspect at WSM. We examined the forest dynamics over the past century and a half in the lower oak forest matrix on southeastern-facing slopes below the montane pine barren. Documentation of longer-term forest changes can point to possible changes in fire return interval and other disturbances that can be compared with studies of the pine barren upslope to better understand this complex montane ecosystem. The role of fire in originating and maintaining these communities and their boundaries and spatial distribution is not well understood.

Reconstructing composition and stand structure under varying human-induced and natural disturbances can shed light on long-term successional trends and inform management practices. Aldrich et al. 2010 described the return interval on southwest slopes of nearby Mill Mountain, Virginia (about 15 km from WSM) as every five years before 1930, and every 16 years since anthropogenic fire control regimes were put into place. Only a few areas of southern Appalachian Virginia have well known fire histories; most are less understood (Hessl et al. 2011). In addition, logging has been an important forest disturbance, but its impacts are spatially and temporally variable and often not well documented through historical or other means (Burke 2011).

The land managers at WSM, in this case The Nature Conservancy and the U.S. Forest Service, are interested in management implications related to fire ecology and how fire affects species composition and forest structure of southeastern slopes in the oak/pine forests downslope from pine barren. Fire-dependent and fire-tolerant species are abundant in the area, but fire-intolerant trees are also present. Fuel loads have increased with the lack of fire, and stands containing large amounts of coarse woody debris can ignite causing very large, stand replacing, crown fires (Brose et al. 2001). Although humans have inhabited the region for centuries, the steep, remote, mountainous terrain has limited both timber management and scientific research. Spatial and temporal analysis can help to uncover vegetation patterns that relate to fire and other disturbances.

Dendrochronology can provide annual resolution data on tree ages and can be implemented to understand fire-vegetation interaction and tree response to other disturbances across the landscape. In this study, tree-ring data is combined with vegetation analysis and location data (from a global positioning system) to provide a high resolution and long-term view of vegetation dynamics. The main objective of this research is to document the spatial and temporal variation in vegetation composition and structure on southeastern-facing slopes along a gradient from higher elevations below the ridgeline to the toe of slope of Warm Springs Mountain and relate forest dynamics to disturbance history as best possible, including the impacts of changing fire regimes.

A brief review of some relevant literature follows this introduction and Chapter 2 presents the study in the form of a manuscript in preparation for submission to a journal.

1.2. Literature Review

1.2.1. Fire and Forests

Many factors contribute to vegetation dynamics and must be considered in any examination of changes in patterns over time. Weather patterns and forest composition are complex phenomena that need to be documented before understanding how fire regimes changes over space and time (Bergeron et al. 2002). Abrams and Copenheaver (1999) examined succession in complex forest ecosystems in the northern Virginia Piedmont to assess the relationship between species composition and disturbance events like fire, insect infestation, and anthropogenic disturbances like coal mining and charcoal production. They noted, along with others (Delcourt et al. 1986, Brose et al. 2001) that the eastern United States has a long history of anthropogenic disturbance from both Native Americans and European settlers. Their study used dendrochronology to make inferences about the ecological history of the site over the last few hundred years. The loss of *C. dentata*, and the advent of fire suppression in the southern Appalachians caused growth in some old oak species (*Quercus*) species and they speculated that the lack of fire has allowed these late-successional species to advance into oak/pine forests (Abrams and Copenheaver 1999).

Other research has shown that reintroducing fire into stands that have been suppressed for long periods of time does not always lead to an increase in fire-dependent species or a decrease in fire-intolerant species (Alexander et al. 2008). Alexander et al. (2008) focused on oak/fire relationships in Daniel Boone National Forest in eastern Kentucky. They hypothesized that adding fire to stands long suppressed should allow oak and pine species to regenerate by removing aggressive colonizing species such as Tulip-poplar (*Liriodendron tulipifera* L.) (Alexander et al. 2008). The authors attempted to establish a fire regime over a six year period. Fire was introduced three times at one site, once at another site, and suppressed at a control site. Oak species did not take advantage of the open seedbed and species like sassafras (*Sassafras albidum* Nutt.), and red maple (*Acer rubrum* L.), recruited heavily. Maple species showed a greater ability to adapt to short term changes in light and moisture availability (Alexander et al. 2008). The authors mention that a longer study would be better to capture the ability of oak species to survive in harsh climates due to resources being put into belowground biomass as opposed to height. White oaks (*Quercus alba* L.) showed a tendency to put resources underground and displayed the ability to regenerate better when fire was introduced after canopy releases (Alexander et al. 2008).

A study by Bergeron et al. (2002) focused on several disturbance regimes present in complex forest ecosystems at Lake Duparquet Research and Teaching Forest in northwestern Quebec. The authors took cross sections of several boreal species to determine insect infestation patterns and related those to climate patterns using tree rings. They described fire as the most important stand level disturbance. The second most important factor to stand disturbance regimes was insect infestation. The authors used the cross sections not only for tree ring dating, but to look for damage from insects. By using tree rings to date the infestations of species like the Gypsy moth, *Lymantria dispar*, more accurate representations of forest dynamics were represented (Bergeron et al. 2002).

Dendrochronology is a tool useful in reconstructing climate (Lafon et al. 2005), but the potential for its use in reconstructing fire history has not been fully realized in eastern forests due to wood decay in more humid climates and land fragmentation (Shumway et al. 2001). Fire scarred trees can be studied through dendrochronology to determine the return interval of fire at a very fine scale (Everett 2008). A western study used cross sections of 33 fire scarred trees to develop a fire history in the San Jacinto Mountains of southern California. Since all fires do not scar all trees and many species display a high rate of mortality when exposed to fire, simply walking around looking for fire scars was impractical. The authors used GIS to determine the most likely locations (slope and aspect) of fire scarred trees. Once the trees were designated as fire scarred, they were cut and cross sections were taken for analysis. The study was able to construct a 653 year history with 126 total fire scarred cross sections. Research of this type is common in California (Taylor and Skinner 1998) and many other parts of the west (Grissino-Mayer and Swetnam 2000, Knapp et al. 2002, Everett 2008) but more studies are needed in the eastern United States. New research will add to the existing spatial and temporal knowledge of the fire history of the southern Appalachian forest. There are challenges in the eastern U.S. including a lack of old trees and fewer incidences of fire scars.

1.2.2. Anthropogenic Disturbance in the Southern Appalachians

Brose et al. (2001) developed a broad scale history of the Appalachian region in terms of fire regime changes. The authors did not develop a history of the Appalachians prior to European settlement but the three major shifts in fire regime adequately summarize fire in the region. The first change occurred with the arrival of humans, Native Americans, who began using fire to manipulate forests in order to gain hunting grounds, masts, and agricultural land (Delcourt et al. 1986). The second regime change occurred with the introduction of large scale commercial logging practices beginning during the Industrial Revolution (Brose et al. 2001). The last major regime change, according to the authors, came with the idea that fire was unimportant to the forest ecosystem. Fire suppression led to the advance of fire-intolerant species and the retreat of ecosystems like pine savannahs. Many of these forests have been without fire for close to a century and future forest dynamics are difficult to predict. Oak/hickory forests provide abundant resources to both the forest biota and humans. Since fire suppression, forests have been increasingly composed of thick understory species that have made regeneration difficult for species not equipped for this type of change (Abrams and Copenheaver 1999, Brose et al. 2001). Many have seen prescribed fire as the savior of Appalachian forests, but the land has been so fragmented by human settlements, roads, and different land uses that large scale fire management changes could be a difficult undertaking (Brose et al. 2001).

The relationship between human presence and fire has been documented in the Lower Boston Mountains of Arkansas (Guyette and Spetich 2003). This study took place in a mixed pine/oak forest with well documented land use changes. The main objective of the study was to understand how population density relates to fire intervals and fire regime changes. Historical accounts of the area such as logging records and Native American interviews were incorporated to further develop the history of the forest. State and county population records for both European settlers and Native Americans gave the authors a picture of how the area might have looked in terms of density. They used tree stumps and trees with visible fire scars to develop a dendrochronological record for the area. As previously mentioned, fire scars can be difficult to find, but for this study, data collection in the area was concentrated in areas with a history of commercial logging. Forty-five (45) shortleaf pine (*Pinus echinada* Mill.) stumps were dated using accepted dendrochronology techniques (Stokes and Smiley, 1996). This project did not develop a chronology past the early portion of the 19th century. Additional studies are needed to piece together the relationship between fire regimes to include the modern suppression era. Their conclusion stated that population density directly affected fire-return intervals. Periods of high population density were characterized as having longer fire return intervals and periods of low population density displayed more rapid fire occurrence (Guyette and Spetich 2003).

Another study compared the size and extent of anthropological fires to size and extent of naturally occurring fires in the southern Appalachians (Lafon et al. 2005). They used aerial photographs from 1970–2003 to look at over 1800 fires to determine how flame height, fire intensity, ignition source, and ignition density contribute to the overall dynamics of forest fires. Natural fires in their study only burned 5362 ha. Most fires in Eastern forests are small, surface fires (Pyne 1982) but a few large fires did most of the damage burning over 50% of the total area burned. Large areas in the study did not burn for many years at a time. Seasonality had an influence on fire return where late summer and fall displayed the highest number of fires due to decreased precipitation and dry material on the forest floor. Spring also brought fire into the forest as changing winds and increased lightning activity added to the fuel load that had dried out during the winter. The Palmer Drought Severity Index (PDSI) and Palmer Hydrological Drought Index (PHDI) showed the correlation between fire and periods of drought. The climate variable showed a strong correlation with regard to drought but the varied landscape made regional prediction impossible (Lafon et al. 2005).

Powers (2010) examined a montane pine barren at Warm Springs Mountain. The vegetation dynamics present in this rare ecosystem are interesting to the present study since the location is upslope from our own sites. Two hypotheses were discussed on how these barrens are able to remain stable. The first discussed the necessity of frequent fire in order to maintain pine barrens of this type (Latham et al. 1996, Maurice et al. 2004). The second suggests that harsh environmental conditions, soil type, wind patterns, etc., are the main factor in the stability of pine barren ecosystems that lack consistent fire events (Motzkin et al. 2002). Powers used dendrochronology and soil charcoal methods were used to examine the fire history and vegetation dynamics in the pine barren. Dominant species on this site included *Pinus rigida* Mill. and *Quercus rubra* L. Cone serotiny is one of the adaptations *P. rigida* has developed living in the presence of fire. The Nature Conservancy was interested in the regeneration of *P. rigida* in the barren. They were also interested in developing management techniques that could be beneficial to maintaining the pine barren. The author found no *P. rigida* seedlings. Shade tolerant species, like *A. rubrum*, showed high numbers of recruitment in the understory. This study concluded that since fire suppression practices began in the 1930s, *P. rigida* has ceased to regenerate and shade-tolerant, fire-intolerant species have begun to take their place (Powers 2010).

The most relevant study to this research was conducted at nearby Mill Mountain, Virginia by Aldrich et al. (2010). Mill Mountain is about 15 kilometers from Warm Springs Mountain, the site of our study. Aldrich et al. (2010) documented the fire regime on Mill Mountain based on dendrochronological techniques to date fire scarred trees. Two hypotheses about fire history in the southern Appalachians were discussed in this study. The first stated that fire was infrequent before European settlement and fire-adapted species only existed on sites too poor for hardwoods to compete (Williams 1998). According to Williams (1998), fire-adapted species, like *P. rigida*, moved into lower elevations and more favorable sites after commercial logging practices began in the later part of the 19th century. The second discussed a “polycyclic” (Frost 1998) pattern of frequent fire in the southern Appalachians.

In order to examine the spatial extent of fire in this region, Aldrich et al. (2010) set up four 10 X 20 meter plots, located on the northwest side of the mountain. Fire scarred trees were sampled by taking cross sections and tree cores. *Pinus* species, including *P. rigida, P. pungens* Lamb., and *P. virginiana* Mill. were favored because of the ease of dating the samples compared to *Quercus* species (McEwan et al. 2007). Hardwood species in the study sites included *Nyssa sylvatica* Marsh., *Quercus montana* Willd., and *A. rubrum.* The seedlings and saplings were counted to gain a better understanding of regeneration on the site.

Aldrich et al. (2010) were able to construct a 299-year fire history from 209 scars on 63 individual trees. They concluded that surface fire was more frequent than the previously thought 7–10 year return interval, and that large stand replacing fires occurred at a return interval around 75 years, consistent with Frost (1998). Other study sites were mentioned (Hoss et al. 2008) as being different from this site because Mill Mountain is more remote. However, the authors believe as much fire existed on Mill Mountain even with the lowered chance for anthropogenic fire (Aldrich et al. 2010). Their study produced a relatively strong chronology for Eastern forests but they emphasized that more research needs to be conducted to fully understand the dynamics involved with fire in the southern Appalachians.

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Chapter 2: Manuscript

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2.1. Introduction

Natural and anthropogenic disturbances are major drivers of vegetation dynamics in the southern Appalachians (Brose et al. 2001). Climate, pathogens, land use patterns, and altered fire regimes are among the most important in changing or maintaining the forests of this region (Williams 1998). These disturbances can drastically alter species composition, especially when multiple cycles are disrupted simultaneously.

Fire, for example, has been an important disturbance in eastern deciduous forests for millennia (Lorimer 1985, Abrams and Nowacki 1992, Fesenmeyer and Christensen 2010). Although lightning has been known to cause fire in the southern Appalachians (Barden and Woods 1976), anthropogenic fires are much more common (Lafon et al. 2005). Fires were ignited by Native Americans, who burned forests for increased mast production, game trails, and agricultural land (Pyne 1983, Rufner and Abrams 2002). Later, European colonization brought about higher populations that continued burning for similar reasons. Fire became more common when commercial logging and charcoal production became a frequent practice in the eastern United States during the Industrial Revolution (Rufner and Abrams 2002).

The southern Appalachians have undergone major changes since European settlers first began using the forests (Brose et al. 2001). Not only has the long dominant American chestnut (*Castanea dentata* L.) been removed by the chestnut blight (*Cryphonectricia parasitica*), but extensive fire suppression has also had major impacts on forest ecosystems of the eastern United States (Elliot and Swank 2008). Some southern Appalachian forest canopies were estimated to contain 40–45% *C. dentata* before the blight (Reed 1905, Keever 1953). Another human disturbance, heavy commercial logging, coincided with the chestnut decline to dramatically change the forests of the southern Appalachians (Burke 2011, Aldrich 2010). Some areas have extensive records regarding logging activity and other land use changes (Guyette and Spetich 2003, Signell et al. 2005), while others, especially remote areas, are less well known.

Later, the U.S. government began actively suppressing fire because of proximity to settlements, towns, and factories. Management practices and land fragmentation effectively removed fire from the landscape by the early 20th century; prior to fire suppression by the U.S. Forest Service, many forests would have looked dramatically different than today's forests (Brose et al. 2001). Many parts of the southern Appalachians have never been heavily populated but a mountainous landscape, land fragmentation, and fire suppression since the 1930s underlie a complex change in vegetation that has not yet been well documented.

Altered fire regimes and climatic changes influence vegetation and are further complicated by increased fuel loads, land use changes, and topography (Lafon et al. 2005, Flatley et al. 2010). Other regions have experienced changes in vegetation composition and structure, such as the longleaf pine savannas of the southeastern coastal plains, which have been converted to closed canopy forests with increased deciduous components and understory density under the current fire suppression regime (Frost 1993). In the southern Appalachians, complex topography is a prominent factor affecting both climatic and fire patterns and needs more study.

Our study area on Warm Springs Mountain (WSM), located in Bath County, Virginia, is among the largest and most biologically significant forest blocks in the southern Appalachians (Thorne and Anderson 2003). Over 3600 hectares of the mountain has been owned by The Nature Conservancy (TNC) since 2002 and managed as the non-profit conservation organization's largest nature preserve in Virginia. The Warm Springs Mountain Preserve shares an eastern border 21 km in length with the George Washington National Forest and effectively provides an 8000 ha conservation area collaboratively managed as a public-private partnership.

Only scant information exists about vegetation change for the higher elevations at WSM. The pine barrens on the ridgetop flats are the only expression of this vegetation type in Virginia and one of only a few in the world. Warm Springs Mountain is located in the southern Appalachian region and the delicate pine barrens on its flattened peaks have been described as strongly influenced by fire, in terms of stand structure and composition (Powers 2010). Forests downslope from the pine barrens of this mountainous region consist of a mosaic of community types, with pines and oaks as important canopy components. Fires have been suppressed in the WSM pine barrens for at least 80 years (Marek Smith, personal communication) and the pre-suppression fire regime on the lower slopes, especially southeastern aspects, is unknown. Aspect may alter the return interval due to changes in vegetation dynamics, sun exposure, and other environmental variables on different facing slopes. Most work on fire history in the region has been carried out on southwestern slopes, which are driest and considered most likely to burn. We were interested in slightly more mesic southeastern slopes and the long-term vegetation change associated with this aspect at WSM. We examined the forest dynamics over the past century and a half in the lower oak forest matrix on southeastern-facing slopes below the montane pine barren. Documentation of longer-term forest changes can point to possible changes in fire return interval and other disturbances that can be compared with studies of the pine barren upslope to better understand this complex montane ecosystem. The role of fire in originating and maintaining these communities and their boundaries and spatial distribution is not well understood.

Reconstructing composition and stand structure under varying human-induced and natural disturbances can shed light on long-term successional trends and inform management practices. Aldrich et al. 2010 described the return interval on southwest slopes of nearby Mill

Mountain, Virginia (about 15 km from WSM) as every five years before 1930, and every 16 years since anthropogenic fire control regimes were put into place. Only a few areas of southern Appalachian Virginia have well known fire histories; most are less understood (Hessl et al. 2011). In addition, logging has been an important forest disturbance, but its impacts are spatially and temporally variable and often not well documented through historical or other means (Burke 2011).

The land managers at WSM, in this case The Nature Conservancy and the U.S. Forest Service, are interested in management implications related to fire ecology and how fire affects species composition and forest structure of southeastern slopes in the oak/pine forests downslope from pine barren. Fire-dependent and fire-tolerant species are abundant in the area, but fire-intolerant trees are also present. Fuel loads have increased with the lack of fire, and stands containing large amounts of coarse woody debris can ignite causing very large, stand replacing, crown fires (Brose et al. 2001). Although humans have inhabited the region for centuries, the steep, remote, mountainous terrain has limited both timber management and scientific research. Spatial and temporal analysis can help to uncover vegetation patterns that relate to fire and other disturbances.

Dendrochronology can provide annual resolution data on tree ages and can be implemented to understand fire-vegetation interaction and tree response to other disturbances across the landscape. In this study, tree-ring data is combined with vegetation analysis and location data (from a global positioning system) to provide a high resolution and long-term view of vegetation dynamics. The main objective of this research is to document the spatial and temporal variation in vegetation composition and structure on southeastern-facing slopes along a gradient from higher elevations below the ridgeline to the toe of slope of Warm Springs

Mountain and relate forest dynamics to disturbance history as best possible, including the impacts of changing fire regimes.

2.2. Methods

2.2.1. Study Area

The study site is located on Warm Springs Mountain (WSM, Figure 1) in Bath County, near the village of Hot Springs in west central Virginia. The site spans part of a 3751-ha nature preserve owned by The Nature Conservancy and part of the Warm Springs Ranger District of the George Washington National Forest. WSM is about 45 km in length, 21 km of which forms a common boundary between the Conservancy's largest preserve in the state and the national forest. Reaching 1288 m (Bald Knob) in elevation, WSM is part of the Ridge and Valley physiographic province, characterized by alternating linear ridges and valleys with a trellis drainage pattern and mountains formed by eroding streams carving out carbonate rock types (USGS 2012). The ridgeline of WSM is capped by relatively resistant Silurian sandstone and quartzite (USGS 2012).

The weather station for Hot Springs, Virginia is located approximately 3 km from the village at an elevation of 762 m on WSM. The average precipitation is 1087 mm annually, well distributed across months (NCDC 2012), and the mean monthly temperature ranges from -1°C in February to 22°C in July. Our sites are on the southeastern slopes of WSM about 5 km from Hot Springs and about 150 m higher in elevation. The forest matrix at WSM is characterized as oak-dominated with xeric pine-oak stands along the ridgetops and spur ridges. Narrow coves contain mesic species along the many intermittent and perennial streams. Shade intolerant species such as pitch pine (*Pinus rigida* Mill.), chestnut oak (*Quercus Montana* L.), and white oak (*Quercus* *alba* L.) dominate the upper slopes with *Q. montana*, scarlet oak *(Quercus coccinea* Munchh.), and red maple (*Acer rubrum* L.) on the lower, mesophytic slopes and valleys. *A. rubrum*, black gum (*Nyssa sylvatica*), sassafras (*Sassafras albidum* Nutt.), mountain laurel (*Kalmia latifolia* L.), and rhododendron (*Rhododendron catawbiense* Michx.) are common subcanopy components.

Native Americans inhabited the region before European settlement, mainly along the Cowpasture River to the east, but there is no evidence of large scale communities or agriculture (Geier et al. 1982). The land was granted to Robert Douthat in 1795 by the Commonwealth of Virginia as part of the Douthat Survey. Parts of WSM were owned by numerous parties throughout the 19th century and eventually purchased by the Douthat Land Company in 1926. In 1929, the property was purchased by the Ingalls family, owners of the Homestead Resort in Hot Springs, on the northwestern lower slopes of WSM. The U.S. Forest Service purchased a large area of the mountain in the 1930s (Marek Smith, personal communication). The Nature Conservancy has owned their portion of the mountain since 2002. A book about Virginia hot springs by the one of the Ingalls family members (Fay Ingalls) contains several mentions of fires in the area between 1929 and 1943 (Ingalls, 1949).

2.2.2. Plot Selection

We randomly established six 20 x 50 meter plots on southeastern slopes (aspects ranged between 90° and 180°) to document and assess vegetation characteristics and change on the southeastern slopes of WSM. We used a stratified random sampling system to select three higher elevation plots and three lower elevation plots (Table 1). Low elevation plots were placed at random elevations between 500 and 800 m. High elevation plots were chosen randomly between 900 and 1200 m. We recorded all plot points to include plot center, corners, and top and bottom boundary lines with a Trimble® ProXRT GPS unit.

2.2.3. Vegetation Sampling

We identified and measured tree diameters at breast height (DBH) for all individuals >5 cm DBH. DBH was defined as 1.4 m from the forest floor on the uphill side of the tree. We established a 20 x 20 m subplot within the larger plot (Figure 2). In the subplot we tallied seedlings and saplings of all tree species. Seedlings were defined as <1 meter in height and individuals >1 meter in height but less than 5 cm DBH were counted as saplings. All stems inside the subplot were cored once at breast height using increment bores, and all stems >20 cm DBH outside the subplot. Cores were taken from the stem with the largest DBH on multi-trunk stems. We took cores from each stem on both sides when possible. We took cross sections of remnant stumps and fire marked trees inside and near our plots with a chainsaw. We also recorded notes on ground cover, fire scars, and evidence of logging, along with aspect measurements.

2.2.4. Data Preparation and Analysis

Relative density and relative dominance were calculated by dividing each species total by the total of each metric and multiplied by 100. Density refers to the number of trees in a given area. Dominance refers to the basal area of each species in a given area. Relative importance was calculated by averaging relative dominance and relative density. We used two-tailed T-tests to examine differences between seedlings and saplings at the high site versus the low, and differences in high and low site dominance.

Tree cores and sections were dried, mounted, and sanded with progressively finer sandpaper to make tree rings visible under a microscope (Stokes and Smiley, 1996). Trees were dated to determine the age structure of the stand and then combined with the data from different stands to reconstruct vegetation change on southeastern slopes. Tree rings document tree ages, and thus show a pattern of recruitment that can be related to fire history and vegetation change over time. Each core was measured with the Velmex unislide tree-ring measuring system, visually cross dated using the Yamaguchi (1991) method, and statistically checked using the COFECHA software package (Holmes 1983).

2.3. Results

2.3.1. Species composition

Importance values (Table 2) were highest for *Q. montana* (30.03) at the lower site at WSM followed by *A. rubrum* (20.81) and *N. sylvatica* (18.75). The high site important species were *Q. alba* (30.38), *Q. montana* (24.55), and *P. rigida* (21.44). *Q. montana* accounted for more than 25% of total stems with a combined 54.58 importance value. All 54 *Q. alba* stems located on the high elevation site were located in a single plot near the montane pine barren. Of those *Q. alba* individuals, 25 were multi-trunk trees where the split occurred below breast height. Five multi-trunk trees were made up of more than two stems. *Q. alba* had higher importance values at the higher site (30.38) than the lower site (1.13); however, there was no significant difference between basal area or stem density in *Q. alba* between the two sites (Table 3).

P. rigida was much more important at the higher site (21.44) than the lower site (1.45) but showed no significant difference in density or basal area between sites (Table 3). *A. rubrum* on the other hand was more important (20.81) on the lower site compared to a 2.13 importance value at the higher site. Like *Q. alba* and *P. rigida*, there was no significant difference in basal area or stem density for *A. rubrum* between the high and low sites. There were several *C. dentata* stems throughout both sites but most were small stump sprouts. In terms of size structure, our sites exhibited an inverse-J pattern typical of uneven age stands (Figure 2).

2.3.2 Age Structure

We collected 97 cores and 46 cross sections for age analysis. Of these, only 90 cores were of sufficient quality for dating. The *P. rigida* series correlation was 0.459. The *Quercus* species were grouped and the series correlation was 0.527. We removed the four oldest cores from this correlation due to difficulties matching the series. The series correlation for *A. rubrum* cores was 0.516. We did not date fire-scarred cross sections due to the small sample size of recorded fires; most samples were from stumps and much of the material was rotten and not identifiable to species. In all six plots combined, we found 27 trees with char around their bases, indicating burning in the past. The trees we cored established at dates between 1670 and 1966 (Figure 3). *Q. montana* was the species with the oldest individuals including trees that established around 1670, 1767, 1779, and 1780. *Q. alba* establishment in our study site began in 1710. *Quercus* species were present throughout the last three centuries but began heavily recruiting in the 1920s. *P. rigida* was first represented on our sites in the late 1800s. *P. rigida* steadily recruited from the 1880s until the 1930s when recruitment ceased. *A. rubrum* establishment by 1915 and recruited steadily through the 1960s, our latest establishment dates on core trees (>10 cm DBH). Overall, peak recruitment occurred between 1920 and 1950. Stem density decreases with size at WSM (Figure 4). R^2 values for age/DBH were calculated for each species using a standard least squares equation in the statistical software JMP®. The r^2 value for all *Quercus* species combined was 0.77. The r^2 value for *A. rubrum* was 0.14 and 0.20 for *P. rigida*.

Seedling densities were far higher than sapling densities across all plots but were highest in lower plots (Table 4). Conversely sapling densities were higher in the high site. In the low site three species had especially high sapling densities: *A. rubrum* (24,700), *Q. montana* (23,700), *S. albidum* (22,600). We found no *P. rigida* seedlings on either site and found only eight total saplings in all plots. We found no significant correlations for seedling/sapling density between the high and low elevation sites (Table 4).

2.4. Discussion and Conclusion

Composition at WSM has changed over time, most notably within the last century. *Q. montana* has been present in the area for some time but now dominates much of the forest. The chestnut (*C. dentata*) decline during the early part of the 20th century (Braun 1950, Keever 1953) and much reduced regeneration likely promoted *Q. montana* to greater dominance at WSM. *A. rubrum* has moved into a more dominant role in low elevation areas. Species promotion since chestnut decline has been documented in several areas of the Appalachain Mountain chain. Ireland, Oswald, and Foster (2011) found *Betula* spp. taking the place of the once dominant *C. dentata* in New England. In sites closer to ours, Vandermast and Van Lear (2002) found *C. dentata* replaced by *A. rubrum*, *Q. alba, Q. prinus, Q. rubra*, *Tsuga Canadensis* L., *L. tulipifera*, and *Betula lenta* L. in southern Appalachian sites of North and South Carolina. Our results exhibit oak promotion most similar to Elliot and Swank (2008) who found a rise in *Q. prinus* and *A. rubrum* in the western North Carolina Appalachians since 1934.

While our study indicates a high relative density and relative importance for *Q. alba* (calculated from all plots), it is notable that a single plot (of six) contains the vast majority of stems at our sites. About half of these *Q. alba* stems were multi-trunked splitting below breast height. Because we cored only the largest stem at breast height, we did not capture the date of establishment of the original stems, which may be much older than those of the stems growing from remnant stumps. Many hardwoods are capable of producing multi-trunk trees and is most likely caused by severing the parent stem (Oliver and Larson 1996), such as with logging. New stems can form from preformed basal buds after the main stem is severed (Stone and Cornwell 1968). Our *Q. alba* stem dates cluster around 1943, probably indicating a logging event around that time. Boring, Monk, and Swank (1981) found that after a clear cut, multiple stump sprouts of *Q. alba* occurred and represented the majority of new biomass after disturbance.

Periods of logging have clearly influenced the compositional patterns at WSM by removing certain species and clearing space for others to advance. We recorded 33 stumps in or around our plots that generally could not be assigned to species due to decomposition of bark and outer wood, but their presence is clear evidence of earlier logging. The history of logging may be the driver of the regeneration peak at WSM that lasted from 1920 to 1940 (Figure 4) through opening the canopy and providing conditions conducive for regeneration. Both *P. rigida* and *A. rubrum* recruited and moved into the canopy over the next 70 years, but only *A. rubrum* has continued to recruit in recent times, as evidenced by the high density of saplings in our plots. Since we did not core trees with a DBH <20 cm, some evidence of recruitment is likely be missing from our results. In contrast to *A. rubrum*, we found very little pine establishment dates after the beginning of the fire suppression era, and zero saplings and seedlings, even at our higher elevation site, *where P. rigida* makes up about 20% of mature stems. Since *P. rigida* is considered to be fire-dependent (Williams 1998) and *A. rubrum* fire-intolerant, it seems clear that a lack of fire since the 1930s is driving the inverse recruitment trends of *P. rigida* and *A. rubrum* (Table 3). *A. rubrum* has also become a dominant species in the forest matrix of our sites during the 20th century. *A.rubrum* is shade tolerant, which has enabled it to establish even in the absence of canopy disturbances and steadily ascend to the main canopy. It is also fire-intolerant providing a marker of fire absence in sites where it is found.

Aldrich et al. (2010) and Powers (2010) reported the same lack of *P. rigida* recruitment during recent decades in nearby sites—Powers (2010) in reporting on the WSM montane pine barren, and Aldrich et al. (2010) on southwestern facing slopes about 15 km away from our sites. Both authors attributed the lack of pitch pine recruitment to fire suppression. Our plots were situated on the slightly more mesic southeastern slopes of Warm Springs Mountain, but we expect that fire was still a more frequent occurrence before management practices began to limit burning. The charred trees found in our plots were probably caused by a small wildfire in and around our plots that occurred at the beginning of the last decade. Our findings are in agreement with Abrams and Copenheaver (1999) and Lafon et al. (2005), who also discussed the retreat of fire-dependent species and advance of more shade tolerant, fire-intolerant species in sites at Mill Mountain, Virginia and Great Falls National Park, Virginia. The oldest individuals in our plots were *Q. montana,* a species that can tolerate low intensity surface fires (Lafon et al. 2007). *Quercus* sp. basal area dominance in our sites could be explained by its ability to withstand some fire and to recruit without fire.

P. rigida remains an important part of the canopy in our site nearer the main WSM ridgeline, where fire may have been more prevalent and where edaphic or climatic conditions limit the growth of some species. Since *A. rubrum* does not compete well on the slopes at higher elevations at WSM, other species of trees and shrubs may replace the *P. rigida* if regeneration continues to decline and we believe the less aggressive regeneration on plots three, four, and five reflect this aspect of succession. We noted heavy cover of the understory Ericaceous shrubs, *Rhododendron catawbiense* and *Kalmia latifolia*, in all plots at the higher elevations. These species were present in the lower plots but at much lower densities. Without fire, these shrubs could become the dominant species at the higher elevations at WSM imposing further barriers to regeneration of pines and all tree species (Cain 1930, Whittaker 1956, Brose and Waldrop 2010). The presence of Ericaceous shrubs could explain the lack of *P. rigida* regeneration in the understory as *R. catawbiense* and *K. latifolia* produce thick brush and prevent light from reaching the shade intolerant *P. rigida* seedlings. Prescribed fire is one way of limiting hardwood understory growth and allowing fire-dependent species to remain intact in the region (Waldrop and Brose 1999). Since the southern Appalachians are in close proximity to large human populations, fire management is difficult and often dangerous when fuel loads are high.

It is increasingly important to document long-term forest dynamics using dendrochronological techniques to document changes in forest composition and structure through time (Bergeron et al. 2002, Everett 2008, Aldrich et al. 2010). Reconstructing forest dynamics at high resolution sheds light on drivers of ecological change and allows prediction of future changes. This study has shown the decline and eventual cessation of pitch pine (*P. rigida*) regeneration and an increase in fire-intolerant species such as red maple (*A. rubrum*). Without fire management, most of these stands will likely experience dramatic increases in deciduous tree importance over the next several decades. The central and southern Appalachian region with its highly variable and diverse forests deserves further study at all scales (Abrams and Copenheaver 1999, Elliott et al. 1999, Lafon et al. 2005, Hoss et al. 2008, DeWeese et al. 2010, Aldrich et al.

2010). This study illustrates how changes in disturbance regimes may influence the long-term appearance of forests in this area.

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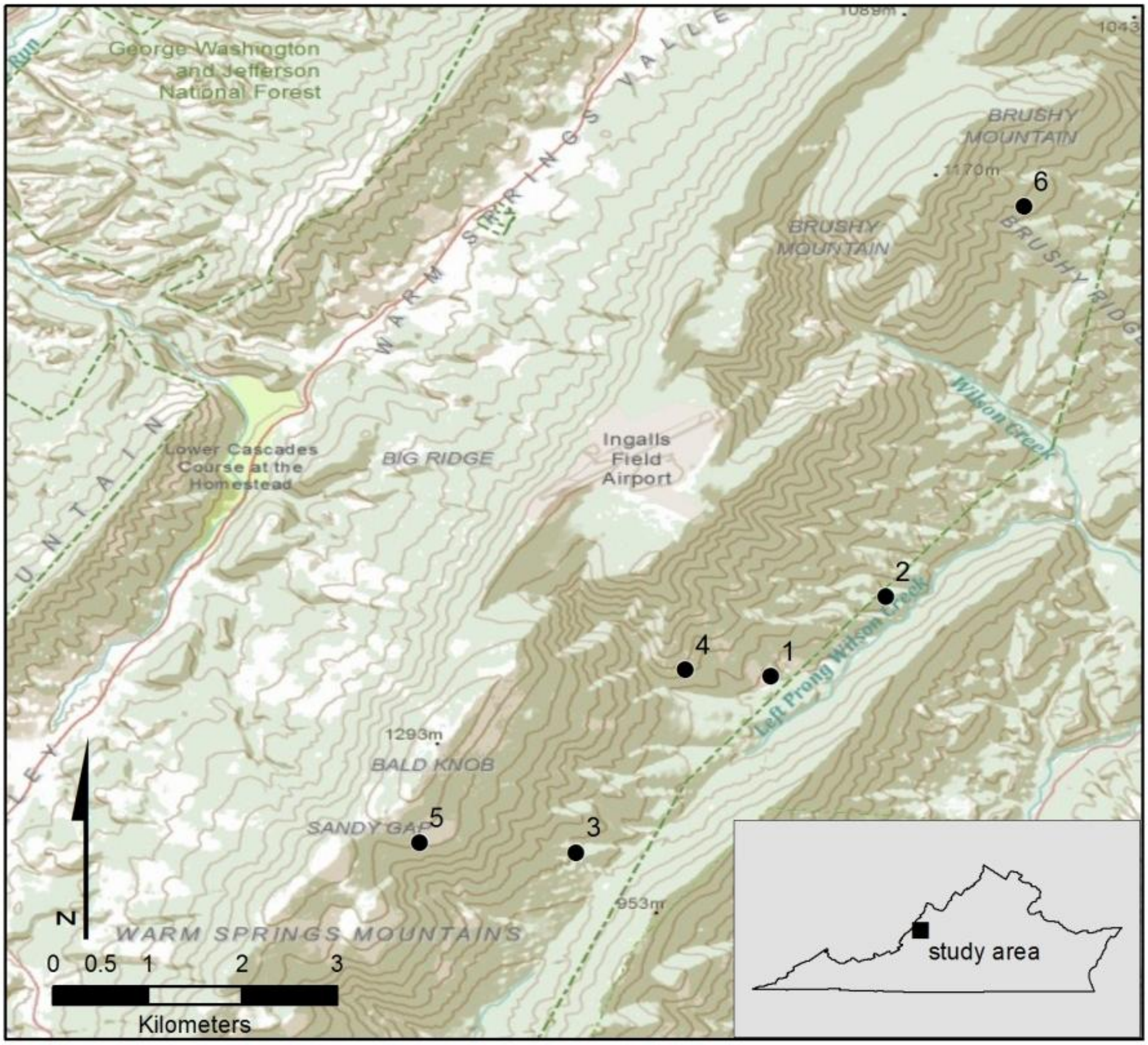

Figure 1. The study area at Warm Springs Mountain, Virginia with plots 1–6 shown on United States Geological Survey Hot Springs local quadrangle topographic map. Contour interval = 20 m.

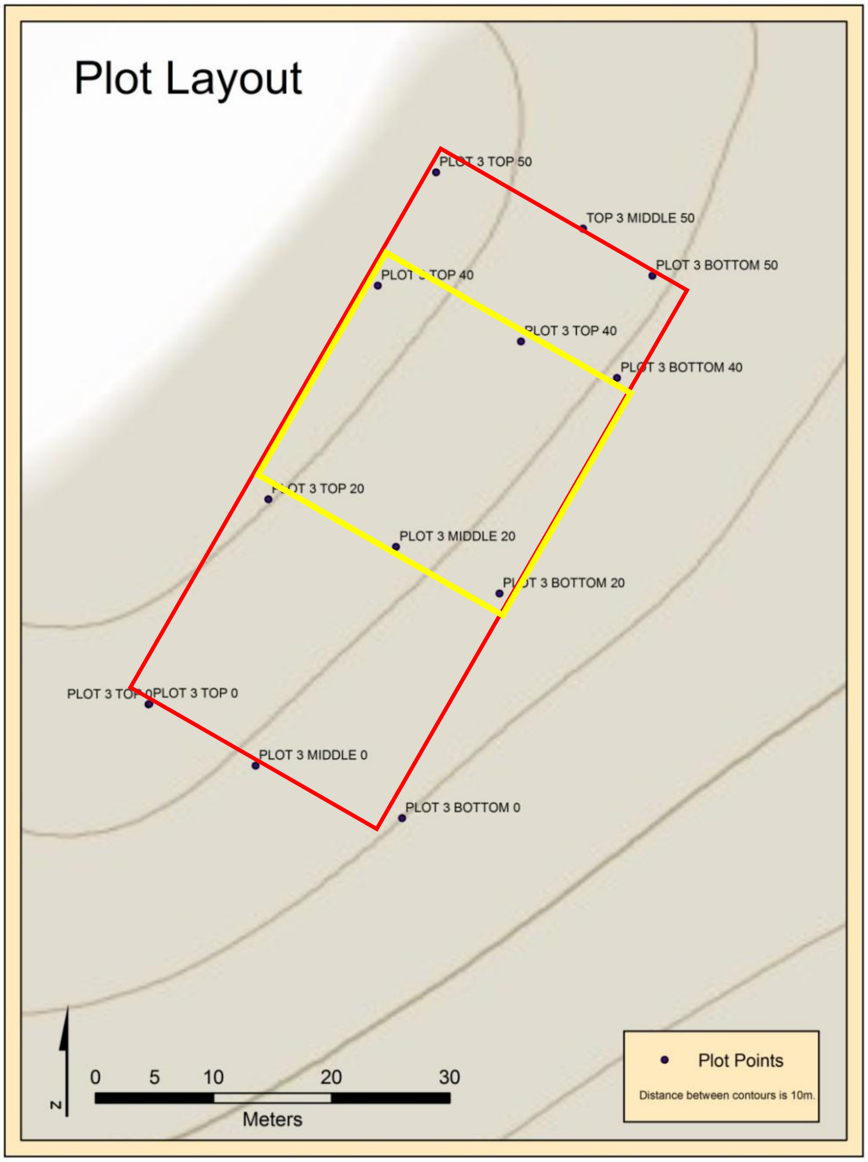

Figure 2. Layout of a typical plot showing 20 x 50 m boundaries (in red) and 20 x 20 subplot boundaries (in yellow).

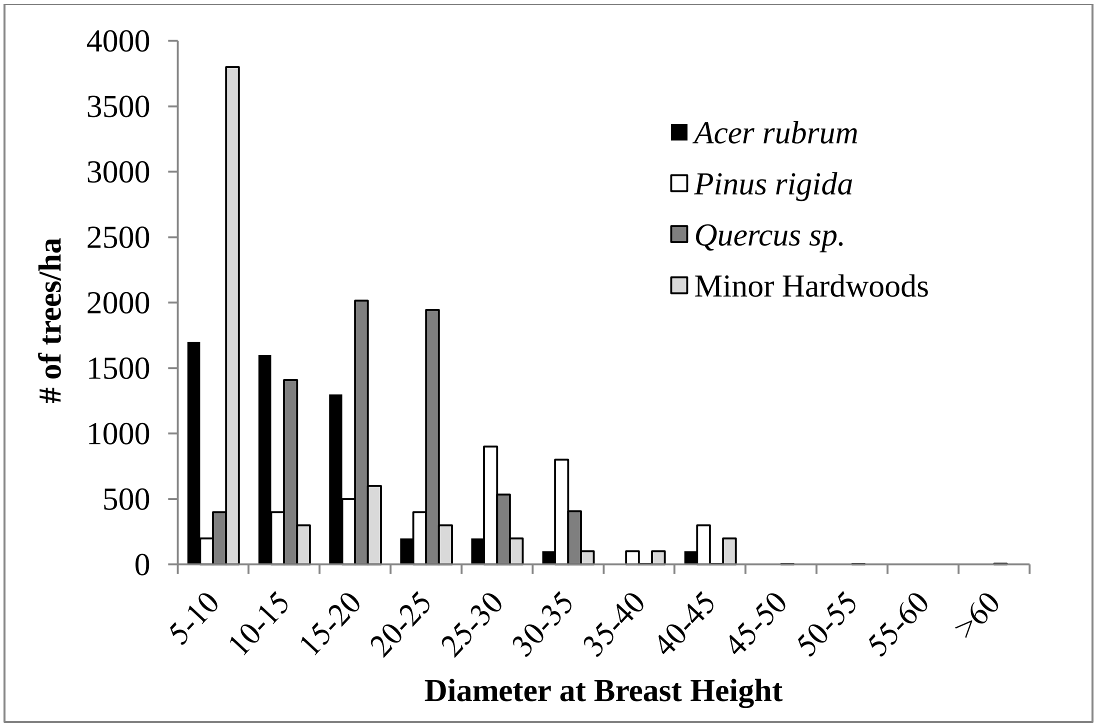

Figure 3. Size structure for all plots at WSM.

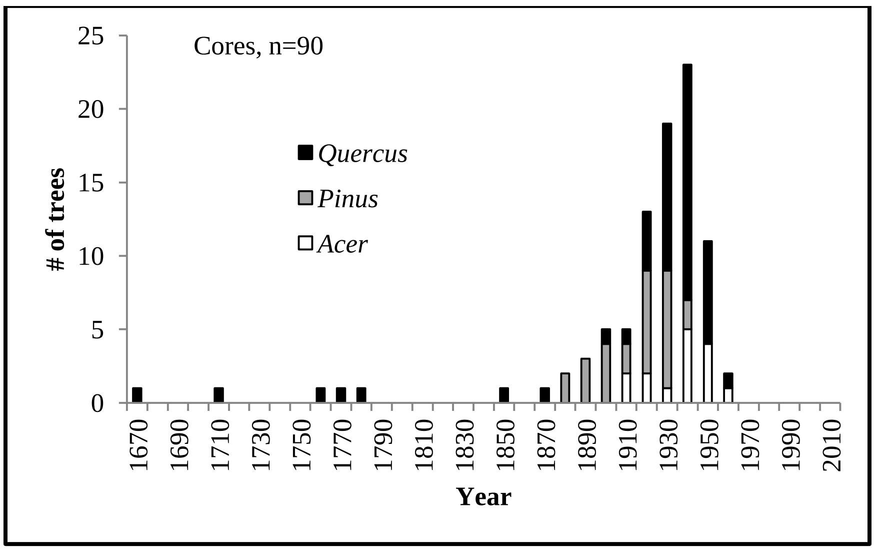

Figure 4. Establishment dates for trees (>20 cm diameter at breast height) cored in all plots. Trees <20 cm DBH were not cored and could explain the lack of recruitment since the 1960s. *Seven minor hardwood species cores are not shown.

Figure 5. Age/diameter relationship for all cored individuals in all plots at WSM. *Seven minor hardwood cores are not shown.

Table 1. Plot center locations and elevations of each plot. Plots 1, 2, and 3 represent the lower elevation plots (elevation <800m). Plots 4, 5, and 6 represent the higher elevation plots (elevation >800m).

Plot	Latitude (N)	Longitude (W)	Elevation (m)
1	37.93130°	79.82022°	701
2	37.93827°	79.80907°	640
3	37.94425°	79.80115°	610
4	37.93927°	79.82426°	945
5	37.91499°	79.85344°	1189
6	37.97562°	79.79606°	914

Table 2. Relative density, relative dominance, and relative importance values converted to # per hectare for both sites. Density and dominance values were relativized by dividing each species total by the total for each site. Relative importance was calculated by averaging relative density and relative dominance. (Note: 25 *Quercus alba* stems on the high elevation site had multiple trunks that were tallied at breast height.)

Species	Relative Density (%)	Relative Dominance (%)	Relative Importance (IV)
a) Low elevation			
Quercus montana	23.7	36.35	30.03
Acer rubrum	24.7	16.88	20.81
Nyssa sylvatica	23.2	14.3	18.75
Quercus rubra	10.8	13.87	12.35
Quercus coccinea	10.8	12.98	11.90
Sassafras albidum	3.6	1.34	2.47
Pinus rigida	1.0	1.87	1.45
Quercus alba	0.5	1.75	1.13
Carpinus caroliniana	1.0	0.43	0.73
Castanea dentata	0.5	0.22	0.37
b) High elevation			
Quercus alba	31.2	29.55	30.38
Quercus montana	23.1	25.97	24.55
Pinus rigida	19.6	23.23	21.44
Nyssa sylvatica	11.5	8.84	10.2
Quercus rubra	4.6	5.05	4.84
Quercus coccinea	2.8	3.02	2.96
Castanea dentata	2.8	1.46	2.18
Acer rubrum	2.3	1.95	2.13
Sassafras albidum	1.7	0.93	1.33

Table 3. Seedling/sapling distribution (converted to #/hectare) for both sites at WSM.

Site and Species	Seedling density #/ha	Sapling density #/ha
a) Low elevation		
Acer rubrum	24,700	–
Quercus montana	23,700	200
Sassafras albidum	22,600	100
Quercus coccinea	13,900	–
Quercus rubra	6,900	–
Nyssa sylvatica	2,000	500
Castanea dentata	700	–
Quercus alba	100	–
Pinus rigida	–	–
Total	94,600	800
b) High elevation		
Sassafras albidum	11,300	300
Quercus montana	7,400	1,600
Quercus alba	6,500	800
Quercus coccinea	5,800	300
Acer rubrum	5,000	300
Castanea dentata	2,000	900
Nyssa sylvatica	1,300	200
Quercus rubra	700	0
Pinus rigida	–	800
Total	40,000	5,200