EFFECTS OF SPRING PRESCRIBED FIRE ON CHIPMUNK HOME RANGES, AND A WOODLAND SALAMANDER COMMUNITY IN A CENTRAL APPALACHIAN HARDWOOD FOREST

by

ELLA ROWAN

Under the Direction of Steven Castleberry

ABSTRACT

Forest managers wish to improve oak regeneration in many Appalachian forest types through the use of prescribed fire, however the effects of prescribed fire on fauna communities in these systems are unknown. The objectives of my studies were to examine impacts of spring prescribed fire on eastern chipmunk home range attributes and the woodland salamander community. I used radiotelemetry to examine chipmunk burrow use, home range and core area sizes, and extent of overlap between conspecifics. Results showed prescribed fire had no effect on these attributes, and there were no differences between the sexes. I used coverboard arrays, pre-fire and post-fire surveys, and control groups, to assess the impacts of fire on woodland salamander relative abundance, species richness, body condition and seasonal trends in the presence of different species. Results suggest spring prescribed fire did not have negative short-term effects on the salamander community, with no decreases in relative abundance or species richness. Seasonal changes in the presence of different species aboveground were evident.

INDEX WORDS: Prescribed fire, Chipmunk, Radiotelemetry, Home range, Salamander
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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW
INTRODUCTION

Eastern forest plant species composition has changed over the past century due to timber harvesting, grazing, introduced pests, increased white-tailed deer (*Odocoileus virginianus*) populations, and changes in both natural and anthropogenic fire regimes. Oak (*Quercus* spp.) abundance has been declining in many Appalachian forest types, and forest managers wish to improve regeneration for both economic and ecological reasons (Schuler 2004). The combined use of prescribed fire and shelterwood harvests to improve regeneration appears promising (Brose et al. 1999), however the re-introduction of fire to ecosystems that have been under fire suppression for a century will have unknown effects. In addition, many mesic regions where prescribed fire could be employed, have unknown historic fire intervals. It is important to monitor the effects of silvicultural manipulations on flora and fauna communities to ensure the integrity of the ecosystem is not diminished. Retention of native species, species diversity, and abiotic factors associated with species requirements are important factors for overall ecosystem health (Brose et al. 2001).

Regenerating oak forests through the use of prescribed fire will ultimately benefit many species of wildlife that utilize oaks for mast. However, the initial effects on fauna survival, community structure, and population dynamics within these burned oak habitats are unclear. The eastern chipmunk (*Tamias striatus*) represents an ideal study organism for examining the effects of prescribed fire on mammalian fauna in the eastern hardwood forest, because it is a common ground-dwelling species, heavily reliant upon oak mast, but which also consumes large quantities of alternative forage during spring and summer months.

Although chipmunk survival post-prescribed fire is of interest, data about changes in home range size and territoriality post-prescribed fire also are lacking in the eastern hardwood forests.
Chipmunk home ranges are not static, with size being related to resource availability (Allen 1938, Mares et al. 1976, Getty 1981b, Mares et al. 1982, Mares and Lacher 1987, Bowers et al. 1990) and reproductive condition (Bowers and Carr 1992). As larder-hoarding animals, caches are vital to survival during low forage/food availability periods, so cache defense may limit movement distances from burrows (Elliott 1978, Anderson 1978, Taylor 1988). Predation risk also is a factor influencing the distance animals will venture from their burrows (Elliott 1978, Anderson 1978, Taylor 1988). Prescribed fire can eliminate, or alter, forest understory structure and composition, and may decrease spring and summer forage resources for chipmunks. According to previous research, decreased resource availability causes an increase in home range size for some mammals (Harestad and Bunnell 1979, Mares et al. 1982, Litvaitis et al. 1986, Ostfield 1986, Ims 1987, Jones 1990, Castleberry et al. 2001). However, chipmunks may decrease home range size and increase territoriality to protect their remaining hard mast cache, as well as in response to increased predation risk.

The effects of prescribed fire on woodland salamander communities are virtually unknown in the mesic areas of the Allegheny Plateau. Pilliod et al. (2003) summarized studies conducted in the eastern United States, and found variable results in salamander response to fire, depending on region and season. Additionally, high white-tailed deer densities in the eastern deciduous forests can cause dramatic changes in forest structure and plant composition (Tilghman 1989, Trumbull and Zielinski 1989, Waller and Alverson 1997), which have unknown effects on the salamander community. Habitat changes that reduce leaf litter and vegetative cover, increase solar exposure, and decrease humidity on the forest floor can increase salamander physiological stress and predation risk. Similarly, silvicultural manipulations can alter invertebrate prey resource availability and increase competition in salamander communities (Pilliod et al. 2003).
My first experiment investigated the effects of spring prescribed fire on chipmunk home range size and territoriality. The specific objectives were to:

1. Determine the effects of spring prescribed fire on eastern chipmunk home range size and conspecific home range overlap.
2. Determine differences in home range size and extent of overlap between sexes.

My second experiment investigated the effects of spring prescribed fire and deer browsing on a woodland salamander community. The objectives included:

1. Determine the effects of spring prescribed fire on woodland salamander abundance and species richness.
2. Determine the effects of deer browsing on woodland salamander abundance and species richness.
3. Examine seasonal trends in the abundance of various woodland salamander species.

LITERATURE REVIEW

Oak Regeneration

The Central Hardwood Region of the eastern United States encompasses over 89 million hectares and includes the deciduous broadleaf forests of the Allegheny Plateau in the central Appalachians. Historically, this region is thought to have experienced an increase in oak dominance during the early Holocene (10,000 years B.P.) because of warmer, drier conditions and an increase in fire occurrence (Watts 1979, Davis 1985, Webb 1988). Lightning strike fires were largely responsible for maintaining mixed-oak forests in their mid-successional seral stage (Day 1953, Watts 1979, Lorimer 1985, Patterson and Sassaman 1988, Martin 1990, Abrams 1992, Delcourt and Delcourt 1998), although fire also was used by Native Americans to clear
land for agriculture and to maintain open understories for improved hunting (Pyne 1982, DeVivo 1991). While Native American-set fires generally were localized around habitation, occasional wildfires occurred, leading to widespread favorable conditions for oak forests (Russell 1983, Patterson and Sassaman 1988).

European settlement within the Central Hardwood Region began in the late 1700’s, and led to numerous forest disturbances. Settlers continued the Native American tradition of using fire to clear land for agriculture and grazing, but also began the charcoal-iron industry, particularly in the central Appalachians in Virginia and Pennsylvania (VanLear and Waldrop 1989, Abrams 1992). The late 1800’s and early 1900’s also saw a logging boom. This series of disturbances is thought to have increased oak levels beyond pre-settlement levels (Abrams 2001). The loss of American chestnut (*Castanea dentata*) to chestnut blight (*Cryphonectria parasitica*) was one of the most ecologically and economically devastating occurrences of the past century, however, this too allowed oak species to increase in dominance after 1930 (Weitzman 1949, Abrams 2001, Johnson et al. 2002).

Factors allowing for increased prevalence of oak until the 1930’s within eastern forests were counteracted by 3 factors decreasing oak abundance after this time. Fire suppression efforts in the United States, since the 1930’s, are thought to have allowed an increase in later-successional tree species in forest understory (Abrams 1992 and 1998, Brose et al. 2001, Schuler 2004). Also, gypsy moth (*Lymantria dispar*), introduced to New England in 1869, eventually caused large reductions in oak abundance, with its spread to the south and west. Oaks appear to be a favored forage plant (Bess et al. 1947, Gasner and Herrick 1987), and outbreaks have caused extensive defoliation and high mortality rates within forests. Lastly, other suspected causes of oak regeneration declines include insect and vertebrate predators on acorns and oak seedlings.

Recently, forest managers have noted a continued decrease in understory oak as a component of eastern hardwood forests (Pallardy et al. 1988, Spencer and Kingsley 1991, Stephens and Ward 1992). Schuler and Gillespie (2000) investigated species composition in a mesic Allegheny Plateau forest, and found significant decreases in species diversity over the past 100 years. They found northern red oak (Q. rubra) levels have decreased in the understory since 1948, and forest understories are increasingly dominated by sugar maple (Acer saccharum). The abundance of sugar maple in the understory will allow this species to fill in overstory gaps in the future, and slowly replace northern red oak as a co-dominant species in the overstory.

Decreasing oak dominance may be a problem for many wildlife species that rely upon oaks for mast (Wentworth et al. 1992, Johnson et al. 1995), as well as for the timber industry which finds many oak species, particularly northern red oak, economically desirable. For these reasons, forest managers are experimenting with prescribed fire and partial-harvesting techniques in an attempt to enhance oak regeneration to ensure oaks continue to comprise a significant proportion of future stocking.

Bringing oaks back into dominance has proved to be a difficult task for silviculturalists. Clearcut and shelterwood cutting alone does not enhance regeneration, but instead allows faster-growing, shade-intolerant species such as yellow poplar (Liriodendron tulipifera) to take over the new stands (Loftis 1983, Beck and Hooper 1986, Lorimer 1993). Partial harvesting also can increase dominance of more shade-tolerant species, such as sugar maple and red maple (A.}
rubrum), in the understory (Abrams 1992, Schuler and Miller 1995, Schuler and Gillespie 2000). In areas where mountain laurel (Kalmia latifolia) and rhododendron (Rhododendron sp.) inhibit oak regeneration in the understory, intense fires, which open the overstory, appear beneficial to oak regeneration efforts (Moser et al. 1996). Kruger and Reich (1997) in a mesic hardwood forest of Wisconsin, and Brose and Van Lear (1998) in the Piedmont of Virginia both found opening the overstory with partial harvesting, followed by prescribed fire, significantly improved oak regeneration. Nonetheless, Vose (2000) cautioned land managers to carefully examine current ecosystem conditions, and apply appropriate prescribed fire techniques to achieve a healthy and sustainable ecosystem. Following the long period of fire suppression, the reintroduction of prescribed fire within Central Appalachian forests to facilitate oak regeneration and survival requires a full understanding of the impacts on wildlife and herbaceous plant communities.

*Eastern Chipmunk Natural History and Responses to Silviculture in the Appalachians*

The eastern chipmunk is one of 22 species of chipmunks inhabiting North America. Its range encompasses most of eastern North America, from southern Canada to the Gulf States (Whitaker and Hamilton 1998). They are solitary and diurnal animals, spending the majority of their time foraging on the forest floor (Snyder 1982), and are most commonly associated with mature deciduous hardwood forest habitats (Whitaker and Hamilton 1998).

Eastern chipmunks are larder-hoarding animals that rely on both hard and soft mast, vegetative matter, fungi, and invertebrates for forage (Wrazen and Svendsen 1978). Over 60% of their diet consists of hard mast (i.e., acorns) they store in their underground burrows for year-round consumption and during periods of low-forage availability (Elliott 1978, Wrazen and

Numerous studies have examined eastern chipmunk population responses to timber harvesting, and have found them to be positively associated with closed overstory and open understory forests (Dueser and Shugart 1979, Bowers 1995). Although clearcutting removes all overstory trees, research has shown mixed results with eastern chipmunk population responses. Krull (1970) in New York, and Kirkland (1977) in West Virginia, both found chipmunk abundance to decrease the first five years post-clearcut in deciduous forests, whereas Lovejoy (1975) found chipmunk abundance to increase in New Hampshire. Possible causes for abundance decreases include a reduction in mast and an increase in predation risk due to the loss of overstory trees. Kirkland (1977) found relative abundance returned to uncut forest levels by the 6th year after clearcutting, and Mahan and Yahner (1998) in Pennsylvania and Krull (1970) found densities returned to pre-cut levels by the 10th year. Dense, woody regeneration from stump sprouts and seedlings that established by the 6th year provided chipmunks with significant quantities of soft mast and cover during the summer. Kirkland et al. (1985) found chipmunk abundance along clearcut peripheries in Pennsylvania actually increased due to soft mast availability.

The effects of prescribed fire on chipmunk populations in the northcentral and northeastern United States are virtually unknown. Tester (1965), and Krefting and Ahlgren (1974) conducted fire studies in savannas and mixed conifer-hardwood forests of Minnesota, but found impacts on
chipmunk abundance unclear. Studies of other small mammal species, such as shrews, mice, and voles, in the eastern hardwood region of the central Appalachians in Pennsylvania (Kirkland et al. 1996), and the southern Appalachians in North Carolina (Ford et al. 1999), found the effects of prescribed fire on abundance and species assemblage to be minimal or transitory, lasting less than 8 months. Although their focal small mammal species differ from chipmunks in forage types, foraging strategies, burrow depths, and escape capabilities, similarities may exist relative to animal responses to predation exposure, forage loss, and fire-escape behavior. In North Carolina, Ford et al. (1999) found trees and large coarse woody debris (which provide small mammals with cover and forage substrate) were not affected by the fire in most areas. They also suggest the fossorial nature of many small mammal species may have allowed animals to escape fire by seeking underground refuge. A 10 cm soil depth could provide safe temperature conditions for animals that seek refuge in their underground burrow systems during fires (Lawrence 1966). Howard et al. (1959) found animals survived as long as burrow conditions allowed vapor pressures below 40 mm Hg during a fire in a California chaparral community. Chipmunks may survive fires by seeking refuge within their burrow systems, or by running ahead of the flames to unburned areas. Once the fire has passed, they may return to their normal home range areas, usually within a few hours (T.M. Schuler, USDA Forest Service, pers. comm.).

Regenerating oak forests, through the use of prescribed fire, will ultimately benefit many species of wildlife that utilize oaks for mast. However, the initial effects upon fauna survival, community structure, and population dynamics in burned oak habitat are unclear. The eastern chipmunk represents an ideal study organism for examining the effects of prescribed fire on mammalian fauna in the eastern hardwood forest, because it is a common ground-dwelling
species that is heavily reliant upon oak mast, but also consumes large quantities of alternative forage during spring and summer months.

Although direct chipmunk survival post-prescribed fire is of interest, data about changes in home range size and territoriality post-prescribed fire are also lacking in the eastern hardwood forests. Previous trapping studies in unburned habitat of the northeastern United States and Canada found home ranges to encompass 0.05 – 1.25 ha (Blair 1942, Smith 1942, Manville 1949, Yerger 1953 and 1955, Forbes 1966, Dunford 1970, Forsyth and Smith 1973, Ickes 1974, Mares et al. 1976, Elliott 1978, Lacher and Mares 1996). However, Hackett and Trevor-Deutsch (1982) in Ontario, Canada, and Bergstrom (1988) in Colorado found trapping underestimated various *Tamias* species’ home range size by 100% - 600% when compared to estimates obtained using radiotelemetry. Most chipmunk research has shown a considerable amount of overlap exists among individual home ranges (Blair 1942, Yerger 1953, Forsyth and Smith 1973, Ickes 1974, Elliott 1978, Getty 1980 and 1981a). Still, many researchers have concluded chipmunks are territorial (Dunford 1970, Ickes 1974, Yahner 1978, Elliott 1978), or at least antagonistic toward home range intruders (Fraleigh 1929, Harper 1929, Blair 1942, Manville 1949). Getty (1980 and 1981a) found them to be territorial toward non-neighbors, whereas neighbors generally avoided each other by time-sharing their habitat.

Eastern chipmunk home ranges are not static, with sizes related to resource availability (Allen 1938, Mares et al. 1976, Getty 1981b, Mares et al. 1982, Mares and Lacher 1987, Bowers et al. 1990) and reproductive condition (Bowers and Carr 1992). As larder-hoarding animals, caches are vital to survival during low forage/food availability periods, so cache defense may limit movement distances from burrows (Elliott 1978, Anderson 1978, Taylor 1988). Predation risk also is a factor influencing the distance ventured from burrows (Elliott 1978, Anderson 1978,
Taylor 1988). Prescribed fire can eliminate, or alter, forest understory structure and composition, which may decrease spring and summer forage resources. According to previous research, this should increase home range size for mammals (Harestad and Bunnell 1979, Mares et al. 1982, Litvaitis et al. 1986, Ostfield 1986, Ims 1987, Jones 1990, Castleberry et al. 2001). However, chipmunks may decrease home range size and increase territoriality to protect their remaining hard mast cache, as well as in response to possible increased predation risk.

_Salamander Community Responses to Silviculture in the Appalachians_

The Allegheny Plateau region of the eastern United States provides habitat for over 22 species of salamanders, 15 of which are known to utilize woodlands during some part of their lives (Conant and Collins 1998). Most plethodontid salamanders (Family Plethodontidae) are found on the forest floor, amongst leaf litter and on understory vegetation, where they forage upon invertebrates at night (Hairston 1987, Kramer et al. 1993, Harpole and Haas 1999). Because they are lungless, they require moist, cool microclimates to facilitate cutaneous respiration (Spotila 1972, Feder 1983, Welsh and Droege 2001). Salamander longevity, small territory size, site fidelity, and physiological sensitivity to environmental changes make them ideal organisms for evaluating environmental change due to anthropogenic disturbances.


Clearcutting causes dramatic changes in vegetation structure and composition, the extent of which depends on forest type and site quality. Pough et al. (1987) found fewer salamanders in clearcuts than in old growth forest in New York, whereas Ash (1988) found populations of *Plethodon* species disappeared completely from hardwood forest sites within a few years of clearcutting in North Carolina. Petranka et al. (1993 and 1994), and Harper and Guynn (1999) in North Carolina; Sattler and Reichenbach (1998), Harpole and Haas (1999), and Knapp et al. (2003) in Virginia; Herbeck and Larson (1999) in Missouri; and Knapp et al. (2003) in West Virginia all found salamander numbers decreased significantly post-clearcut in hardwood stands. Ford et al. (2002) studied salamander response to stand age post-clearcutting in cove hardwood forests of Georgia, and found decreased richness and abundance of most species in younger stands. They suggested that at least 50 years is required for the salamander community to recover from the effects of clearcutting. Salamander abundance is suspected to decline in clearcut stands because of the loss of shade-producing trees, which causes increased ground

Shelterwood, leave-tree, and group selection harvests are methods of reducing tree density within a stand, while maintaining some overstory structure and shade-producing trees. Harpole and Haas (1999) found salamander abundance was significantly lower post-harvest in group selection stands (0.5 ha removed/2 ha stand), leave-tree stands (residual 3 – 4 m² basal area/ha), and shelterwood stands (residual 4 – 15 m² basal area/ha), than pre-harvest or in control stands in a low-elevation, xeric hardwood forest of Virginia. Knapp et al. (2003) also found significantly fewer salamanders post-harvest in group selection stands (residual 18 m² basal area/ha), leave-tree stands (residual 8 m² basal area/ha) and shelterwood stands (residual 6 m² - 14 m² basal area/ha) than in control stands in a mid-elevation xeric hardwood forest of West Virginia. However, Bartman (1998) found no significant differences in salamander abundance post-harvest for shelterwood stands (residual 7 m² basal area/ha) compared to control stands, as did Ford et al. (2000) in group selection stands (1 – 3 ha/6.2 ha removed) and 2-aged stands (residual 4 – 8 m² basal area/ha) in a high-elevation, moderately mesic, upland hardwood community of North Carolina. In more mesic hardwood forests, Messere and Ducey (1998) in New York found no significant differences in salamander abundances within 2 years post-harvest (selection cut to create gaps 22 – 94 m² in area), and Brooks (1999) found no significant differences in salamander abundance 12-21 years after partial harvesting (unlisted residual basal area) in harvested stands compared to control stands in Massachusetts. Sattler and Reichenbach (1998) also found salamander abundance in shelterwood-cut (unlisted residual basal area) forests did not differ from uncut forests Virginia. Partially harvesting stands may not be as detrimental to
salamander populations as clearcutting, because some overstory trees are left intact which can allow for cooler (Johnson et al. 1985), more humid microclimates on the forest floor (Harpole 1996) and leaf litter recruitment over the years.

The effects of prescribed fire on salamander communities are virtually unknown in the mesic areas of the Allegheny Plateau. Pilliod et al. (2003) summarized studies conducted in the eastern United States and found variable results in salamander response to fire, depending on region and season. Kirkland et al. (1996) found inconclusive results regarding fires effects upon salamanders in a Pennsylvania oak forest, although Ford et al. (1999) found no significant declines in salamander species richness or abundance for most species 1-2 yrs post-prescribed fire in the southern Appalachians. Floyd et al. (2002) found no significant effects of fire on salamander abundance, diversity, or richness one year post-fire in a hardwood forest of the upper Piedmont of South Carolina. Fire can consume significant quantities of woody debris that provides moist, cool cover for salamanders on the forest floor, but may be retained if the fires do not burn too intensely. In the Appalachians, prescribed fire can consume much of the understory vegetation and leaf litter, however most burns in hardwood forests are variable and patchy, depending on elevation, aspect, site quality, and vegetation community (Ford et al. 1999). Salamanders utilize leaf litter as foraging grounds and for cover when other objects are not available. Research has shown adult salamanders to predominate under cover objects, whereas juveniles have to find cover in the leaf litter (Jeager et al. 1995). If overstory trees are left intact, leaf litter loss should only be temporary, and litter accumulation should begin within 2 seasons. Whether the salamander community can withstand the temporary loss of leaf litter, or additional microclimate changes due to loss of vegetation is unclear.
Forest vegetation changes caused by high white-tailed deer densities are well-documented, including the removal of significant quantities of understory and midstory vegetation, and subsequent altered forest structure and plant species composition (Tilghman 1989, Trumbull and Zielinski 1989, Waller and Alverson 1997). High deer densities have led to reduced small mammal and canopy-gleaning bird abundances (Brooks and Healy 1988, DeGraaf et al. 1991, deCalesta 1997, McShea and Rappole 1997), yet the impacts upon most vertebrates are unclear. Brooks (1999) found high deer densities in Massachusetts led to changes in plant species composition and structure, with increased fern cover, however, redback salamander (*Plethodon cinereus*) abundance did not seem to be affected.

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CHAPTER 2

EFFECTS OF SPRING PRESCRIBED FIRE ON EASTERN CHIPMUNK HOME RANGES
INTRODUCTION

Currently, the hardwood forests of the central Appalachians are experiencing significant oak 
(*Quercus* spp.) abundance declines in newly regenerating forests (Schuler and Gillespie 2000). Oak decline has serious implications for forest management, as well as for native wildlife. Forest managers in the Mid-Atlantic and Northeast United States are experimenting with prescribed fire as a means to inhibit shade-tolerant, later-successional species (sugar maple, *Acer saccharum*; red maple, *A. rubrum*; and American beech, *Fagus grandifolia*), and fast-growing, shade-intolerant species (yellow poplar, *Liriodendron tulipifera*), and to improve oak seedling recruitment (Barnes and Van Lear 1998, Brose and Van Lear 1998, Brose et al. 1999). Although the oak component of forests undoubtedly is beneficial to wildlife, it is important to monitor the effects of silvicultural manipulations on flora and fauna communities to ensure the integrity of the ecosystem as a whole is not diminished.

Fire is suspected to have played a major historic role in the establishment of oaks in eastern hardwood forests. The anatomy and physiology of oak species is such that it encourages the spread of fire through its leaf structure, and is protected from mortality through its bark structure. Site characteristics such as elevation, climate, slope, aspect, and soil substrate are factors dictating fire intervals, and these characteristics differ between forest types. Many mesic oak ecosystems have unknown fire intervals, but have gone greater than a century without fire due to suppression efforts initiated in the early 20\textsuperscript{th} century. Prescribed fire has many uses in land management including timber management, improvement of wildlife habitat, and restoration of natural communities. However, the reintroduction of fire after greater than a century in mesic oak communities will have unknown effects upon ecosystem attributes.
Chipmunks are a potential model species for assessing prescribed fire effects as they are common components of the forest floor and their diets include understory vegetation, fruits, fungi, and invertebrates (Aldous 1941, Elliott 1978, Wrazen and Svendsen 1978, Mahan and Yahner 1996, Martin et al. 1951), which may be affected by understory fire. Chipmunks also serve valuable functions in forest ecosystems as dispersers of acorns and other plant seeds (Steele et al. 2001), as prey items for numerous vertebrate species (Whitaker and Hamilton 1998), and as consumers of forest pests (ie. *Lepidoptera*, Aldous 1941). Two studies conducted by Tester (1965), and Krefting and Ahlgren (1974), in savannas and mixed conifer-hardwood forests of Minnesota, were inconclusive in determining the effects of fire on chipmunk abundance. Kirkland et al. (1996) studied the effects of wildfire on other small mammals, such as shrews, mice, and voles, in an oak-dominated forest of south central Pennsylvania, and found effects upon abundance to be transitory, lasting less than 8 months. Ford et al. (1999) also studied the effects of prescribed fire on multiple small mammal species (ie. shrews, mice, voles) in a hardwood forest of southwest North Carolina, and found few discernible effects for most species. While other small mammal species may differ from chipmunks in forage types, foraging strategies, burrow depths, and escape capabilities, similarities may exist relative to animal responses to predation exposure, forage loss, and fire-escape behavior.

trapping underestimated various *Tamias* species home range size by 100% - 600% when compared to estimates obtained using radiotelemetry. Most research has shown a considerable amount of overlap exists among individual chipmunk home ranges (Blair 1942, Yerger 1953, Forsyth and Smith 1973, Ickes 1974, Elliott 1978, Getty 1980, 1981a), although no studies have actually quantified the extent of overlap or whether this extent changes in response to changes in resource availability.

Eastern chipmunk home ranges are not static, with size related to resource availability (Allen 1938, Mares et al. 1976 and 1982, Mares and Lacher 1987, Getty 1981b, Bowers et al. 1990) and reproductive condition (Bowers and Carr 1992). As larder-hoarding animals, hard-mast caches are vital to survival during low-forage/food availability periods. Cache-defense and predation risk may limit distances animals will venture from their burrows (Elliott 1978, Anderson 1978, Taylor 1988). Although they rely heavily upon mast, their utilization of plants and fungi during the non-winter months may be severely depleted after a prescribed fire. There also may be increased exposure to predators due to the loss of vegetative cover. Fire may decrease spring and summer forage resources, which causes an increase in home range size for some mammals (Harestad and Bunnell 1979, Mares et al. 1982, Litvaitis et al. 1986, Ostfield 1986, Ims 1987, Jones 1990, Castleberry et al. 2001). It is possible, however, that chipmunks may decrease home range size and increase territoriality to protect their remaining hard mast cache, as well as in response to increased predation risk.

The objectives of my study were to examine the effects of prescribed fire on chipmunk home range size and core area size, the extent of home range overlap, and the extent of core area overlap between animals. An additional objective was to examine the effects of prescribed fire on burrow use. I hypothesized home range sizes would increase in areas having undergone
prescribed fire due to potential decreased alternative forage resources, and core areas would decrease in size due to increased territoriality and the need to protect remaining burrow caches. I also hypothesized the extent of home range overlap would increase, while core area overlap would decrease due to prescribed fire, with chipmunks becoming more territorial. Lastly, I hypothesized burrow use would be unaffected.

METHODS

Study area

My study was conducted on the Fernow Experimental Forest in Tucker County, West Virginia (39.03 N, 79.67 W). The 1900-ha forest was established in 1934, and is under the management of the United States Department of Agriculture Forest Service Northeastern Research Station. The Fernow is adjacent to the 368,000-ha Monongahela National Forest, and is part of the Central Appalachian Broadleaf Forest (McNab and Avers 1994). The study sites on the Fernow ranged in elevation from 730-800 m. Annual temperatures average 8.9º Celsius and annual precipitation averages 143 cm.

All stands used in this study were oak-dominated Allegheny hardwood sites containing northern red oak (Q. rubra), chestnut oak (Q. prinus), white oak (Q. alba), and yellow poplar, American beech and sugar maple in the overstory. Midstories were dominated by sugar maple, American beech and striped maple (Acer pensylvanicum). Groundcover included: Christmas fern (Polystichum acrostichoides), ferns (Dennstaedtia spp., Dryopteris spp.), greenbriar (Smilax spp.), black berries (Rubus spp.), blueberries (Vaccinium spp.), violets (Viola spp.), wood nettle (Laportea canadensis), wild yam (Dioscorea quaternata), and numerous lesser representatives.
One control (unburned) and one treatment (burn stand) stand were chosen as study sites in each of 2002 and 2003 (4 stands total). Stands were selected based on the presence of chipmunks, tree species composition, tree age and size classes, stand topography, and proximity to each other. All stands were established from natural regeneration and ranged from 70-95 years old. The 2003 burn stand had been thinned approximately 20 years prior to the study to a 60-75% stocking level (Schuler and Miller 1995). Each burn stand had undergone a low-intensity, one day prescribed fire in the April preceding its use for my study. The 2002 fire encompassed approximately 12 ha and the 2003 fire encompassed about 19 ha.

Capture and Telemetry

In March 2002 and May 2003 I established 0.35 ha trapping grids in each control and treatment stand. Trapping grids consisted of 6 x 8 trapping arrays with traps spaced 10 m apart (48 traps/stand). Each stand had a >20 m buffer strip surrounding the grids. I captured chipmunks using #102 Tomohawk (Tomahawk Live Trap, Tomahawk, WI) and Sherman (H.B. Sherman Traps, Inc., Talahassee, Florida) traps baited with peanut butter and rolled oats. All chipmunks were sexed, weighed, measured, and assessed for reproductive condition. Female reproductive condition was determined by palpating the abdomen and examining teat condition for signs of lactation. Male reproductive condition was determined by examining testes position. All chipmunks were classified as juvenile, subadult, or adult based on weights, measurements, and sexual characteristics (Forbes 1966, Tryon and Snyder 1973).

I tagged all chipmunks captured with uniquely numbered Monel #1 eartags (National Band and Tag Co., Newport, Kentucky), and attached radiocollars to all adults (≥ 76 g) and 2 subadults (≥ 70 g). Adults (2002: 9M, 8F, 2M subadult; 2003: 6M, 10F) were lightly anesthetized with halothane gas (Halocarbon Laboratories, River Edge, New Jersey), and were fitted with PD-2,
whip antenna radiotransmitter collars (Holohil Systems, Ontario, Canada). Radiotransmitters were attached with either a tubing and wire collar or tubing and modified zip-tie collar. The combined mass of the transmitter and collar (3.5 - 4.2 g) was ≤ 5% of body weight. All chipmunks were given 48 hours to adjust to their collars and all bait was removed from traps before telemetry locations were taken. Upon completion of each study season, attempts were made to recapture each collared chipmunk to retrieve collars and assess condition. All animals were handled under the auspices of West Virginia Division of Natural Resources Collection Permit numbers 2002.108 and 2003.054, and University of Georgia Institutional Animal Care and Use number A2002-10016-c2.

Locations of radiocollared chipmunks were determined by triangulation with radio receivers and hand-held Yagi antennas (Wildlife Materials, Carbondale, Illinois). Two compass bearings were taken simultaneously from pre-determined observer stations located outside of each trapping grid. Stations were established to maintain triangulation angles as near 90° as possible. Telemetry was conducted from June through August of 2002 and 2003, for 1-3 hours/day, and encompassing all possible time periods between 10:00 – 18:00 hours. Locations were taken on each chipmunk every 10-14 minutes during telemetry sessions. I determined telemetry error by estimating locations on test-transmitters located on the study area (White and Garrott 1990). Mean telemetry error was 22° (SD = 18.6; n = 16) at a mean distance of 84 m from the test-transmitters.

**Home Range Analysis**

I obtained Universal Transverse Mecator (UTM) coordinates for all observer stations by using a mobile global position system unit (GPS; Trimble Navigation Ltd., Sunnyvale, California) and Pathfinder software (PathFinder Software, Glendale, Arizona). All GPS locations were
differentially corrected. I used program LOCATE (J.G. Kie, Fresno, California) to determine chipmunk locations by converting compass bearings from known station UTM coordinates into location UTM coordinates. I entered locations into ArcView (ESRI, Redlands, California), and the Animal Movement Extension (Hooge and Eichenlaub 2000) was used to determine home range and core areas.

I accepted only animal locations that created angles of intersection with the observers of 35° - 145° to improve accuracy. Each animal’s acceptable locations were then subject to a 5% outlier removal using the harmonic mean method (Dixon and Chapman 1980). Only animals possessing a minimum of 28 acceptable locations were retained for subsequent home range analysis (Seaman et al. 1999). I used the fixed kernel method with least squares cross validation as a smoothing parameter to construct 95% and 50% contour utilization distributions (UD; Silverman 1986, Worton 1989). I also created Minimum Convex Polygons (MCP’s) from acceptable locations for comparison with previous studies of chipmunk home range size. Neil’s Ute Extension (Free Software Foundation, Inc., Boston, Massachusetts) in ArcView was used to determine the extent of overlap between home ranges (95% UD), and core areas (50% UD) for each possible pair of chipmunks collared in the same trapping grid. Neil’s Ute gives an Overlap Coefficient, which is the area that is overlapped between the pair of animals as a percentage of the total union of the areas.

**Burrow Use**

I tracked radiocollared chipmunks to their burrow chambers after sunset between May and August of 2002 and 2003. Burrow chambers were located an average of 9 times/month, with haphazard sampling. All burrow chambers were marked with metal stakes, and UTM
coordinates were determined using GPS. I used only animals that had $\geq 30$ days of burrow chamber data for analysis.

**Statistical Analysis**

I used SAS (SAS Institute, Inc., Cary, North Carolina) for data analysis, setting statistical significance at $\alpha = 0.05$. I assessed normality using the Shapiro-Wilks test and data were log-transformed when necessary to meet assumptions of normality. When transformations were unsuccessful, non-parametric tests were utilized.

I used a blocked, 2-factor ANOVA to analyze home range attribute data (Sokal and Rohlf 1987). Blocks were years (2002 or 2003) and main effects were treatment type (Control or Burn) and sex (Male or Female). Response variables included: home range size, core area size, extent of home range intrasex (M-M or F-F) overlap, and extent of core area intrasex overlap.

I also performed a blocked 1-factor ANOVA on combined-sex (M-M, M-F, F-F) overlap values for both home ranges and core areas. The data were blocked by year, the main effect was treatment type, and the response variable was the overlap coefficient.

Mann-Whitney U-tests were used to analyze burrow data because data transformations were unsuccessful (Sokal and Rohlf 1987). The factor was treatment type, and the response variable was the number of burrow chambers used/30 days.

**RESULTS**

I tracked 19 chipmunks in 2002 (9M, 8F, 2M subadult) and 16 in 2003 (6M, 10F). The final number of Control chipmunks used in home range analyses included: 3 males and 0 females in 2002, and 2 males and 3 females in 2003. The final number of Burn chipmunks included: 4 males and 4 females in 2002, and 2 males and 3 females in 2003.
**Home Range and Core Area Sizes**

For both years combined, mean home range size for males was 0.39 ha (SE = 0.16; Table 2.1) in the Burn areas and 0.55 ha (SE = 0.13) in the Control areas. Mean female home range size was 0.58 ha (SE = 0.14) in the Burn areas and 0.76 ha (SE = 0.10) in the Control areas. Mean core area size for males was 0.04 ha (SE = 0.01) in the Burn areas and 0.08 ha (SE = 0.03) in the Control areas. Mean female core area size was 0.09 ha (SE = 0.02) in the Burn areas and 0.16 ha (SE = 0.06) in the Control.

Comparisons of home range size between sexes and treatment types showed there was a block effect ($P = 0.04$), indicating differences between years or areas. There was no interaction ($P = 0.56$), and no difference in home range size by sex ($P = 0.52$) or treatment type ($P = 0.37$).

Results for core area sizes between the sexes and treatment types showed no interaction ($P = 0.74$) or block effect ($P = 0.24$), and no difference in core area size by sex ($P = 0.18$) or treatment type ($P = 0.25$).

**Extent of Home Range Overlap and Core Area Overlap**

For both years combined, mean male-male home range overlap coefficient was 0.60 (SE = 0.06; Table 2.2) in the Burn areas and 0.50 (SE = 0.06) in the Control areas. Mean female-female home range overlap coefficient was 0.46 (SE = 0.04) in the Burn areas and 0.54 (SE = 0.05) in the Control areas. Mean male-male core area overlap coefficient was 0.40 (SE = 0.14) in the Burn areas and 0.30 (SE = 0.09) in the Control areas. Mean female-female core area overlap coefficient was 0.35 (SE = 0.08) in the Burn areas and 0.15 (SE = 0.05) in the Control areas. Mean combined-sex home range overlap coefficient was 0.52 (SE = 0.02) in the Burn areas and 0.51 (SE = 0.03) in the Control areas. Mean combined-sex core area overlap coefficient was 0.37 (SE = 0.05) in the Burn areas and 0.21 (SE = 0.03) in the Control areas.
Results for intrasex home range overlap showed no interaction ($P = 0.11$) or block effect ($P = 0.54$), and no difference by sex ($P = 0.70$) or treatment type ($P = 0.90$). Results for intrasex core area overlap showed no interaction ($P = 0.79$) or block effect ($P = 0.36$), and no difference by sex ($P = 0.79$) or treatment type ($P = 0.25$).

There was no difference in the extent of combined-sex home range overlap by treatment type ($P = 0.13$) and there was no block effect ($P = 0.11$). There was no difference in the extent of combined-sex core area overlap by treatment type ($P = 0.80$) and there was no block effect ($P = 0.61$).

**Burrow Use**

The final number of Control chipmunks used for burrow analysis included: 4 males and 2 females in 2002, and 2 males and 5 females in 2003. The final number of Burn chipmunks used included: 3 males and 2 females in 2002, and 3 males and 4 females in 2003.

Control males averaged 1.33 (SE = 0.21) burrows/30 days (Table 2.3) and Control females averaged 1.14 (SE = 0.14). Burn males averaged 1.33 (SE = 0.21) burrows/30 days and Burn females averaged 1.33 (SE = 0.21). After combining sexes by treatment type, the Mann-Whitney U-tests showed no difference in the number of burrows used between treatment types ($P = 0.60$).

**DISCUSSION**

Chipmunks readily utilized area that had undergone prescribed fire, and were rarely seen moving into control areas. Chipmunks living on the outskirts of the burned areas were occasionally seen leaving the burn area to enter an unburned area, although no telemetry locations were outside of burned areas. My home range sizes derived from telemetry fell within previously reported estimates based on trapping for eastern chipmunks within the eastern
deciduous forests of North America (Blair 1942, Smith 1942, Manville 1949, Yerger 1953 and 1955, Forbes 1966, Dunford 1970, Forsyth and Smith 1973, Ickes 1974, Mares et al. 1976, Elliott 1978, Lacher and Mares 1996). I found no difference in home range size or core area size between the sexes. This finding also is consistent with previous chipmunk research (Blair 1942, Lacher and Mares 1996, Bowers et al. 1990), although is unusual for mammals in general (Harestad and Bunnell 1979, Swingland and Greenwood 1983). Previous research has shown home range size for many mammal species to be inversely related to habitat productivity (Harestad and Bunnell 1979, Litvaitis et al. 1986, Ostfield 1986, Ims 1987, Jones 1990, Castleberry et al. 2001), and chipmunks are known to adjust home range size according to resource availability (Mares et al. 1976, Getty 1981b, Mares et al. 1982, Mares and Lacher 1987, Bowers et al. 1990). I found no difference in home range or core area sizes between Control and Burn chipmunks, which may indicate there may be no real difference in resource availability between the two treatment types following prescribed fire. Understory vegetation, fruits, fungi, and invertebrates are components of chipmunk spring and summer diets (Aldous 1941, Elliott 1978, Wrazen and Svendsen 1978, Mahan and Yahner 1996, Martin et al. 1951), but there is a possibility chipmunks relied upon remaining hard-mast caches, until vegetation and fungi grew back. The spring fires killed much of the understory vegetation (< 2 m) on my study sites, but I observed seedlings and herbaceous plants sprouted rapidly throughout the late spring and summer. Fungi were noted in the stands, but not quantified, while soft-mast may have been significantly reduced during the summer. My results suggest the brief impact on vegetation and fungi, due to the low-intensity spring fires, did not affect chipmunks and alternative forage items must have been available.
Eastern chipmunks are known to have broadly overlapping home ranges (Blair 1942, Yerger 1953, Forsyth and Smith 1973, Ickes 1974, Elliott 1978, Getty 1980 and 1981a), and my results demonstrate spring prescribed fire did not alter the extent of overlap between animals. Opposite sex animals were just as likely to overlap home ranges as same sex animals, even in their core areas. Considering other small mammal species, Jones (1990) found female meadow voles (*Microtus pennsylvanicus*) increased distance to nearest female neighbor in response to low forage availability, while Ims (1987; *Clethrionomys rufocanus*) and Ostfield (1986; *Microtus californicus*) found female vole home ranges overlapped to a greater extent when there was more food available. Lima et al. (2002) found northern short-tailed shrews (*Blarina brevicauda*) had well-defined territories, which prevented food resource depletion during harsh winter months, and that population dynamics were dominated by intraspecific competition in the form of aggression, non-overlapping home ranges, territoriality, and solitary nesting. The similarity in extent of overlap between Control and Burn chipmunk home ranges and core areas indicates there was no increased territoriality and no additional competition for resources in the Burn area.

I found both sexes to utilize 1-2 burrow systems/30 days. Of the 7 animals that used 2 burrows, 4 of them made permanent changes to the newer burrow. Three animals switched to a new burrow for a short time period, and then returned to their previously used burrow. Chipmunks will rapidly overtake a “better” burrow, if the original owner dies or emigrates (Elliott 1978, Yahner 1978), and this was witnessed by 2 of my collared animals. A radiocollared female was killed in a trap and her burrow was taken over by a neighboring radiocollared male the next day. As for the animals that switched burrows temporarily, it has been suggested that animals might move to new burrows to escape parasites (fleas) for a short time period (Yahner 1978). Burrow chambers were often found to lie within “mounds” created
by old fallen trees, whose root systems created tiny hills on the landscape, however many chambers also were located beneath flat areas (Peterson et al. 1990).

Spring prescribed fire appears to be an acceptable forest management tool regarding its effects upon chipmunk survival and home range attributes in the mesic forests of West Virginia. The conclusions of this study may be extended to other small mammal species with similar forage requirements and population dynamics. Species that rely heavily upon spring vegetation (i.e., voles) may be negatively affected, because they do not have large mast caches to rely upon.

The effects of fires on juvenile and subadult chipmunks are unknown, as are the effects of prescribed fire during alternate seasons. Autumn is a critical season for acorn-dependant species and how chipmunks would respond to prescribed fire during this time period is unknown. Hard mast is vital to chipmunk over-winter survival and any interference with mast acquisition could negatively impact local populations. Prescribed fire, however, generally occurs in limited areas and re-population through immigration by adjacent populations may occur quickly.

Future research is needed to investigate changes in chipmunk spatial use after prescribed fire occurring in different seasons, as well as demographic changes. Additionally, the amount of mast contained within chipmunk burrow-caches after winter consumption is of interest. The effects of back-to-back silvicultural manipulations upon chipmunk survival and home range structure is of interest because of forest industry’s interest in conducting shelterwood cuts within a short time after prescribed fire.

LITERATURE CITED


Table 2.1. Mean (± SE) home range size (ha) and mean (± SE) core area size (ha) among Burn and Control treatment eastern chipmunks on the Fernow Experimental Forest, West Virginia, 2002 and 2003.

<table>
<thead>
<tr>
<th>Treatment Type/Sex</th>
<th>n</th>
<th>Home Range</th>
<th>Core Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male</td>
<td>6</td>
<td>0.39 (0.16)</td>
<td>0.04 (0.01)</td>
</tr>
<tr>
<td>Female</td>
<td>7</td>
<td>0.58 (0.14)</td>
<td>0.09 (0.02)</td>
</tr>
<tr>
<td>Control</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male</td>
<td>5</td>
<td>0.55 (0.13)</td>
<td>0.08 (0.03)</td>
</tr>
<tr>
<td>Female</td>
<td>3</td>
<td>0.76 (0.10)</td>
<td>0.16 (0.06)</td>
</tr>
</tbody>
</table>
Table 2.2. Mean (± SE) overlap coefficients for Control and Burn treatment eastern chipmunk home ranges and core areas, including male-male overlap, female-female overlap, and combined-sex (male-male, male-female, female-female) overlap on the Fernow Experimental Forest, West Virginia, 2002 and 2003.

<table>
<thead>
<tr>
<th>Treatment Type/group</th>
<th>n</th>
<th>Home Range</th>
<th>Core Area</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Burn</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male-male</td>
<td>4</td>
<td>0.60 (0.06)</td>
<td>0.40 (0.14)</td>
</tr>
<tr>
<td>Female-female</td>
<td>5</td>
<td>0.46 (0.04)</td>
<td>0.35 (0.08)</td>
</tr>
<tr>
<td>Combined-sex</td>
<td>23</td>
<td>0.53 (0.02)</td>
<td>0.37 (0.05)</td>
</tr>
<tr>
<td><strong>Control</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male-male</td>
<td>4</td>
<td>0.50 (0.06)</td>
<td>0.30 (0.09)</td>
</tr>
<tr>
<td>Female-female</td>
<td>3</td>
<td>0.54 (0.05)</td>
<td>0.15 (0.05)</td>
</tr>
<tr>
<td>Combined-sex</td>
<td>13</td>
<td>0.51 (0.03)</td>
<td>0.21 (0.03)</td>
</tr>
</tbody>
</table>
Table 2.3. Mean (± SE) number of burrows used per 30 days by eastern chipmunks on the Fernow Experimental Forest, West Virginia, 2002 and 2003.

<table>
<thead>
<tr>
<th>Treatment Type/Sex</th>
<th>n</th>
<th>Burrows</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burn</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male</td>
<td>6</td>
<td>1.33 (0.21)</td>
</tr>
<tr>
<td>Female</td>
<td>6</td>
<td>1.33 (0.21)</td>
</tr>
<tr>
<td>Control</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Male</td>
<td>6</td>
<td>1.33 (0.21)</td>
</tr>
<tr>
<td>Female</td>
<td>7</td>
<td>1.14 (0.14)</td>
</tr>
</tbody>
</table>
CHAPTER 3

EFFECTS OF SPRING PRESCRIBED FIRE AND DEER EXCLUSION ON A WOODLAND SALAMANDER COMMUNITY
INTRODUCTION

Similar to much of the eastern hardwood forests throughout the United States, the mixed mesophytic forests of the Central Appalachians are experiencing significant oak (*Quercus* spp.) abundance declines in regenerating forests (Schuler and Gillespie 2000). Oak decline has serious implications for forest and wildlife management, due to the loss of economically profitable species (i.e., northern red oak, *Q. rubra*) and valuable mast-production for wildlife. Therefore forest managers are experimenting with both pre- and post-timber harvest prescribed fires as a means to improve both oak advance regeneration establishment and seedling recruitment. Research thus far has shown prescribed fire to inhibit shade-tolerant, later-successional species such as red maple (*Acer rubrum*), sugar maple (*A. saccharum*) and American beech (*Fagus grandifolia*), which are abundant as advance regeneration, as well as fast-growing, shade-intolerant species such as yellow poplar (*Liriodendron tulipifera*) and sweet birch (*Betula lenta*) that respond rapidly to overstory removal (Brose and Van Lear 1998). White-tailed deer (*Odocoileus virginianus*) populations increased greatly over the last half of the 20th century, and are thought to contribute to oak declines through increased browsing pressure (Grisez 1957, Shafer et al. 1961, Jordon 1967, Marquis 1975 and 1981, Hughes and Fahey 1991, Abrams 1998). Forest managers also are utilizing deer exclusion fences in hopes of protecting oak acorns and seedlings, thereby improving regeneration.

It is unclear what effects prescribed fires and deer over-browsing have on forest floor-dwelling vertebrates, such as woodland salamanders. Woodland salamanders are thought to account for greater biomass than all other vertebrates combined in mesic regions of the eastern United States (Burton and Likens 1975, Hairston 1987), and they serve as valuable prey for birds, mammals and reptiles (Brodie 1977, Brodie and Brodie 1980, Ducey and Brodie 1983).
Most plethodontid salamanders (Family Plethodontidae) are found on the forest floor, where they forage upon invertebrates amongst leaf litter and on understory vegetation (Pough 1983, Hairston 1987, Kramer et al. 1993, Harpole and Haas 1999). Because they are lungless, they require moist, cool microclimates to facilitate cutaneous respiration (Spotila 1972, Feder 1983, Welsh and Droege 2001). Prescribed fire consumes much of a forests leaf litter and groundcover, and thereby may alter microclimates. Salamanders are thought to be ideal environmental indicators because of their sensitivity to microclimate temperature, pH, and moisture (Herbeck and Larson 1999).


Salamander abundance is suspected to decline in clearcut stands because of the loss of shade-producing trees, with concomitant increased surface temperatures (Johnson et al. 1985), desiccation and loss of leaf litter, reduced moisture levels in surface soil (all of which may increase salamander desiccation risk), and decreased foraging opportunities for salamanders (Jaeger 1980, Pough et al. 1987, Ash 1988, Petranka et al 1993, Ash 1995 and 1997). The effects of partial-harvesting are not fully understood, however, because numerous studies have
shown conflicting effects upon salamander abundance. Harpole and Haas (1999) found salamander abundance was significantly lower post-harvest in group selection stands (one 0.5 ha group removed/2 ha stand), leave-tree stands (residual 3 – 4 m² basal area/ha), and shelterwood stands (residual 4 – 15 m² basal area/ha), than pre-harvest or in control stands in a low-elevation, xeric hardwood forest of Virginia. Knapp et al. (2003) also found significantly fewer salamanders post-harvest in group selection stands (residual 18 m² basal area/ha), leave-tree stands (residual 8 m² basal area/ha) and shelterwood stands (residual 6 - 14 m² basal area/ha) than in control stands in a mid-elevation xeric hardwood forest of West Virginia. However, Bartman (1998) found no significant differences in salamander abundance post-harvest for shelterwood stands (residual 7 m² basal area/ha) compared to control stands, as did Ford et al. (2000) in group selection stands (1 – 3 ha group removed/6.2 ha) and 2-aged stands (residual 4 – 8 m² basal area/ha) in a high-elevation, moderately mesic, upland hardwood community of North Carolina. In more mesic hardwood forests, Messere and Ducey (1998) in New York found no differences in salamander abundances within 2 years post-harvest (selection cut to create gaps 22 – 94 m² in area), and Brooks (1999) found no differences in salamander abundance 12-21 years after overstory-harvesting (unlisted basal area) in harvested hardwood-pine stands compared to control stands in Massachusetts. Sattler and Reichenbach (1998) also found salamander abundance in shelterwood-cut (unlisted basal area) forests did not differ from uncut forests in Virginia. Partial-harvesting stands may not be as detrimental to salamander populations as clearcutting because some overstory trees are left intact which can allow for cooler (Johnson et al. 1985), more humid microclimates on the forest floor (Harpole 1996) and continued leaf litter recruitment. It is also possible that more mesic habitats ameliorate the effects of overstory partial-harvesting, which causes decreased soil moisture and humidity in xeric habitats.
The effects of prescribed fire on salamander communities are virtually unknown in the Appalachians, and particularly in the mesic forests of the Allegheny Plateau. Pilliod et al. (2003) summarized studies conducted in the eastern United States, and found variable results in salamander response to fire, depending on region and season. Kirkland et al. (1996) found inconclusive results regarding fire effects on salamanders in a Pennsylvania oak forest, although Ford et al. (1999) found no declines in species richness, or salamander abundance for most species 1-2 yrs post-prescribed fire in the southern Appalachians. Floyd et al. (2002) also found no significant effects of fire on salamander abundance, diversity, or richness one year post-fire in a hardwood forest of the Upper Piedmont of South Carolina. In the Appalachians, prescribed fire can consume much of the understory vegetation and leaf litter, however most burns in hardwood forests are variable and patchy, depending on elevation, aspect, site quality, and vegetation community (Ford et al. 1999). The extent of substrate consumption and amount of cover objects left for salamander use are determined by the intensity of fire, that in turn is determined by climatic and environmental factors. If overstory trees are left intact, however, leaf litter loss should only be temporary, and litter accumulation should begin the following autumn. Whether the salamander community can withstand the temporary loss of leaf litter, or additional microclimate changes due to loss of vegetation is unclear.

Forest vegetation changes caused by high white-tailed deer densities are well-documented, and include the removal of significant quantities of understory and midstory vegetation which alters forest structure as well as plant species composition (Tilghman 1989, Trumbull and Zeilinski 1989, Waller and Alverson 1997). High deer densities reduce small mammal and canopy-gleaning bird abundances (Brooks and Healy 1988, DeGraaf et al. 1991, deCalesta 1997, McShea and Rappole 1997), yet the impacts upon most vertebrate types are uncertain. Brooks
(1999) found high deer densities in Massachusetts led to changes in plant species composition and structure, with increased fern cover; however, redback salamander (*Plethodon cinereus*) abundance was not affected. Synergistic effects from the combination of deer browsing and prescribed fire have not been studied.

My objectives were to examine the effects of spring prescribed fire and deer herbivory on woodland salamander relative abundance, species richness, body condition, as well as the effects of season on the abundance of individual salamander species. I hypothesized salamander abundance, species richness, and body condition would decline immediately post-prescribed fire, due to the loss of leaf litter that provides moist cover and foraging opportunities. I also hypothesized deer exclusion areas would have less drastic decreases in salamander abundance, species richness, and body condition than unfenced areas, due to increased vegetation coverage that would provide cooler microclimates and forage sites. Lastly, I hypothesized seasonal shifts in the abundances of different species would occur, as demonstrated by previous researchers.

METHODS

*Study area*

My study was conducted between April and November of 2001 - 2004 on the Fernow Experimental Forest in Tucker County, West Virginia (39.03 N, 79.67 W). The 1900-ha forest was established in 1934, and is under the management of the Northeast Research Station of the United States Forest Service. The Fernow lies adjacent to the 368,000-ha Monongahela National Forest, and is part of the Central Appalachian Broadleaf Forest (McNab and Avers 1994). Study sites on the Fernow ranged in elevation from 730 - 800 m. Annual temperatures average 8.9° Celsius and annual precipitation averages 143 cm.
The forest stands used for my study were oak-dominated Allegheny hardwood sites containing northern red oak (*Q. rubra*), chestnut oak (*Q. prinus*), white oak (*Q. alba*), and yellow poplar, with lesser amounts of American beech and sugar maple. Lower mid-stories were dominated by striped maple (*Acer pensylvanicum*) and American beech, and groundcover included: Christmas fern (*Polystichum acrostichoides*), ferns (*Dryopteris* spp., *Dennstaedtia* spp.), wood nettle (*Laportea canadensis*), greenbriar (*Smilax* spp.), black berries (*Rubus* spp.), violets (*Viola* spp.), wild yam (*Dioscorea quaternata*), and numerous lesser representatives.

Ten prescribed fire (Burn) stands, 10 prescribed fire with deer exclosures (Fenced-burn) stands, and 4 Control (unburned - unfenced) stands were randomly assigned to the three treatment types (24 stands total). Stands were 0.81 ha (0.20 ha plus 30 m buffer strips) each, naturally regenerated and approximately 70 years old, with ten mid-slope stands having established following old-field succession and 14 ridgetop stands having established after initial logging of the virgin forest. The 14 ridgetop stands underwent thinnings in 1983 to achieve 60 - 75% overstory stocking levels (Schuler and Miller 1995). The mid-slope Burn (*n* = 4) and Fenced-burn (*n* = 4) stands underwent a low-severity prescribed fire in April 2002 that encompassed approximately 12 ha. The ridgetop Burn (*n* = 6) and Fenced-burn (*n* = 6) stands underwent a low-severity prescribed fire in April of 2003 that encompassed about 19 ha.

**Surveys**

Terrestrial salamanders were surveyed six times each year, between April and November of 2001 - 2004. Surveys took place after precipitation events whenever possible, and were conducted every 3 - 5 weeks, between 0800 and 1700 hours. Surveys were conducted by examining salamander presence under coverboard arrays (DeGraaf and Yamasaki 1992), which consisted of 1m² hardwood composite boards cut into 9 identical squares, abutted next to each
other and placed flush with the soil or rock. Three arrays were located along the central axis of each stand, and were separated by 10 m spacing. All coverboard arrays were allowed to “weather” for 6 months before surveys began.

All salamanders captured were identified to species, weighed, and measured for snout-vent lengths (SVL; Heyer et al. 1994). Salamanders were then returned to the coverboard square from which they were captured. All animals were handled under the auspices of West Virginia Division of Natural Resources Collection Permit numbers 2002.108 and 2003.054, and University of Georgia Institutional Animal Care and Use number A2002-10016-c2.

Statistical Analysis

SAS (SAS Institute, Inc., Cary, North Carolina) was used for data analysis, and significance was set at $\alpha = 0.05$. Twenty-four surveys were conducted between 2001 and 2004. However, because half of the stands were burned in 2002 and half in 2003, the 2002 Burn sites had one additional year of post-fire surveys and the stands burned in 2003 had one additional year of pre-fire surveys. These 6 extra surveys have been eliminated from the analyses. The following 18 surveys were used in the analyses: immediate pre-fire surveys ($n = 6$), immediate post-fire surveys ($n = 6$), and one-year post-fire surveys ($n = 6$).

Mean relative abundance of salamanders/subplot (coverboard array) was determined for each stand and each survey. Means for each stand were then averaged over the 6 pre-fire surveys, giving each stand an overall pre-fire mean/subplot ($n = 20$). Overall means/subplot were also determined for the 6 immediate post-fire surveys ($n = 20$), and the 6 one-year post-fire surveys ($n = 20$). Species richness was calculated as the total number of species/stand. The 6 pre-fire values were averaged together for each stand, giving a pre-fire mean richness/stand ($n = 20$), as were the 6 immediate post-fire values ($n = 20$) and the 6 one-year post-fire values ($n = 20$).
I assessed the effects of spring prescribed fire and deer exclusion on salamander combined-species relative abundance, relative abundance by species and species richness using mixed-design ANOVA’s (Sokal and Rohlf 1987). The between subject factor was fencing (fenced or unfenced), and the within subjects factor was fire (pre-fire or post-fire). Body condition was examined using analysis of covariance (ANCOVA) to test for differences between treatment types, with individual salamander weight as the dependant variable and SVL as a covariate. Pre-fire salamander values were compared with immediate post-fire values. All 20 stands that were burned were used in the analysis, regardless of whether or not they were fenced. I examined seasonal changes in relative abundances of different species pre-fire and immediately post-fire, by graphing each salamander species mean relative abundance/subplot against survey month.

RESULTS

I conducted 18 woodland salamander surveys over 4 years, during which 1,238 salamanders were captured/recaptured and released. These included 777 redback salamanders (*P. cinereus*), 257 Allegheny mountain dusky salamanders (*Desmognathus ochrophaeus*), 151 northern slimy salamanders (*P. glutinosus*), 9 northern two-lined salamanders (*Eurycea bislineata*), 5 red-eft stage of eastern newt (*Notophthalmus viridescens*), and 39 unknown escapee salamanders. Because of their small sample sizes or unknown species, eastern newts, northern two-lined, and the 39 unknowns were excluded from analyses related to species-specific responses.

Species Abundance and Richness Among Treatments

There was no fire x fencing interaction immediately post-fire (*P* = 0.35) or one-year post-fire (*P* = 0.31) for combined-species salamander relative abundance. Abundance increased immediately post-fire (*P* = 0.001; Table 3.1), but returned to pre-fire levels one-year post-fire (*P*
= 0.86). There was no effect of fencing immediately post-fire ($P = 0.50$) or one-year post-fire ($P = 0.52$).

There was no fire x fencing interaction for redback salamander relative abundance immediately post-fire ($P = 0.96$) or one-year post-fire ($P = 0.63$), and no effect of fencing immediately post-fire ($P = 0.73$) or one-year post-fire ($P = 0.84$). Redback salamander abundance increased immediately post-fire ($P = 0.01$; Table 3.1), but returned to pre-fire levels one-year post-fire ($P = 0.31$).

There was no fire x fencing interaction ($P = 0.44$) or effect of fencing ($P = 0.49$) for dusky salamanders immediately post-fire. Abundance increased immediately post-fire ($P = 0.004$; Table 3.1), but returned to pre-fire levels one-year post-fire ($P = 0.55$). There was no fire x fencing interaction ($P = 0.91$) or effect of fencing ($P = 0.60$) one-year post-fire.

There was no fire x fencing interaction immediately post-fire ($P = 0.06$), but there was one-year post-fire ($P = 0.01$), such that there were more slimy salamanders in unfenced stands than fenced stands post-fire. Slimy salamander relative abundance did not change immediately post-fire ($P = 0.49$; Table 3.1), but was greater one-year post-fire than pre-fire ($P = 0.009$). There was no effect of fencing immediately post-fire ($P = 0.17$) or one-year post-fire ($P = 0.33$).

There was a fire x fencing interaction immediately post-fire ($P = 0.02$) for species richness, with higher richness in unfenced stands than fenced stands post-fire. There was no interaction one-year post-fire ($P = 0.07$). Richness increased immediately post-fire ($P = 0.02$; Table 3.1), but returned to pre-fire levels one-year post-fire ($P = 0.38$). There was higher richness in unfenced stands than fenced stands immediately post-fire ($P = 0.02$), but there was no difference one-year post-fire ($P = 0.28$).
**Body Condition**

Body condition for Allegheny mountain dusky and northern slimy salamanders did not differ immediately post-fire from pre-fire values ($P = 0.30$ and $P = 0.79$, respectively; Table 3.2). Redback salamander body condition was not different immediately post-fire compared to pre-fire values ($P = 0.30$), however salamanders had greater SVL ($P = 0.001$) post-fire.

**Seasonal Trends in Salamander Relative Abundance**

Graphs of mean relative abundance/subplot of salamanders over the six survey months per year show pre-fire trends in combined-species relative abundance closely parallels post-fire trends (Figure 3.1). Redback salamanders were most abundant during early spring and early autumn, with lowest abundance during the summer (Figure 3.2). This was true both pre-fire and immediately following fire. Allegheny mountain dusky salamanders were most abundant during the summer months, with lower abundance during the spring and autumn (Figure 3.3). This was true both pre-fire and immediately following fire. Northern slimy salamanders were more abundant during the summer, than in spring or autumn, and this was true for pre-fire and immediately following fire (Figure 3.4).

**DISCUSSION**

My research indicates spring prescribed fire and deer herbivory on the Fernow Experimental Forest probably do not negatively impact woodland salamander abundance or species richness in the short-term. Fire had no negative effect on salamanders, with increases seen immediately post-fire in species richness and abundances of combined-species, redback salamanders and dusky salamanders. Deer herbivory also had little effect on salamanders, with an increase in species richness post-fire in unfenced stands compared to fenced stands. Most increases seen in
variables returned to pre-fire levels by one-year post-fire. Similarly, Means and Campbell (1981) found amphibians in fire-adapted communities of Florida did not experience population declines post-fire even though they were associated with leaf litter and cover objects. Also, Ford et al. (1999) found salamanders in the southern Appalachians did not exhibit declines post-prescribed fire, and attributed this to the retention of overstory trees and the mosaic of vegetation left unburned by the fire.

Many woodland salamanders are likely able to descend into underground burrows to escape fire. Their low metabolism allows them to persist without food for long periods of time (Feder 1983), and they may survive underground for extended periods until surface conditions improve. None of my coverboard were burned more than superficially, and may indicate loss of coarse woody cover objects was not a problem. The increases in abundance of salamanders under coverboards immediately post-fire may have occurred due to the loss of leaf litter cover and the movement of salamanders under available cover objects. Heatwole (1962) found salamander density increased under logs during dry summer months, presumably because forest floor conditions became too dry. Northern slimy salamander abundance may have increased one-year post-fire because leaf litter would have accumulated on the forest floor, activity above ground may have increased, and they may have been claiming cover objects.

Species richness increased temporarily due to fire, and this may be attributed to the few species captured on my study area. Only 5 species utilized coverboards during the study, with eastern newts and northern two-lined salamanders being uncommon. DeGraaf and Yamasaki (1992) also found these two species to rarely utilize surface cover objects. The increase in richness may have been a function of leaf litter loss driving eastern newts or northern two-lined salamanders under coverboards more often. By one-year post-fire, species richness returned to
pre-fire levels, and this may have been due to the return of leaf litter during the autumn, and movement back to the leaf litter.

There were slight increases in species richness and northern slimy salamanders in unfenced stands compared to fenced stands post-fire, but these were the only effects of deer herbivory. Salamanders are known to use vegetation as forage sites, where they catch more prey than conspecifics on the forest floor (Jaeger 1978). It is still unclear how extensively they use vegetation, but the presumed decrease in vegetative cover outside of my deer exclosures may not have been significant enough to hinder salamander use of vegetation. Brooks (1999) also found high deer densities in a hardwood forest of Massachusetts had no impact on redback salamander captures.

Body condition remained unchanged post-fire for northern slimy and Allegheny mountain dusky salamanders. Redback salamander weights were the same pre- and post-fire, although SVL was greater post-fire. Longer SVL post-fire may indicate smaller redback salamanders were displaced from cover objects by larger individuals. Jeager et al. (1995) found adult salamanders are able to defend their cover objects from smaller individuals, thereby forcing smaller salamanders to seek cover in the leaf litter. Capturing salamanders with longer SVL post-fire also may indicate smaller individuals were killed by fire, resulting in a shift in the size distribution.

Seasonal trends in the presence of different salamander species were evident, and were similar both pre-fire and post-fire. Shifts in species may be due to differences in the timing of breeding and egg laying, temperature and moisture thresholds, or may be a mechanism for competition avoidance. Allegheny mountain dusky salamanders were at their highest abundance during the summer months, which Moore et al. (2001) also found to be true in the Allegheny
Plateau region. High redback salamander abundance, primarily during early spring (late April) and early autumn (Sept/Oct), may be a function of their high moisture thresholds. Other studies have found similar seasonal trends for redback salamanders (Brooks 1999, Moore et al. 2001, DeGraaf and Yamasaki 2002, Hicks and Pearson 2003). Allegheny mountain dusky salamanders are known to aggressively defend cover objects from redback salamanders, whereas the converse is rarely true (Smith and Pough 1994). Adult redback and adult northern slimy salamanders are not aggressive toward each other (Jaeger et al. 1998, Price and Shields 2002), and were often found sharing coverboard arrays. Northern slimy salamanders were present throughout the summer months, likely a reflection of their potentially lower moisture thresholds due to their larger body size (Cunningham 1960, Maiorana 1976, Feder 1983).

Some concerns about the use of coverboards as a method for surveying salamander population responses to prescribed fire exist. First, is the possible increase in salamander density under boards due to the loss or desiccation of leaf litter, leading to a false observation of an increase in abundance. Moreover, some species, such as northern two-lined salamanders and eastern newts, rarely utilize coverboards (DeGraaf and Yamasaki 1992). Second, there is an inability to properly monitor age structure within the population, because cover objects often are defended by adults and juveniles are generally left to live in the leaf litter (Jaeger et al. 1995). Lastly, it is uncertain whether equal representation by both sexes occurs under coverboards. Long term monitoring is needed to see whether juvenile salamanders survived the prescribed fires, or additional survey methods are needed.

Future research concerning prescribed fire’s effects upon salamander communities could investigate the effects of back-to-back manipulations, such as overstory-harvesting before or
after burning. Also, studies should also look at age/sex structure within the population, which may require additional methods of surveying.

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Table 3.1. Mean (± SE) combined-species relative abundance/subplot, relative abundance/subplot by species and species richness/stand for salamanders on fenced and unfenced stands at pre-fire, immediate post-fire, and one-year post-fire time periods on the Fernow Experimental Forest, West Virginia, 2001 – 2004.

<table>
<thead>
<tr>
<th>Variable</th>
<th>n⁷</th>
<th>Pre-fire</th>
<th>Immediate Post-fire</th>
<th>One-year Post-fire</th>
</tr>
</thead>
<tbody>
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<td>Combined-spp. Abundance⁸</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Unfenced</td>
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<td>1.22 (0.15)</td>
<td>0.85 (0.08)</td>
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<td>1.04 (0.09)</td>
<td>0.70 (0.09)</td>
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<td>Plethodon cinereus⁹</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Unfenced</td>
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<td>0.48 (0.11)</td>
<td>0.69 (0.19)</td>
<td>0.44 (0.12)</td>
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<td>0.43 (0.09)</td>
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<td>Plethodon glutinosus¹⁰</td>
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<tr>
<td>Unfenced</td>
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<td>0.23 (0.06)</td>
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<td>0.11 (0.03)</td>
<td>0.03 (0.02)</td>
<td>0.11 (0.04)</td>
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<tr>
<td>Desmognathus ochrophaeus¹¹</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Unfenced</td>
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<td>Species Richness¹²</td>
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<td>1.18 (0.13)</td>
<td>1.18 (0.08)</td>
<td>1.10 (0.11)</td>
</tr>
</tbody>
</table>

- a: Immediate post-fire sig. greater than pre-fire
- b: One-year post-fire sig. greater than pre-fire
- c: Unfenced sig. greater than fenced
- d: $P \leq 0.05$
- e: $P \leq 0.01$
- f: d.f. = 1, 18 for all comparisons
Table 3.2. Mean (± SE) weight by species (g) and length (mm) for salamanders pre-fire and immediate post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2004.

<table>
<thead>
<tr>
<th>Variable</th>
<th>n</th>
<th>Pre-fire</th>
<th>n</th>
<th>Immediate Post-fire</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Weight</strong></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Plethodon cinereus</em></td>
<td>173</td>
<td>0.78 (0.03)</td>
<td>248</td>
<td>0.73 (0.02)</td>
</tr>
<tr>
<td><em>Plethodon glutinosus</em></td>
<td>37</td>
<td>2.35 (0.37)</td>
<td>27</td>
<td>2.08 (0.38)</td>
</tr>
<tr>
<td><em>Desmognathus ochrophaeus</em></td>
<td>57</td>
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<tr>
<td><strong>Length</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Plethodon cinereus</em></td>
<td>173</td>
<td>33.20 (0.49)</td>
<td>248</td>
<td>34.70 (0.38)</td>
</tr>
</tbody>
</table>

*Immediate post-fire sig. greater than pre-fire; \( P \leq 0.01 \)
Figure 3.1. Mean relative abundance/subplot for combined-species of salamanders for six surveys pre-fire and six surveys immediate post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2003. Means are derived from two different years, depending on subplot.
Figure 3.2. Mean relative abundance/subplot for *Plethodon cinereus* for six surveys pre-fire and six surveys immediate post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2003. Means are derived from two different years, depending on subplot.
Figure 3.3. Mean relative abundance/subplot for *Desmognathus ochrophaeus* for six surveys pre-fire and six surveys immediate post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2003. Means are derived from two different years, depending on subplot.
Figure 3.4. Mean relative abundance/subplot for *Plethodon glutinosus* for six surveys pre-fire and six surveys immediate post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2003. Means are derived from two different years, depending on subplot.
CHAPTER 4

CONCLUSIONS AND MANAGEMENT IMPLICATIONS
Forest plant species composition in eastern North America has changed over the past century due to timber harvesting, grazing, introduced pests, increased white-tailed deer populations, and changes in both natural and anthropogenic fire regimes (Abrams 2001). Oak (*Quercus* spp.) abundance has been declining in many Appalachian forest types, and forest managers wish to improve regeneration for both economic and ecological reasons (Schuler 2004). The combined use of prescribed fire and shelterwood harvests to improve regeneration appear promising (Brose et al. 1999), however the re-introduction of fire to ecosystems that have been under fire suppression for a century will have unknown effects. In addition, many mesic regions where prescribed fire could be employed have unknown historic fire intervals, or long fire-intervals. It is important to monitor the effects of silvicultural manipulations on flora and fauna communities to ensure the integrity of the ecosystem is not diminished. Retention of native species, species diversity, and abiotic factors associated with species requirements are important factors for the overall health of an ecosystem (Brose et al. 2001).

The results from this study have shown a low-intensity, spring prescribed fire in a central Appalachian hardwood forest had little short-term impact on eastern chipmunk home range attributes or a woodland salamander community. Chipmunk home range size, core area size, the extent of home range overlap, the extent of core area overlap, and the number of burrows used did not differ between animals living in burned sites and control sites. Results also indicate no differences between the sexes for all variables analyzed. Chipmunks readily used areas that had undergone prescribed fire, and the ephemeral changes in understory vegetation, fungi, vertebrates, and invertebrates may not have been profound enough to interfere with forage requirements. Eastern chipmunk diet consists primarily of hard-mast, which is hoarded during the autumn, and eaten throughout the following year. If decreases in alternate spring and
summer forage availability occurred, chipmunks may have compensated by relying on remaining burrow caches.

The effects of prescribed fire on salamander communities are virtually unknown in the mesic areas of the Allegheny Plateau. Pilliod et al. (2003) summarized studies conducted in the eastern United States and found variable results in salamander response to fire, depending on region and season. Additionally, high white-tailed deer densities in the eastern deciduous forests can cause dramatic changes in forest structure and plant composition (Tilghman 1989, Trumbull and Zielinski 1989, Waller and Alverson 1997), which have unknown effects on the salamander community. Habitat changes that reduce leaf litter and vegetative cover, increase solar radiation, decrease humidity on the forest floor, or alter invertebrate prey resource availability can increase physiological stress, predation risk, and competition in salamander communities (Pilliod et al. 2003).

The results from my study indicate woodland salamander abundance and species richness were not negatively affected by one spring prescribed fire. Initial increases in abundance and species richness were observed, but may have been due to the loss of leaf litter and animals moving under available cover objects. These effects were temporary, with values returning to pre-fire levels by one year post-fire. Negative effects of deer herbivory were not observed, with higher species richness in unfenced areas than fenced areas immediately post-fire. Seasonal shifts in species present were evident, with redback salamanders more abundant in the early spring and autumn. Northern slimy and Allegheny mountain dusky salamanders were more prevalent during the summer. These shifts may reflect differences in reproductive strategies, moisture thresholds, or competition avoidance mechanisms. The results from this study may not represent all age and sex classes because of the territorial nature of salamanders. Cover objects
often are defended by older or larger animals, which may exclude juveniles. Also, no attempts at sexing individual salamanders were made. The mesic nature of this forest type may have ameliorated the desiccating effects of prescribed fire, and conclusions should not be extended to other forest types. Long-term research is needed to fully understand the impacts of low-severity prescribed fire on this community, and future research regarding back-to-back silvicultural manipulations is essential.

LITERATURE CITED


Appendix A. Vascular plant presence pre-fire and 0 – 6 months post-fire on the Fernow Experimental Forest, West Virginia, 2001 – 2003. Half of sites underwent prescribed fire in April 2002, and half in April 2003, therefore pre-fire species may be from 2001 or 2002, and post-fire species may be from 2002 or 2003.

<table>
<thead>
<tr>
<th>Plant Species</th>
<th>Pre-fire</th>
<th>0 - 6 Months Post-fire</th>
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</thead>
<tbody>
<tr>
<td><strong>Trees</strong></td>
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