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Northern Myotis Roosting Ecology and Use of Prescribed Fire and Herbicide Treatment to Enhance Roost Availability in Hardwood Forests

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**Northern Myotis Roosting Ecology and Use of Prescribed Fire and Herbicide Treatment to
Enhance Roost Availability in Hardwood Forests**

Milu Karp

**Thesis submitted to the
Davis College of Agriculture, Natural Resources and Design
at West Virginia University
in partial fulfillment of the requirements
for the degree of**

**Master of Science
In
Wildlife & Fisheries Management**

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Division of Forestry and Natural Resources

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Keywords: Northern myotis; *Myotis septentrionalis*; Decay stage, Prescribed fire; Roost use;
Snag dynamics, Survival, West Virginia
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ABSTRACT

Northern Myotis Roosting Ecology and Use of Prescribed Fire and Herbicide Treatment to Enhance Roost Availability in Hardwood Forests

Milu S. Karp

Recent declines in myotis populations throughout the mid-Atlantic region due to habitat degradation and the spread of white-nose syndrome have led to the increased need for understanding of roost ecology to aid in population recovery. Land management agencies in the Central Appalachian Hardwood Region can contribute to the conservation and possible recovery of northern myotis through habitat creation by means of prescribed fire. Within the Fernow Experimental Forest in West Virginia, three separate compartments were subjected to prescribed fire to better understand the effects of disturbance on summer roost tree use. During the summer of 2011 and 2012, Northern myotis were captured using mist nets and attached with radio transmitters. In burned and unburned areas, 38 roost trees were located and measurements were taken on the individual roost tree and the area surrounding each roost tree, including species, size, canopy gap, distance to neighboring trees, aspect and slope at roost tree locations. Black locust was the most utilized tree species, followed by red maple and red oak. Roosts were located in trees with cavities, loose bark, and crevices; 57% of female and 40% of male roosts were in snags. Both female and male northern myotis were found roosting in burned and unburned areas, while female northern myotis used roosts surrounded with larger canopy gaps overhead. Roosts in unburned areas had a smaller percent canopy closure and were located on steeper slopes. Slope at roost locations was significantly different between both female and male roost locations as well as between roosts in burned areas and unburned areas. As the burned compartments within the FEF continue to decay and forest regeneration is allowed to progress, the effect of fire disturbance on roosting ecology can be better quantified and this may allow forest managers to develop predictive models for roost tree creation and use on a local and landscape scale. Snags with cavities and exfoliating bark are an important component of stand structure for wildlife in hardwood forests. Understanding changes in decaying trees over time is needed to manage cavity-dependent biota. I quantified short-term (2006 – 2012) rates of initial tree death and progression through decay stages following fire and herbicide treatments initially applied in the spring of 2007 in the Alleghany Plateau of West Virginia. Red maple, chestnut oak, northern red oak and sugar maple were the most prevalent species found in measured plots. Northern red oaks were found to live the longest following fire treatment while black locusts were found to decay fastest after treatment. The greatest decline in trees in all treatments was in 2008, one year following initial fire and/or herbicide treatment, with a more gradual decline following in 2012. Transition of trees among decay classes differed by species and size. Prescribed fire alone had the least impact on overall mortality and decay progression for the treatments. Fire tolerance was also positively correlated with tree size, with yellow-poplar having the lowest mortality rate after prescribed fire treatment. Prescribed fire remained the least impactful treatment in rapid progression of trees through decay stages. Plots subjected to herbicide treatment for all non-oak and hickories found with >12.7 cm dbh showed a more rapid rate of decline than both fire only and midstory treatment areas. Snag longevity was significantly greater with increasing diameter for all treatments. Silvicultural practices that increase tree mortality rates and snag retention

over long periods of time could be a beneficial approach to management for cavity-dependent wildlife in hardwood forests.

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INTRODUCTION, JUSTIFICATION, AND RESEARCH OBJECTIVES

In recent decades, significant attention has been given to the management and conservation of forest-dwelling bats to promote the recovery of populations under threat from habitat degradation, disturbance to hibernacula, and disease (Tuttle 1979, Fenton 2003). This focus is due to their important ecological roles as nocturnal insectivores and as possible indicator species of forest ecosystem health (Fenton 2003). Myotis bats have suffered substantial declines and represent the largest and most widely distributed group of bats in North America (Reid 2006). Human disturbance of hibernacula, environmental pollution, and loss of summer habitat (Tuttle 1979, Pierson 1998, Fenton 2003) have led to population declines, resulting in the listing of several bat species under the Endangered Species Act, as well as listing as species of concern (Harvey et al. 1999). Recent declines in myotis populations in the eastern United States and throughout the mid-Atlantic region due to habitat degradation and the spread of white-nose syndrome (*Geomyces destructans*), have led to a need for increased understanding of roost ecology (winter hibernacula and summer roost trees) to aid in population recovery. The primary focus has been protection of myotis hibernacula in an attempt to increase population size. Attempts to protect hibernacula have led to population increases for species such as gray myotis (*Myotis grisescens*), Indiana myotis (*M. sodalis*), and Virginia big-eared bats (*Corynorhinus townsendii virginianus*) (Clawson 2002, Martin et al. 2003). Although the protection of winter hibernacula is important, more recent effort has focused on understanding the broader range of habitat needs, including summer roost habitat. A complete range of habitat needs, both during winter and summer seasons, is necessary to understand the full scope of habitat requirements when planning for management and protection of species.

Information on day-roost selection is necessary for guiding and assessing forest management decisions to improve and protect declining bat populations (Arnett 2003). Various intensities of forest harvesting, ranging from clear-cutting to single-tree selection, have the potential to create forest structure that can affect roost selection of bats (Owen et al. 2004). One silvicultural practice that is being considered beneficial for both forest regeneration and habitat creation in hardwood forests is prescribed fire (Johnson et al. 2009). As prescribed fire continues to aid in promoting oak regeneration in the Central Hardwood Forest region, this practice may also benefit bats (Johnson et al. 2010). Although fire may reduce the longevity or utility of individual roost trees, it may also kill or scar other trees in the vicinity, creating suitable bat habitat in the form of loose bark or snags (Russell et al. 2006). Moreover, trees killed by fire will leave gaps in the canopy as they lose limbs and fall, thus increasing solar radiation that can reach lower surfaces of surrounding trees, where bats may be found (Menzel et al. 2001). Increased solar radiation may be a preferred characteristic of roost sites, as warmer temperatures facilitate more rapid development of neonates (Sedgeley 2001, Boyles and Aubrey 2005).

As the use of prescribed fire in forest management gains acceptance in eastern forests, little is known about its implication for forest-dwelling wildlife. For myotis bats, fire will serve to improve roosting habitat by increasing snag availability as well as increasing solar radiation at roost sites (Brooks and Ford 2006, Johnson et al. 2010). More specifically, several species of bats that occur in the central Appalachians of West Virginia rely on trees and snags for summer maternity roosts as well as roosts for individual male bats, including the northern myotis (*Myotis septentrionalis*) and the federally endangered Indiana bat. Declining populations of these bats are attributed to complex, multi-factor causes, such as exposure to pesticides (Clark 1988), alteration of foraging habitat (Hill and Smith 1984), disturbance of hibernacula (Tuttle 1979),

and destruction of maternity roost habitat, which may outweigh all others as the ultimate cause for the decline in many species (Kunz 1982). The northern myotis has been found to use roosts in disturbed landscapes with increased snags and larger canopy gaps (Menzel et al. 2002); whether these roosts will support increased productivity and recruitment remains to be fully determined.

The potential of using prescribed fire to create suitable roost conditions, in either live trees or snags, for bats is a largely unexplored, but a potentially valuable area of research (Leput 2004, Boyles and Aubrey 2005, Loeb and Waldrop 2008). Recently, prescribed fire has served a dual purpose in the Appalachian Mountains and Central Hardwoods Forest regions: one of hardwood regeneration and shade-tolerant tree species suppression. Insufficient retention and/or regeneration of hard-mast producing oaks (*Quercus* spp.) and hickories (*Carya* spp.) in managed and unmanaged forest stands are major concerns of forest and wildlife managers. The failure to adequately maintain oak has been attributed in part to fire suppression and uneven-age management practices that favor shade-tolerant tree species such as maples (*Acer* spp.) (Clark 1993, Schuler and Fajvan 1999, Brose et al. 2001, Schuler 2004, Nowacki and Abrams 2008). Oak regeneration can benefit from a combination of prescribed fire and forest canopy thinning by tree removal, girdling, or herbicide of competitors, to augment oak seedling initiation and survival through reduced competition and increased canopy openness (Miller 1993, Brose et al. 1999, Brose et al. 2001). The use of fire and herbicide for the regeneration of oak is therefore important in the disturbance it creates. These methods increase snag abundance that in turn favor snag-dependent wildlife species, including bats (Brose et al. 2001, Hallett et al. 2001, Albrecht and McCarthy 2006, Boyles and Aubrey 2005).

Research Objectives

The primary objectives of this study were to:

- (1) Examine the differences in roost sites of northern myotis in forested stands subjected to prescribed fire (burned) and in unburned forested stands.
- (2) Compare roost site use between female and male northern myotis in burned and unburned stands.
- (3) Identify quantitative changes in snag decay and survival dynamics in a second- and third-growth hardwood forest.
- (4) Estimate the probability of snag survival using decay stage.
- (5) Quantify the effect of fire and herbicide on tree progression through stages of decay.

I determined if bats were using areas of burn and herbicide treatment more consistently than areas of control, and if this may was due to creation of adequate cavity/bark characteristics in areas of prescribed fire. I predicted that (1) northern myotis would select roosts in areas subjected to prescribed fire and herbicides, and that use would be related to tree decay class, canopy exposure, solar radiation, and tree species; (2) northern myotis would use newly-created snag and scarred trees (including exfoliating bark) for roosting within burn treatments more than non-treatment reference areas and (3) roost trees selected by female northern myotis would differ from male myotis in canopy exposure and tree species.

I predicted that (1) size class of hardwood trees would be positively related to fire tolerance and that all species allowed to reach a larger size class would remain tolerant to prescribed fire and herbicide regardless of bark thickness, (2) trees injected with herbicide would decline at a rapid rate compared to trees subjected to fire only, and (3) herbicide would be effective in eliminating smaller diameter trees of various species.

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CHAPTER 1
BAT ECOLOGY AND SILVICULTURE
LITERATURE REVIEW

LITERATURE REVIEW

Silviculture and Snag Creation

After decades of fire suppression in the forests in Appalachia, prescribed fire is now gaining acceptance as a management technique to aid in regeneration of oak in hardwood forests. Very little is known about the impact of fire on sensitive forest-dwelling wildlife (Boyles et al. 2005). The northern myotis is one such forest-dependent species. At a time when populations are being dramatically affected by the white-nose syndrome and habitat loss due to the roost degradation and human development, it is imperative to gain a better understanding of the impacts of forest management and silviculture on sensitive species.

Silviculture can be defined as the science and art of manipulating a stand of trees toward a desired future condition (Guldin et al. 2007). Contemporary forest management goals include health, diversity, productivity, and sustainability of ecosystems. A silviculturist sees a forest stand as a collection of individual trees, each of which uses resources according to its size and species attributes (Guldin et al 2007). Silvicultural practices enable trees to grow faster, so that desired stand conditions can be achieved more rapidly. Unwanted competitors can be removed through felling, girdling, use of herbicides, top-killing with fire, or any other practice that effectively reduces or eliminates the ability of undesirable plants to compete with desired individuals. Different silviculture practices will promote different wildlife and foraging habitats, which will favor different species of bats. For example, the little brown bat (*M. lucifugus*) forages along edges of clear-cuts while the northern myotis prefers to forage in intact forests (Patriquin and Barclay 2003). Ideally, these differing habitat regimes can be created through particular silvicultural techniques.

The limited information available does suggest a positive relation between number of large-diameter snags and the abundance of bats (Hayes and Loeb 2007). Along with exposure to pesticides (Clark 1988), alteration of foraging habitat (Hill and Smith 1984) and disturbance of hibernacula (Tuttle 1979), destruction of maternity roost habitat used in summer may outweigh all others as the ultimate cause for the decline in many species (Kunz 1982). The availability of suitable snags therefore is considered to be the most important factor in sustaining populations of cavity-nesting species (McClelland et al. 1979). One of the most immediate and direct ways that forest management activities influence bat habitat is their impact on number of roosts in an area. Removal of actual and potential roost sites is common during timber harvest (Hayes and Loeb 2007). Humes et al. (1999) found that bat presence was higher both in old growth stands and in thinned stands than in unthinned stands. Structural changes caused by thinning may benefit bats by creating habitat structure in young stands that bats are able to use more effectively (Humes et al. 1999).

Prescribed fire is often designated as a tool to release nutrients stored in leaf litter and woody material and to top-kill seedlings and stump sprouts not adapted to fire (Guldin et al. 2007). In the Central Hardwood Forests region of West Virginia, fire is becoming a common denominator for the development of oak forests (Abrams 1992). Prescribed burning also appears to initially lead to creation or restoration of favorable cavity-dwelling bat habitat and its continual implementation perpetuates an open sub-canopy (Boyles and Aubrey 2005). Prescribed fire in forest management may create additional roost resources, depending on spatial context (Johnson et al. 2010).

Fire initially leads to an increase of dead and dying trees and results in greater light availability, and a decrease of canopy and sub-canopy tree density. These characteristics are

beneficial to many forest-dwelling vertebrates like cavity-roosting bats (Boyles and Aubrey 2005) including the northern myotis. Further research indicates that prescribed fires alter forest structure in a way that is favorable to bats that roost in tree cavities or under exfoliating bark (Boyles and Aubrey 2006). An increase in light availability may release bats from one of the constraints suggested for many forest-dwelling bat species in roost tree selection: that of sun exposure (Boyles and Aubrey 2005). Just as quality of burned forests varies dramatically with time-since-fire for cavity-nesting birds (Saab et al. 2004), rapid changes in condition of dead trees in years after fire probably influence suitability of snags for bats (Hayes and Loeb 2007). Another challenge is that prescribed fires create relatively few snags (Horton and Mannan 1988) and many of these are small in diameter (Tiedmann et al. 2000). Small diameter snags are relatively poor roost sites for most species of bats (Barclay and Kurta 2007).

Stand age alone is not an adequate predictor of habitat use by bats (Humes et al. 1999), but forest managers can influence roost abundance by augmenting snag availability via snag creation. It is suspected that significant use of created snags by bats does not occur for several years following their creation because of the time lag associated with deterioration in integrity of bark (Hayes and Loeb 2007). Cavity-nesting species use anthropogenically-created snags for foraging and nesting, but there are differences in this use due to treatment method, degree of decay, and tree species (Hallet et al. 2001). Silviculture can also affect the environmental context in which a roost occurs – through alteration of conditions around a roost, and consequently the thermal characteristics of roost sites (Hayes and Loeb 2007).

Because bats spend considerable time roosting, thermal dynamics play a critical role in ecology of bats (Barclay and Kurta 2007). Sedgeley (2001) compared the microclimate characteristics of bat roosts in New Zealand and found that the temperatures both in used and

potential roosts were relatively stable despite large fluctuations in ambient conditions. The temperature and humidity in occupied roosts were consistently higher than in potential but unoccupied sites and used roosts had a relative humidity that was almost constantly 100% (Hayes and Loeb 2007). In roosts located in tree hollows or beneath loose bark, thermal benefits were influenced by wall thickness and state of decay (Kurta 1996). Also, roost orientation on a particular tree and position of the tree within the forest canopy will influence the amount of sunlight reaching the roost and the degree to which the roost will be heated (Vonhof and Barclay 1996).

Thermal characteristics of trees are influenced by many factors, including size, density, conductivity, specific heat of wood, color and characteristics of bark, and exposure of trees to solar radiation and wind (Potter and Andresen 2002). Johnson et al. (2010) determined that within areas subjected to fire, northern myotis maternity colonies were more likely located in cavity trees that were smaller in diameter, higher in crown class, and located in stands with lower basal area, gentler slopes, and higher percentage of fire-killed stems than random trees. Northern myotis also responded favorably to prescribed fire by exploiting trees, mostly black locust (*Robinia pseudoacacia*) that were located in canopy gaps that were created by the senescence and decay of surrounding overstory trees (Johnson et al. 2010). Overall, understanding quantitative relations of abundance and viability of bat populations with habitat characteristics at multiple spatial scales is central to understanding the response of bats to forest management and to shaping management recommendations (Hayes and Loeb 2007).

Roost Ecology in Forest Management

The structure of forest habitat affects the composition of bat communities directly because of constraints such as echolocation and morphology and indirectly through the types of insects

present (Brigham 2007). A number of studies in various parts of the world have indicated that there is a relation between wing morphology and feeding ecology. Aldridge et al. (1987) found significant correlations between wing morphology and echolocation call design, and maneuverability and habitat use in insectivorous bats in North America. In another study of old world fruit bats, small differences in wing morphology were found to be ecologically significant to the vertical stratification of bats (Hodgkison et al. 2004). Buford et al. (1995) proposed that vegetative structure and plant diversity are related to the production of prey for bats. The possibility that bats select foraging areas based on vegetation structure raises questions about habitat use (Brigham 2007).

Studies have provided evidence that tree-roosting bats select roost trees with particular attributes from among the trees that are available (Barclay and Kurta 2007). Older forests, thinned mature forests with reduced mid-stories, and greenbelts (areas of largely undeveloped, wild, or agricultural land otherwise surrounded by urban development) retained in harvested areas are all important roosting habitats. The variation in habitat preferences emphasizes the importance of open forest conditions and a diversity of stand types for bat communities in southeastern United States (Barclay and Kurta 2007). Vonhof and Barclay (1996) found that myotis bats in British Columbia will select taller trees surrounded by an open canopy, compared to other trees available in the area. Reproductive female bats may benefit energetically by selecting roosts that are heated by the sun for at least part of the day. The orientation of roosts to the sun (i.e. southern and western-facing exposure) will contribute to the amount of heat reaching roosts. Absolute tree height is also a good discriminator between roost and available trees, but height of the tree relative to the canopy is not substantial enough to impact selection. It is suggested that bats may select tall trees to reduce risk of predation from ground predators, and

that bats select trees of particular species and in particular stages of decay. Roost tree species selection may simply be due to tree having appropriate conditions for cavity formation (due to physiological stresses from natural forces including weather, insect attack, bacteria or fungi, and cavity-forming vertebrates) (Vonhof and Barclay 1996).

Studies directly addressing bats' selection of particular roosts are limited and new approaches for evaluating selection of roosts are needed (Brigham 2007). Forest managers need to identify general features of roosts and the surrounding habitat necessary for bats that are relevant to the manager's particular ecosystem type (Marcot 1996). Recent reports address the hypothesis that a combination of reproduction and thermoregulation needs promote differences in site selection by males and females (Brack et al. 2002). For example, female long-eared bats (*Nyctophilus spp.*) will continue to roost under exfoliating bark on the sunny side of trees and in shallow tree cavities (Turbill and Geiser 2008). Most evidence suggests that bats choose roost sites based on characteristics at a stand level, likely as a result of availability of potential roosts. Also, individual bats frequently change roosts and they require multiple trees during a season (Barclay and Kurta 2007).

From a management perspective, roost switching has a substantial implication that cannot be overemphasized (Barclay and Kurta 2007). In some species, groups of bats switch their roosts almost daily and use up to 100 different roosts within one year, which suggests that suitable roosts are often not severely limited (Kerth 2008). Roost switching frequency and distances moved by northern and Indiana myotis among roost trees were similar between burned and unburned stands in one study (Johnson et al. 2010). Daytime roosting is important for the energy maintenance of small, tree-roosting bats like the northern myotis (Turbill and Geiser 2003). Although frequently changing roosts, myotis bats roosted within a particular area that

ranged in size from approximately 0.5 to 6.25 ha (Turbill and Geiser 2003). Although stand dynamics are important, managers must make recommendations that concentrate on conserving entire forested areas, because conserving specific sites alone may not always suffice (Brigham 2007).

Roost fidelity was related to the type of roost occupied in a study comparing 43 different species of bats: high fidelity was directly related to roost permanency and inversely related to roost availability (Lewis 1995). Behaviors such as dispersal, migration, and movement between alternate dens or roosts are of interest because they potentially involve costs associated with leaving a familiar area. Such movements increase the risk of predation, lower familiarity with foraging opportunities, and may disrupt social bonds (Lewis 1995). Proximity to resources may affect larger-scale selection of roosting areas by bats. By roosting close to areas with high-quality foraging opportunities and sites for drinking, bats can minimize commuting times and distances, thereby reducing energy costs (Barclay and Kurta 2007). Small-scale local stand and site factors are perhaps more important than larger-scale landscape features (Broders et al. 2006). It is hypothesized that characteristics of the roost itself are more important than time and energy aspects. Distances traveled between roosts vary greatly among species, and may be related to differences in habitat within home range of various colonies (Barclay and Kurta 2007). Bats often display fidelity to a group of roosts that are clumped in the environment. This may occur in part because clumped trees tend to be similar in age, therefore similar in size. Foraging activity is high in older forests (Crampton and Barclay 1996) and often very low in sapling and regenerating stands (Buford and Lacki 1995). Reduction in numbers of large trees and snags within forests may affect roost availability and may increase the distance between sites used for roosting, foraging, and drinking (Erickson and West 1996).

The true importance of snags and cavity trees to forest dwelling bats as night roosts, distinct from maternity sites, is not known (Ormsbee et al. 2007), but female northern myotis roost communally in summer months, presumably, to reduce thermoregulation costs (Broders et al. 2006). Compared to random trees, maternity roosts of northern myotis have been found within intact forests, signifying that these bats make extensive use of cavities and crevices in areas of low disturbance (Carter and Feldhamer 2005). Kurta et al. (1996) found that individuals that form maternity colonies collectively may use several trees to provide the total resources needed for roosting, including cover and correct temperature, during varying environmental conditions. Colonies usually have from one to a few “primary” trees or “focal” trees (O’Donnell and Sedgely 1999) that provide proper roosting conditions most of the time. In Illinois Carter et al. (2005) found northern myotis roosted in areas with higher canopy closure than in random plots and move shorter distances between consecutive roosts (<100 meters apart). In the Midwest, northern myotis commonly occur in more contiguous, interior bottomland forest habitats (Carroll et al. 2002). Menzel et al. (2001) found that all roosts located for northern myotis were located in or below the forest canopy, which suggests that solar exposure may not be as critical for selection of roost trees by maternity colonies as compared to other species of tree-roosting bats at more northern latitudes in North America.

Maternity colonies of northern myotis have been found to roost in intensively managed stands in South Carolina (Menzel et al. 2001). There, roost trees were not larger and did not extend above the surrounding forest canopy more than randomly located cavity trees; in fact, several roost trees were shorter than surrounding trees. This suggests that increased roost visibility by potential predatory species may not be of primary importance in roost tree selection (Menzel et al. 2001). Foster and Kurta (1999) hypothesized that increased maneuverability of

northern myotis allowed that species to exploit roosts in cluttered environments to avoid niche overlap with other bat species, particularly congeneric groups. Overall, mature forests with an abundance of tree cavities seem a requisite for productive populations of northern myotis (Broders et al. 2006).

Roosts are critically important for bats because they provide cover from environmental conditions and shelter from predators, as well as locations to mate, raise young, hibernate, digest food, and socially interact (Brigham 2007). During winter, northern myotis are found in hibernacula, which are commonly caves or abandoned mines (Griffin 1940; Hiteckcock 1965; Caire et al. 1979). They will return to the same hibernacula although not always in sequential seasons. Northern myotis will hibernate along with other species (*M. lucifugus*, *Eptesicus fuscus*, and *Perimyotis subflavus*) in hibernacula, but are usually a small portion of the total population (Caire et al. 1979). During summer, northern myotis occupy a variety of day and night roosts, and have been shown to favor tree roosts (Barbour and Davis 1969), and have also been found in anthropogenic structures (Foster and Kurta 1999); sexes roost separately, with reproductive females forming small maternity colonies of <60 individuals. Maternity groups roost in trees, under shingles, and in buildings (Brandon 1961; Clark et al. 1987).

Research on forest-dwelling bats has shown that the size and type of roost trees selected for roosting varies from large diameter trees in early stages of decay (Sasse and Pekins 1996) to in crevices, hollows, or under bark of live and dead deciduous trees (Foster and Kurta 1999). Structurally, roosts tend to be closer to neighboring trees than suitable roosts in randomly located cavity trees; in South Carolina, northern myotis were found to roost in nine different species of trees (Menzel et al. 2001). Tree species is an important factor that influences roost tree selection, with *Robinia pseudoacacia* being used significantly more than expected based on their

abundance across the managed forest landscape (Menzel et al. 2001). In managed Allegheny hardwood - northern hardwood forests of the central Appalachians, short-lived early successional species such as black locust may provide suitable roosting habitat for northern myotis, as well as other species of bats (Johnson et al. 2009).

Bat Ecology and Life History

Bats represent about 20 percent of all classified mammalian species worldwide, with over 1,200 different bat species divided into two suborders: Megachiroptera and Microchiroptera (Tudge 2000). In North America, 47 different species of bats can be found, represented by four groups: the leaf-nosed, ghost-faced, vesper bat, and free-tailed families (Kunz 1998). Thirteen different species of insectivorous bats can be found in the Appalachian hardwood regions alone. The genus myotis is represented by 17 different species within the North American continent, more than any other genus.

Understanding the life history and ecology contributes greatly to the protection and management of an organism. In recent years, more information on the elusive life cycle of insectivores has been uncovered. Social systems of bats are far more complex than had been previously understood (Kerth 2008). For instance, in most species of bats, females form maternity colonies to rear young communally, whereas males are solitary, form groups of their own, or join female groups. In several bat species, researchers are discovering patterns of cryptic colony and social structure in the form of fission-fusion systems (temporary splitting of colonies into several subgroups) previously known mostly from some primates, carnivores, and cetaceans (Kerth 2008).

Great variation has been observed in the mating season for bats, due mostly to factors such as migration versus hibernation, and because of the vastly different habitats they live in.

Roosts can be divided into day and night roosts, hibernacula, summer roosts, maternity roosts, feeding roosts and transient roosts. Bats undergo periods of facultative heterothermy (during periods of hibernation), whereby they control their temperature to match surroundings (Wilson 1997). For bats that hibernate, such as the northern myotis, the vital functions (i.e. pulse rate, breathing rate, and metabolic rate) slow down dramatically.

For their size, bats are the slowest reproducing mammals (Austad et al. 1991). Copulation for many forest bats takes place in fall, when swarming behavior occurs near hibernacula for hibernating species. Hibernation exerts a profound influence on the physiology of reproductive bats. The impact of hibernation on the female reproductive cycle appears to arrest the progress of reproductive events begun in the autumn and to postpone their completion until spring. Vespertilionidae display delayed fertilization, where mating takes place and sperm is stored in fall, but fertilization and embryo implantation does not occur until spring (Oxberry 1979). The gestation period lasts 3–4 weeks. Females of most bat species produce a single pup, but particular species give birth to twins, in June and July (Laubach et al. 1994). Young are born naked and helpless, although it has been observed that the newborns assist in their emergence and will pull themselves free before crawling to a feeding position on the mother. Young are relatively large at birth, and emerge feet first to avoid entanglement of wings. Northern myotis bats for example give birth to one young at a time, and lactation lasts on average 1–2 months (Caire et al. 1979). Male bats have been shown to play no significant role in rearing young and females care for developing young (Wilson 1997). Bat pups first exhibit spontaneous, drop-evoked flapping behavior at 10 days, short horizontal flight at 17 days, and sustained flight at 24 days of age (Powers et al. 1991). During the maternity season, particularly during the lactation period, females return to their diurnal roosts several times during the night to nurse their young

(Ormsbee et al. 2007). The earliest-born young are usually volant by early July, and the nursery colonies disband around this time (Laubach et al. 1994).

Bats have enormous potential as bioindicators: they show taxonomic stability, trends in their populations can be monitored, short- and long-term effects on populations can be measured and they are distributed widely (Jones et al. 2009). Bats provide several ecosystem services, and hence reflect the status of the plant populations on which they feed and pollinate as well as the productivity of insect communities. Bat populations are affected by a wide range of stressors that affect many other taxa. In particular, changes in bat numbers or activity can be related to climate change (including extremes of drought, heat, cold and precipitation, cyclones and sea level rise), deterioration of water quality, agricultural intensification, loss and fragmentation of forests, fatalities at wind turbines, disease, pesticide use and overhunting (Jones et al. 2009).

Northern Myotis Ecology

Within the Vespertilionidae family, the northern myotis bat (*order Chiroptera, family Vespertilionidae*) is a member of the myotis genus, the largest and most widely distributed group of bats in North America (Reid 2006). The word *Myotis* is derived from the Greek for “mouse eared”. *Septentrionalis* comes from the Latin for “northern” (Nargosen and Brigham 1993). Other common names for this bat are northern long-eared bat and northern bat (Foster and Kurta 1999). The northern myotis is an insectivorous hibernating bat, widely distributed in Canada and eastern North America. It is commonly found in the New England states and in the southeastern United States. Its range also extends from Florida and west into Alberta, Canada, Montana, and eastern Wyoming (Kurten and Anderson 1980).

Longevity record for northern myotis, based on bats banded in the wild, is 18.5 years (Hall et al. 1957). It is this trait of longevity that separates them from other similar-sized

mammals. In combination with natal philopatry (the behavior of remaining in or returning to location of birth), their longevity leads to an overlapping of generations that may share the same roosts, and thus to the formation of multigenerational social groups (Kerth 2008). In northern myotis, female and male bats roost separately and females form small maternity colonies (Brandon 1961). They may return to the same hibernaculum year after year, and will occupy a variety of day roosts during warmer seasons (Griffon 1940). These qualities in turn make the northern myotis a valuable indicator species.

The northern myotis is an insectivorous bat (Lacki et al. 2007) and can be found in the Fernow Experimental Forest. Previously, this myotis showed no evidence for selection of any order of insects and perhaps this species is closest to what might be considered a “foraging opportunist” (Lacki et al. 2007), but a recent prey identification study found a disproportionate number of lepidopteran prey in fecal matter, and that the northern myotis differentiates itself from other similar insectivorous bats in the prey that it selects (Dodd et al. 2012). A suite of morphological factors influences foraging behavior in insectivorous bats, including body mass, and the size and shape of the skull, jaws, and wings (Lacki et al. 2007). The importance of wingspan, wing area, and body mass for flight is widely recognized (Findley et al. 1972). For example, the size and shape of wings determine flight speed and maneuverability (Norberg 1981), which in turn, affect where a bat is likely to forage. Bats with lower wing loading exhibit greater maneuverability and are more capable of exploiting foraging space that is cluttered with vegetation (Aldridge 1986). The northern myotis has a body designed for feeding in forest canopies or near cluttered vegetation (Lacki et al. 2007).

Foraging areas are also a necessary component in the management of insectivorous bat habitat. The few estimates of foraging areas for forest dwelling bats clearly limits generalizations

of foraging area requirements, but one commonality among studies is that foraging areas usually encompass a body of water or a riparian corridor (Waldien and Hayes 1999). Northern myotis feed on flying insects often found near bodies of water, but also glean prey from substrates (Faure et al. 1993). Gleaning allows for a broader diet, including those insects that can detect echolocation calls of aerial hawking northern myotis (Faure et al. 1993). Northern myotis have been observed using behaviors that suggest that this species is specialized for the task of gleaning prey (Ratcliffe and Dawson 2002). Northern myotis forage under the forest canopy, at small ponds or streams, along paths and roads, or at the forest edge (Caire et al. 1979; van Zyll de Jong et al. 1980; Fenton et al. 1983).

Snag Density and Dynamics

Many wildlife species require standing dead trees as part of their habitat. The ability to predict future density, distribution, and condition of snags is essential in forest management (Morrison and Raphael 1993). Most land-management agencies have established requirements for retaining a minimum number and size of snags (Morrison et al. 1986). But achieving and maintaining densities of snags is challenging because of the dynamic nature of snag recruitment (i.e. tree mortality), decay, and loss, and the desire to maximize economic return from the forest by salvaging recently dead trees (Morrison and Raphael 1993). Many eastern hardwood forests are mainly 30 – 70-year-old second-growth stands that have received little or no management. Clear-cutting is an accepted method of regenerating shade-intolerant tree species in the central Appalachians, but clear-cutting would remove, for many years, the large snags needed by some snag-dependent fauna (McComb and Muller 1983).

Under natural conditions, snags are created primarily by fire, weather, insects, and disease. Individual snags decay and fall at different rates, based on species, diameter, height,

cause of tree death, and exposure to wind (Raphael and Morrison 1987). Most models of snag dynamics assume constant recruitment and decay and loss rates, but meeting such an assumption is unlikely because of the fluctuating nature of abiotic and biotic factors (Morrison and Raphael 1993). Previous research on snag longevity has identified tree species, tree size, decay class, crown scorch, and stand density as important factors determining snag longevity (Bull 1983; Morrison and Raphael 1993; Garber et al. 2005). The species of tree is relatively less important with regard to snag longevity than the method of creation and tree diameter (Raphael and Morrison 1987). Characteristics of individual snags and stand-level information are the most influential factors in predicting snag longevity (Russell et al. 2006). Snags in advanced stages of decay usually fall within five years and snags created by fire decay rapidly and fall quicker (within 10 years) than those on unburned plots. Snags that are larger in diameter, shorter in height, and less decayed remained standing the longest (Russell et al. 2006). Snags in large burned areas are directly exposed to wind; probably causing them to fall sooner than individual snags surrounded by live trees (Gratkowski 1956; Lohmander and Helles 1987).

Small-scale variables (age, height, DBH, tree species, decay, and snag density) are the best predictors of persistence rates (Russell et al. 2006). Age of snag was also an important variable predicting snag persistence (older snags more likely to fall). Current models of snag dynamics designed to assist land managers in maintaining wildlife habitat include the snag recruitment simulator (SRS; Marcot 1992), snag dynamics projection model (SDPM; McComb and Ohmann 1996), and Table Interpolation Program for Stand Yields (forest vegetation simulator; TIPSYS; Stone 1996). The models rely on information regarding snag longevity to estimate how land management practices will affect the availability of snags for wildlife (Garber et al. 2005). Developing snag longevity models based on larger-scale, remotely sensed data will

be necessary to provide more easily implemented management guidelines for maintaining wildlife habitat in postfire forests (Russell et al. 2006).

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CHAPTER 2

ROOST TREE USE OF NORTHERN MYOTIS (*Myotis septentrionalis*) IN A CENTRAL APPALACHIAN HARDWOOD FOREST

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ABSTRACT

Recent declines in myotis populations throughout the mid-Atlantic region due to habitat degradation and the spread of white-nose syndrome have led to the increased need for understanding of roost ecology to aid in population recovery. Land management agencies in the Central Appalachian Hardwood Region can contribute to the conservation and possible recovery of northern myotis through habitat creation by means of prescribed fire. Within the Fernow Experimental Forest in West Virginia, three separate compartments were subjected to prescribed fire to better understand the effects of disturbance on summer roost tree use. Female and male northern myotis were captured during 11 nights, attached with radio transmitters and located during daylight hours using radio telemetry. During the summer 2011 and 2012, 38 northern myotis roost trees were located in burned areas and unburned areas. Measurements were taken on the individual roost tree and the area surrounding each roost tree, including species, size, canopy gap, distance to neighboring trees, aspect and slope at roost tree locations. Black locust was the most utilized tree species (21% of roosts), followed by red maple and red oak. Roosts

were located in trees with cavities, loose bark, and crevices; 57% of female and 40% of male roosts were in snags. Both female and male northern myotis were found roosting in burned and unburned areas. Female roost trees were surrounded by smaller neighboring trees further in decay and lower in crown class than trees used by male bats. Female northern myotis also used roosts surrounded with larger overhead gaps in the canopy than male northern myotis roost trees. Males roosted on less steep slopes than females. Roosts trees of both female and male northern myotis in burned areas were surrounded by trees further in decay and higher in crown class. Roosts in burned areas had a greater canopy gap and were located on steeper slopes. Slope at roost locations was significant in both female and male roost selection and well as for roosts in burned areas and unburned areas, as was the amount of gap present in the canopy surrounding roost trees. Northern myotis exploited areas treated with prescribed fire, but were found using roost trees in both burned and unburned areas. As the burned compartments within the FEF continue to decay and forest regeneration is allowed to progress, the effect of fire disturbance on roosting ecology can be better quantified and allow forest managers to develop models for roost tree creation and use on a local and landscape scale.

KEY WORDS: Northern myotis, prescribed fire, roost selection, West Virginia

INTRODUCTION

In recent decades attention has been given to the management and conservation of forest-dwelling bats to promote the recovery of declining populations (Tuttle 1979; Fenton 2003). This focus is due to their important ecological roles as nocturnal insectivores and as a possible indicator species of forest ecosystem health (Fenton 2003). Myotid bats have suffered substantial declines and represent the largest and most widely distributed group of bats in North America (Reid 2006). Human disturbance of hibernacula, environmental pollution, and loss of summer

habitat (Tuttle 1979, Pierson 1998, Fenton 2003) have led to population declines, resulting in the listing of species under the Endangered Species Act (Title 16 United States Code, Sections 1531-1544), as well as listing as species of concern (Harvey et al. 1999). Recent declines in myotis populations throughout the mid-Atlantic region due to habitat degradation and the spread of white-nose syndrome (WNS) (*Geomyces destructans*) have led to the increased need for understanding of roost ecology (winter hibernacula and summer roost trees) to aid in population recovery.

The northern myotis (*Myotis septentrionalis*) is a small (5–10 g) forest-dwelling bat found throughout much of the eastern United States (Barbour and Davis 1969; Caceres and Barclay 2000; Ford et al. 2005). Northern myotis were once common in Central Hardwood Forests and their recent decline associated with WNS support a need to examine their roost site ecology. Northern myotis roost in trees in various stages of decay and size class (Lacki et al. 2001). They roost under exfoliating bark or in crevices or cavities of live and dead or dying trees (Campbell et al. 1996, Brigham et al. 1997, Weller and Zabel 2001, Menzel et al. 2002). Northern myotis use a variety of tree species for roosts including pin oak (*Quercus palustris*), and elm (*Ulmus spp.*) in southern Illinois (Carter et al. 2005) and short-leaf pine (*Pinus echinata*) and sourwood (*Oxydendron arobreum*) in northeastern Kentucky (Lacki et al. 2001). In the Central Appalachian region of West Virginia, northern myotis select black locust (*Robinia pseudoacacia*) in greater proportion than available (Menzel et al. 2002; Johnson et al. 2009). Black locust is a highly decay resistant species (Panshin et al. 1964; Menzel et al. 2002) that can develop cavities and other suitable roost characteristics as a persistent snag (Lewis 1995, Menzel et al. 2002, Owen et al. 2002, Ford et al. 2006, Johnson et al. 2009).

Male northern myotis are less selective than females regarding roost tree characteristics (Perry et al. 2007). Perry et al. (2007) compared roost selection between female and male northern myotis in central Arkansas (Perry et al. 2007) and found fewer large hardwoods and small snags surrounding female roost sites than at random sites. Male roost sites were more common in unharvested stands than female roost sites while female roosts had fewer surrounding midstory trees than male northern myotis sites (Perry et al. 2007). Female northern myotis form maternity colonies of <100 individuals in cavities and under bark (Sasse and Pekins 1996; Foster and Kurta 1999; Menzel et al. 2002). These sites provide shelter during lactation and rearing of young, as well as predator avoidance and support of thermoregulation (Hamilton and Barclay 1994; Willis et al. 2006). In the central Appalachian Mountains and in southern Illinois, northern myotis maternity colonies were found in roost trees within intact forests (Carter et al. 2005). Alternately, in West Virginia, Johnson et al. (2009) found northern myotis maternity colonies readily exploiting areas where prescribed fire had created large canopy gaps.

Prescribed fire is beneficial in regulating and maintaining forest ecosystems of the eastern United States (Boyles and Aubrey 2006). Fire assists in the regeneration of some hardwood species (Clark 1993). The lack of regeneration and retention of oaks and hickories (*Carya* spp.) in hardwood forests remains a concern for both wildlife and forest managers. This inadequacy is partly attributed to the suppression of fire and uneven-aged management practices that have allowed shade-tolerant species such as maples (*Acer* spp.) and beech (*Fagus* spp.) to dominate the landscape (Clark 1993; Schuler and Fajvan 1999; Brose et al. 2001; Schuler 2004). Oaks (*Quercus* spp.) are valuable tree species for wildlife, including bats. Acorn production by oaks is an important food resource for wildlife in many deciduous forests (McShea et al. 2010) and many species of forest dwelling bats including northern myotis have been found roosting in oak

trees (Menzel et al. 2002; Lacki et al. 2009; Johnson et al. 2009). Alterations to forest structure that reduces competition between oaks and more shade tolerant species and increases forest canopy gap size can aid in oak retention and regeneration (Miller, 1993; Brose et al. 2001; Johnson et al. 2009). Such alterations to forest structure could lead to an increase in snag abundance that may also serve to provide additional roosting habitat for northern myotis (Johnson et al. 2009). In the Central Hardwood Region, prescribed fire has been used to aid in oak regeneration and retention (Miller 1993; Brose et al. 2001), but little is known about the impact of fire on the modification of northern myotis roosting habitat. It is suggested that maternity roost trees (Lacki et al. 2001) may be selected in areas where longer periods of solar radiation exist, such as those found under gaps in the canopy created by fire disturbance. Whether recruitment and survivorship of female and male bats is improved through changes to forest stands remains to be determined (Johnson et al. 2009); however, the implementation of silvicultural practices such as prescribed fire may help to sustain a variety of tree species, tree size classes, and snag decay stages necessary to sustain populations of northern myotis (Lacki et al. 2001). For example, fire can accelerate the creation of snags and increase the size of existing canopy gaps or create new gaps (Johnson et al. 2009) as well as increase insect abundance (Lacki et al. 2009). As managers attempt to improve oak regeneration through the use of prescribed fire, bat habitat may also be enhanced through the creation of suitable roosts (Boyles and Aubrey 2006; Keyser and Ford 2006; Johnson et al. 2009).

Consequently, I examined (1) differences in roost sites of northern myotis in forested stands subjected to prescribed fire (hereafter, burned) and in unburned forested stands, and (2) compared roost site use between female and male northern myotis in burned and unburned stands. I hypothesized that prescribed fire would increase the availability of potential roost sites

resulting from dead and dying trees, and therefore predicted that northern myotis roosts would be found more often in stands subjected to prescribed fire. Moreover, I hypothesized that an increase in tree mortality in burned areas would result in the creation of canopy gaps and associated increase in solar exposure. I predicted that roosts of female northern myotis would be found more often in burned areas because of beneficial thermal conditions due to larger gaps in the canopy.

METHODS

Study Area

Research was conducted on the Fernow Experimental Forest (FEF) (39.03° N, 79.67° S) located in Tucker County, West Virginia. The FEF is managed by the United States Forest Service, Northern Research Station. It is located in the central Appalachians approximately 5 km south of Parsons, West Virginia, in the Unglaciaded Allegheny Plateau Region subsection of the Appalachian Highlands Physiographic Province (Fenneman 1938; Kochenderfer et al. 2007). The FEF was established in 1934 following the designation of the Ellick Run watershed for research and demonstration of forestry practices. When established, the FEF comprised 1470 ha; it was expanded to about 1900 ha in 1974 (U.S. Forest Service Experimental Forest and Range Network). Elevation on the FEF ranges of 530 to 1115 m, with Ellick stream running approximately south to north through the forest. Several tributaries intersect Ellick Run, creating over 5.5 km of streams (Madarish et al. 2002). Mean annual precipitation is estimated at 145.8 cm per year and the mean annual temperature is 9.2 ° C (Kochenderfer 2006).

The FEF vegetation consists of second- and third-growth hardwood stands. The second-growth forest is classified as mixed mesophytic, having characteristics of both mixed-oak and northern hardwood forest types (U.S. Forest Service Experimental Forest and Range Network). Vegetation on the FEF has been managed by even (patch clear-cut) and uneven-aged (e.g.,

single-tree selection), along with two-age and other partial harvesting and silvicultural practices since the mid-20th century, or left undisturbed following initial harvesting in the Elklick watershed from 1903 to 1911 (Schuler and Fajvan 1999). This combination has created a rich mosaic of stand structures. This ecoregion is dominated by mixed mesophytic vegetation; it is identified as a mixed mesophytic forest type (Hicks 1998). Tree species dominating the FEF include: sugar maple (*Acer saccharum*), red maple (*A. rubrum*), yellow-poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), sweet birch (*Betula lenta*), black cherry (*Prunus serotina*), and basswood (*Tilia americana*) (Schuler 2004).

Bat species found in the FEF include northern myotis, little brown bat (*Myotis lucifugus*), big brown bat (*Eptesicus fuscus*), tri-colored bat (*Pipistrellus subflavus*), eastern red bat (*Lasiurus borealis*), silver-haired bat (*Lasionycteris noctivagans*), and hoary bat (*Lasiurus cinereus*), all of which can be found from late spring to late fall (Owen et al. 2004). The federally listed Indiana bat (*Myotis sodalis*) has also been identified in the FEF, and utilizes Big Springs cave during winter months for hibernation (Ford et al. 2005) along with northern myotis. Big Springs Cave, located within the FEF, is part of an extensive cave system underlying or within the Greenbrier limestone formation.

Recently, prescribed fire has been utilized to encourage oak regeneration in areas where clear-cutting and fire have both been underutilized and suppressed (Schuler 2004). In April and May of 2007, three compartments were treated with prescribed fire for 1 day using a strip-head fire technique to understand the effects of fire on regeneration and habitat enhancement. A prescribed fire treatment was applied to 152 ha, consisting of compartment 45 (127 ha), 13 (13 ha) and 21 (12 ha). All 3 compartments were retreated with fire in a following year (compartments 13 and 21 in 2008; compartment 45 in 2009) (Appendix 2.a).

Bat capture and radio telemetry

Bat capturing. —Bats were captured between May and September 2011 and 2012 using mist nets (Avinet, Inc., Dryden, NY), erected in various locations over and near small bodies of water (springs, streams, and artificial weir ponds). A single harp trap was placed at the entrance of Big Springs Cave. Locations for netting were chosen in an attempt to sample in both burned and unburned areas (as well as near cave entrances in an attempt to capture northern myotis). Nets were set up <2 hours after sunset, in suitable weather as to avoid rain and wind. Specific data collected for each bat captured followed Menzel et al. (2002) and included: species, sex, age, forearm length, weight and reproductive condition (if possible) (Menzel et al. 2002).

Radio-telemetry. —Following capture and weighing, I attached radio transmitters (Lotek Wireless Inc, Newmarket, Ontario, Canada) between the scapulae of adult northern myotis, after trimming fur to expose skin. I used skin bond glue (SkinBond®, Smith & Nephew Inc., Largo, FL) to attach the 0.35 gram radio transmitters. Capture and handling of bats followed established guidelines of the American Society of Mammalogists (ACUC, 1998). The estimated battery life of transmitters was 12 – 15 days. I used radio receivers and 3-element Yagi antennae to locate bats (Wildlife Materials, Inc., Murphysboro, Illinois). I tracked individuals and located diurnal roosts for the length of transmitter battery life or until transmitters fell off. I used a Garmin GPSmap 60CSx global positioning units to document each roost tree location.

Habitat and Vegetation Measurements

Roost trees. — I recorded the following characteristics of each individual roost tree: species, dbh (cm) (DBHR), decay class (DECAYR) (1 = live, 2 = declining, 3 = recent dead, 4 = loose bark, 5 = no bark, 6 = broken top, 7 = broken bole, 8 = down material, 9 = stump; Maser et al. 1979), crown class (CROWNR) (4 = suppressed, 3 = intermediate, 2 = co-dominant, 1 =

dominant; USDA Forest Service 2002), tree height (m) (HEIGHT), and ocular estimate of remaining bark (BARK; 0, 25, 50, 75, 100%; Lacki et al. 2001; Carter et al. 2005; Perry et al. 2007; Johnson et al. 2009; Lacki et al. 2009). I recorded percent slope (SLOPE) and degree aspect (ASPECT) at the base of each roost tree using a compass. I determined canopy coverage at each roost tree using photographs taken with a Nikon Coolpix 8400 camera with fish-eye lens (Melville, New York) attached to a self-leveling tripod. I placed the camera on the east side of the roost tree, with top of camera pointed north and lens directed up with camera horizontal to ground level. I used Winscanopy and XLS canopy software (Regent Instruments, Inc., Canada) to determine percent canopy coverage on 4 levels: percent canopy gap as whole (TOTALGAP), 45° wedge off of the horizon (HORIZGAP), 90° wedge centered on midpoint (OVERHEADGAP), and a 90° wedge centered on the four cardinal directions (NORTHGAP, EASTGAP, SOUTHGAP, WESTGAP). Basal area surrounding the roost tree was measured with a 20-factor prism (m^2/ha ; BASAL), using the roost tree as plot center. For the four trees closest to the roost tree, I recorded and averaged dbh (DBHN), crown class (CROWN), and decay class (DECAYN); distance of each tree to the roost tree was also recorded (DISTROOST) (Appendix 2.b).

Statistical Analysis

Roost site use. —A Wilcoxon/Mann-Whitney U-test was used to test for differences between roost tree use in burned and unburned areas (objective 1) and between female and male roost trees (objective 2) (JMP, Version 10. SAS Institute Inc. 2010. Cary, NC: SAS Institute Inc.). Characteristics used in determining differences in roost trees included: dbh, decay stage, crown class, tree height, and percent bark remaining. I compared the distance of the roost tree to the four closest trees, dbh, decay stage and crown class of four nearest trees, and stand basal

area. I compared percent canopy cover in the four cardinal directions and total gap coverage. I then compared aspect and slope at the roost tree. Measurements of slope and remaining bark were arcsine-transformed and degree (aspect) was cosine-transformed to better approximate normality; however, untransformed means are reported for ease of interpretation. A Pearson's χ^2 test for goodness of fit was used to determine difference in frequency between female and male roost use in burned and unburned areas and between live trees and snag usage. G-values were adjusted using William's 1979 correction equation (McDonald 2009) due to low sample size. Because of small sample size, I present only summary statistics of snag and live roost tree species usage between male and female northern myotis in burned and unburned areas.

RESULTS

During summer 2011, mist-netting was conducted for 5 nights and resulted in 89 captures including: 52 northern myotis, 15 tri-colored bats, 11 little brown myotis, 8 Indiana myotis, 2 eastern red bats, and one Rafinesque's big-eared bat. Radio transmitters were attached to 32 northern myotis (11 female, 21 male). Sixteen northern myotis (8 female, 8 male) roost trees were located: three of the female roost trees were located outside the boundary of the FEF. During summer 2012, mist-netting was conducted for 6 nights and resulted in 25 bat captures including: 21 northern myotis, 3 northern red bats, and one big brown bat. Transmitters were attached to 11 northern myotis (8 female, 3 male) and 22 roost trees (15 female, 7 male) were located.

Within burned and unburned areas, I located 38 northern myotis roost trees consisting of 10 different tree species of various sizes (Figure 2.1). Red maple, black locust, and red oak were common tree species used for roosting by northern myotis (Table 2.1). Female roosts were most often (6 of 23) located in black locust while males roosted more often (5 of 15) in red maple. Six

of 8 black locusts identified as roost trees were relic snags, and not the result of prescribed fire based on their lack of bark or branches. Roosts used by female and male northern myotis were located in snags (47%) and in live trees (53%; Table 2.2). Females roosted in snags more often (57%) than in live trees (43%); males roosted in live trees more often (60%) than in snags (40%) (Table 2.2). Frequency of snag and live tree use by female and male northern myotis was similar (Table 2.3).

Female and male northern myotis roost trees were similar in size (dbh cm) (female = $34.86 \text{ cm} \pm 2.86 \text{ SE}$; male = $30.87 \text{ cm} \pm 4.04 \text{ SE}$; $Z_{36} = -1.16$; $P = 0.244$), decay stage (female = $3.3 \pm 0.40 \text{ SE}$; male = $2.93 \pm 4.04 \text{ SE}$; $Z_{36} = -0.565$; $P = 0.572$), crown class (female = $2.90 \pm 0.10 \text{ SE}$; male = $2.5 \pm 0.27 \text{ SE}$; $Z_{36} = 1.14$; $P = 0.256$), and tree height (female = $18.88 \text{ m} \pm 2.12 \text{ SE}$; male = $19.87 \text{ m} \pm 2.34 \text{ SE}$; $Z_{36} = 0.00$; $P = 1.00$). The mean distance of female and male roost trees from the four nearest surrounding trees (female = $3.96 \text{ m} \pm 0.21 \text{ SE}$; male = $4.5 \text{ m} \pm 2.19 \text{ SE}$; $Z_{36} = 1.26$; $P = 0.209$) was also similar. Crown class of the four nearest trees to the roost tree (female = 2.01 ± 0.93 ; male = $2.57 \pm 0.67 \text{ SE}$; $Z_{36} = -3.58$; $P = 0.0003$) and the size (dbh) of the four nearest trees (female = $29.46 \text{ cm} \pm 1.70$; male = $34.5 \text{ cm} \pm 2.19 \text{ SE}$; $Z_{36} = -1.06$; $P = 0.045$), were different between sexes. Female northern myotis used roost trees surrounded by smaller trees that were lower in crown class than trees surrounding male roost trees (Table 2.4). Overhead canopy gap (female = $19.17\% \pm 2.93 \text{ SE}$; male = $9.44\% \pm 1.82 \text{ SE}$; $Z_{36} = -2.151$; $P = 0.0315$) was larger for sites used by females than sites used by males. Female northern myotis used roost trees on steeper, (female = $33.30\% \pm 2.74 \text{ SE}$; male = $12.07\% \pm 2.41$; $Z_{36} = -4.09$; $P = <0.0001$) more northern-facing slopes than male roost trees (Table 2.4).

In 2011, 6 roost trees (male) were found within burned areas and 10 (8 female, 2 male) were found in unburned areas. In 2012, 5 roost trees (4 female, 1 male) were found in burned

areas. Roost trees not located in burned areas ($n = 27$) were located in unmanaged areas of forest in the FEF not currently under manipulation for fire disturbance (unburned). Two roost characteristics found to be significantly different between sexes were also found to be significantly different between burned and unburned areas: crown class of the 4 nearest trees (burned = 2.63 ± 0.13 SE; unburned = 2.07 ± 0.09 SE; $Z_{36} = -3.10$; $P = 0.0019$) and slope (burned = $18.27\% \pm 6.24$; unburned = $27.63\% \pm 2.43$; $Z_{36} = -2.195$; $P = 0.0281$). Decay stage of 4 nearest trees (burned = 2.05 ± 0.27 SE; unburned = $1.24 \pm .07$ SE; $Z_{36} = 2.57$; $P = 0.010$), and percent northern gap at roost locations was different between burned and unburned areas (burned = $18.63\% \pm 2.68$ SE; unburned = $10.45\% \pm 1.63$ SE; $Z_{36} = 2.74$; $P = 0.0062$). Neighboring trees were more decayed and higher in crown class in burned areas. Roost trees used in burned areas had a greater canopy gap in a north-facing direction, and were located on less steep slopes than roosts located in unburned areas (Table 2.5).

DISCUSSION

Female northern myotis roosted in 10 different species of hardwood trees, whereas males used 6 different species. This finding coincides with previous studies that report northern myotis using a variety of tree species as roosts (Carter and Feldhamer 2005; Menzel et al. 2002; Lacki et al. 2009). Northern myotis on the FEF readily used relic black locust snags. Menzel et al. (2002) Ford et al. (2006), and Johnson et al. (2009) also reported northern myotis selection of black locusts snags on the FEF. Because black locust was not as abundant historically (Abrams and McCay 1996, Schuler and Gillespie 2000, Ford et al. 2006), the selection of locust in present Central Hardwood Forests may demonstrate flexibility in roost tree species given the species meets the roost requirements of the northern myotis.

Roost fidelity can be a measure of roost use based on characteristics other than tree species per se and can give insight into whether a tree species is important. Lewis (1987) found roost fidelity was related to the type of roost occupied: high fidelity was directly related to roost permanency and inversely related to roost availability. Because black locust was not a common tree species in the Central Appalachian Mountains prior to the widespread harvesting that occurred regionally from the end of the 1800s into the 1920s (Schuler and Fajvan, 1999), the observed use by northern myotis for roosting may be a relatively new phenomenon. Northern myotis may be selecting for particular cavity and bark characteristics provided by standing dead black locust trees. Moreover, the decay resistance of black locust snags (i.e., permanency) may allow northern myotis to develop strong fidelity to roost sites. Other studies have observed northern myotis using roost areas repeatedly and, to a lesser extent, reusing specific roost trees (e.g. Cryan et al. 2001, Johnson 2009). Benefits of such fidelity may include increased fitness due to familiarity with high-quality roosts and the maintenance of social bonds (Cryan et al. 2001).

Northern myotis on the FEF roosted in both live trees and snags, which is consistent with past findings (Sasse and Pekins 1996; Foster and Kurta 1999; Lacki and Schwierjohann 2001). A similar percent use of live trees (53% of roosts) was observed on the FEF as reported elsewhere in hardwood forests (53%; Foster and Kurta 1999). In burned areas, female northern myotis roosted only in snags, while male roosts were equally distributed among snags and live trees. Snags used by female northern myotis had obvious cavities and lacked bark, indicating a high level of decay. Broders et al. (2004) reported female northern myotis using decayed trees 5 times more often than live or recently dead trees. In West Virginia, Menzel et al. (2002) found that female northern myotis formed maternity colonies more often in cavities than under exfoliating bark.

Female northern myotis on the FEF roosted in snags with greater overhead canopy gap percentage than males. Large canopy gaps allow for increased solar radiation, which may be a selected characteristic of female roosts, as warmer temperatures facilitate more rapid development of neonates (Sedgeley 2001, Boyles and Aubrey 2006, Lausen and Barclay 2006). Johnson et al. (2010) found daily mean and maximum temperatures were higher in areas treated with prescribed fire, which could be due to decay and subsequent falling of trees damaged by fire, creating gaps in the canopy. Female northern myotis were found to roost in areas with greater slope than males. Greater slope could be related to greater fire intensity resulting in more snags and possibly a greater presence of bats. Although of possible biological importance, previous research on the FEF, however, did not find slope to be a predictor of roost selection in northern myotis (Menzel et al. 2002, Johnson et al. 2009).

The location of roost trees in both burned and unburned areas demonstrated a variability of tree usage by northern myotis on the FEF, exploiting characteristics found in both area types. Depending on need and availability, northern myotis may exploit roost trees in areas of burn with larger canopy gaps when present, while also exploiting areas that provide similar habitat in unburned areas where less solar exposure is required (i.e. seasonal or temperature shifts). Broders et al. (2004) found female northern myotis roosting in trees in mid-levels of decay more than live or recently decayed. Such decay stages could be more rapidly acquired through the use of prescribed fire. Female bats responded to alterations to the forest structure by using roost trees located in burned areas. Males also were found exploiting burn areas. Male bats were found using an even proportion of roost locations in burn and unburned areas which is similar to other findings where male northern myotis were found to be less selective in roost habitat (Perry et al. 2007), including roost tree size and location when comparing areas treated with fire to areas left

unburned. Contrary to prediction, the majority of females were found using unburned areas more than burned areas, with a relatively even distribution between live trees and snags. This may be due to suitable roost tree characteristics within unburned areas (e.g. location of tree in canopy gap, cavities present) and that the necessary requirements for adequate roosts remain in unburned areas even when altered roost trees, such as those exposed to fire, are available in burned areas. Specifically in the FEF, significant roost habitat characteristics included canopy gaps, and size and decay stage of surrounding trees. These characteristics, as well as snag longevity and increased solar radiation in forest canopy gaps can be created through the use of fire.

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Table 2.1. Roost tree species used by *Myotis septentrionalis* on the Fernow Experimental Forest, 2011 and 2012.

Tree species	Female		Male	
	(n)	%	(n)	%
<i>Robinia pseudoacacia</i> (Black locust)	6	26.1	2	13.3
<i>Acer rubra</i> (Red maple)	5	21.7	5	33.3
<i>Quercus rubra</i> (Red oak)	3	13.0	3	20.0
<i>Acer saccharum</i> (Sugar maple)	2	8.7	2	13.3
<i>Betula lenta</i> (Black birch)	2	8.7		
<i>Quercus prinus</i> (Chestnut oak)	1	4.3	2	13.3
<i>Sassafras</i> sp. (Sassafras)	1	4.3	1	6.7
<i>Magnolia fraseri</i> (Fraser magnolia)	1	4.3		
<i>Carya ovata</i> (Shagbark hickory)	1	4.3		
<i>Liriodendron tulipifera</i> (Yellow-poplar)	1	4.3		

Table 2.2. Distribution of *Myotis septentrionalis* roosts in live trees and snags in the Fernow Experimental Forest, 2011 and 2012.

Tree species	Female		Male	
	Snag	Live	Snag	Live
<i>Acer rubra</i> (Red maple)	2	3	2	1
<i>Robinia pseudoacacia</i> (Black locust)	4	2	2	
<i>Quercus rubra</i> (Red oak)	2	1		5
<i>Acer saccharum</i> (Sugar maple)	2			3
<i>Betula lenta</i> (Black birch)	1	1		
<i>Quercus prinus</i> (Chestnut oak)		1	1	
<i>Sassafras sp.</i> (Sassafras)		1	1	
<i>Magnolia fraseri</i> (Fraser magnolia)	1			
<i>Carya ovata</i> (Shagbark hickory)		1		
<i>Liriodendron tulipifera</i> (Yellow-poplar)	1			
All species	13	10	6	9

Table 2.3. *Myotis septentrionalis* roost trees located in burned and unburned areas in the Fernow Experimental Forest, 2011 and 2012. Pearson's g-test for goodness of fit between sex and burn with adjusted g-values for small sample size.

	Female							Male						
	Snag	Live	Total	<i>g</i>	<i>q</i> *	df	<i>P</i> -value	Snag	Live	Total	<i>g</i>	<i>q</i> *	df	<i>P</i> -value
Burned	4	0	4	2.667	3.0	1	0.102	3	4	7	0.032	4.5	1	0.858
Unburned	8	11	19	0.476	10.5	1	0.490	3	5	8	0.101	5.0	1	0.750

*Williams 1979 corrected *g*-value: $q=1+(a^2-1)/6nv$ (*a* = number of categories; *n* = sample size; *v* = degrees of freedom) (McDonald, J.H. 2009. Handbook of Biological Statistics (2nd ed.). Sparky House Publishing, Baltimore, Maryland)

Table 2.4. *Myotis septentrionalis* roost tree and site characteristics on the Fernow Experimental Forest, 2011 and 2012.

	Female (<i>n</i> = 23)		Male (<i>n</i> = 15)		Z-score	<i>P</i> -value
	Median	SE	Median	SE		
DBHR	29.72	2.86	24.89	4.04	-1.16489	0.2441
DECAYR	4.00	0.40	2.00	0.61	-0.56528	0.5719
CROWNR	3.00	0.10	2.00	0.27	1.13713	0.2555
HEIGHT	23.16	2.12	20.42	2.34	0.00000	1.0000
BARK	75.00	5.41	100.00	7.66	1.80531	0.0710
DISTN	0.30	0.21	0.38	2.19	1.25723	0.2087
DBHN	26.04	1.70	29.72	2.11	-1.06029	0.0445*
DECAYN	1.00	0.15	1.00	0.60	-1.94396	0.0591
CROWN	2.00	0.93	2.00	0.67	-3.58287	0.0003*
TOTALGAP	15.39	1.88	9.36	0.95	-1.85167	0.0641
NORTHGAP	10.11	2.24	13.18	1.75	0.44799	0.6542
EASTGAP	9.53	11.07	9.04	5.16	-0.95570	0.3392
SOUTHGAP	10.98	2.48	8.02	0.96	-1.43355	0.1517
WESTGAP	13.71	2.55	6.05	1.53	-1.85167	0.0641
OVERHEADGAP	20.58	2.93	7.78	1.82	-2.15056	0.0315*
HORIZGAP	19.21	2.75	11.60	1.41	-1.55302	0.1204
ASPECT	28.00	18.98	135.00	15.32	-1.79312	0.0730
SLOPE	28.00	2.74	10.00	2.41	-4.08899	<0.0001*
BASAL	7.00	0.50	7.00	0.48	-0.54320	0.5870

Table 2.5. *Myotis septentrionalis* roost tree and site characteristics in burned and unburned areas in the Fernow Experimental Forest, 2011 and 2012.

	Burned (<i>n</i> = 11)		Unburned (<i>n</i> = 27)		z-score	<i>P</i> -value
	Median	SE	Median	SE		
DBHR	25.40	2.84	29.72	3.02	-1.19111	0.2336
DECAYR	4.00	0.57	2.00	0.42	0.27992	0.7795
CROWNR	2.00	0.40	2.00	0.14	-0.6253	0.9501
HEIGHT	18.14	1.91	23.62	1.82	-0.90115	0.3675
BARK	75.00	7.43	75.00	5.71	-0.08387	0.9332
DISTN	0.38	1.01	0.30	0.65	1.88119	0.0599
DBHN	30.36	2.36	26.42	1.61	1.28386	0.1992
DECAYN	1.00	0.27	1.00	0.07	2.57488	0.0100*
CROWNN	2.00	0.13	3.00	0.09	-3.10410	0.0019*
TOTALGAP	11.27	2.45	11.65	1.51	0.19313	0.8469
NORTHGAP	21.14	2.68	10.00	1.63	2.73604	0.0062*
EASTGAP	9.04	1.08	10.31	1.98	-0.32189	0.7475
SOUTHGAP	7.00	2.51	9.16	2.05	-1.19098	0.2337
WESTGAP	9.75	4.19	7.47	1.75	0.77253	0.4398
OVERHEADGAP	7.76	4.58	13.94	2.26	-0.90138	0.3674
HORIZGAP	11.90	4.00	14.26	2.08	-0.19313	0.8469
ASPECT	120.00	23.88	188.00	15.75	-1.73933	0.0820
SLOPE	10.00	6.24	28.00	2.43	-2.19545	0.0281*
BASAL	7.00	0.57	7.00	2.30	-0.39030	0.6963

Figure 2.1. Roost tree species used by female and male northern myotis by size (dbh cm) in 2011 and 2012 in the Fernow Experimental Forest.

Figure 2.2. Northern myotis roost tree locations within the Fernow Experimental Forest, Tucker County, West Virginia, 2011 and 2012.

Appendix 2.a. Treatment type and timeline of treatment application for 3 compartments in the Fernow Experimental Forest, West Virginia, 2006 – 2012.

COMPARTMENT 13 & 21	
Date	Activity
2006	plots established & measured
2007a	prescribed fire
2007b	plots remeasured
2007c	herbicide treatment
2008a	plots remeasured
2008b	herbicide treatment
2009a	prescribed fire*
2009b	plots remeasured
2010	plots remeasured
2012	Plots remeasured

COMPARTMENT 45	
Date	Activity
2006	plots established & measured
2007a	prescribed fire
2007b	plots remeasured
2007c	herbicide treatment
2008a	prescribed fire*
2008b	plots remeasured
2008c	herbicide treatment
2009	plots remeasured
2010	plots remeasured
2012	plots remeasured

* Compartment 45 received re-application of fire one year prior to compartment 13 and 21.

Appendix 2.b. Total and divided area of compartments and areas of treatments by hectare, including percent area of treatment plots, established in 2006 in the Fernow Experimental Forest.

Compartment	Hectares	% of Total
13	13.0	8.9
21	12.0	8.2
45	121.0	82.9
Total compartments	146.0	100%

Treatment plots	Hectares	% of Total (146 hectares)
20-m radius plot (each)	0.13	0.09
Burn only	3.0	2.1
Burn + herbicide	6.0	4.2
Total plot acreage in compartments	9.0	6.3%

Appendix 2.c. Decay stage designation and the characteristics of trees in those decay stages
(based on decay of Douglas-fir snags as adapted by Maser et al. 1979).

Decay stage	Characteristics
1	Live
2	Live with few limbs broken
3	Dead with all limbs present
4	Dead, few limbs missing, loose bark
5	Few limbs remaining, no bark
6	Broken bole, down material, no limbs
7	Decomposed

Appendix 2.d. Crown class designation and associated characteristics of trees in those classes
(*USDA Forest Service*).

Crown class	Definition	Characteristics
1	Dominant	Crowns receiving full light from above and partly from the side; usually larger than average trees or shrubs in stand, with crowns that extend above the general level of canopy
2	Codominant	Crowns receiving full light from above, but comparatively little from sides; crowns usually form general level of canopy
3	Intermediate	Receiving little direct light from above, none from the sides; usually with small crowns considerably crowded on sides that are generally either below or extending into canopy formed by codominant trees or shrubs
4	Suppressed	Crowns receiving no direct light either from above or from sides, and that will not respond to release; usually crowns are entirely below general level of the canopy

Appendix 2.e. Characteristics of roost trees used by *Myotis septentrionalis* in areas that were and were not subjected to prescribed fire on the Fernow Experimental Forest, 2011 and 2012.

Variable	Burned (<i>n</i> = 11)			Unburned (<i>n</i> = 27)		
	Mean	Median	SE	Mean	Median	SE
ROOSTTREE						
DBHR (cm)	28.03	25.40	2.84	35.42	29.72	3.02
DECAYR	3.27	4.00	0.57	3.11	2.00	0.42
CROWNR	2.63	2.00	0.40	2.75	2.00	0.14
HEIGHT (m)	17.73	18.14	1.91	20.85	23.62	1.82
BARK (%)	70.45	75.00	7.43	69.44	75.00	5.71
TREECLUSTER						
DISTN (m)	0.37	0.38	0.03	0.32	0.30	0.02
DBHN (cm)	33.47	30.36	2.36	30.62	26.42	1.61
DECAYN	2.05	1.00	0.27	1.24	1.00	0.07
CROWNN	2.63	2.00	0.13	2.07	3.00	0.09
GAP						
TOTALGAP (%)	13.27	11.27	2.45	12.79	11.65	1.51
NORTHGAP (%)	18.63	21.14	2.68	10.45	10.00	1.63
EASTGAP (%)	8.68	9.04	1.08	12.40	10.31	1.98
SOUTHGAP (%)	8.70	7.00	2.51	12.86	9.16	2.05
WESTGAP (%)	15.07	9.75	4.19	11.19	7.47	1.75
OVERHEADGAP (%)	14.28	7.76	4.58	15.76	13.94	2.26
HORIZGAP (%)	16.94	11.90	4.00	16.94	14.26	2.08
SITE						
ASPECT (°)	146.27	120.00	23.88	185.52	188.00	15.75
SLOPE (%)	18.27	10.00	6.24	27.63	28.00	2.43
BASAL (sq.m./ha)	6.73	7.00	0.57	6.96	7.00	0.44

Appendix 2.f. Characteristics of roost trees used by female and male *Myotis septentrionalis* on the Fernow Experimental Forest, 2011 and 2012.

Variable	Female (<i>n</i> = 23)			Male (<i>n</i> = 15)		
	Mean	Median	SE	Mean	Median	SE
ROOSTTREE						
DBHR (cm)	34.86	29.72	2.86	30.87	24.89	4.04
DECAYR	3.30	4.00	0.40	2.93	2.00	0.61
CROWNR	2.90	3.00	0.10	2.50	2.00	0.27
HEIGHT (m)	18.88	23.16	2.12	19.87	20.42	2.34
BARK (%)	64.13	75.00	5.41	78.33	100.00	7.66
TREECLUSTER						
DISTN (m)	3.96	0.30	0.21	4.50	0.38	2.19
DBHN (cm)	29.46	26.04	1.70	34.50	29.72	2.11
DECAYN	1.66	1.00	0.15	1.00	1.00	0.60
CROWNN	2.01	2.00	0.93	2.57	2.00	0.67
GAP						
TOTALGAP (%)	15.27	15.39	1.88	9.35	9.36	0.95
NORTHGAP (%)	12.95	10.11	2.24	12.62	13.18	1.75
EASTGAP (%)	13.31	9.53	2.24	8.27	9.04	1.03
SOUTHGAP (%)	14.49	10.98	2.48	7.32	8.02	0.96
WESTGAP (%)	15.01	13.71	2.55	8.18	6.05	1.53
OVERHEADGAP (%)	19.17	20.58	2.93	9.44	7.78	1.82
HORIZGAP (%)	20.34	19.21	2.75	12.16	11.60	1.41
SITE						
ASPECT (°)	191.39	28.00	18.98	147.73	135.00	15.32
SLOPE (%)	33.30	10.00	2.74	12.07	10.00	2.41
BASAL (sq.m./ha)	7.00	7.00	0.50	6.73	7.00	0.48

CHAPTER 3

**EFFECT OF PRESCRIBED FIRE AND HERBICIDE TREATMENT ON SNAG
DYNAMICS IN A HARDWOOD FOREST IN WEST VIRGINIA**

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EFFECT OF PRESCRIBED FIRE AND HERBICIDE TREATMENT ON SNAG DYNAMICS IN A HARDWOOD FOREST IN WEST VIRGINIA

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ABSTRACT

Snags with cavities and exfoliating bark are an important component of stand structure for wildlife in hardwood forests. Understanding changes in decaying trees over time is needed to manage cavity-dependent biota. I quantified short-term (2006 – 2012) rates of initial tree death and progression through decay stages following fire and herbicide treatments initially applied in spring 2007 in the Alleghany Plateau of West Virginia. Red maple, chestnut oak, northern red oak and sugar maple were the most prevalent species found in measured plots. Northern red oaks were found to live the longest following fire treatment while black locusts were found to decay fastest after treatment. The greatest decline in trees in all treatments was in 2008, one year following initial fire and/or herbicide treatment, with a more gradual decline following in 2012. Transition of trees among decay classes differed by species and size. Prescribed fire alone had the least impact on overall mortality and decay progression for the treatments. Fire tolerance was also positively correlated with tree size, with larger yellow-poplar having the lowest mortality rate after prescribed fire treatment. Prescribed fire had the least amount of impact of the treatments in the rapid progression of trees through decay stages. All non-oak and non-hickory

trees found with >12.7 cm dbh in plots subjected to herbicide treatment showed a more rapid rate of decline than both fire only and midstory treatments. Snag longevity was significantly greater with increasing diameter for all treatments. Silvicultural practices that increase tree mortality rates and retain snags over long periods of time could be a beneficial approach to management for cavity-dependent wildlife in hardwood forests.

KEY WORDS: Herbicide, mortality, prescribed fire, snag dynamics, West Virginia

INTRODUCTION

The dynamics of snag creation, size, and density are subject to various influences throughout the life of the stand (Jones et al. 2009). Forest management practices have traditionally attempted to manipulate structure such that mortality is minimized (Vanderwel et al. 2006). More recently, the recognition of the ecological role of dead trees for wildlife has brought attention to the need for the provision of snags in forest practices (Harmon 2001). Sustainable forestry certification systems have emerged to ensure that the ecological impacts of management are studied and results integrated into management models (Jones et al. 2009). In order to ensure that snags and decaying trees are maintained in the landscape over time, it is important to understand the dynamics of snag creation and their progression over time.

Snag dynamics and the processes that create them in hardwood forests are an important component of wildlife conservation (McComb and Lindenmayer 1999). In many regions of the United States, prescribed fire is a common restoration and management technique in creating snags in forest ecosystems (Ladd 1991). Prior to European settlement in the eastern United States, much of the hardwood forests were thought to have been shaped through fires set by Native Americans (Cottam 1949, Nuzzo 1986, Ladd 1991). Fire is critical in regulating and maintaining a variety of forest ecosystems (Huddle and Pallardy 1996; van Lear 2002) and the

use of prescribed fire reintroduces an historic disturbance that may be lacking in other silvicultural practices (Boyles and Aubrey 2005). Fire can alter a variety of characteristics in a forest ecosystem, including acting as a mortality agent, which can increase the abundance of snags for wildlife. (Huddle and Pallardy 1996; Arthur et al. 1998; Hartman and Heumann 2003; Aubrey 2004). Fire also alters pathogen susceptibility, which can expedite the formation of cavities (Paulsen 1957; Smith and Sutherland 2001), and opens the forest canopy to more light penetration (Anderson and Brown 1986; Mcarty 1998; Aubrey 2004), which is known to affect roost and den selection (Kurta et al. 1993; Brigham et al. 1997). These characteristics along with a decrease in canopy coverage can be beneficial to cavity nesting and roosting fauna (Van Lear and Watt 1993).

Snags are an important structural component of many forest ecosystems (Harmon et al. 1986). Many wildlife species require standing dead trees as part of their habitat (Loeb and O'Keefe 2006) and maintaining adequate snag abundance and stages of decay is critical to conservation of biodiversity (DeLong et al. 2008). In the eastern United States specifically, The Forest Stewardship Council's forest certification standards (Principle 6.3.2b) requires managers to provide for a diversity of habitats such as declining trees and snags in an attempt to provide habitat for a variety of species (Forest Stewardship Council, 2005). Under natural conditions snags are created primarily by weather, fire, insects and disease (Morrison and Raphael 1993). Prescribed fire can lead to the creation and restoration of favorable cavity-dwelling habitat in addition to these natural processes (Boyles and Aubrey 2006). But the effect of prescribed fire on different species of trees varies based on fire intensity and bark thickness. Several studies have found that large snags tend to remain standing longer than small snags (Landrum et al. 2002; Garber et al. 2005). Snag longevity is greater with increasing diameter and decreases with

increasing stand density (Garber et al. 2005). Moreover, the agent of mortality and the type of treatment applied (e.g. prescribed fire or herbicide) can be important in snag survival (Garber et al 2005). Analyses of snag dynamics should include the species and condition of trees becoming snags as well as the factors causing the tree to die (Morrison and Raphael 1993).

Several studies have examined the effect of fire as well as the present condition of existing snags in hardwood forests. McComb et al. (1983) found higher snag densities in second-growth stands than in older-growth forests. Although particular species have been found more prone to the formation of snags (e.g. *Robinia pseudoacacia* and *Sassafras albidum*; McComb et al. 1983), diameter class remains a common determining factor in snag creation and persistence. Russell et al. (2012) found that the size of a snag was significant in estimating its survival, and snag survival increased by as much as 24% for each 10-cm increase in dbh. This was dependent on species, but due to the suppression of fire in the Central Appalachian forests in the eastern United States, species previously intolerant of fire have been allowed to exceed size classes large enough to tolerate cool to moderate burns (Franklin et al. 2003).

Species composition in hardwood stands can be controlled by selecting trees of desirable form and species and removing undesired trees (Erdmann 1983). In an ideal system, selected crop trees should grow faster and have a greater chance of obtaining and then maintaining dominance in a stand. Obstacles that affect regeneration in a stand include the large variety of species, stem quality, stem density, and growth differences among tree species (Wendel and Lamson 1987). Application of glyphosate (*N*-(phosphonomethyl)glycine), a broad-spectrum systemic herbicide, can be an effective way to eliminate competition around selected crop trees (Wendel and Lamson 1987). In previous studies conducted in West Virginia, common Appalachian hardwoods were top-killed and did not resprout after two years following injection

of glyphosate (Wendel and Kochenderfer 1982), and in the five years following, effects of release were still evident. Herbicide application can reduce sprouting by killing competing trees and leaving desired trees (e.g. oaks and hickories) free from competition. Hardwoods killed by herbicide also become snags (Dickson et al. 1995). Conner et al. (1981) found that herbicide-injected trees created snags with fungal heart rots that were capable of providing suitable cavities for wildlife species. Injection of herbicides can also be a safe and clean method of application, compared to the indirect spraying of herbicide, as herbicide is applied directly to the target trees, therefore eliminating some environmental effects (Wendel and Lamson 1987).

There is a need to improve knowledge of the impact of silvicultural actions, including herbicide use, on snag and coarse woody debris (CDW) dynamics in order to develop reliable data connecting management to stand-level habitat features and wildlife communities (Jones et al. 2009). The estimation of snag persistence, which can be defined as the probability of snag survival to a given age, is vital in the creation of management regimes that ensure their availability to wildlife (Garber et al. 2005).

The primary objectives of this study were to (1) identify quantitative changes in snag decay and survival dynamics in a second- and third-growth hardwood forests, (2) estimate the probability of snag survival using decay stage, and (3) quantify the effect of fire and herbicide on tree progression through stages of decay. I predicted that size class of hardwood trees would be positively related to fire tolerance and that all species allowed to reach a larger size class would remain tolerant to prescribed fire and herbicide regardless of bark thickness. I also predicted that trees injected with herbicide would decline at a rapid rate compared to trees subjected to fire only, and that herbicide would be effective in eliminating smaller diameter trees of various species.

METHODS

Study Area

Research was conducted on the Fernow Experimental Forest (FEF) (39.03° N, 79.67° S) located in Tucker County, West Virginia. The FEF is managed by the United States Forest Service, Northern Research Station. It is located in the central Appalachians approximately 5 km south of Parsons, West Virginia, in the Unglaciaded Allegheny Mountains subsection of the Appalachian Plateau Physiographic Province (Kochenderfer et al. 2007). The FEF was established in 1934 following the designation of the Elklick Run watershed for research and demonstration of forestry practices. When established, the FEF comprised 1,470 ha; it was expanded to approximately 1,900 ha in 1974 (U.S. Forest Service Experimental Forest and Range Network). Geographic features of the forest include an elevation range of 530 to 1115 meters, with Elklick stream running approximately south to north through the forest. Several tributaries intersect Elklick Run, creating over 5.5 kilometers of streams throughout the forest (Madarish et al. 2002). Mean annual precipitation is 145.8 cm per year (9.7 cm mid-October to 14.4 cm in June). The mean annual temperature is 9.2 ° C (18.0 in January to 20.6 in July) (Kochenderfer 2006).

The Fernow Experimental Forest vegetation consists of second- and third-growth hardwood stands. The second-growth forest is classified as mixed mesophytic, having characteristics of both mixed-oak and northern hardwood forest types (U.S. Forest Service Experimental Forest and Range Network). Vegetation on the FEF has been managed by even-aged (patch clear-cut), two-age, and uneven-aged (e.g., single-tree selection), and other partial harvesting and thinning silvicultural practices since the mid-20th century, or left undisturbed following initial harvesting in the Elklick watershed from 1903 to 1911 (Schuler and Fajvan 1999). These practices created a rich mosaic of stand structure. Tree species dominating the

forest include many shade-tolerant species such as maples (*Acer spp.*) and beech (*Fagus grandifolia*). Presently, the variety of tree species dominating this region consists of sugar maple (*Acer saccharum*), red maple (*A. rubrum*), yellow-poplar (*Liriodendron tulipifera*), American beech (*Fagus grandifolia*), sweet birch (*Betula lenta*), black cherry (*Prunus serotina*), and basswood (*Tilia americana*) (Schuler 2004) (Appendix 3.a).

Recently, prescribed fire has been utilized to encourage oak regeneration and accelerate snag creation in areas where clear-cutting and fire have both been underutilized and suppressed (Schuler 2004). Beginning in 2006, three separate experimental compartments were burned for one day using a strip-head fire technique in the FEF to aid in a better understanding of the effects of fire on regeneration and habitat enhancement. A prescribed fire treatment was applied to 152 ha, consisting of compartment 45 (127 ha), 13 (13 ha) and 21 (12 ha) (Figure 3.6). A total of 72 random plots were established throughout these 3 compartments: compartment 13 and 21 contained 6 plots each and compartment 45 contained 60 plots (Schuler, personal communication 2011). Each 20-m radius plot had plot center marked with a steel bar and the plot number painted on a tree. All trees >12.7 cm dbh were measured within each plot and identified with a nail and metal tag with a number at breast height. Species, size (cm), and decay stage was recorded for each tagged tree.

Additionally in 2007, 48 of the 72 plots were randomly selected and an herbicide (glyphosate) injection via “hack-squirt” or girdling method treatment applied. The 48 random plots were then identified as one of two treatment types: MIDSTORY or OVERSTORY (Appendix 3.f.). MIDSTORY plots ($n = 24$) consisted of herbicide application to trees with dbh of 12.7 – 27.9 cm, excluding all oak and hickory species. OVERSTORY plots ($n = 24$) consisted of herbicide application to trees with dbh >12.7 cm excluding all oak and hickory species. The

remaining 24 plots received only prescribed fire (FIRE treatment). All initial treatments were established in 2006 with subsequent re-treatment and re-measurement in 2008, 2009, 2010, and 2012 (Appendix 3.g.).

Data Collection. —In summer 2012, I re-measured all trees found within each of 65 of 72 plots (6 plots: compartment 13, 6 plots: compartment 21, 53 plots: compartment 45) (Figure 3.5). Seven plots were not re-measured due to inability to locate plots after regeneration of understory and elimination of plot boundary after tree-fall on steep slopes. I recorded: species, dbh, and decay stage (Appendix 3.d) for each tagged tree. Crown class was recorded for live trees (Appendix 3.e). Trees without tags were identified by relative location to surrounding tagged trees, species, and prior dbh recorded in 2010. If the tree number could not be identified based on previous data and the tree was standing and adequate in size (>12.7 cm dbh), data was excluded from analysis. If the tree was not located, it was assumed to have fallen and was recorded as coarse woody debris (CWD). In addition, potential roost type was recorded as cavity or adequate bark present. Decay stage and dbh were recorded first in 2006, before initial treatments were applied in 2007. Decay stage was based on previously utilized classification system by the Forest Service to indicate condition of trees (1 = alive, 2 – 7 = declining snag) (Appendix 3.d).

Quantifying Mortality Rates. —All tree species subjected to the prescribed fire treatment (FIRE) were placed into one of four categories based on fire tolerance according to Dey (2002): very sensitive, sensitive, intermediate, and resistant (Appendix 3.b). I only used tree species occurring at $\geq 1\%$ relative frequency in all plots combined in analyses. Tree diameter was divided into two groups: small diameter and large diameter, in order to determine rate of mortality based on size within species. Mean dbh for each species was used to place individuals into either the small diameter or large diameter category; those at or below the mean were placed

into small and those above were large. Mortality rates were calculated using total number present of a particular species, number of snags present in 2006 (before treatment) minus number of snags present in 2012. The final number of snags in 2012 was divided by the total number of live trees. These percentages were then averaged for each fire-tolerance level. The mortality rate for individual trees subjected to the FIRE treatment was calculated in order to quantify the effect of prescribed fire without herbicide injection on trees species. All oak and hickory species were included in the FIRE treatment mortality estimates. Oaks and hickories were removed from analysis of MIDSTORY and OVERSTORY treatment effects, as oaks and hickory species were not treated with herbicide as part of study design. Percent mortality for FIRE treatment plots was also calculated in order to quantify effects of prescribed fire alone without additional herbicide application on species.

Quantifying snag progression and survival. —Progression of all trees through decay stages from 2006 to 2012 was determined for each treatment. Trees that were not located in 2012 were considered no longer standing, and were classified as coarse woody debris (CWD). The percentage of trees identified in each decay stage was calculated based on total number of trees present (n) in 2006 and again in 2012. These progression percentages through decay stages were separated by treatments. Oak and hickory species were removed from count analysis of MIDSTORY and OVERSTORY treatments.

To determine the effect of tree size (dbh cm) on tree mortality and survival, all species present with $\geq 1\%$ frequency were divided into 1 of 6 size classes: size class 1 (10 – 19.9 cm dbh), size class 2 (20 – 29.9 cm dbh), size class 3 (30 – 39.9 cm dbh), size class 4 (40 – 59.9 cm dbh), size class 5 (60–79.9 cm dbh), and size class 6 (80 – 100 cm dbh) (Appendix 3.h.). The number of years a particular tree remained alive after treatment (YEARSALIVE) was calculated

based on decay stage. Trees identified as live (1) in 2006 were given a decay stage classification for every year following (with the exception of 2011). A tree was identified as live until given a decay classification of 2 or higher.

I calculated survival rate for each treatment using a Kaplan-Meier estimator (JMP, Version 10. SAS Institute Inc., Cary, NC, 2010). Survival rate was estimated for various size classes. Survival rate was also estimated for size class by each treatment type from 2006 to 2012. Oak and hickory species were excluded from survival analysis. An analysis of variance (ANOVA) was used to determine if the effect of treatment type on all sizes and species (excluding oak and hickory) was different and if survival of trees differed based on size class. The average number of years that a tree remained living after treatment in 2007 within each size class was calculated to determine the effect of size on each treatment type.

RESULTS

Mortality Rates. —Within the FIRE treatment, yellow-poplar (21.4%), black oak (*Quercus velutina*) (28.6%), and bitternut hickory (*Carya cordiformis*) (28.6%) had the lowest percent mortality for the small dbh size class. Black locust (100%) and scarlet oak (*Quercus coccinea*) (90%) had the highest percent mortality for small-diameter trees. For large-diameter trees, black gum (*Nyssa sylvatica*) (0%) and yellow-poplar (12.5%) had the lowest percent mortality, while black locust (100%) and American beech (77.3%) had the highest percent mortality (Table 3.1). Trees designated as resistant to fire had the lowest mean percent survival (23.3%), while trees grouped as having intermediate resistance to fire had the highest mean percent mortality (66.7%) (Table 3.1).

Snag progression and survival. —Within treatments, all species were assigned a decay stage value based on condition from 2006 to 2012 (Figure 3.1). The total number of trees

subjected to FIRE ($n = 788$), MIDSTORY ($n = 782$), and OVERSTORY ($n = 789$) were aggregated by decay stage. In 2006, FIRE treatment consisted of 91% alive, MIDSTORY had 91% alive, and OVERSTORY had 88% alive prior to treatment. By 2012, 41% of trees in FIRE treatment remained in decay stage 1 (alive), 17% of MIDSTORY, and 1% of OVERSTORY trees remained living. For FIRE treatment, the largest percent of trees remained in the first 2 decay stages (decay stage 2 = 19%; recently dead/declining). FIRE treatment also resulted in the lowest percent of CWD trees (20%) (Table 3.2). For MIDSTORY treatment, the largest percentage of trees remained in decay stage 4 (14%; broken top/loose bark) and 5 (18%; few remaining limbs/no bark) (Table 3.3). For OVERSTORY treatment, the majority of trees also remained in decay stage 4 (24%), 5 (14%) and 6 (14%; broken bole/down material/no limbs) (Table 3.2). Trees subjected to OVERSTORY treatment also had the greatest percentage of downed trees (CWD = 39%).

Based on the Kaplan-Meier survival estimate values for the three treatments, FIRE treatment maintained the highest rate of survival from 2006 to 2012 (Figure 3.2). The lowest survival rate was in 2012, where 66% of trees in the FIRE treatment remained alive while 22% of those in MIDSTORY treatment and 11% in OVERSTORY treatment remained living 4 years after initial treatment (Table 3.2). After 2008, survival rates declined most severely, with a change from 73% to 23% in MIDSTORY and 68% to 2% in the OVERSTORY treatments, respectively. FIRE treatment allowed for the slowest rate of decline over 5 years, with most substantial loss occurring after 2010 (Figure 3.2). MIDSTORY and OVERSTORY treatments resulted in similar rates of decline following treatments in 2007, and resulted in the most substantial loss after 2008. After 2009, both MIDSTORY and OVERSTORY treatments resulted

in more gradual decline of trees (Figure 3.2). The survival of trees among years was significantly different ($P < 0.001$) among treatment types ($\chi^2 = 528.16$, $df = 2$).

A Kaplan Meier survival estimate was calculated for trees in each of the 6 size classes (Figure 3.3). Size class 1 (10–19.9 cm dbh) trees had the lowest probability of survival after 5 years of treatment: 88% survival rate in 2006 to 9% survival rate in 2012 (Figure 3.3). Trees in size class 6 (80 – 100 cm dbh) had 100% survival rate after 5 years of treatment. The probability of survival of trees among years was significantly different ($P < 0.001$) among size classes based on the Wilcoxon non-parametric test ($\chi^2 = 294.42$, $df = 5$).

Because I found evidence of significant differences among both treatment types and size classes, I further analyzed the effects of size class within each treatment type (Figure 3.4). Size class 1 had the lowest probability of survival for all three treatment types compared to all other size classes (FIRE = 0.27, MIDSTORY = 0.00, OVERSTORY = 0.00) after 6 years, while higher size classes had a higher probability of survival, regardless of treatment type (e.g. size class 4: FIRE = 0.74, MIDSTORY = 0.73, OVERSTORY = 0.05). After 2008, OVERSTORY herbicide treatment had a detrimental effect on all trees, regardless of size class. FIRE treatment had the most gradual effect on decline of all trees (Figure 3.4).

DISCUSSION

Tree mortality rates are one of the key factors controlling the long-term dynamics of forest ecosystems (Dietze et al. 2011). Prescribed burning and its effect on mortality has been shown to alter succession in various communities (Luken, 1990). Several of the mesophytic species normally considered fire intolerant have attained large enough diameter in eastern deciduous forests to be tolerant of cool to moderate burns (Franklin et al. 2003). The mortality rates of larger trees in the FEF (>60 cm dbh), regardless of species or previously designated fire-

tolerance level, were found to resist, at least in the short-term, the detrimental effects of fire and herbicide application. Smaller diameter trees were eliminated from the plots 6 years following treatment of fire and herbicide. This rate of mortality on smaller diameter trees is effective in eliminating small, shade-tolerant species such as *Acer* spp., allowing for the regeneration and competitive advantage of *Quercus* spp. The overall low mortality rate of trees such as yellow-poplar can be associated with the relatively larger average mean dbh of those present in the FEF.

Based on the rate of decay observed in the FEF, it was also determined that the rate at which larger trees decay is slower than that of smaller trees. Large trees have a greater proportion of decay-resistant heartwood than smaller ones (Sellin 1994). This characteristic may extend the time larger trees are able to withstand the effects of decay from rot caused by fire and herbicide before falling (Vanderwel et al. 2006). Inherent differences in wood density and the concentration of lignin may also result in different rates of decay among hardwood species (Harmon 2001). The agent of mortality can be an important factor in determining snag longevity (Garber et al. 2005). On the FEF, both the size of trees and the treatment (fire alone or fire in combination with the application of herbicide) was significantly associated with survival and rate of decay. Trees killed by fire decayed slower than those injected with herbicide after being subjected to fire. Results from this study suggest that for smaller diameter trees, fire can be an effective tool in snag creation, while herbicide treatment in addition to fire leads to a more rapid decline and subsequent rapid falling of the majority of trees (<5 years). Such expedited decline to a CWD stage can be ineffective for the creation of snags for wildlife requiring standing trees with cavities, crevices, and exfoliating bark. For larger trees, regardless of bark thickness, fire alone may provide a management technique for snag creation of larger, longer standing trees. Based on the observed rate of decline among treatments, fire is effective in eliminating small

diameter trees, which can be beneficial in the elimination of shade-tolerant species (e.g. *Acer spp.*). Fire treatment also allowed for larger diameter trees to remain standing longer than those treated with herbicide injection. These longer-standing, declining snags are necessary in creating habitat for wildlife species that demonstrate philopatry to particular locations and particular trees in hardwood forests.

Although responses varied among species treated with fire alone and fire with herbicide, all treatments were effective in the creation of snags. Comparatively, prescribed fire alone leads to a slower decline in snag development. Yellow-poplar, a tree identified as resistant to fire in mature stage responded to FIRE treatment with the lowest rate of decline through decay stages (21.4% mortality for small trees; 12.5% for large trees; size class determined by mean).

Following the application of herbicide, yellow-poplar responded with a mortality rate similar to FIRE (17.0%). My results indicate that small hardwood trees (smallest diameter = 12.7 cm) treated with FIRE had the highest mortality rate of all three treatments, which suggests that prescribed fire alone can be effective in eliminating small diameter trees without the use of herbicide. Mortality rates for smaller dbh trees remained consistently higher for all species after FIRE treatment as compared to larger dbh trees. Russell et al. (2012) found that snag survival and decomposition is related to snag dbh and average time in a given decay stage rather than tree species. These results found on the FEF also suggest that fire is successful in achieving a slower decline rate than the use of herbicide.

Rates of survival are a helpful tool in predicting rate of decline over time. The ability to estimate snag persistence requires adequate time for analysis. Estimated rates of mortality and decay stage progression using short-term records (<20 years) (Morrison and Raphael 1993; Mellen and Ager 2002) will likely result in the overestimation of snag progression rates (DeLong

et al. 2008). Morrison et al. (1993) found that snags tended to remain in the most decayed stages for at least 5–10 years before falling. Although there was a net increase in snag creation in the FEF from 2006 to 2012, most of the increase was due to the decay of smaller (10–30 cm dbh) trees, regardless of species. Species of snag was found to be relatively less important with regard to snag longevity than the method of creation or tree diameter. For OVERSTORY treatments, a smaller percentage of trees remained in decay stage one, indicating a faster progression through snag stages (only 21.8% of trees remained alive after treatment while 47.2% of trees remained in the decay stage 1 after FIRE treatment) with a near complete loss of all trees, with the exception of two larger-diameter trees. Trees subjected to FIRE showed slower progression into higher decay stages. MIDSTORY treatment plots had the highest transition to mid-decay stage (4) at 40.8% of total trees measured. This progression towards a mid-level stage of senescence may be beneficial. Rapid loss (large number of trees moving to high decay stages) as observed in OVERSTORY treatment plots without replacement of snags could be counterproductive with the loss of potential living trees that could be recruited into the snag population (Morrison et al. 1993).

A tree species' fire rating will vary depending on fire intensity, duration of fire, time of year and the maturity of the trees (Lambert 2001). Trees of different ages can vary in their resistance, thus requiring size to become a critical variable. Fire damage can also be affected by the season in which the burn is applied. Hot, intense, slow-burning fires will kill more trees than less intense, fast-burning fires (Lambert 2001). Fire treatment on the FEF was not monitored for intensity (i.e., temperature), yet the impact of fire combined with herbicide injection had a substantial impact of snag creation on a variety of species. Based on rate of progression through

decay stages and number of years a tree remained standing after treatment, FIRE had the most gradual effect on tree senescence.

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Table 3.1. Percent mortality for tree species (>1% frequency) subjected to FIRE treatment in the Fernow Experimental Forest, West Virginia, 2012 (fire tolerance based on Gleason 1913: *The Relation of Forest Distribution and Prairie Fires in the Middle West*).

Fire tolerance	Number of stems	Small diameter (cm)	SE	% mortality	Large diameter (cm)	SE	% mortality
Very sensitive							
Sourwood (<i>Oxydendron arboreum</i>)	131	16.3	0.2	75.0	24.8	0.7	53.7
Black locust (<i>Robinia pseudoacacia</i>)	20	17.7	1.0	100.0	21.9	2.5	100.0
				87.5			76.9
Sensitive							
Sweet birch (<i>Betula lenta</i>)	139	17.7	0.5	60.5	31.5	1.2	54.8
Cucumbertree (<i>Magnolia acuminata</i>)	37	22.9	0.7	63.2	36.1	1.7	41.2
Sugar maple (<i>Acer saccharum</i>)	120	16.3	0.3	51.5	31.9	1.5	39.2
Red maple (<i>Acer rubrum</i>)	221	16.7	0.2	60.2	30.1	0.7	54.7
American beech (<i>Fagus grandifolia</i>)	80	15.7	0.4	60.0	22.4	1.1	77.3
				59.1			53.4
Intermediate							
Scarlet oak (<i>Quercus coccinea</i>)	19	33.0	2.0	90.0	53.3	2.6	33.3
Resistant							
Yellow-poplar (<i>Liriodendron tulipifera</i>)	60	30.7	1.9	21.4	50.6	1.3	12.5
Black gum (<i>Nyssa sylvatica</i>)	10	17.2	1.4	66.7	35.1	7.6	0.0
Black oak (<i>Quercus velutina</i>)	14	36.1	1.7	28.6	51.2	2.0	28.6
Bitternut hickory (<i>Carya corfiiformis</i>)	11	19.6	2.0	28.6	45.7	5.0	25.0
White oak (<i>Quercus alba</i>)	47	20.3	1.0	60.9	43.7	2.5	23.5
Chestnut oak (<i>Quercus prinus</i>)	159	26.3	0.7	30.3	43.6	0.7	26.8
Northern red oak (<i>Quercus rubra</i>)	83	30.6	1.3	43.2	56.9	1.5	35.6
				40.0			21.7

Table 3.2. Comparison of decay stage by treatment excluding oaks and hickories for plots measured between 2006 and 2012 in the Fernow Experimental Forest.

Treatment		Decay Stage															
FIRE <i>n</i> = 788	1	%	2	%	3	%	4	%	5	%	6	%	7	%	CWD	%	
	2006	719	91	5	1	5	1	16	2	14	2	27	3	1	0.00	0	0.00
	2012	326	41	150	19	63	8	29	4	28	4	30	4	0	0.00	161	20
MIDSTORY <i>n</i> = 782	1	%	2	%	3	%	4	%	5	%	6	%	7	%	CWD	%	
	2006	712	91	5	1	7	1	20	3	11	1	23	3	4	1	0	0.00
	2012	132	17	35	4	27	3	131	17	141	18	108	14	0	0.00	208	27
OVERSTORY <i>n</i> = 789	1	%	2	%	3	%	4	%	5	%	6	%	7	%	CWD	%	
	2006	698	88	4	1	10	1	26	3	10	1	37	5	4	1	0	0.00
	2012	5	1	4	1	44	6	189	24	111	14	114	14	11	1	311	39

Figure 3.1. Percent trees present in decay stage in 2006 and 2012 and progression among decay stages by treatment type excluding oaks and hickories in the Fernow Experimental Forest, 2006–2012.

Figure 3.2. Probability of trees **excluding oaks and hickories** surviving from 2006 to 2012 based on response to one of three treatments, on the FEF. FIRE = fire only; MIDSTORY = fire + herbicide (all non-oaks 12.7-27.9 cm dbh); OVERSTORY = fire + herbicide (all non-oaks 12.7+ cm).

Figure 3.3. Probability of trees **excluding oaks and hickories** surviving from 2006 to 2012, based on size clas, on the FEF. Size class: 1 = 12.7-19.9 cm, 2 = 20-29.9 cm, 3 = 30-39.9 cm, 4 = 40-59.9 cm, 5 = 60-79.9 cm, 6 = 80-100 cm.

Figure 3.4. Probability of trees **excluding oaks and hickories** surviving from 2006 to 2012, based on size class within treatment, on the FEF. Size class: 1 = 12.7-19.9 cm, 2 = 20-29.9 cm, 3 = 30-39.9 cm, 4 = 40-59.9 cm, 5 = 60-79.9 cm, 6 = 80-100 cm.

Fernow Experimental Forest Treatment Plots Tucker County, WV

Figure 3.5. Three compartments established in 2006 on the Fernow Experimental Forest and fire/herbicide plots, Tucker County, West Virginia.

Appendix 3.a. Common species and shade tolerance ratings of trees found in the Fernow Experimental Forest, West Virginia, 2011 and 2012 (*Silvics of Forest Trees of the United States, USDA Forest Service 1965*).

Species	Species code	Tolerant	Mid-tolerant	Intolerant
American beech (<i>Fagus grandifolia</i>)	32	X		
Black gum (<i>Nyssa sylvatica</i>)	85	X		
Red maple (<i>Acer rubrum</i>)	79	X		
Sugar maple (<i>Acer saccharum</i>)	76	X		
Bitternut hickory (<i>Carya cordiforma</i>)	21		X	
Black oak (<i>Quercus velutina</i>)	47		X	
Chestnut oak (<i>Quercus prinus</i>)	38		X	
Cucumbertree (<i>Magnolia acuminata</i>)	53		X	
Northern red oak (<i>Quercus rubra</i>)	40		X	
Shagbark hickory (<i>Carya ovata</i>)	25		X	
Sourwood (<i>Oxydendrum arboreum</i>)	88		X	
Sweet birch (<i>Betula lenta</i>)	29		X	
White oak (<i>Quercus alba</i>)	34		X	
Black locust (<i>Robinia pseudoacacia</i>)	71			X
Sassafras (<i>Sassafras spp.</i>)	58			X
Scarlet oak (<i>Quercus coccinea</i>)	41			X
Yellow-poplar (<i>Liriodendron tulipifera</i>)	56			X

Appendix 3.b. Common hardwood species in the Fernow Experimental Forest (>1% frequency) by vulnerability, to long-term repeated burning (*Gleason 1913, Garren 1943, Curtis 1959, Heinselman 1973, Grimm 1984, Harmon 1984, Lorimer 1985, Burns and Honkala 1990, Boyer 1990, 1993, Nowacki and Abrams 1992, Orwig and Abrams 1994, Huddle and Pallardy 1996, Hicks 1998, Brose and Van Lear 1998*).

Very sensitive	Sensitive	Intermediate	Resistant
Sourwood* (<i>Oxydendron arboreum</i>)	Red maple (<i>Acer rubrum</i>)	Scarlet oak (<i>Quercus coccinea</i>)	Hickories (<i>Carya spp.</i>)
Black locust* (<i>Robinia pseudoacacia</i>)	Sugar maple (<i>Acer saccharum</i>)		Yellow-poplar (mature) (<i>Liriodendron tulipifera</i>)
	Sweet birch (<i>Betula lenta</i>)		White oak (<i>Quercus alba</i>)
	American beech (<i>Fagus grandifolia</i>)		Chestnut oak (<i>Quercus prinus</i>)
	Cucumbertree (<i>Magnolia acuminata</i>)		Northern red oak (<i>Quercus rubra</i>)
	Fraser magnolia (<i>Magnolia fraseri</i>)		Black gum (<i>Nyssa sylvatica</i>)
	Sassafras (<i>Sassafras spp.</i>)		Black oak (<i>Quercus velutina</i>)
	American basswood (<i>Tilia Americana</i>)		

*Species added to previous table based on short-term resistance to burning in the FEF, 2006-2012.

Appendix 3.c. Total and divided area of compartments and areas of treatments by acre and hectare, including percent area of treatment plots established in 2006 in the Fernow Experimental Forest.

Compartment	Hectares	% of total
13	13.0	8.9
21	12.0	8.2
45	127.0	82.9
Total	152.0	

Treatment	Hectares	% of total
Each 20-m radius plot	0.13	0.1
FIRE only plots	3.0	2.0
FIRE + HERBICIDE plots	6.0	3.9
Total plots w/in compartments	9.0	5.9

Appendix 3.d. Decay stage designation and the characteristics of trees in those decay stages
(USDA Forest Service based on decay of Douglas-fir snags as adapted by Maser et al. 1979).

Decay stage	Forest Service code	Characteristics
1	1 – 2.15	alive (1 = good stem, 2.0 – 2.15 = poor stem)
2	2.16	recently dead, declining
3	2.17	dead, few limbs missing
4	2.18	broken top, loose bark
5	2.19	few limbs remaining, no bark
6	2.20	broken bole, down material, no limbs
7	2.21	decomposed

Appendix 3.e. Commonly used crown classes and the characteristics of trees in those classes (*Kraft 1884; Oliver and Larson 1990; Smith et al. 1997; Helms 1998*).

Crown class	Definition	Characteristics
1	dominant	<ul style="list-style-type: none"> • crown extends above the general canopy layer for the stand • crown intercepts direct sunlight across the top and along sides of the upper branches
2	co-dominant	<ul style="list-style-type: none"> • crown within and helping to form the main crown canopy for the stand • crown intercepts direct sunlight across the top, but only at tips of the upper side branches
3	intermediate	<ul style="list-style-type: none"> • crown extends somewhat into the lower part of the main canopy • crown intercepts direct sunlight only at a limited area on the top and none at the sides
4	suppressed	<ul style="list-style-type: none"> • crown entirely below the main canopy and covered by branches of taller trees • no direct sunlight strikes at any portion of the crown

Appendix 3.f. Treatment type used in the three treatment compartments in the FEF (*USDA Forest Service*).

Treatment	Characteristics
FIRE	<ul style="list-style-type: none">• prescribed fire only, no herbicide
MIDSTORY	<ul style="list-style-type: none">• all non-oaks 12.7-27.9 cm dbh treated with herbicide + fire
OVERSTORY	<ul style="list-style-type: none">• all non-oaks >12.7 cm dbh treated with herbicide + fire

Appendix 3.g. Treatment type and timeline of treatment application for 3 compartments in the Fernow Experimental Forest, 2006 – 2012.

COMPARTMENT 13 & 21	
Date	Activity
2006	plots established & measured
2007a	prescribed fire
2007b	plots remeasured
2007c	herbicide treatment
2008a	plots remeasured
2008b	herbicide treatment
2009a	prescribed fire*
2009b	plots remeasured
2010	plots remeasured
2012	PLOTS REMEASURED

COMPARTMENT 45	
Date	Activity
2006	plots established & measured
2007a	prescribed fire
2007b	plots remeasured
2007c	herbicide treatment
2008a	prescribed fire*
2008b	plots remeasured
2008c	herbicide treatment
2009	plots remeasured
2010	plots remeasured
2012	PLOTS REMEASURED

* Compartment 45 received prescribed fire in alternate year from compartments 13 and 21.

Appendix 3.h. Diameter at breast height (dbh) in cm of trees found in the Fernow Experimental Forest.

Size class	Definition
1	12.7-19.9 cm dbh
2	20-29.9 cm dbh
3	30-39.9 cm dbh
4	40-59.9 cm dbh
5	60-79.9 cm dbh
6	80-100 cm dbh