Effects of Wildfire Intensity on Invasives, Stand Structure and Fuel Loading in Shenandoah National Park

by

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(ABSTRACT)

As invasive species are so prominent, the influence of wildfire intensity on fuel loading, invasives, species richness, diversity, and evenness were studied at Shenandoah National Park. Most National Parks identify invasive species as the biggest threat to their goal of maintaining native ecosystems. Eight study sites were stratified into three burn classes (high intensity, low intensity, and control), and three transects were randomly located so that nested plots and fuel transects were measured at a distance of 50 ft (15 m), 150 ft (45 m), and 250 ft (75 m) from a road or trail. Field sampling was conducted between May 15, 2004 and June 30, 2004. A subsample of these plots were used to determine specific gravity and quadratic mean diameter for each size class of fuel and to determine the bulk density of the duff and litter layers. High intensity wildfires initially reduced species diversity and evenness in the tree and herbaceous strata, but after 14 years tree species diversity and evenness returned to levels found in unburned areas, while herbaceous strata diversity was not associated with time since burn. Low intensity wildfires resulted in the greatest impacts in the shrub stratum. Presence of invasive species was associated with more even and diverse vegetation in all strata, perhaps because invasive species were relatively sparse. Fuel loadings were reduced initially by high intensity

wildfires, but quickly returned to the same level as unburned areas. Although these initial findings indicate that invasive species will not persist after wildfire disturbance, continual monitoring by National Parks would be prudent.

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Introduction and Justification

Exotic species pose a serious threat to the Park Service's goal of maintaining native ecosystems. Hester (1991) states "National Park Service policy generally prohibits the introduction of exotic species into natural zones of National Parks;….exotic species that threaten park resources or public health, are to be managed or eliminated if feasible". The National Park Service considers an exotic species to be one that occurs out of its natural area because of some action by humans whether it be direct or indirect (Hester 1991). Invasive species are those which have a negative impact on the ecosystem where they reproduce, expand their range, establish themselves into a new area, and interrupt native species (Mack et al. 2000). Most U.S. National Park managers state that non-native plant species are a problem within their parks (Knapp and Canham 2000). When National Park managers were surveyed about threats to National Park resources, the majority of parks ranked invasive species or exotics as the biggest threat (Hester 1991). All federal park and forest lands have reported the presence of invasive species (Miller 1997). Invasive species have also caused negative ecological impacts in over half of the National Parks in Canada (MacQuarris and LaCroix 2003).

The majority of non-native species were introduced for food, fiber or ornamental purposes perceived to be beneficial uses of the species (Windle 1997; Pimentel 2002). Of all species introduced into the United States, at least 5,000 have escaped their intended cultivation sites and have invaded natural ecosystems (Pimentel 2002). In general, invasive species are pioneers in disturbed areas. Disturbances are common, whether they are natural or human caused. Intensification of natural disturbances, including wildfire, has been crucial in many of the largest biotic invasions (Hutchinson and Vankat 1997). The intensity, frequency and size of

such disturbances and the type of landscape system all influence the impact on the ecological community and the recovery process (Dale and Adams 2003). Less frequent disturbances of a greater magnitude tend to have the greatest impacts in terms of potential for invasibility.

It is important to understand the ecological implications of invasive species so that native ecosystems may be restored, rather than focusing simply on eradicating the invasive species (Hester 1991). Virginia's Department of Conservation and Recreation (1999) identifies five reasons to control invasive species: invasives disrupt complex ecosystems, reduce biodiversity, jeopardize endangered species, degrade habitat, and invasive species may transmit new diseases which the native species may lack a defense against.

A species' invasive potential must be studied in conjunction with its target community, since the manner in which the invasive species and the community interact determines the success of the invasion (Lodge 1993). Additionally, an understanding of community ecology is important to predict influences of an invasion in a community (Heffernan 1998). The ability of a species to invade other communities is the best prediction of its ability to invade in a new location (Heffernan 1998). Persistence and commitment is the key to success of controlling an invasive species (Mack et al. 2000).

Post-wildfire vegetation response within Shenandoah National Park has been studied by Regelbrugge (1988) and Wilson (1987). Research on invasive species present after a wildfire disturbance has not occurred, but the park has performed general surveys of invasive species within the park's boundaries (Akerson 2003). Neither Regelbrugge (1988) or Wilson (1987) included research on fuel load accumulation after varying levels of wildfire intensity or whether invasive species affected the accumulation rate by changes in stand structure.

The occurrence of invasive species has the potential to cause negative ecosystem and financial impacts, resulting from slowed growth in native species and decreases in species diversity (Heffernan 1998). It is known that invasive species invade after a disturbance, but it is not known how the level of wildfire intensity affects the presence of invasive species after a disturbance. It is also not understood how the stand structure is affected and how long invasive species may persist within the stand. An understanding of how invasive species affect native stand structure and composition is important for land managers to be able to respond after a wildfire in an effort to reduce negative ecological and financial impacts. Currently, tens of thousands of dollars are being spent on recommendations from Burned Area Emergency Rehabilitation Teams after large wildfires, to deal with potential invasive species problems. These resources may be better utilized once affects of invasive species are better understood.

Objectives

There are five major objectives for the proposed research.

1) Assess the density and dominance of invasive species that occur in Shenandoah National Park after wildfire disturbance.

2) Determine the affect of wildfire intensity on invasive species.

3) Determine the effect of time since wildfire on invasive species.

4) Measure native species composition and fuel loadings under varying levels of wildfire intensity and time since wildfire occurred.

5) Determine if the presence of invasive species is affecting native species composition, structure, and fuel loadings.

Literature Review

Ecological Framework

Cover Types

According to Braun (1950), the oak-chestnut cover type is the predominate forest type in Shenandoah National Park. American chestnut (*Castanea dentata* (L.) Mill.) was widespread in the area until the early 1900's when the chestnut blight killed all of the overstory. The lower slopes having been harvested in the period around 1900, are comprised of second growth forests. Black locust (*Robinia pseudoacacia* L.) and sassafras (*Sassafras albidum* (Nutt.) Nees), both pioneer species, are often found in old fields and old clearings, while yellow poplar (*Liriodendron tulipifera* L.), preferring moist conditions, will be found in the coves. Eastern red cedar (*Juniperus virginiana* L.) is common on areas of limestone parent material that exists on the lower slopes of the western side of the park (Braun 1950). Birch (*Betula spp*. Marsh.) is common on upper slopes, as is a component of sugar maple (*Acer saccharum* Marsh.), basswood (*Tilia americana* L.), and birch with northern red oaks (*Quercus rubra* L.).

The composition of species greatly depends on the slope and aspect and whether the slope is concave or convex. Yellow poplar is often found on the north facing coves of ridges less than 2500 ft (760 m) in elevation. On the knobs and convex slopes chestnut oak along with some pockets of pine (*Pinus spp*. L.) are common (Braun 1950). The very rocky ridges and accompanying slopes have sparse trees. Spruce (*Picea spp*. (Moench) Voss) and fir (*Abies spp*. (L.) Mill.) can be found in the park but are limited to the summit of Hawksbill Mountain. The majority of the canopy in Shenandoah National Park is comprised of northern red oak, sugar maple, hemlock (*Tsuga canadensis* (L.) Carr.), chestnut oak (*Quercus prinus* L.), white oak (*Quercus alba* L.), and basswood (Braun 1950). Based on forest inventory data (obtained from

Dan Hurlbert, Shenandoah National Park, on January 12, 2004) of Shenandoah National Park, completed in 1985, chestnut oak comprises 49% of the overstory, yellow poplar comprises 16%, cove hardwood makes up 15%, northern red oak comprises 9%, pine makes up 5%, and black locust makes up 4%. Less than 1% of the area is rock, open areas, or other species.

White Oak Canyon, an area of undisturbed forest, is located on the eastern slopes of the Blue Ridge Mountains within Shenandoah National Park. The cover type comprised of white oak, red oak, chestnut oak, hemlock and other species, varies within the area but can basically be separated into the areas with sugar maple and the areas without sugar maple present in the overstory canopy. Braun (1950) notes that the species composition in the late successional forest containing sugar maple is more diverse with some 17 tree species, compared to only 8 tree species where sugar maple is absent from the overstory.

Land Use History

The pre-European settlement upland forests in this area had many sections with trees fairly widely spaced, making for open conditions with a grass understory. The swampy areas were more dense with thick understory vegetation (Martin 1973). There were many open wooded areas, as well as grass lands and areas that had been farmed by the Native Americans (Day 1953). Portions of the Shenandoah Valley were entirely void of trees during Native Americans presence (Day 1953). Native American use of fire altered the species composition of upland forests (Day 1953).

Pre-European settlement forests in the Shenandoah National Park area of Virginia were dominated by white oak, northern red oak, chestnut oak, hickory (*Carya spp*. (Mill.)), and pine (Abrams 1992). The oak-chestnut forests ranged from New Jersey to central Pennsylvania and south to Virginia. On a site in New Jersey for the period 1641 to 1711, a mean fire interval of 14

years was reported; after 1711, near the time of European settlement, no fire scars were recorded (Abrams 1992). DeVivo (1991) states Native Americans inhabited the vast open fields and used fire to maintain the landscape prior to the area being settled by Europeans. The open areas have been linked to the Native American use of fire for many uses ranging from cooking, heat, hunting, field preparation, killing woody vegetation, improving travel, war purposes, and increasing the abundance of grass seeds and berries (DeVivo 1991; Day 1953). Native Americans also made use of the forest for firewood. Thus, the longer a Native American village was inhabited the larger the opening in the forest became. Abrams (1992) states that there is controversy as to whether the Native American use of fire increased the frequency of fire above natural levels, but some accounts and charcoal studies support the claim. As the Native American population decreased, pioneer species took over the fertile farmland allowing forest succession to occur.

A post-European settlement cycle of repeated logging and fire through the 1800's for charcoal iron production and land clearing aided the dominance of oak in the Mid-Atlantic area (Abrams 1992). White oak and northern red oak dominated near iron furnaces in post-European settlement times in central Virginia.

Frequent fires reduced the understory and midstory components allowing oak to become established as advanced regeneration. Oaks currently dominate on the mesic to xeric sites that were harvested and burned around 1900 (Van Lear 1991). Long term burning regimes, killed the understory and reduced the midstory and overstory, as a result of fire wounds, and led to open oak stands where oaks dominated the understory. The regime of frequent fire allows oak to establish but is also necessary to control competition while oak establishes itself (Van Lear 1991). The frequent burning also leads to better growth form amongst oaks as many stump

sprouts are susceptible to rot. The lack of fire has resulted in changes in species composition. Mesic species of trees and shrubs are able to persist on drier sites that had been oak dominated when fires occurred more frequently (Van Lear 1991). The lack of fire has allowed rhododendron (*Rhododendron maximum* L.) and mountain laurel (*Kalmia latifolia* L.) to create thickets preventing the oaks from regenerating (Van Lear 1991). After European settlement, the oaks dominated most areas because of their ability to adapt to drier sites and the reduction in fire frequency. Oaks have needed fire to aid their development, but in this case, savannahs or very open forests existed, allowing the sun to reach oak seedlings enabling them to grow and fill in the canopy to form a closed oak forest (Abrams 1992). The eastern United States has long been dominated by oak species, whose presence is associated with recurring fire. Now with the lack of fire to create openings, conditions are too shady for oaks to mature beyond the seedling stage (Abrams 1992).

Fire Regimes

The southern Appalachians have historically been maintained in a regime of frequent fire (Van Lear 1991). The Appalachian mixed-oak forests have been subjected to three different fire regimes over the last 500 years (Brose et al. 2001). First, the periodic low intensity fires utilized by Native Americans; second, the high intensity, stand replacing wildfires associated with industrial logging of the area; third, wildfire suppression during the past 80 – 100 years. Each of these regimes has caused ecological changes; the most dramatic changes can be associated with the regime of wildfire suppression which has been in place for over 80 years (Brose et al. 2001).

In the mid 1700's, Native Americans would annually burn the Shenandoah Valley to keep it an open grassy area to use for farming. For the most part, the Native Americans applied frequent low intensity surface fires to the landscape to achieve their goals. Some areas were

burned more frequently than others, but on average the woodlands burned every 10 years (Brose et al. 2001). Van Lear and Waldrop (1989) state it is suspected these fires progressed to the surrounding forest areas along the mountains, where the moist cove areas would eventually cause the fire to die out. This type of burning would have led to open forests. According to Martin (1973) light surface fires set by the Native Americans, caused litter to turn to ash, improved soil fertility and reduced the chances for high intensity fires to occur. Historical accounts show the southern Appalachians burned more frequently which kept them more open than the northern Appalachians (Van Lear and Waldrop 1989).

The Europeans settled in the Appalachian area in the mid-1700's to early 1800's, but did not dramatically change the fire regime, as they probably saw how well it was working for the Native Americans (Pyne et al. 1996). The change in fire regime did not occur until the period between 1880 and 1930 when industrial harvesting took place throughout the area. The lumber companies installed railroad tracks to the most accessible areas first and then continued to the less accessible tracts. The harvest of millions of hectares left large areas of dried slash which was easily ignitable. The steam powered locomotives caused sparks along the railroad tracks which accounted for 71% of wildfires according to a 1907 survey in West Virginia (Brose et al. 2001). The intensive harvesting and use of locomotives led to very intense and large stand replacing wildfires. These wildfires are responsible for reducing the extent of montane coniferous forests within the Appalachians and converting them to hardwoods. At the height of the intensive logging in Pennsylvania almost 1,000,000 acres (404,700 ha) burned annually; in comparison, currently only 3300 acres (1336 ha) burn on an annual basis (Brose et al. 2001).

With the establishment of the U.S. Forest Service came the regime of wildfire suppression and when the timber supply was depleted, locomotives were less common,

eliminating the source of ignition associated with many wildfires (Brose et al. 2001). In the 1920's, the U.S. Forest Service opposed all fires, and even though some people would still burn their land, it led to a much longer interval between fires (Van Lear and Waldrop 1989).

The reduction in fire has had both positive and negative ecological impacts. On a positive note, the lack of fire has allowed oaks to grow into and close the canopy creating high quality hardwood stands. However, after 80 years without fire, many shade tolerant and non-fire resistant species comprise the overstocked understory, and there is minimal oak regeneration occurring (Brose et al. 2001). The dominance of oak was further aided in the early 1900s by the loss of the American chestnut and the exclusion of fire. Presently, many oak dominated forests are transitioning to include more late successional species in the overstory, even though oaks are still present in the understory. If left alone, the oaks will eventually die off and the shade tolerant, less fire resistant species (such as red maple (*Acer rubrum* L.) and blackgum (*Nyssa sylvatica* Marsh.)) will become the next overstory (Christensen 1977). This could mean these stands are more at risk for canopy replacing burns, which leads to major canopy gaps creating light conditions necessary for invasive species to grow (Major 1996). Where fire has been excluded, the loss of oak dominance, acts as indirect evidence to show that periodic fires were important in maintaining oak dominance.

Two peaks in wildfire activity occur in the Appalachians, one in late spring to early summer and the other in the fall. Major (1996) found human caused wildfires to be more important in shaping the landscape than lightning caused wildfires in the Southern Appalachians. Irving (1983) states prescribed fire will determine the fire regimes of the future. It is vital for land managers to understand the way fire shaped the ecosystem in the past to predict the consequences of future fire management (Irving 1983).

Fire Dynamics

Fuels

Forest fuels direct the ignition, buildup, and behavior of wildfire more than any other variable (Brown and Davis 1973). Fuel is added to the forest floor each year as the forest grows and leaves or needles are shed. The litter and duff layers make up the fine fuels. These fine fuels can ignite easily and burn quickly as they direct the behavior of the wildfire, because they take the least amount of time to dry out (Brown and Davis 1973).

Fuels are classified by time-lag classes, which define the time required for the moisture content of a piece of fuel to equilibrate with the surrounding air (Brown and Davis 1973). Smaller fuels have a shorter time-lag than large fuels, hence their importance in determining wildfire behavior. One-hour fuels are those < 0.25 in (< 0.6 cm) in diameter, 10-hr fuels are 0.25 – 1 in (0.6 cm – 2.5 cm), and 100-hr fuels are 1 – 3 in (2.5 cm – 7.6 cm) in diameter. 1000-hr fuels are those between 3 in and 8 in (7.6 cm – 20.3 cm) in diameter. Often the accumulation rate of fine fuels will approximately equal the decomposition rate which balances the fuel load, however an ice storm, hurricane, or disease outbreak has the potential to greatly increase larger fuels very quickly (Brown and Davis 1973).

Amongst leaves and needles, the ability to ignite depends upon the arrangement, quantity, and the physical characteristics of the fuel (Brown and Davis 1973). Deciduous leaves quickly decompose and therefore have less influence than conifer needles which may persist for 10 years. The litter orientation in terms of its exposure to the air greatly influences its flammability. Different tree species drop leaves at varying rates and their leaves also decompose at varying rates; therefore, equal volumes of fuel may behave differently during a wildfire depending on

their stage of decomposition. Shrubs and small trees less than 6 ft (2 m) in height may speed or hamper the spread of a wildfire depending on their moisture content (Brown and Davis 1973).

Brown (1974) and Brown et al. (1982) outline the common practice of the planar intersect technique which is used to characterize fuel loadings in forested stands. This is a nondestructive method, which is quick and avoids the need for weighing large samples of woody debris. The sampling technique consists of three sampling planes, one for each size class of fuel, along a transect. Duff and litter depth are also measured to incorporate these fuels into the total fuel loading per acre. Litter depth is measured from the top of the fresh litter fall to where the litter begins to decompose. The duff layer lies underneath the litter and ends at the top of the mineral soil. The result of the method is a value in tons per acre for different fuel classes, which is the necessary input for fire behavior models (Rothermel 1972). Harmon and Sexton (1996) suggest the use of fixed area plots for measuring fuels. However, the fixed area plot method is focused more towards inventories of mass and nutrient stores. As such, it has a more detailed resolution in terms of decay classes and is geared toward long-term monitoring of change. The planar intersect technique is used by many forest managers and in the National Forest Inventory and Analysis program.

Fire Behavior

Fuel loads and the moisture content of that fuel determine the duration of elevated temperatures and maximum temperature reached during wildfires (Huddle and Pallardy 1996). In general, fire spreads unevenly through a stand as it goes through varying terrain and litter (McGee et al. 1995), however, prescribed fires are usually of lower intensity and only partially consume the litter, while a wildfire is often patchier but consumes the entire litter layer. Wildfires usually occur in the dry season and burn hotter than a prescribed fire (Rieske 2002).

Fire Intensity

The generally accepted lethal temperature for plant tissue is 140ºF (60ºC) (Sweezy and Agee 1991). The height to which scorching occurs is directly related to fire line intensity (Wyant et al. 1986). Crown damage is most dependent upon fire intensity, but bole and root damage is most dependent upon duration and amount of heat (Ryan and Reinhardt 1988). Van Wagner (1973) stated except where prolonged burnout is occurring, fire intense enough to girdle a tree is probably also intense enough to scorch most, if not all, of its foliage. The height of charring along the bole relates directly to the amount of heat which impacted the tree. The total amount of char does not indicate the maximum temperature or duration of heating, but the change in char around the stem of the tree relates to the change in heating around the bole.

The intensity and extent to which fire affects the tree bole changes depending on the position around the bole. For example, fuels tend to accumulate on the uphill side which potentially can lead to longer burnout or smoldering, which keeps the heat in contact with the bole of the tree (Ryan 1983). In other situations, where fuel is not accumulated more on one side of the tree bole than another, the side of the tree facing the fire front is impacted the most.

Depending on the intensity, a single wildfire may have varied results on the vegetation present. Regelbrugge (1988) in a study in Shenandoah National Park, found high intensity wildfires decreased community diversity and low intensity wildfires had no significant affect on the overstory. No significant change in species richness was found at either intensity level. High intensity wildfires resembled clear cuts, as more canopy mortality occurred allowing for more sunlight to reach the forest floor. Areas which experience the highest intensity wildfire also have the highest overstory mortality (Elliot et al. 1999). Increased light intensities, soil temperatures, and surface temperatures will affect species present after the wildfire (Watt 1992). Groeschl et

al. (1991) found high intensity wildfires may reduce site nutrients. Species which can adapt to low fertility will be the first to dominate the post-wildfire site. Consumption of the forest floor fuels is directly correlated to the intensity of the wildfire (Groeschl et al. 1991). Many studies have shown that prescribed fires, due to their lower intensity, do not cause a significant reduction in the forest floor. Nitrogen increases on the site following a low intensity fire because incomplete combustion occurs, whereas a loss of organic matter leads to a reduction in carbon content on a site (Groeschl et al. 1991). Major (1996) found with a low intensity burn, no real changes in understory species composition or abundance occurred.

Fire Disturbances

Understanding the composition and structural changes in a stand after a wildfire is important to making management decisions (Schmalzer and Hinkle 1992). Wildfire can remove competitors and open the canopy to increase light levels allowing exotic species to invade in the period following the disturbance (Reichard and Hamilton 1997). Groeschl et al. (1991), in a study conducted at Shenandoah National Park, states wildfire may result in a species shift, as nutrient levels are reduced, some species may not be able to thrive and as such those species no longer occupy the site. Periodic burns in the Missouri Ozarks resulted in more damage than annual burns, because it allowed more fuel and litter to accumulate between burns (Huddle and Pallardy 1996). For white oaks, the annual burning regime resulted in significantly lower diameter growth than in the periodic and no burn regime. For the hickory and northern red oaks there was no significant difference in diameter growth among the three treatments (Huddle and Pallardy 1996).

The extent of the disturbance affects light levels, seed sources of colonizers, and the growth of survivors after the wildfire. Across the landscape, the vegetation becomes patchy as

wildfire intensity and frequency varies, and as such may increase species diversity. The frequency, extent, and intensity of a wildfire determines how the individual plant and community of plants will be affected (Grime 1977). As the frequency of wildfire increases, vegetative dominance will shift from fast growing species to species more resistant to wildfires (Grime 1977). Populations under stress often have different responses to disturbances, such as: variable occurrence, increase, decrease, invade, or retreat (DeSelm and Clebsch 1991). When woody plants are top killed, resprouting occurs from suppressed or adventitious buds. As burning increases, tree cover will likely be reduced, but grass, forb, shrub, and vine cover will increase (DeSelm and Clebsch 1991).

Wildfire modifies vegetation in several ways including alteration of photosynthetic rates and efficiency, and modifying vegetation physiology, which in turn, influences competitive ability and rate of succession (Rieske 2002). Physiological changes may stem from a variety of sources such as changes in photosynthetic capacity caused by an increase in root-shoot ratio resulting from reduced foliar biomass, increase in availability of soil nutrients leading to changes in foliar nutrient concentrations, changes in leaf gas exchange, and a decrease in source-sink ratio in post-burn vegetation. Such changes in physiology of vegetation after a wildfire may cause a reduction of oak competition, increases in oak regeneration growth, or changes in overstory tree growth. Wildfires have many benefits for oak, including: aiding oak regeneration, hindering presence of fire-intolerant species, increasing stand vigor, and modifying species composition (Rieske 2002).

In a prescribed fire at Nantahala National Forest, fire reduced the abundance of red maple and increased the growth and establishment of oaks. Overall the non-woody species increased while woody species decreased in terms of relative percent cover after the fire (Elliot et al.

1999). Heffernan (1998) states fire stimulates the growth of some invasives such as garlic mustard (*Alliaria petiolata* (Bieb.) Carara & Grande) and Chinese lespedeza (*Lespedeza cuneata* (DuMont) G. Don.). Canopy opening disturbances are ideal for oak canopy accession, however these are also good conditions for some invasives to grow (Watt 1992). Burning may encourage germination of seed stored in the soil. In a loblolly pine (*Pinus taeda* L.) stand one year after fire, the number of plant species increased significantly (Watt 1992). Winter burning, while increasing diversity of herbaceous species, did not increase the total number of herbs, shrubs, or vines compared to the unburned areas (Watt 1992).

The disturbance history of an area can explain some of the invasion of exotic species. Prescribed fires in hardwood stands of the southeast have been found to increase the presence of herbaceous plants and vines (Wendel and Smith 1986). After a fire, herbaceous species increase for a period, but woody species composition often stays the same other than the above ground portions being killed by the fire (Matlock et al. 1993). In ecosystems adapted to frequent wildfires, woody species often have many dormant meristems. Wildfire also may modify the seed bed enabling new plants to germinate or sprout from buried buds (Matlock et al. 1993). Stem density quickly returns to pre-wildfire levels, probably by self-thinning among young shoots.

The overall impact of wildfire on vegetation has been primarily seen as setting back the developmental process of succession to an earlier stage. In oak-saw palmetto (*Serenoa repens* (Bartr.) Small) habitat, the percent bare ground was reduced to near zero 18 months after a fire, and post fire vegetation greater than 1.5 ft (0.5 m) equaled the preburn levels in 3 years after the burn. Species richness increased after the fire, but the increase was not statistically significant. The season of the burn and fire intensity can affect resprouting recovery. Many studies have

documented severe changes in species composition after a fire, but others have shown no impact after a fire (Abrahamson 1984). This difference is likely due to species response to different intensities of fire.

Major (1996) found in the herbaceous layer, 14 species existed prior to the burn and 29 species were present one growing season after the fire, however most were present at a low frequency. The majority of herbaceous vegetation declined by the fifth year following fire because of a reduction in light resulting from canopy closure (Godsey 1988). Godsey (1988) found an increase in grasses and legumes after periodic and annual burning in the Missouri Ozarks, but perennials were only able to become established after periodic burning.

Tree mortality and species response

A tree's resistance to bole damage is highly correlated to bark thickness, which varies by species, age, height above the ground, and the pre-fire health and vigor of the tree (Brown and Smith 2000). Bark's insulative properties depend on its moisture content, density, structure, and composition. Trees with thick bark tend to protect the cambium from fire injury, but the cambium can still be damaged by prolonged burnout or smoldering near the base of the tree. The density of bark is influenced by the amount of cork present. Trees having denser bark, have higher thermal diffusivity. Bark density will also vary by species. The heat capacity of bark is very close to the heat capacity of the actual wood (Reifsnyder et al. 1967). According to Rouse (1986), chestnut oak has the thickest bark of the oaks, black oak (*Quercus velutina* Lam) and northern red oak are intermediate in thickness, and white oak has the thinnest bark. Besides simply bark thickness, the configuration of the bark, such as the fissures and ridges, and the chemical makeup of the bark are also important in determining fire resistance among oaks (Rouse 1986).

Trees which have previous fire injured cambium or mechanical injuries are more at risk for additional fire damage to occur. The degree to which injury harms the tree is directly related to the tree's ability to recover and continue functioning (Brown and Smith 2000). Certain species such as black oak are able to compartmentalize around a wound which prevents infection from entering at the fire scar and spreading into live healthy tissue (Brown and Smith 2000). Bole damage among southern pines may take some time to result in the death of a tree. For example, if the phloem is affected by lethal temperatures and the xylem is not affected, then water will still be able to be transported from the roots to the crown. It may take 1 – 2 years for fully developed root systems of healthy trees to use up their food supplies and eventually die (Wade and Johansen 1986).

Cambial damage is a more important factor leading to mortality in small diameter trees, while crown injuries are much more important in larger diameter trees (McHugh and Kolb 2003). Amongst pines which suffer crown scorch in the spring, variation in survival is related to bud type. Loblolly pine, longleaf pine (*Pinus palustris* Mill.), shortleaf pine (*Pinus echinata* Mill.), and slash pine (*Pinus elliottii* Engelm.) do not have preformed buds, which allow them to have multiple successive needle flushes within a growing season. On the other hand, species such as eastern white pine (*Pinus strobus* L.) have preformed buds, so they only have one flush of needles each year, meaning the tree must wait until the following growing season before producing new foliage (Wade and Johansen 1986).

Oaks are a fire adapted species by having the ability to repeatedly resprout after being top-killed by fires and mature oaks survive by having a thick layer of bark. Annual burning of hardwood species will eventually kill all the hardwoods, but oaks will be the most persistent. Oaks are able to survive annual winter burnings. The peak of carbohydrate reserves in the

winter, allows oaks to sprout more vigorously after a winter fire than after a growing season fire. With fire, oak will comprise a larger proportion of the number of stems in a stand (Van Lear 1991). When oak dominated stands are harvested, they regenerate to other species in the absence of fire. Fire acts to remove the shade tolerant and fire intolerant species.

In general, as bark thickness increases, the temperature gradient decreases and the maximum cambial temperature will be lower. Mid-successional oak species being replaced by climax maple species may result from reduced fire frequency (Hengst and Dawson 1994). Unless competing vegetation is controlled, this trend will likely continue. The thin bark of maple, leading to reduced survival following fire, suggests prescribed burning may reduce the transition from oaks to maples. Hengst and Dawson (1994) found that upland species in Illinois tended to have thicker bark for a given diameter than bottomland species. While there are thermal and physical differences of bark between species, the bark thickness differences are considered to be most important in protecting the tree from cambial injury. In addition bark thickness is much easier to measure than thermal properties (Hengst and Dawson 1994). The specific gravity of bark is higher for species with thin bark such as sugar maple than for species with thick bark such as oak. According to Hengst and Dawson (1994), the moisture content of the inner bark is generally accepted to be higher than in the outer bark. Species with thin bark have high thermal diffusivity, increasing the time necessary for ignition to occur. As bark thickness increases the time required to reach maximum temperature and then return to ambient conditions also increases. Proportion of bark tissue varies by species, ranging from 4.6-6.5% in white oak, to 5-6.5% in black oak, 5-5.6% in scarlet oak (*Quercus coccinea* Muenchn.), 4% in hickories, and 3.5-3.7% in red maple. As a result, the critical diameter for white oak to survive is smaller than for red maple to survive (Huddle and Pallardy 1996). Fire resistance of species

also depends on the heat capacity of bark, ability to sprout as seedlings, and storage of nutrients. Storage ability tends to increase with tree size, but the ability to sprout decreases as trees get larger.

Low intensity fires in combination with lightning caused wildfires perpetuated the oaks and chestnuts. The oaks were favored by fire because of their thick bark, ability to compartmentalize rot and sprouting ability. In addition, fires freed up resources to be used by the oaks thus, eliminating the more fire sensitive species competing with the oaks. Fire may also have aided the oaks by reducing insects known to hinder acorn germination and preparing the necessary seedbed for oaks (Brose et al. 2001; Abrams 1992).

Elliot et al. (1999) found, after a prescribed burn in the Nantahala National Forest, that the relative percent cover of growth forms differed from before the burn. For example, prior to the burn non-woody species represented 6% of the relative cover and after one growing season since the burn, non-woody species accounted for 22% of the relative cover (Elliot et al. 1999).

Watt (1992) states the future species composition after a prescribed fire in South Carolina depends as much on the frequency of fires as it does the intensity of fires. Watt (1992) found very young root stocks to be more susceptible to fire damage than older root stocks. McGee (1980) found the large increase in stems after fire created a canopy opening was mainly a result of multiple sprouting from existing root stocks rather than new seedlings. A study involving a spring fire in New York found perennial forb species were not negatively impacted by the fire, and that within one growing season most had either met or surpassed their preburn densities (McGee et al. 1995). Groeschl et al. (1991) found in Shenandoah National Park after a wildfire in a pine-hardwood forest that mortality averaged 20% after the first year and 40% after the second year, suggesting that delayed mortality may occur (Major 1996). Trees which are self

pruners may be more equipped to resist fire damages because the branches and foliage are further from the flames. Dense stands may also enable the trees to survive better because of self pruning and poor air circulation (Godsey 1988).

Invasive Species Threat

Ecological Niche

Eighty percent of invasive species originated from Europe or Asia and most were introduced as ornamental species (Hutchinson and Vankat 1997). Currently more is understood about how invasive species enter the country than how these species manage to spread outside their original ranges within the U.S. (Windle 1997). Invasive species often become established by way of human activity, transport or habitat alterations which provide new opportunities for species establishment (Windle 1997). Human activities include the creation of disturbed sites through processes of agriculture, fire, logging, or grazing (Sakai et al. 2001).

Recent studies suggest that species diversity and scale of disturbance are just as important as the extent of the disturbance in understanding invasive potential (Sakai et al. 2001). Areas of high disturbance may be more susceptible to invasion (Horvitz et al. 1998). A study on Prince Edward Island by MacQuarrie and LaCroix (2003) looked at depth of forest edge and invasion patterns. They found that edges tend to be good areas for establishment of exotic species because compared to interior portions of the forest, the edge has warmer air and soil temperatures, higher light intensity, greater wind velocity, greater species richness, higher vegetation density, drier soils and smaller diameter trees (MacQuarris and LaCroix 2003). The most extensive plant species changes will likely occur within 160 ft (50 m) of the edge (Brothers and Spingam 1992) to 330 ft (100 m) into the interior of the undisturbed forest (Chen et al. 1992). Plots adjacent to agricultural fields had higher soil temperatures and less canopy cover

than the forested plots. Plots closest to the edge had more exotic species and a greater proportion of exotics, and native species had lower richness and proportion compared to the interior plots (MacQuarris and LaCroix 2003). Most exotic species (65%) were limited to the nonforest adjacent habitat. The proportion of total species considered exotic decreased at the more interior plots. The depth of edge is greater between a grass area and forest than between a 15-year old and an 80-year old stand. This is true because the continued disturbance in a grass area or some other area allows for more opportunities for intentional or accidental introduction of exotic species (MacQuarris and LaCroix 2003). Exotics showed a negative correlation to the increase in canopy cover, thus disturbances creating openings in the canopy may aid invasion by exotics.

After a disturbance, revegetation is rapid, but species composition changes. There may be an increase in opportunistic species such as yellow poplar, black locust, and red maple according to Elliot et al. (1997) who conducted a study in the Coweeta Basin in western North Carolina. In eastern forests there tends to be an initial increase in species diversity and richness, and then a decrease as the forest matures. Initial species abundance after a disturbance, is determined partially by stress tolerance, growth rate, and ability to competitively capture nutrients (Elliot et al. 1997). Prolific and early production of wind disseminated seeds is usually linked with the ability to compete after a large disturbance. The soil often contains seeds of early successional plants that remain viable for years, and sprout following a disturbance (Bazzaz 1979). The seeds of early successional species are often sensitive to light and unable to sprout under shaded conditions (Bazzaz 1979). Effects on the forest ecosystem in terms of regeneration or revegetation, structure and productivity, depend on the frequency, intensity and scale of disturbance (Elliot et al. 1997). In looking at the recovery process for an ecological community,

not only are the characteristics of the invasives important, but the characteristics of the plant species which survived the disturbance are equally important (Dale and Adams 2003).

Elliot et al. (1997) found that a large disturbance such as a clear-cut in the Coweeta Basin, causes forest floor microclimate changes which lasted for 3 years following the disturbance when canopy closure occurred. Such changes included increased temperatures at the boundary between the litter layer and soil, lower litter moisture, and increased soil moisture. The increased soil moisture likely occurred due to the removal of trees resulting in less water uptake allowing the water table to rise closer to the surface (Elliot et al. 1997).

The invasion potential of a species depends on a combination of that species growth habits as well as the scale of the disturbance and the influence of the disturbance on the growth habits of the species (Runkle 1985). Canopy gaps will increase the amount of light reaching the forest floor, until the canopy closes again. Knapp and Canham (2000) concluded that gap size, in a New York study, was irrelevant in terms of the amount of light reaching the forest floor between gaps 190 - 410 ft^2 (17 – 37 m^2) in size.

In a central Indiana study of undisturbed old growth forests, exotic species richness and frequency decreased rapidly as one moved from the forest edge toward the interior. Low light levels and low disturbance may possibly be contributing to the reduction in exotic species presence (Brothers and Spingarm 1992). Micro-environmental changes along the forest edge may allow some exotics to take hold, but this usually occurs in open and disturbed forests. They found going from the agricultural field to 6 – 10 ft (2 – 3 m) inside of the forest caused air temperatures to decrease by 3.2º F (1.8º C), soil temperature to decrease by 6.7º F (3.7º C), and relative humidity was 9% higher. Most exotic species were found right at the forest edge because low light was the most limiting factor (Brothers and Spingarm 1992). The disturbances

which drive succession also create ideal conditions for invasive species to establish (Gordon 1998). Maintaining an intact canopy has aided some forest stands from being penetrated by invasive species, but not all stands with intact canopies have remained protected (Mack et al. 2000). Hutchinson and Vankat (1997) conducted a study in southwestern Ohio which suggests that early to mid-successional communities are more susceptible to invasion than late successional communities, because disturbance reinitiates succession processes.

McGee et al. (1995) states that community stability may be attributed to the inability of invasive species to persist. Usually invasives are considered to thrive on disturbed sites, but several non-native trees and shrubs have successfully invaded intact forest habitats. Examples include Norway maple (*Acer platanoides* L.), glossy buckthorn (*Rhamnus frangula* L.) and autumn olive (*Eleagnus umbellate* Thunb.) (Sanford et al. 2003). Generally environmental variability is higher in early successional stages and decreases in late successional stages (Bazzaz 1979). Plants trying to establish themselves in a clear-cut or an open field experience more extremes in air temperature, soil temperature, and soil moisture, which they must adapt to, compared to conditions under an existing forest canopy. Sakai et al. (2001) states that more research is needed to grasp the extent to which human disturbances influence establishment of invasives.

Many studies have discussed the impacts of invasive species on native species and natural stand structure (Dale and Adams 2003, Gordon 1998, Williamson 1996, Wilcove et al. 1998, Parker et al. 1999, and Stein et al. 2000). The potential for invasive species to spread diseases to native species is of high concern (Dale and Adams 2003, Virginia Department of Conservation and Recreation 1999, Parker et al. 1999), as is the potential of invasives to disrupt or modify ecosystem properties, such as nutrient cycling or water cycling (Gordon 1998).

However, more research is needed in terms of evolution, genetics and interactions of invasive species on native species in the invaded communities, to make predictions about the susceptibility of ecosystems to invasion by exotic species (Sakai et al. 2001). The process of invasion has many steps, including: native elsewhere, survival in transport, establish in new areas, lag period, spread, ecological impact, and human impact. There are questions to ask at each of these steps to aid managers in controlling invasive species (Sakai et al. 2001). Baker (1974) discussed traits which make a species a good invader. Traits included: reproductive ability both sexually and asexually, rapid growth from germination to sexual maturity, adaptive ability to environmental stress, and high tolerance to environmental heterogeneity. These traits are discussed in literature (Sakai et al. 2001; Newsom and Noble 1986), but little empirical data exists to support Baker's (1974) invasive species traits. Reichard and Hamilton (1997) in their analysis of traits of introduced woody plants, found that high potential for invasion was linked to species with vegetative reproduction, no need for seed treatment, perfect flowers, and persistent fruit on the plant. Kolar and Lodge (2001) state that common characteristics of invading species include pioneer habit, short germination time, high growth rate and high fecundity. Baker (1974) says that the smaller seed is linked with higher seed crops, faster growth rate and no need for seed treatment prior to germination, while Forcella (1985) states that the heavier seeds germinate faster and therefore are better invaders. When looking at invasion failures in both plants and animals, reasons include inability to pre-adapt to new climate, disturbances, competition and predation from native species and diseases.

Lack of Parasitic Agents

When an exotic species is introduced to a new area, its natural predators are usually not introduced with it, thus allowing the species to grow free from predation (Miller 1997). The lack

of predators for an invasive species enables it to grow uncontrollably and better utilize the available resources, because energy is not being devoted to defense mechanisms (Sakai et al. 2001; Mack et al. 2000). There are few dominant vine species native to the southeastern United States. Most of the vines present are invasive species, which are successful in a new ecosystem because there are no natural controls and the invasives' competitive nature allows them to out compete the native vegetation (Schweder 1993). Despite an environment free of natural predators, extinction of invasive species is more common than successful establishment (Mack et al. 2000).

Fast Growth Rate

Some vines can grow much faster than trees, since in woody stems, increase in height must be matched by increase in volume and diameter, while in vines the apical meristem is where growth is concentrated for increasing leaf surface area (Schweder 1993). Prolific seed production and below ground rhizomes are a means of spreading (Miller 1997). Invasives may be able to allocate more biomass to growth since none or very little is needed for defense mechanisms as there are usually no natural predators for an invasive. This gives invasives a growth advantage over native species (Sakai et al. 2001).

Because of kudzu's (*Pueraria montana* (Lour.) Merr. Var. *lobata* (Willd.) Maeser & S. Almeida) high specific leaf area and its small allocation to stem biomass, it has a high collection-support ratio. The collection-support ratio is the ratio of light and CO_2 collecting surface area to the dry weight of the tissues supporting this surface. It is also a means of determining a plant's efficiency at building its canopy given its growth form. Trees often reach taller heights and have comparable photosynthetic rates, but their low collection-support ratio prevents them from growing over kudzu (Wechsler 1977). More research is needed to be able to identify when the

lag phase behavior is occurring or is going to occur as it seems to vary among species. This is necessary so that a low abundance of invasives is not mistaken for a non-invasive that will eventually cause a problem (Frappier et al. 2003). Once a species becomes invasive it will continue to spread until it reaches the limits of its range (Mack et al. 2000).

A study by Pattison et al. (1998) in Hawaiian forest species showed that in full sun 5 invasive species, including *Schinus terebinthifolius* Raddi, *Citharexylum caudatum* Mill ex. L., *Cestrum nocturnum* L., *Psidium cattleianum* Sabine, and *Bidens pilosa* L., attained 2.5 times higher growth rates than native species. Four of these species are evergreen perennials, while *Bidens pilosa* L. is a herbaceous plant. This was achieved because of the invasive species' greater photosynthetic capacity (Sanford et al. 2003). Rapid growth is a common trait amongst invasives, such as tree-of-heaven (Sanford et al. 2003). Carter et al. (1989) in a study of photosynthetic abilities of exotic vs. native vines, found that the extreme growth rates of exotic vines is not highly explained by higher levels of photosynthesis. Their study found that steady state photosynthesis performance was comparable for native and exotic vine species (Carter et al. 1989). Resistance to drought by means of deep roots is a mechanism favoring kudzu's invasibility (Wechsler 1977).

Use of Resources

Once an invasive species has colonized itself in a location, it must then develop a thriving sustainable population (Sakai et al. 2001). A successful invasive species must be able to manipulate local resources to its advantage. After establishment, invasives must spread into new territory. An invasives ability to do so is influenced by factors affecting dispersal mode, number of propagules, and demographics. Miller (1997) conducted research in the southeastern U. S. which showed initial infestations into disturbed areas such as right-of-ways, aids further spread

into surrounding forested areas. Mutualistic relationships between invasive and native species may aid or hinder an invasion. For example, if pollination was low for scotch broom (*Cytisus scoporius* (L.) Link.) it would slow an invasion by this non-native species (Parker et al. 1999). Susceptibility to invasion increases when there is the potential for hybridization to occur between native and invasive species (Levin et al. 1996; Sakai et al. 2001). Through hybridization, an invasive species may be able to attain genes from the native species enabling it to be better adapted to the new environment (Ellstrand 2003).

There is often a time lag between when the invasive species establishes itself and when the population begins to rapidly expand (Sakai et al. 2001; Miller 1997). Potential reasons why lag phases occur include: limits on the detection of a population's growth; the number and arrangement of infestations of immigrants; natural selection among rare or newly created genotypes adapted to the new ranges; the variations of environmental forces. Variations in environmental forces may allow a small population to persist and eventually succeed or it may disappear entirely (Mack et al. 2000). The environmental variability experienced by a plant is based on a number of factors including climate, geographical location, geomorphologic features, the nature of site disturbances, and the abundance and diversity of species present (Bazzaz 1979). Early successional plants must be able to grow quickly above and below ground to grow out of the zone where most variation exists and become established.

Biomass allocation is a key process which may enable a species to resprout. This is especially important in low light situations where the plant balances allocation of resources to roots or to increased leaf surface area (Longbrake and McCarthy 2001). Longbrake and McCarthy (2001) conducted a study, in southeast Ohio, to examine changes in root biomass, to see how a plant adjusts the root-shoot ratio after loosing some above ground biomass. They

found that the plant abandons some of its root biomass, rather than allocating more biomass to above ground portions of the plant. The ability to acclimate rapidly to a wide range of environmental conditions may greatly increase the territory a plant is able to invade (Longbrake and McCarthy 2001). For invasive species to succeed, the ability to resprout is very important. This ability may enable long-lived species to persist and eventually survive and reproduce in the stand. A study conducted in New York, to see how tree-of-heaven is growing into canopy gaps found a short window of establishment because the ages of the trees in a given canopy opening only varied by 2 – 3 years (Knapp and Canham 2000). Invasive woody species can occupy the understory of an intact forest stand by high survivability or dominate the canopy gap by fast growth rates, but invasives are not capable of both (Sanford et al. 2003).

Reichard and Hamilton (1997), whose study involved invasives from North America, disagreed with those who stressed the interactions between the invasive species and the environment as being the most important factor in leading to invasive success. Instead, the characteristics of the individual invasive species controls the invasive potential of a species. Callaway and Aschehoug (2000) states chemical allelopathy may be used by some invasive species to eliminate the native competition. Few habitats are able to resist invasions by exotic species (Gordon 1998). Exotic species invasions often occur when certain resources become available (Gordon 1998). More efficient water and light use allows invasives to out compete native species (Miller 1997). Invasive grasses competing for water have been found to interrupt succession among native perennials (D'Antonio and Vitousek 1992). Seeds are often transported by water, wind, and animals (Sakai et al. 2001). Evolutionary and genetic processes are features that determine whether invasive species thrive and spread. The new invasive population is most

likely less genetically diverse than the population which previously occupied the site (Sakai et al. 2001).

Invasive species share many characteristics common to ruderal species, such as the ability to germinate in numerous environments, rapid growth to sexual maturity, high seed output, vegetative and sexual reproduction, and strong ability to compete. Ruderals are often self-fertilizers, a trait shared by many invasive species (Baker 1974). Vines are also a common growth form of invasive species. If competition for light is crucial then the arrangement of leaves to capture the most light is important. Vines do this by minimizing biomass in support structures, and maximizing biomass for photosynthetic surface area and height growth. This allocation enables vines to reach the necessary light and to increase its chances of dominating the canopy (Schweder 1993).

Negative Implications of Invasive Species

Damage Caused to Ecosystems by Invasives

The 1974 Federal Noxious Weed Act does not have any stipulations requiring new intentionally introduced species to be assessed for their invasive ability. It is also noted that numerous invasive species have a fairly long "lag period" between the time of introduction and when the species begin to function as an invasive (Reichard and Hamilton 1997). A study conducted 15 years after the eruption of Mount Saint Helens, stresses the importance for land managers to understand long-term recovery trends after major disturbances and how non-native species used to aid vegetation recovery will influence the future vegetation (Dale and Adams 2003).

Invasive species limit biodiversity in numerous ways. Invasives out compete native plants for space reducing productivity of native species. With the loss of native plants, also

comes a reduction in wildlife numbers (Weiser 1997). Invasive species, especially vines, tend to form dense infestations which may prevent natural regeneration or slow succession (Miller 1997). Areas which are naturally low in nutrients or other vital resources may be resistant to invasive species because the minimum level of resources necessary for the invasive species is not present (Tilman 1999). Species rich communities may be more resistant to invasion than communities with only a few species (Sakai et al. 2001).

A study of vegetation changes 15 years after the eruption of Mount Saint Helens revealed that shortly after the invasive species introduction non-native species' growth increased, significantly more mortality occurred among conifer species than in areas where the non-natives were not present (Dale and Adams 2003). Initial impacts of invasive species may have lasting effects. For example, 9 years after the eruption of Mount Saint Helens, Spanish clover (*Lotus purshiana* Clem. & E. G. Clem), an introduced non-native herb, was the species with the highest cover. This species was introduced when the avalanche field was aerially seeded to prevent mudslides. Interestingly, plots, where non-native species were dominant, had significantly ($P < .05$) more native species richness than those plots where very few non-natives were present (Dale and Adams 2003). Plots which contained the invasive species had higher mean cover 5 and 14 years after the disturbance, than plots where only native species existed. Over time the native species distribution was affected in different ways by high cover of the non-native species. The number of native species increased so much that now there are more native species present on plots with non-natives than on plots with only natives. This may have occurred because at first the natives and non-natives were competing for resources, with the non-natives winning. Then the shade and soil amelioration provided by the high non-native cover improved conditions enough to allow the native species to establish themselves (Dale and Adams 2003).

After a prescribed fire in the Nantahala National Forest, Elliot et al. (1997) found that invasive species (*Andropogan scoparius*, *Panicum* spp., *Rubus* spp., *Coreopsis major*) were present, but they did not persist for more than a few years because they were shade intolerant species. There is reason to be concerned about invasive species affect on forest stands in the future because tree-of-heaven has been found growing in canopy gaps of mature stands (Knapp and Canham 2000). Exotics that can establish, but do not become dominant are less likely to cause large scale process changes. Those species, such as *Melaleuca quinquenervia* and *Pueraria montana*, that do become dominant are more likely to change environmental conditions and resource availability over larger areas (Gordon 1998). For example a non-native species, such as *Melaleuca quinquenervia* or *Sapium sebiferum*, whose evapotranspiration rate was different from native species, may cause significant changes to the hydrological regime and either raise or lower the water table on the site (Gordon 1998).

Gordon (1998) hypothesizes that many invasive species alter natural systems at several scales. Invasive species have many consequences for the community they invade including: displacement of native plants, threatening native diversity by way of hybridization, causing possible extinction of native species, altering fire regimes, affecting the productivity of the ecosystem, and interrupting the normal processes of nitrogen fixation, water cycles, and sedimentation. Invasives, such as *Casuarina equisetifolia* or *Melaleuca quinquenervia*, can affect ecosystem properties such as geomorphology, hydrology, biogeochemistry, and disturbance. Given that a species can invade and become established, it is likely that they will alter the structure, species composition, and habitat characteristics of the native community (Gordon 1998). Presence of invasive species altering ecosystem processes, results in functional changes in addition to compositional changes. D'Antonio and Vitousek (1992) have described

three ways in which invasives may change ecosystems: 1) altering system level rates of resource supply; 2) altering the trophic structure; 3) altering the disturbance regime. Native structure is altered when a species having a different phenology invades an area, because it is a different life form. Invasive vines which twine around trees may eventually cause the supporting tree canopy to collapse or result in a broken top, possibly resulting in a change of the vertical structure of the community.

Invasive species presence may have direct or indirect ecological impacts on the community it invades. Direct impacts include predation, competition, or mutualism, while indirect impacts may include altered habitat (Sakai et al. 2001). A study by Callaway and Aschehoug (2000) provided an example of allelopathy associated with an invasive species because the carbon uptake of the native species was altered at the site of invasion. Honeysuckle (*Lonicera* spp. Thunb.) is known to inhibit forest regeneration (Carter et al. 1989). It usually alters stand structure where it invades by out competing native species. The presence of invasive species has the potential to cause serious impacts to the ecosystem by changing the dominant species of an environment which consequently impacts plant productivity and processes such as nutrient cycling (Mack et al. 2000).

Impacts of invasive species can be measured at five levels: 1) effects on individuals; 2) genetic effects; 3) population dynamic effects; 4) community effects; 5) effects on ecosystem processes (Parker et al. 1999). The disruption of entire ecosystems is the biggest threat associated with invasive species (Mack et al. 2000). Hutchinson and Vankat (1997) state that persistent dominance of early successional tree species or the conversion to shrub land would cause serious ecological and economical impacts on forests. The impact of an invasive species can be assessed in terms of range, abundance, and effect per unit biomass of the invader (Parker

et al. 1999). A reasoning for use of invader abundance as a measure of impact is that any biomass controlled by the invader constitutes resources no longer available to competitors or prey.

D'Antonio and Vitousek (1992) identify invasive European grasses as one of the major causes of substandard oak recruitment because these grasses extract the soil moisture, better than the native grasses, which reduces the water available for oaks. Invasive grasses compete against woody seedlings and saplings more so than larger trees because grasses have a dense mat of roots at the soil surface, which seedlings must penetrate to gain access to available water. (D'Antonio and Vitousek 1992). The potential exists for invasive species to cause microclimate effects where they become established. For example, slower decomposition rates of invasive grasses, may lead to litter buildup which effects soil temperature and moisture, both of which influence germination of seed growth and nutrient transformation (D'Antonio and Vitousek 1992).

Monetary Costs and Potential Losses

Some invasive species have beneficial uses in agriculture or horticulture (Miller 1997), but the United States spends $125 billion annually dealing with invasive species and the consequences associated with them (MacQuarris and LaCroix 2003). Economic impact equals the cost of control plus the loss in potential economic output (Mack et al. 2000). Approximately 138 non-native tree and shrub species have now invaded once native forest ecosystems in the United States. For example, in the Great Smokey Mountains National Park, 400 of the 1500 plant species in the park are exotic (*Ailanthus altissima* and *Celastrus orbiculatus*) and 10 of these are negatively impacting native species in the park to the point of displacement (Pimentel 2002). The Virginia Department of Conservation's Natural Heritage Program and the Virginia

Native Plant Society have identified 115 invasive species to be present in Virginia (Heffernan 1998). Shenandoah National Park is being invaded along its roadsides and disturbed areas by tree-of-heaven (*Ailanthus altissima* (Mill.) Swingle) (Burch and Zedaker 2003). Of the exotic species causing problems in the United States, 15% have been found to cause major environmental or economic impacts (Windle 1997). When total eradication fails, control efforts usually shift to maintenance of the population at acceptable levels.

Invasives Changing Fire Regime

The presence of vines in a fire prone area, could prevent fire movement where vines have created a thick layer of succulent foliage, partially due to the increase in relative humidity levels (Gordon 1998). The presence of invasive species may alter the wildfire regime to causing more frequent or more intense wildfires, which could harm non-wildfire adapted native species and alter stand ecology (Mack et al. 2000; McGee 1980) D'Antonio and Vitousek (1992) stress that changes in wildfire regime alters the resource availability on a site which impacts competitive interactions. Invasion of exotic species can completely change wildfire regimes. For example, the invasion of European cheat grass (*Bromus tectorum* L.) into shrub-steppe habitat has altered the wildfire return interval from 60 – 110 years to 3 – 5 years; this inhibits the growth of shrubs and changes the species composition (Pimentel 2002).

Materials and Methods

Overall Approach

This research was conducted as a survey to assess the existence of significant correlations between wildfire intensity, vector distance (distance from a road or trail), and time since wildfire occurred on the dominance of invasive species. Study sites were selected where wildfires have occurred since 1980 and there was access either by road or trail. The study sites were stratified by wildfire intensity, either high, low, or none, and vector distance from a road or trail as either 50 ft (15 m), 150 ft (45 m), or 250 ft (75 m). The wildfire intensity classes of high, low, and none were chosen so that the variability between classes would be higher than the variability within the class. Transects were randomly positioned within the strata and nested plots and fuel transects were measured along those transects.

Attributes of Shenandoah National Park

Geographic Location

The proposed research took place in Shenandoah National Park, which was established in 1936, encompasses 183,311 ac (74,186 ha) and has an elevation ranging from 600 ft (183 m) to 4048 ft (1234 m) (Braun 1950). The park lies along a 70 mile (113 km) section of the Blue Ridge Mountains between Front Royal, VA to the north and Rockfish Gap to the south. The park is quite narrow ranging from 3 miles (5 km) to 6 miles (10 km) in width. Skyline Drive, the northern extension of the Blue Ridge parkway, is a famous north-south scenic route, traversing the ridge through the center of the park (Gathright 1976). Shenandoah National Park is in the Blue Ridge physiographic province, with the Piedmont beginning at the eastern edge of the park. To the west of the park lies the Shenandoah Valley and the Massanutten Mountain Range (Gathright 1976).

Shenandoah National Park contains the highest peaks in the Northern Blue Ridge area, Stony Man (4012 ft)(1223 m) and Hawksbill Mountain (4048 ft)(1234 m). European settlers found easy access to the area by way of the Piedmont and the Great Valley (Shenandoah Valley) enabling them to harvest much of the timber in the area. Timber operations in the late 1800's and early 1900's removed the commercially valuable timber (Gathright 1976). When the park was established in 1936, many small farms existed in what is now encompassed by the park. These areas have progressed through stages of succession since abandonment, but are still noticeable and may be important in terms of a seed source for non-native vegetation (Gathright 1976).

Geology

Geology of Shenandoah National Park can be divided into two general areas: the north half along with the eastern slopes of the southern half and the southwest portion. The northern half and eastern ridges of the southern half are underlain by bedrock of granite and basalt, while the southwestern portion of the park is comprised of metamorphosed shale. The southwest portion has thinner soil above the shale and fewer springs than are found in the rest of the park (Gathright 1976). The bedrock underlying Shenandoah National Park can be dated back to the Precambrian, Cambrian, Ordovician, and Triassic periods (Gathright 1976). Iron, manganese, and copper were mined in the past along the western edges of the park from more than 40 abandoned mines.

Climate

Historical climate information is sparse from within Shenandoah National Park. Several weather stations have existed within the park boundaries but have not collected data for long or continuous periods. The only weather station which has collected data on a regular basis, in the

park, is at Big Meadows. Big Meadows is located at 38º 32' N 78º 26' W and at an elevation of 3540 ft (1079 m) (NCDC 2004). An average annual temperature of 48° F (9º C) occurs at Big Meadows, where 52 in (132 cm) of precipitation falls annually. The other closest weather station, which is outside the park, is in Luray, VA. This weather station is located at 38º 40' N 78º 22' W and at an elevation of 1400 ft (427 m). At Luray, the average annual temperature is 54º F (12º C) and 36 in (91 cm) of precipitation occurs annually. The lower elevations experience a modified continental climate with mild winters and warm humid summers. The higher elevation areas have moderately cold winters and cool summers. In July, the maximum temperatures are 11º F (6º C) cooler at Big Meadows compared to the lower elevations. Throughout the park, temperatures range from 19º F to 39º F (-7º C to 4º C) in January and from 57º F to 75º F (14º C to 24º C) in July. The precipitation ranging from 39 to 59 in (100 to 150 cm) is spread evenly throughout the year with a slight increase in July and August from thunderstorms (NCDC 2004).

Field Sampling

Site Selection

All study sites were located within the boundaries of Shenandoah National Park, in Virginia. The park is operated by the United States Department of the Interior, National Park Service. There has not been any timber harvesting within the park since its inception in 1936. The only potential disturbances in these areas were wind or ice damage and insects or disease. Forest cover types were gathered from a 1985 forest inventory conducted for the entire park. A more recent survey was completed in 2003 but will not be published until 2005.

Several GIS layers, including wildfire history, cover type, roads, trails, streams, topography, Digital Ortho Quarter Quad's, boundaries, and Composite Burn Indices, were

provided for the site selection process by Dan Hurlbert of Shenandoah National Park. Utilizing GIS layers of wildfire history in Shenandoah National Park and ESRI ArcGIS 8.2, a query was performed to display only those wildfires since 1980 where no previous wildfires had been recorded since the establishment of the park in 1936. Several recent wildfires, which encompassed greater than 90 ac (35 ha) within the wildfire boundary free of previous wildfire, were also included. The selected wildfires since 1980 were then queried again to select for wildfires greater than 90 ac (35 ha) (Table 1) (Figure 1). This was important to make sure they were large enough to allow a range of intensities to occur, as wildfires of less than 90 ac (35 ha) total were not likely to have had crown wildfires or canopy destroying wildfires. Digital Ortho Quarter Quads (DOQQ's) were used to make sure each wildfire contained different aspects to allow for sampling to capture various levels of intensity within each wildfire. Control sampling was done in an adjacent area of similar slope, aspect, and elevation where no previous wildfires have been recorded since 1936.

Table 1. General characteristics of each study site in Shenandoah National Park, Virginia.

Wildfire	Growing Seasons Since Burn	Acres	Dominant Overstory Species
Rocky Top	1	913	Chestnut Oak, Pine, Cove Hardwoods
Fultz Run	2	4200	Chestnut Oak, Pine, Yellow Poplar, Red Oak
Shenandoah Complex	3	24197	Chestnut Oak, Pine, Yellow Poplar, Red Oak
Bootens Gap	4	1579	Chestnut Oak, Pine, Yellow Poplar, Red Oak
Madison Run	14	210	Chestnut Oak, Pine
Neighbor Mountain	17	130	Chestnut Oak
Big Run	18	4475	Chestnut Oak, Pine, Cove Hardwoods
Piney River	23	680	Chestnut Oak, Pine, Red Oak, Cove Hardwoods

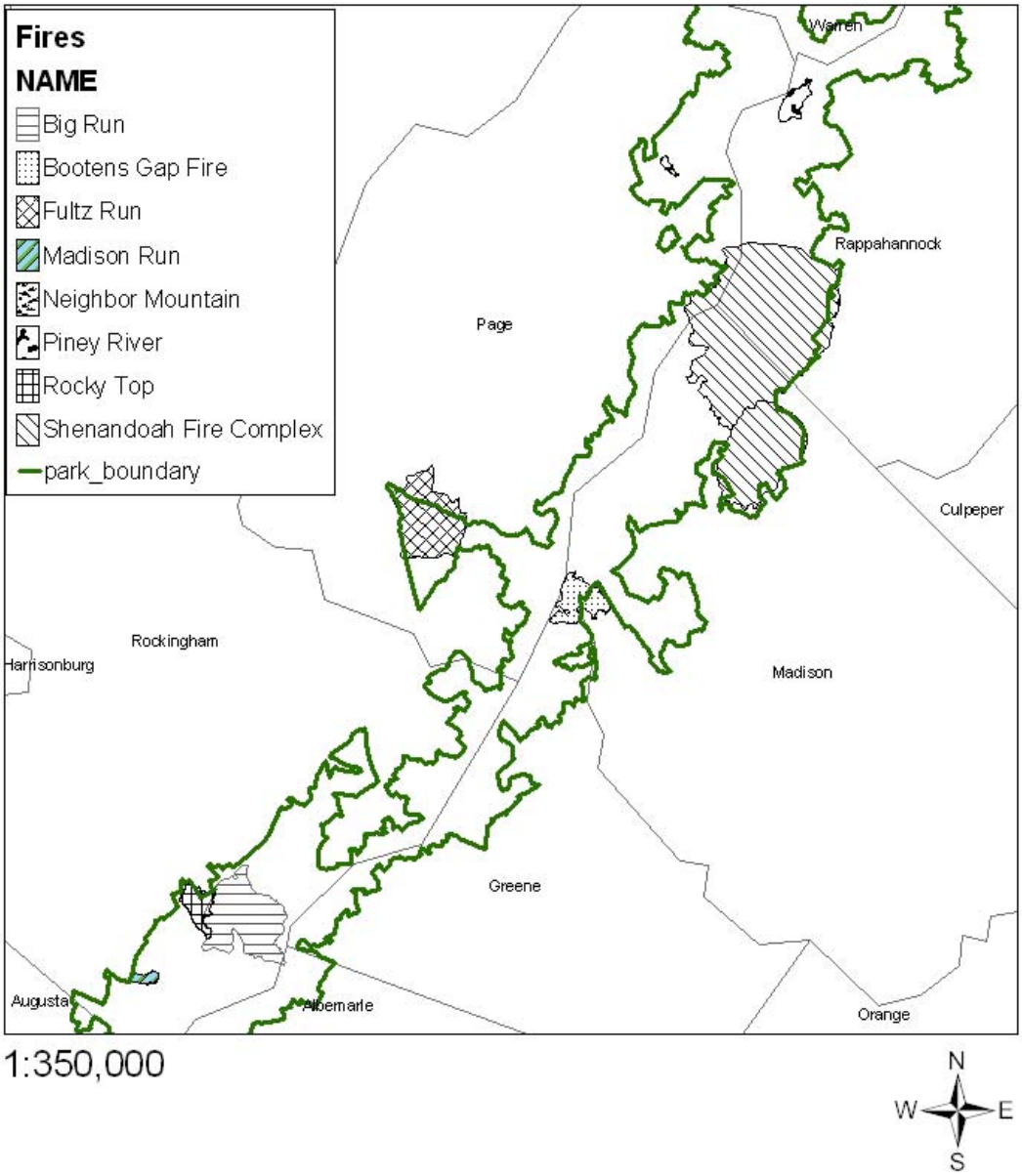

Figure 1 – General location of all study sites in Shenandoah National Park, in the Blue Ridge Physiographic Province, in Virginia.

Site Descriptions

Big Run

The Big Run wildfire (38º 17' N 78º 41' W) encompassed 4475 ac (1811 ha) of Rockingham County, Virginia, between May 2 and May 19, 1986 (Figure 2). The wildfire was initially set by campers and spread rapidly due to low humidity and high winds. Many wildfires have occurred in the north and central portions of the Big Run wildfire area. There is a fairly large area adjacent and north of Patterson Ridge were there is no record of previous wildfire disturbance since the establishment of the park. The overstory within the area consists of cove hardwoods, chestnut oak, bear oak, and pine. Three soil series are present in the area encompassed by the Big Run wildfire. The Drall soil series is Sandy-skeletal, siliceous, mesic Typic Udorthents. The Laidig soil series is Fine-loamy, siliceous, active, mesic Typic Fragiudults and the Sylco soil series is a taxadjunct series described as Loamy-skeletal, mixed, active, mesic Typic Dystrudepts.

The Big Run wildfire is the area where Regelbrugge (1988) conducted his research on the effects of wildfire on the structure and composition of forested stands in the Blue Ridge of Virginia. High overstory mortality was observed most in pitch pine with mountain laurel understory, while moderate mortality was observed to have occurred in oak or oak/pine stands. Low mortality occurred in the chestnut oak stands and no mortality occurred on the north or east slopes where northern red oak/ash (*Fraxinus pennsylvanica* Marsh.)/basswood, red maple/oak, or mixed oak stands were growing.

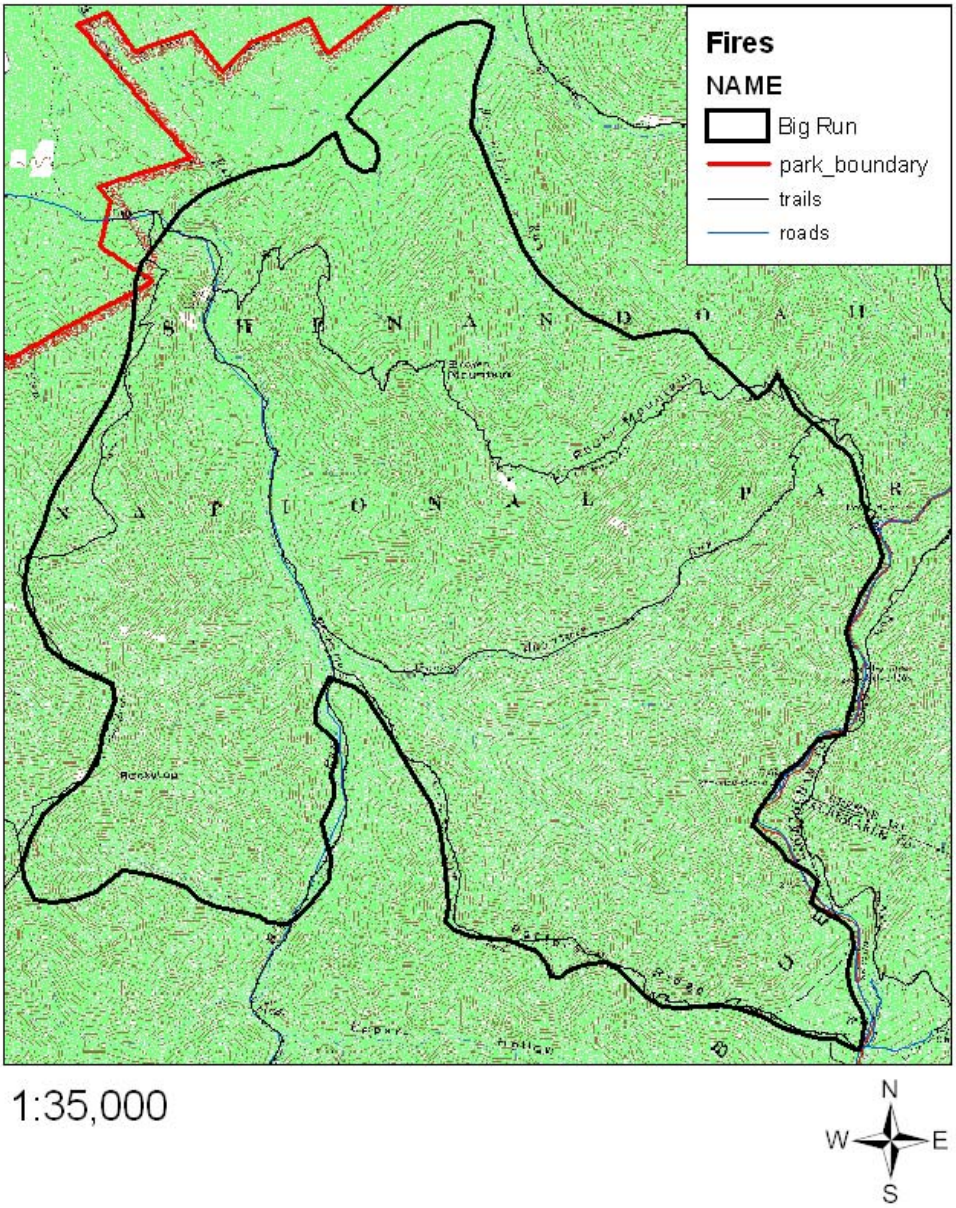

Figure 2 – Specific location of Big Run wildfire in Shenandoah National Park, in Rockingham County, Virginia.

Bootens Gap

The Bootens Gap wildfire (38º 27' N 78º 26' W) burned 1579 ac (639 ha), between November 16 and November 20, 1999 (Figure 3). The wildfire carried in hardwood leaf litter producing flame lengths between 1 ft – 3 ft (0.5 m – 1.5 m), while most were less than 1 ft (0.5 m). The wildfire occurred in portions of Madison and Greene Counties, Virginia, on the eastern side of Shenandoah National Park. Approximately one third of the wildfire occurred outside of the park boundary. There is no record of previous wildfire disturbance since the park inception. The overstory in the Bootens Gap wildfire site consists of yellow poplar, chestnut oak, northern red oak, cove hardwoods, and eastern hemlock. The Bootens Gap wildfire encompassed an area which is classified into three soil series. The Porters and Tusquitee soil series are Coarse-loamy, isotic, mesic Typic Dystrudepts and the Unison soil series is a Fine, mixed, semiactive, mesic Typic Hapludults.

Fultz Run

The Fultz Run wildfire (38º 30' N 78º 33' W) burned 4200 ac (1700 ha), between February 26 and March 3, 2002, in Page County, Virginia (Figure 4). The human caused wildfire was fueled by high winds and low humidity as it burned through bug-killed timber. The wildfire was primarily a surface wildfire, but some crowning did occur. The majority of the area has been previously disturbed by wildfire, however a section along Smith Mountain in the southwest corner of the burn area does not have any record of previous wildfire disturbances. The overstory is composed primarily of northern red oak, chestnut oak, yellow poplar, and pine. Seven soil series exist in the area burned during the Fultz Run wildfire. The Peaks, Sylco, and Dekalb soil series are all Loamy-skeletal, mixed active, mesic Typic Dystrudepts. The Edgemont and Edneytown soil series are Fine-loamy, mixed, active, mesic Typic Hapludults.

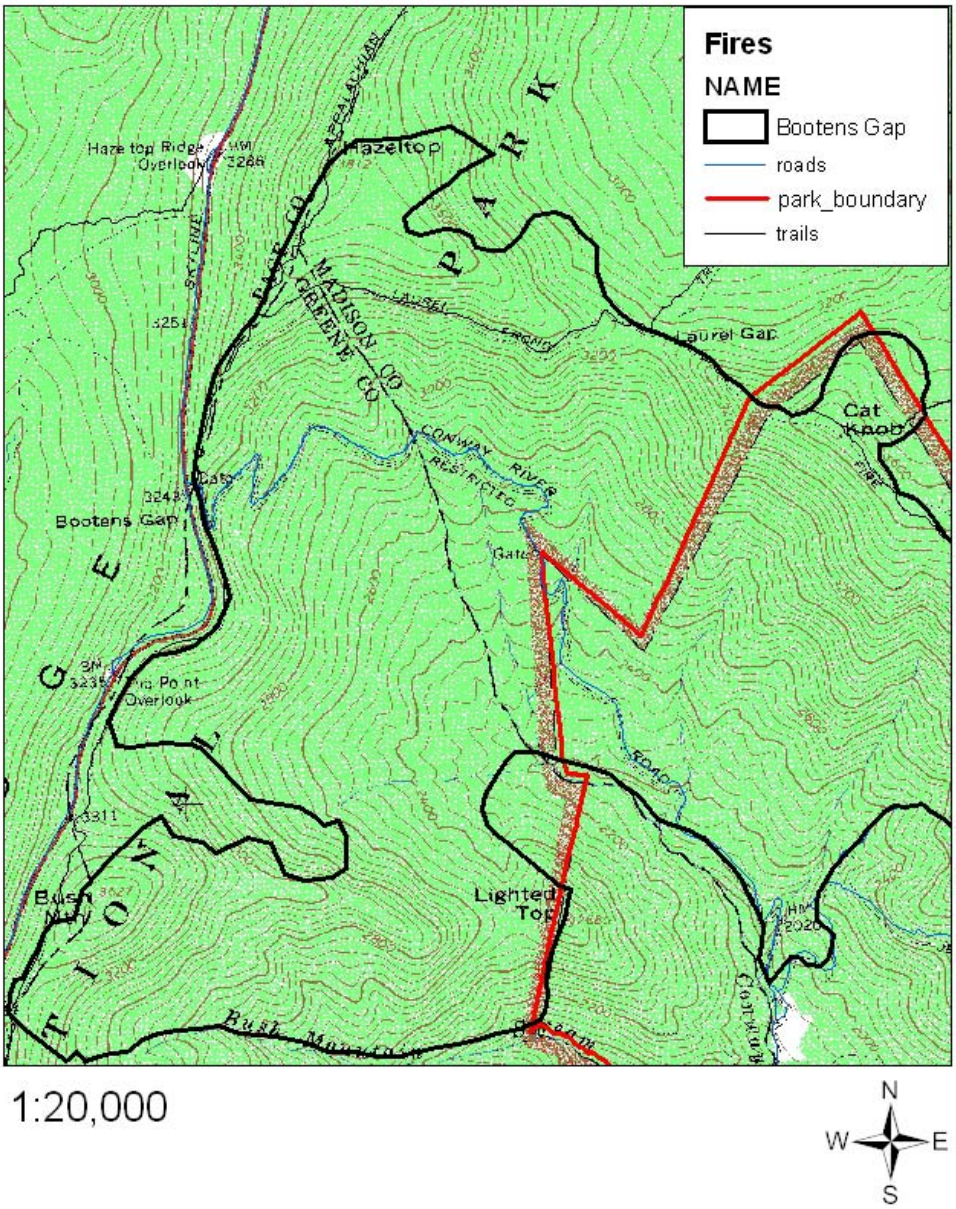

Figure 3 – Specific location of Bootens Gap wildfire in Shenandoah National Park, in Madison and Greene Counties, Virginia.

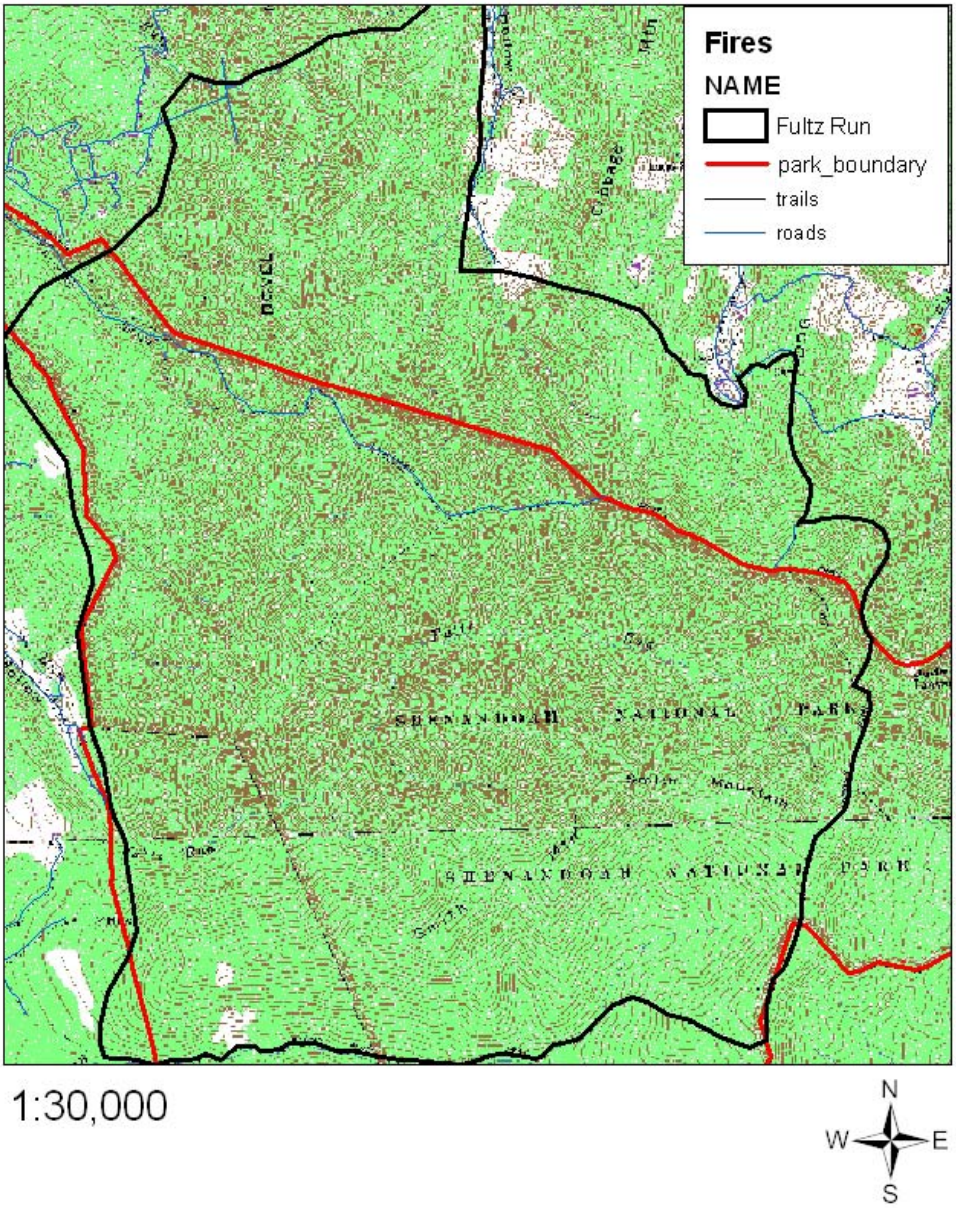

Figure 4 – Specific location of the Fultz Run wildfire in Shenandoah National Park, in Page County, Virginia.

The Craigsville soil series is a Loamy-skeletal, mixed, superactive, mesic Fluventic Dystrudepts, and the Sylvatus soil series is a Loamy-skeletal, mixed, active, mesic Lithic Dystrudepts.

Madison Run

The Madison Run wildfire (38º 15' N 78º 45' W) occurred in 1989, burning 210 ac (85 ha) in Rockingham County, Virginia (Figure 5). Prior to 1989, no wildfires had been recorded in the area, however a 17 ac (7 ha) wildfire occurred in 1999 on the ridge top at the northeast corner of the Madison Run wildfire boundary. The overstory consists of pine and chestnut oak. On the westerly aspects a pure canopy of chestnut oak exists with virtually no understory, while on the south facing slope a few pine overstory trees remain, but the area resembles a recent clearcut, with pine seedlings and mountain laurel making up the understory. The soils in the area are classified as the Drall soil series which are Sandy-skeletal, siliceous, mesic Typic Udorthents.

Neighbor Mountain

The Neighbor Mountain wildfire (38º 42' N 78º 22' W) also occurred in 1986, burning 131 ac (53 ha) in Page County, Virginia, on the western side of the park (Figure 6). There is no record of previous wildfire disturbance since the establishment of the park. The overstory consists entirely of chestnut oak. Six soil series encompass the area burned during the Neighbor Mountain wildfire. The Dekalb and Sylco soil series are both Loamy-skeletal, mixed, active, mesic Typic Dystrudepts. The Catoctin soil series is a Loamy-skeletal, mixed, superactive, mesic Ruptic-Alfic Eutrudepts. The Edgemont soil series is a Fine-loamy, mixed, active, mesic Typic Hapludults. The Myersville soil series is a Fine-loamy, mixed, active, mesic Ultic Hapludalfs and the Sylvatus soil series is a Loamy-skeletal, mixed, active, mesic Lithic Dystrudepts.

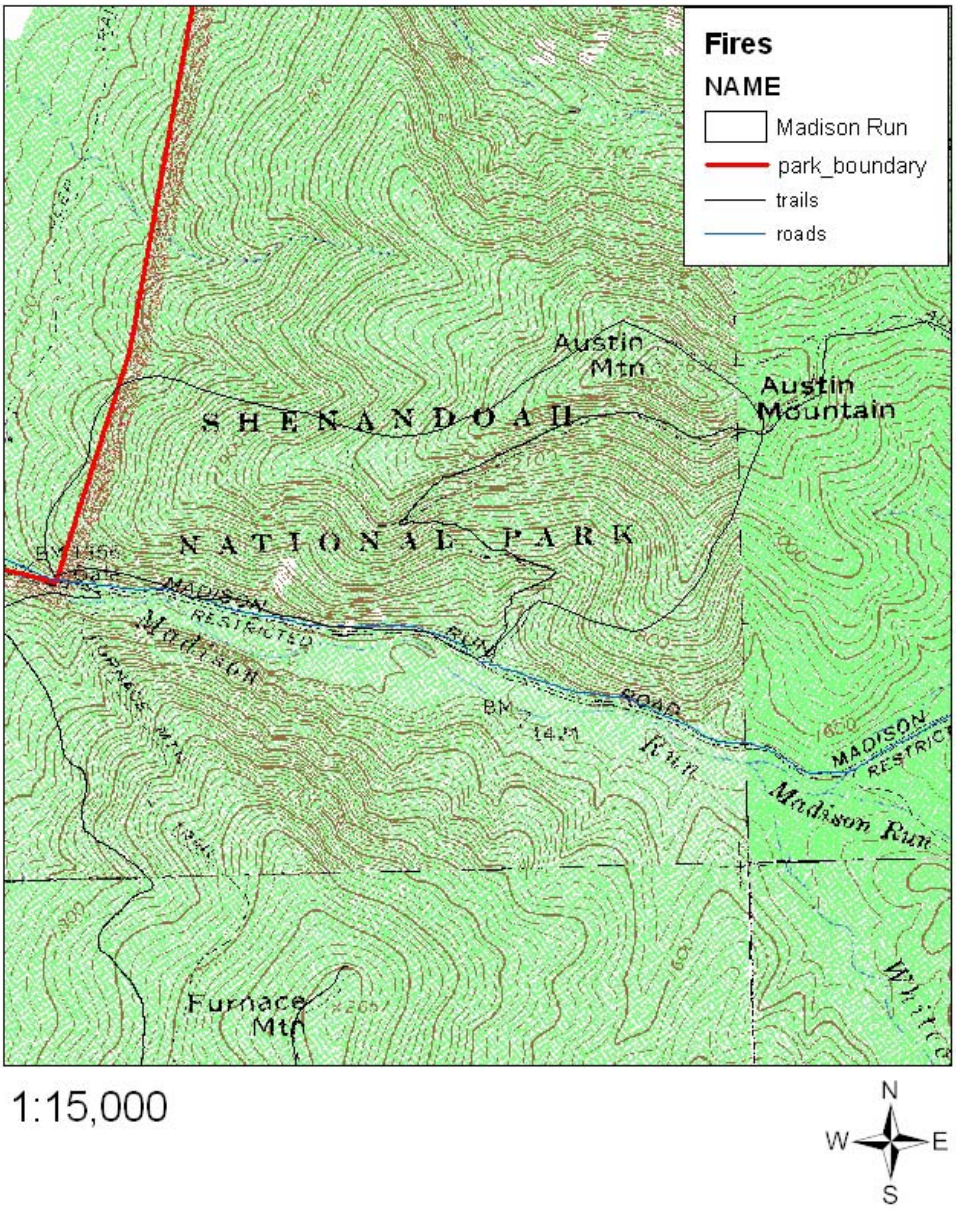

Figure 5 – Specific location of Madison Run wildfire in Shenandoah National Park, in Rockingham County, Virginia.

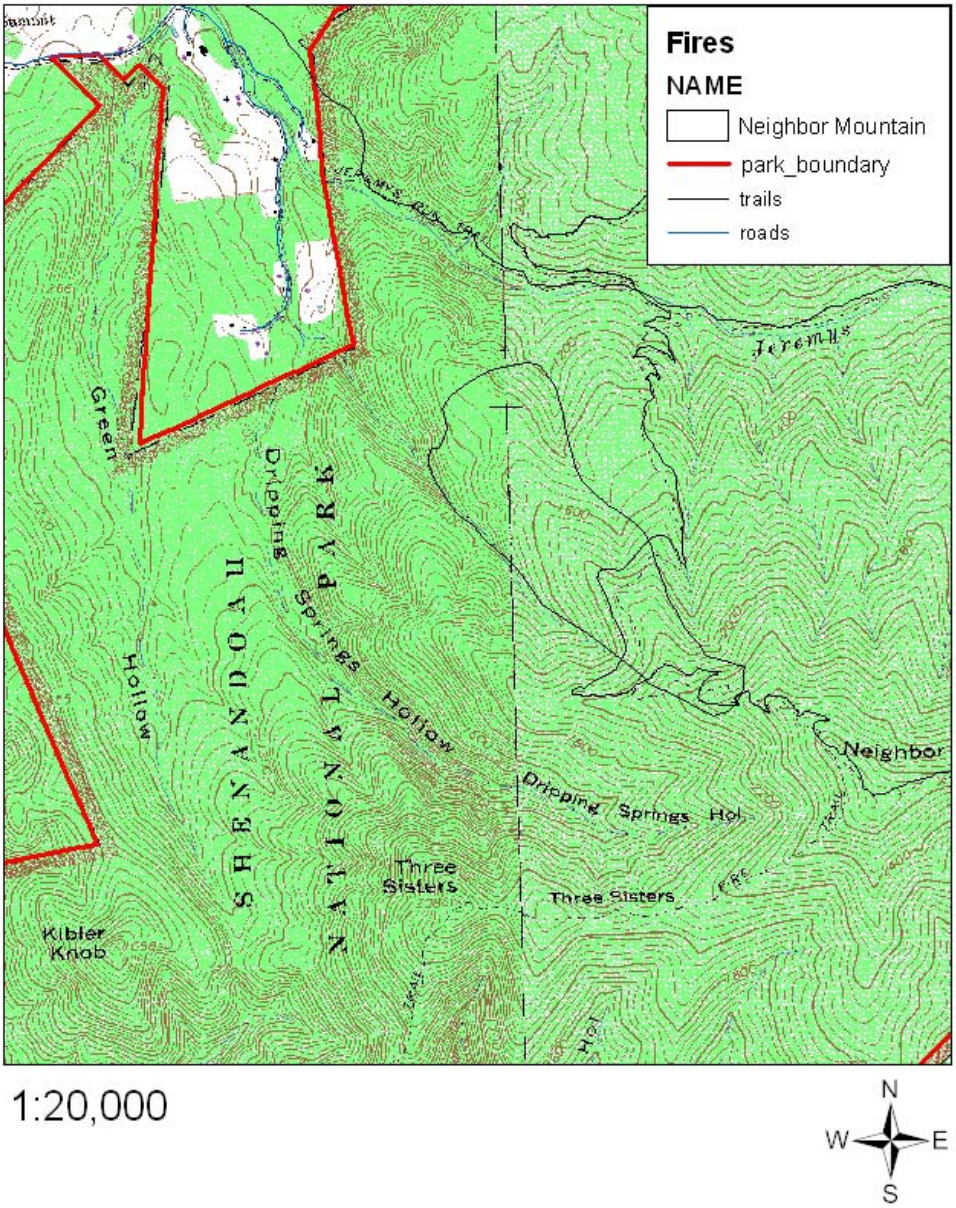

Figure 6 – Specific location of Neighbor Mountain wildfire in Shenandoah National Park, in Page County, Virginia.

Piney River

The Piney River wildfire (38º 44' N 78º 16' W) consists of 680 ac (275 ha), which burned in 1981, on the east side of Skyline Drive in Rappahannock County, Virginia (Figure 7). Two previous wildfires have occurred within the boundary of the Piney River wildfire, however these were both small in area and were avoidable in the sampling. The overstory is composed of cove hardwoods, chestnut oak, northern red oak, black locust and pine. The soils in this area are poorly mapped, and are classified simply as very rocky land.

Rocky Top

The Rocky Top wildfire (38º 17' N 78º 43' W) in 2002, burned 911 ac (369 ha) in Rockingham County, Virginia, on the western side of Shenandoah National Park. The specific location of the burned area is shown in Figure 8. A small portion of this area had previously burned between 1930 and 1950. The remainder of the area has been free of wildfire disturbance since the inception of the park. The overstory of the site is comprised of chestnut oak, pine, and cove hardwood. The area is mapped as having two soil series. The Drall soil series is Sandy-skeletal, siliceous, mesic Typic Udorthents, and the Laidig soil series is Fine-loamy, siliceous, active, mesic Typic Fragiudults.

Shenandoah Complex

The largest wildfire in Shenandoah National Park history, occurred between October 29 and November 12, 2000, encompassing 24,198 ac (9,793 ha), around Old Rag Mountain in Madison County, Virginia (Figure 9 and 10). The Shenandoah Complex wildfire (38º 35' N 78º 17' W) was actually two separate wildfires which came together. Elevations within the burned area range from 840 ft (256 m) to 3730 ft (1137 m). The southern portion was utilized for this research. The area, being so large, has had numerous wildfire disturbances over the last 70

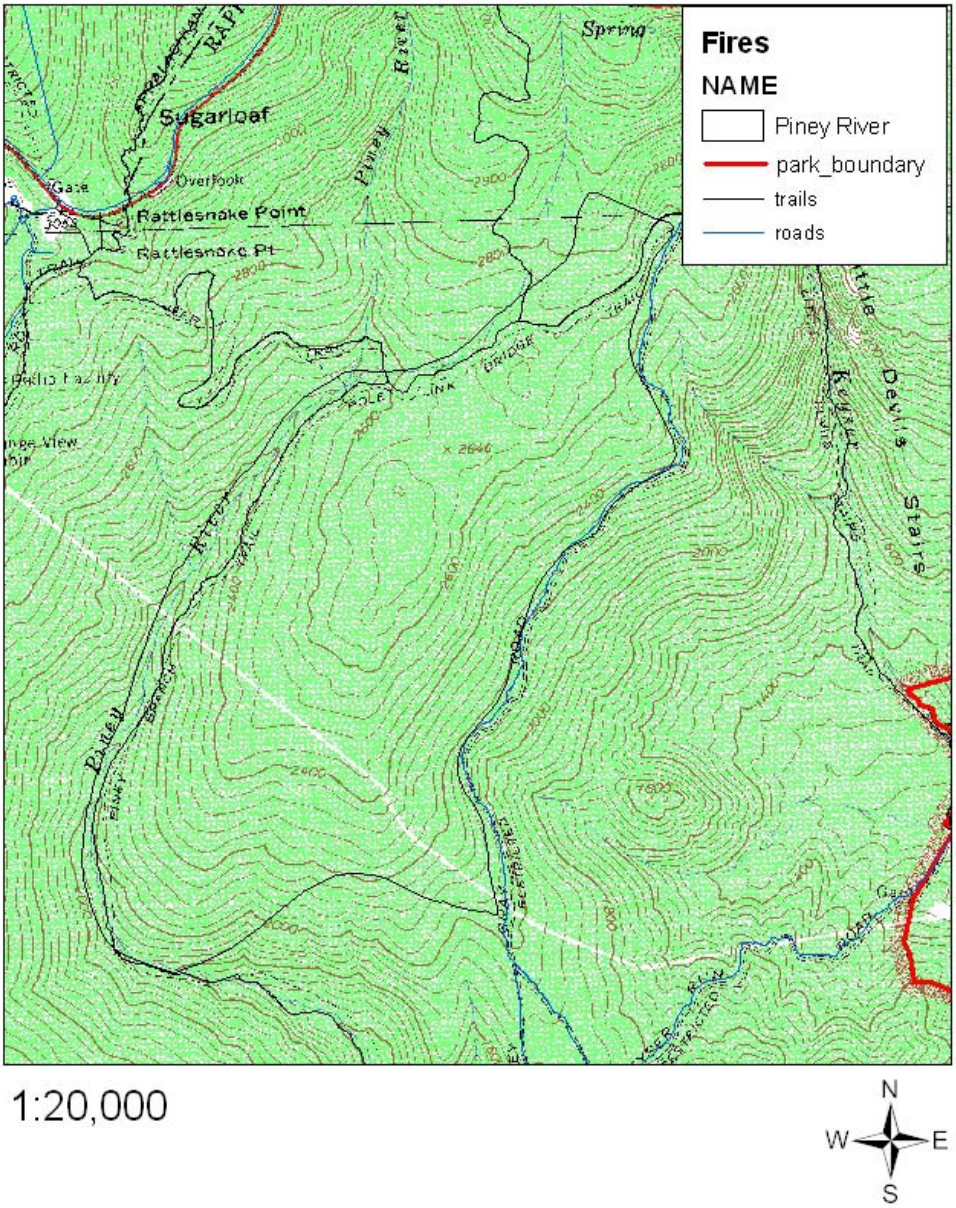

Figure 7 – Specific location of Piney River wildfire in Shenandoah National Park, in Rappahannock County, Virginia.

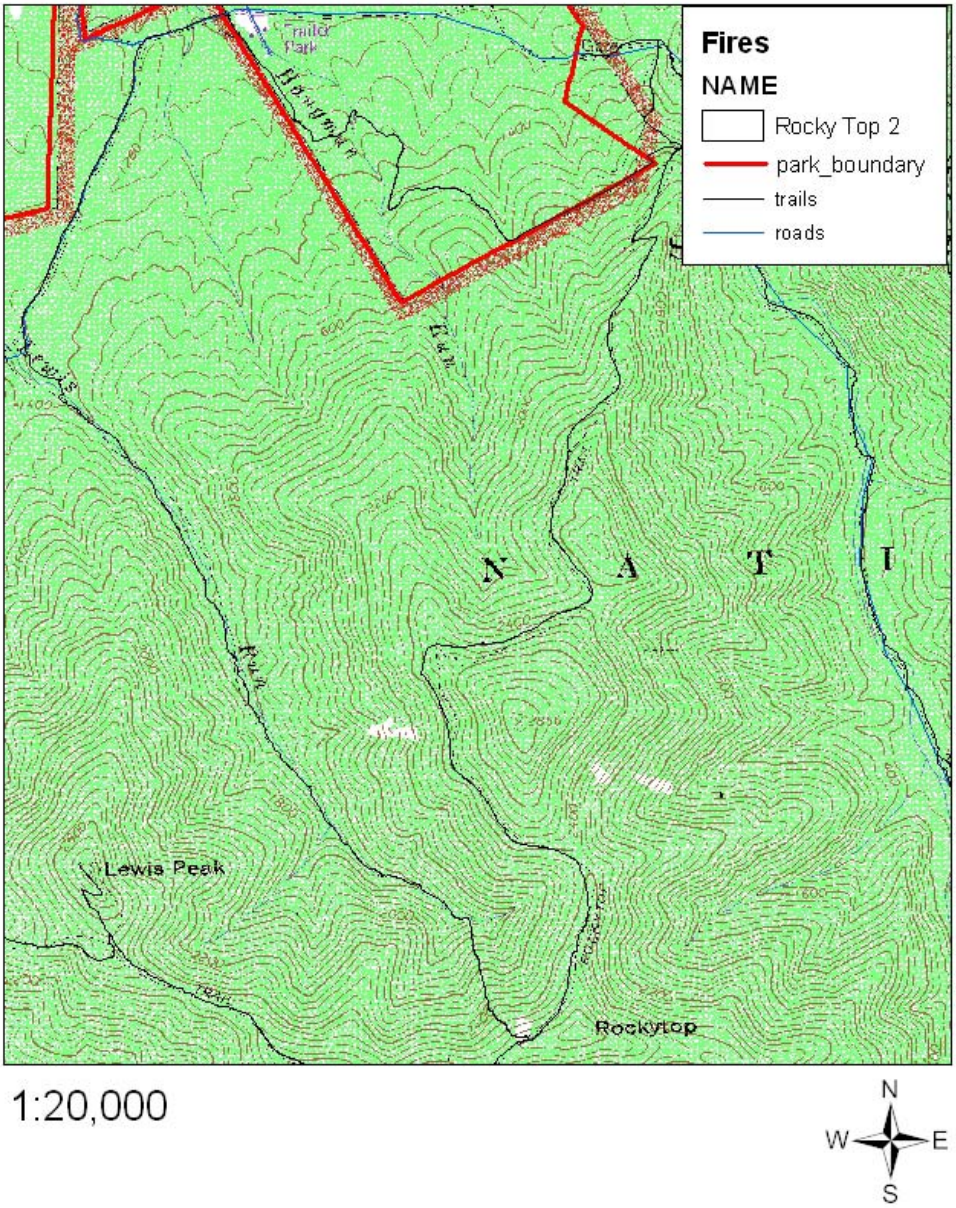

Figure 8 – Specific location of Rocky Top wildfire in Shenandoah National Park, in Rockingham County, Virginia.

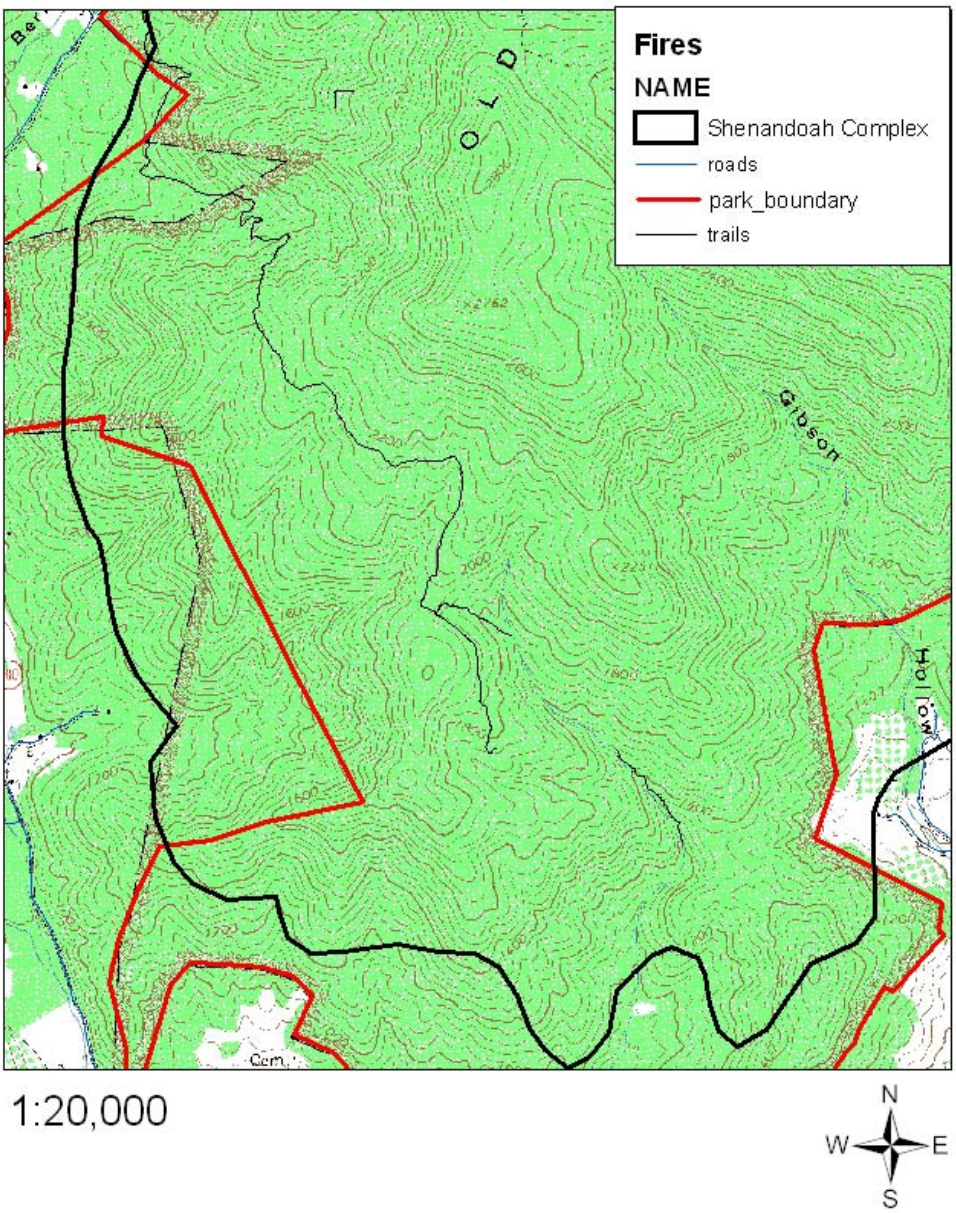

Figure 9 – Southwestern portion of Shenandoah Complex Wildfire in Shenandoah National Park, in Madison County, Virginia.

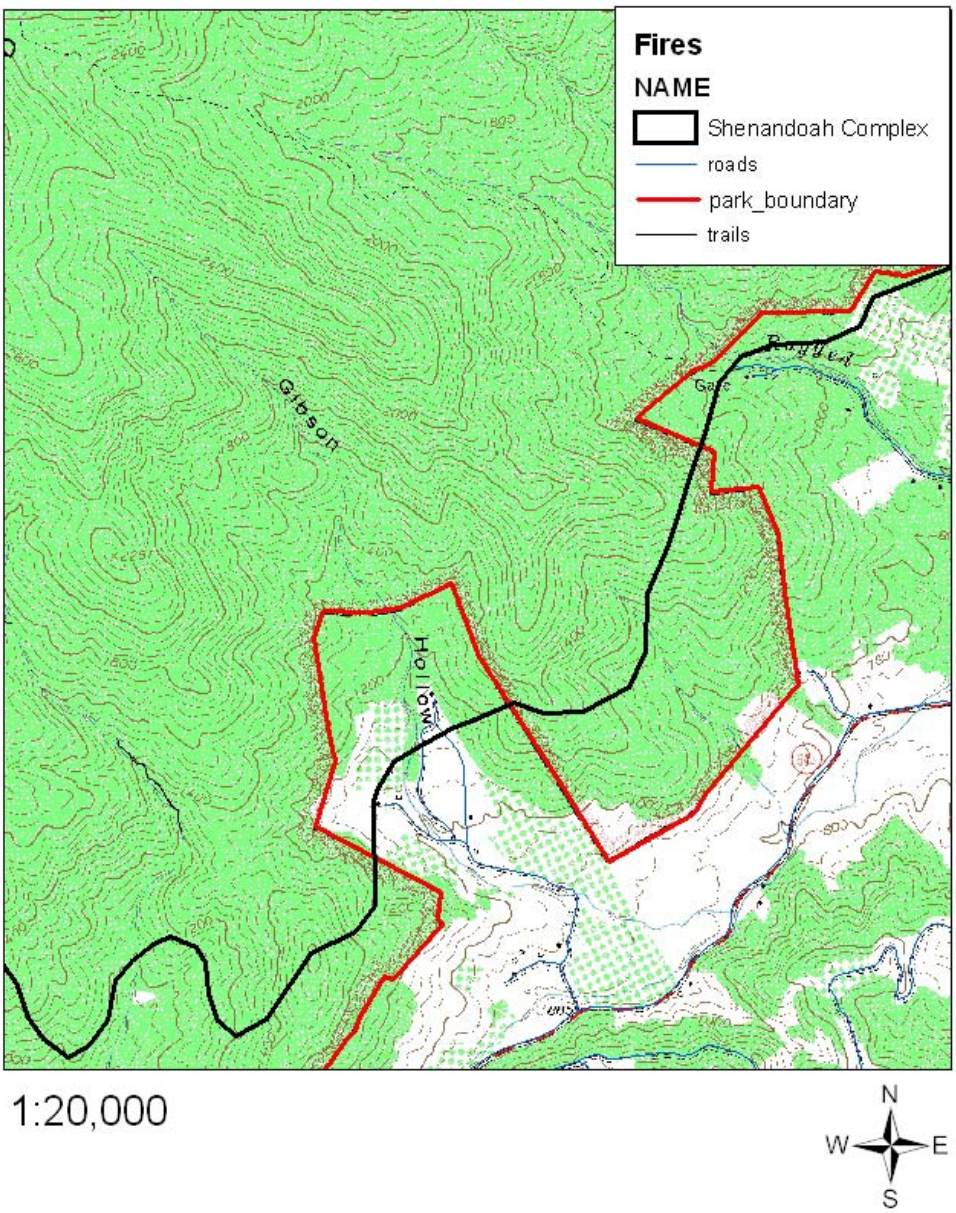

Figure 10 – Southeastern portion of Shenandoah Complex Wildfire in Shenandoah National Park, in Madison County, Virginia.

years, however the southeastern corner of the wildfire does not show any record of previous wildfires occurring. The overstory is comprised of yellow poplar, chestnut oak, northern red oak, cove hardwood, and eastern hemlock. The Old Rag Horse Trail, as well as a hiking trail, exist through the burn area. In addition, there is a road along the southeast boundary of the wildfire. Five soil series are found in the area of the Shenandoah Complex wildfire. The Porters and Tusquitee soil series are Coarse-loamy, isotic, mesic Typic Dystrudepts. The Braddock soil series is a Fine, mixed, semiactive, mesic Typic Hapludults. The Brandywine soil series is a Sandy-skeletal, mixed, mesic Typic Dystrudepts and the Thurmont soil series is a Fine-loamy, mixed, active, mesic Oxyaquic Hapludults.

Due to the size of this wildfire a BAER (Burned Area Emergency Rehabilitation) Team was brought in to assess the damages caused and to make recommendations to aid the recovery of the site. The report found the surface wildfire, driven by high winds, was fueled by grasses, shrubs, and leaf litter and burned in a mosaic pattern across the landscape. The flames averaged less than a meter in height but some crowning occurred on the south side of Old Rag Mountain. The team listed known infestation areas of invasive species to include a power-line right of way on the North wildfire boundary, 5 overlooks along Skyline Drive and ditches in between, the Pinnacles dump site, Old Rag shelter and access road, and the Panorama entrance area. The BAER team found numerous invasive species on site including oriental bittersweet (*Celastrus orbiculatus* Thunb.), Japanese honeysuckle, mullein (*Verbascum thapsus* L.), princess tree (*Paulownia tomentosa* (Thunb.) Sieb. And Zucc. Ex Steud (Scrophuloriaceae)), tree-of-heaven, Japanese stiltgrass (*Microstegium vimineum* (Trin.) A. Camus), spotted knapweed (*Centaurea biebersteinii* DC.), gill-over-the-ground (*Glechoma hederaceae* L.), garlic mustard (*Alliaria*

petiolata [Bieb] Cavara & Grande), and Japanese knotweed (*Polygonum cuspidatum* Sieb. & Zucc.).

Plots

The sampling design was stratified by two variables. The sample sites were classified into high and low wildfire intensity, equating to greater than 70% overstory mortality and less than 30% overstory mortality, respectively. According to Regelbrugge (1988), on the Big Run wildfire, in Shenandoah National Park, in May 1986, high intensity wildfires were classified as reducing pre-wildfire basal area by 67% on average and low intensity wildfires reducing pre-wildfire basal area by 8% on average. Use of highly contrasting wildfire intensity strata prevented the possibility for overlap of wildfire intensities and reduced the sample size necessary to capture the variation within a burned area. The unburned area (control) was in an adjacent stand of similar overstory composition, slope, aspect, and elevation. The high, low, and no burn intensity areas were further stratified by vector distance from a potential seed source such as a road or trail. The levels of the vector variable included 50 ft (15 m), 150 ft (45 m) and 250 ft (75 m) from the road or trail.

For purposes of stratification by intensity of wildfire, slope and aspect maps for each site were created using ESRI ArcGIS 8.2 (Longley et al. 2002). There is a very strong correlation between aspect and intensity of wildfire. Kushla and Ripple (1997) in a model to predict wildfire intensity using terrain variables found slope and aspect to be the two best predictors of wildfire intensity. South and southwest slopes receive more sunlight and more heating which enables fuels present to become drier than on other aspects. In addition, steeper slopes receive more heating, which dries out the fuels, causing wildfires to be more intense.

Development of slope and aspect maps for Shenandoah National Park, began by obtaining Digital Elevation Models (DEM's) produced by the U. S. Geological Survey from the GIS Center at Radford University, for each county within the park. The DEM's were downloaded in a zipped format and then imported to ArcMap using ArcToolBox. Once in ArcMap the Spatial Analyst features were utilized where slope can be selected and a new grid file saved as the slope map. The process was then repeated to create an aspect map for each county. The aspect maps were reclassified as follows:

112.5º – 247.5º	high intensity
0º – 112.5º	low intensity
247.5º – 360º	low intensity

The slope maps were reclassified as follows:

0 – 5%	Flat
5 – 10%	Low
10 – 20%	Moderate
20 – 40%	Steep
40 – 150%	Very Steep

The slope and aspect maps were then overlain to detect areas of steep slope and high intensity from the aspect map, as well as low slopes and low intensity.

Roads and trail layers were also added to the map. At that point a series of three buffers were applied around the roads and trails. The buffers corresponded to the levels of the vector variable, 50 ft (15 m), 150 ft (45 m), and 250 ft (75 m). The map was then queried for areas greater than 2.5 ac (1 ha) in each strata to ensure the area was representative of the condition and sampling would not be influenced by the adjacent condition. The 9 stratified areas corresponding to wildfire intensity levels and vector levels were field verified prior to sampling.

This was an observational study, rather than an experimental study, but statistical techniques were utilized for analysis as though it were an experiment. For purposes of analysis

the data were regarded as being from a randomized complete block design, as the transects were randomly located within predefined strata for sampling. The results of the research allow an assessment of the potential total impact of invasive species by mapping their dominance relative to wildfire intensity and vector distance.

Within each of the field verified strata, a transect was established, from a random starting point, along which a series of nested plots were established and fuel transects were measured. The sampling transect followed the contour within the stratum and plots were established at 2 chain (40 m) intervals along the transect until 3 plots per transect had been sampled. The starting plot of the transect was GPS'd and the location on the ground was monumented temporarily with a PVC stake, an aluminum identification tag and pink flagging. In addition, detailed directions to the transect were recorded from an easily accessible location to enable the sample transects to be revisited in the future. Subsequent plots along the transect were marked in a similar manner.

A nested plot design, similar to Mueller-Dombis and Ellenberg (1974) and Regelbrugge (1988) was utilized for sampling vegetation (Figure 11). This sampling design is also similar to the Long – Term Ecological Monitoring Program in place at Shenandoah National Park (Akerson 2003). All sampling was conducted between May 15, 2004 and June 30, 2004. A $1/10^{th}$ ac (0.04 ha) tree strata circular plot was measured for overstory tree composition, where trees were tallied by species in 2 in (6 cm) dbh classes, for trees greater than 6 in (15 cm) dbh. DBH was measured with calipers at 4.5 ft (1.4 m) above ground line on the uphill side of the tree bole. Measuring dbh allowed basal area to be calculated for assessing the dominance of tree species. Native species were tallied separately from invasive species for comparison purposes on all of the nested plots. A $1/100^{th}$ ac (0.004 ha) shrub strata circular plot was measured for the shrub layer. Shrubs and trees less than 6 in (15 cm) dbh were tallied by species. DBH was

measured as described above for trees and percent cover for the shrub layer was visually assessed using cover class grids. Three herbaceous strata 10.5 ft^2 (1 m^2) square plots were established 10 ft (3 m) from the plot center to count and assess percent cover by species of tree seedlings, grasses, forbs, and sedges. One plot was established in each of the north, south, and east directions.

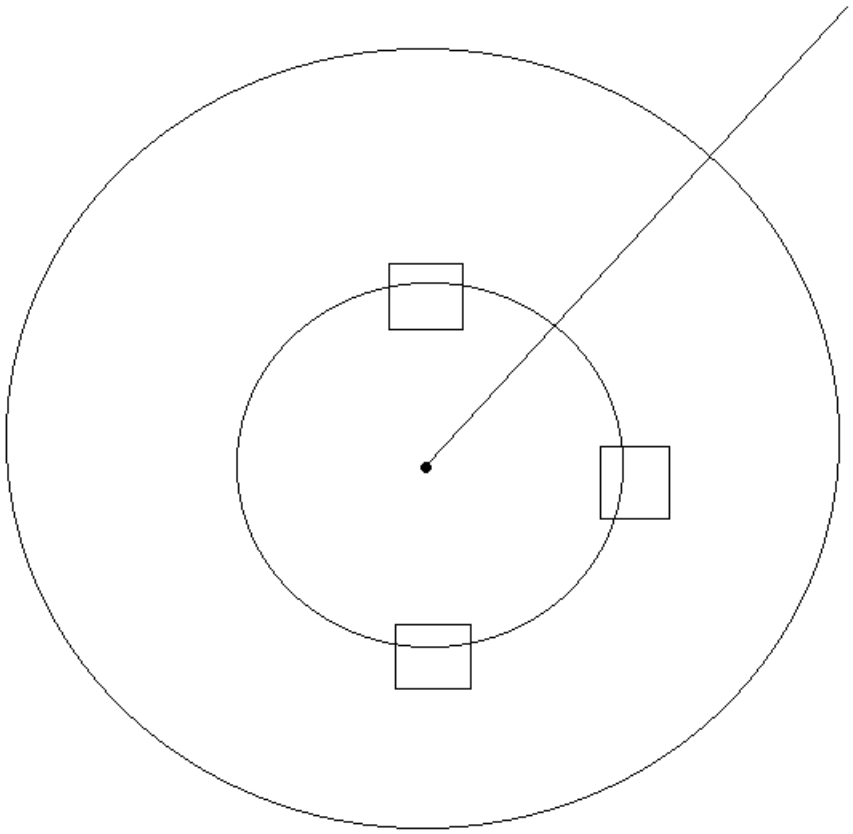

Figure 11 – Sample plot design. Interior plot circle is 1/100th ac (0.004 ha), the exterior plot is 1/10th ac (0.04 ha) and the squares are 10.5 ft^2 (1 m^2) each. A 50 ft (15.2 m) fuel transect was located in a random direction beginning at the center point of the above plot. Three of these plots were systematically installed based on a random starting point in each vector/intensity strata on each of the study wildfires in Shenandoah National Park, Virginia.

All invasive species known to exist in Virginia (Virginia Department of Conservation 1999) and Shenandoah National Park were surveyed for when they would be present and easily identifiable (Table 2). Sampling during the month of June allowed us to capture 74% of the

invasive species, which could have potentially been on site. No single month allows all potential invasive species to be observed.

Table 2. Effectiveness of sampling period for capturing known invasive species in Shenandoah National Park. Flowering period was used as a surrogate for being present and easily identifiable.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Overall	34%	34%	33%	39%	51%	74%	84%	82%	71%	56%	35%	36%
Herbs	8%	8%	5%	12%	31%	62%	79%	77%	61%	38%	9%	10%
Shrubs	89%	89%	89%	95%	95%	95%	95%	89%	89%	89%	89%	89%
Trees	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
Vines	67%	67%	78%	89%	78%	89%	89%	89%	89%	89%	67%	67%

Slope percent was measured twice on each plot using a clinometer and the mean of these two measurements was recorded as the average slope percent of the plot. Aspect was measured using a compass and recorded for each plot. Elevation for plot center was estimated using the GPS points on the topographic map. Elevation was estimated to the nearest 20 ft (6 m).

Fuel Transects

A 50 ft (15.2 m) fuel transect was randomly oriented from each plot center following Brown (1974) and Brown et al. (1982). The transects were randomly oriented using a random number table created from a random number generator, where the last two digits correspond to azimuth in increments of ten. The number of intersections for the < 0.25 in (< 0.6 cm) and 0.25 – 1 in (0.6 – 2.5 cm) size classes were tallied between 0 and 6 ft (0 – 1.8 m) along the transect. The number of intersections for the 1 – 3 in (2.5 – 7.6 cm) size class were tallied between 0 and 12 ft (0 – 3.6 m) along the transect. For pieces greater than 3 in (7.6 cm) in diameter, the diameter was recorded and the piece was classified as sound or rotten along the entire length of the 50 ft (15.2 m) fuel transect. The size classes sampled correspond to the common 1-hr, 10-hr, and 100-hr time-lag fuels used in determining potential wildfire behavior.

Sub-samples

To conduct fuel loading measurements as outlined in Brown (1974) and Brown et al. (1982), measurements of specific gravity were needed for each of the fuel time-lag classes. In addition, a squared average quadratic mean diameter was needed for each fuel class. Bulk density of litter and duff layers were also necessary to complete fuel loading calculations.

This information was obtained by sub-sampling the fuel transects. One fuel transect from each wildfire intensity strata of each wildfire was selected for the sub-sample, for a total of 24 plots. Each piece of dead downed wood intersected on the transect was collected. This ranged from approximately 5 pieces to 30 pieces being collected per site; however the majority were small pieces < 0.25 in (0.6 cm) in diameter, with only a few pieces being greater than 3 in (7.6 cm) in diameter. The pieces were grouped by fuel class and taken back to the lab where measurements for specific gravity and diameter were performed. If the piece was greater than 6 in (15 cm) in length then it was cut with a hand saw or pruners depending on its diameter. If a piece greater than 6 in (15 cm) in diameter was encountered, then a 2 in (5 cm) thick disk was removed and the remaining piece was left in place. In the lab, the diameter was measured with a caliper to the nearest millimeter and then used to compute the average squared quadratic mean diameter. The samples were placed in a cylinder of water to measure the volume of water displaced and then oven dried to determine the oven dry weight (ASTM 2002). The samples remained in the oven for 48 hours at 149° F (65° C). The samples were then individually weighed on a scale to the nearest tenth of a gram. Dividing the oven-dry weight by the volume gave a value for specific gravity.

On each of these 24 fuel transects litter depth was measured to the nearest 0.1 in (0.2 cm) at one sample location. A 1 ft^2 (0.09 m^2) sample of litter was collected and then brought back to

the lab for weighing. The litter volume was calculated by multiplying the depth by 1 ft^2 (0.09 m^2). These samples were also dried in the oven for 48 hours at 149° F (65° C). The samples were then individually weighed on a scale to the nearest tenth of a gram. This enabled the litter component to be added to the total fuel loading. At the same place along the fuel transect, the process was repeated except that the duff depth was measured and a 1 ft^2 (0.09 m^2) sample was collected. The measured litter and duff weights were used for the remainder of the fuel transects sampled within the strata at each site.

The specific gravity measurements were subjected to statistical tests, specifically testing the hypothesis that there is no significant difference in the specific gravity of fuels in a given size class between the sites and relative to wildfire intensity. The hypothesis that there is no difference in average squared quadratic mean diameter relative to wildfire intensity and site was also tested. If the hypotheses held, then a single value for each size class of fuel was used for specific gravity and the average squared quadratic mean diameter. If not, then the site and strata specific values were utilized. The following formulae were used for calculating fuel loading in tons/acre as required by most wildfire behavior models (Brown 1974; Brown et al. 1982):

(a) 0 – 3" material: $= (11.64*n*d^2*s*a*c)/NL$

Where:

11.64 =	a units conversion constant
n =	number of pieces
d^2 =	average squared quadratic mean diameter
s =	specific gravity
a =	nonhorizontal angle correction factor
c =	slope correction factor
N =	number of transects

L = length of line in feet

(b) 3”+ material: = $(11.64*\sum d^2*s*a*c)/NL$

Where: 11.64 = a units conversion constant

$\sum d^2$ = sum of squared diameters

s = specific gravity

a = nonhorizontal angle correction factor

c = slope correction factor

N = number of transects

L = length of line in feet

(c) duff = dd*dbd*44.51

Where: dd = duff depth in inches

dbd = duff bulk density

(d) litter = ld*lbd*44.51

Where: ld = litter depth in inches

lbd = litter bulk density

Formula (a) was calculated for each of the fuel time-lag classes and formula (b) for both rotten and sound material. The calculated values were then summed for the fuel loading in tons/ac.

Data Analysis

All statistical tests were carried out using SPSS 11.0 (SPSS Inc. 2001). Four independent variables; wildfire intensity, vector distance, age class (growing seasons since burn), and invasive presence were tested against the dependent variables as subtests of each hypothesis test. Wildfire intensity was measured at 3 levels; high, low, none. Eight different wildfires were collapsed into two age classes, ≤ 4 growing seasons (1, 2, 3, 4) since wildfire or ≥ 14 growing seasons (14, 17, 18, 23) since wildfire. Vector distance had three levels: 50 ft (15 m), 150 ft (45 m), 250 ft (75 m). The following null hypotheses were tested:

1) Duff depth does not differ relative to each of the independent variables.
2) Litter depth does not differ relative to each of the independent variables.
3) Invasive species dominance does not differ relative to each of the independent variables.
4) Species richness does not differ relative to each of the independent variables.
5) The Shannon Index of Community Diversity does not differ relative to each of the independent variables.
6) Pielou's Evenness Index does not differ relative to each of the independent variables.
7) Tree basal area does not differ relative to each of the independent variables.
8) Fuel loading does not differ relative to each of the independent variables.

To assess plant diversity on each study site, the Shannon Index of Community Diversity was calculated separately for the tree, shrub, and herb strata because of the different plot sizes utilized in sampling. The Shannon Index of Community Diversity was calculated using the following formula (Abrahamson 1984):

$$H' = -\sum_{t=1}^{K}(p_t * \ln p_t)$$

Where:

H' = Shannon Index of Community Diversity

P_t = the relative density of species t

K = number of species in the stand

The calculation of Pielou's Evenness Index allows one to isolate the affect of evenness on community diversity (Magurran 1988). Since the Shannon Index of Community Diversity does not enable one to determine the degree to which each species contributes to diversity (Elliot and Hewitt 1997), Pielou's Evenness Index was calculated using the following formula (Pielou 1966):

$$J' = H' / LN(S)$$

Where:

J' = Pielou's Evenness Index

H' = Shannon Index of Community Diversity

LN = Natural Log

S = Species Richness

Species richness was also calculated for each plot by summing the number of species encountered on that plot. Species richness was calculated for all species on a plot, rather than separately for each stratum.

Basal area (sq.ft./ac) was calculated for each plot, using the number of trees per plot in each DBH class and the mid-point value of the diameter class. All calculations and analysis were based on English units for basal area, but these values can easily be converted to square meters per hectare by multiplying sq.ft./ac by 0.2296. Basal area was used as a measure of

abundance for the tree strata when calculating the Shannon Index of Community Diversity, as well as providing verification that high intensity sites contain less basal area than low intensity and control sites. High intensity sites were defined as having at least 70% canopy removal and low intensity sites having less than 30% canopy removal.

Subsample

One plot from each of the intensity/growing seasons since wildfire combination was used to determine the coefficients necessary to calculate fuel loading (N=24). On each plot, fuels tallied in each size class were collected. In the lab, green volume (cm^3) of each piece was determined using water displacement, and then dried at 149° F (65° C) for a total of 48 hours. After 24 hours a piece from each size class was weighed and then placed in the oven to continue drying. After another 24 hours those pieces were weighed again to ensure that a constant weight had been achieved. All other samples were then weighed at 48 hours which was an adequate drying time to reach equilibrium. Afterwards, oven dry weight (g) was measured. Dividing oven-dry weight by green volume equates to specific gravity for each piece of fuel. An average specific gravity was calculated for each size class on each plot. This was necessary because the fuel loading calculations from Brown (1974) and Brown et al. (1982) contain specific gravities by size class for western species only.

In addition, the quadratic mean diameter was calculated, for each piece in the 1-hr, 10-hr, and 100-hr size classes, as follows:

$$QMD = (\Sigma(d^2)/n)^{1/2}$$

Where d = diameter

n = sample size

Actual diameters were measured in the 1000-hr size class, so the quadratic mean diameter was not applicable. Litter depth and duff depth samples were also collected on each of the subsample plots. Green volume was established by multiplying the 1 sq.ft (0.09 m^2) sample area by depth. The samples were also dried at 149° F (65° C) for 48 hours before being weighed for oven-dry weight to calculate the bulk density of litter and duff. Duff is reported along with fuel loading as a depth, while litter depth is converted to a tons/ac value and combined with 1-hr fuels, by multiplying the litter bulk density by litter depth.

From the subsamples, the average squared quadratic mean diameter for each size class and specific gravity values were incorporated into the fuel loading equation outlined by Brown (1974) and Brown et al. (1982). A total fuel loading for each plot, as well as 1-hr, 10-hr, 100-hr, 1000-hr sound, 1000-hr rotten, and 1000-hr fuel loadings were calculated in tons/ac so that the contribution from each size class to total fuel loading could be assessed.

Prior to being included in calculations for fuel loading, the quadratic mean diameters for each size class, specific gravity for each size class, and bulk densities for litter and duff were subjected to statistical tests of differences between growing seasons since burn, as each occurred on different sites. Assumptions of normality were visually assessed for all dependent variables using a histogram and a normal probability plot (Little and Hills 1978). Natural log transformations were applied as necessary and then reassessed visually to ensure normality existed in the data.

Analysis of subsample

A One-Way Analysis-of-variance (ANOVA) was used to test how much the mean values of each dependent variable differed between each level of the independent categorical variable. The ANOVA assumes that samples are normally distributed and are independent of each other.

Natural log transformations were necessary to meet the normality assumption for duff bulk density, litter bulk density, specific gravity for 10-hr fuels, and specific gravity for 1000-hr sound fuels. Levene's test for equality of variance was utilized to determine if unequal variance existed between values within a level of the independent variable. From the ANOVA, the relative efficiency of the randomized complete block design compared to the completely randomized design was calculated as follows (Ott and Longnecker 2001):

$$RE(RCB, CR) = ((b-1)MSB + b(t-1)MSE)/((bt-1)MSE)$$

When this value was greater than 1.25 it was concluded that blocking was efficient, because to gain the same level of precision, many more observations would have been necessary in the completely randomized design than in the randomized complete block design.

The null hypothesis that no difference in duff bulk density occurs across age class or wildfire intensity, was tested with an ANOVA using the natural log of the duff bulk density values. The relative efficiency test showed blocking was not beneficial. Neither wildfire intensity, age class, nor the interaction of the two were associated ($p > 0.05$) with differences in duff bulk density (Appendix C, Table 1). Therefore the mean value, 0.202 g/cm^3, was used as the duff bulk density for all sites. The null hypothesis that no difference in litter bulk density across age classes or wildfire intensity was tested using similar techniques. Litter bulk density was not significantly associated with intensity or age class and therefore the mean value, 0.073 g/cm^3, was used for all sites (Appendix C, Table 2).

The null hypothesis that no difference in specific gravity occurs for each size class relative to intensity or age class was tested the same way. A separate ANOVA was run for each size class (1-hr, 10-hr, 100-hr, 1000-hr sound and 1000-hr rotten) (Appendix C, Tables 3-7). There was no significant association ($p > 0.05$) between specific gravity and intensity, growing

seasons since burn (using each individual age, or age class (≤ 4 or ≥ 14 years) for the 1-hr, 100-hr, 1000-hr sound and 1000-hr rotten fuel classes. The relative efficiency test for the 10-hr fuel size class showed blocking by growing seasons since burn was beneficial. This, left only 3 observations per level of growing season since burn, so while intensity was not statistically significant ($p > 0.05$), it may have been affected by a small sample size. Since blocking by growing season since burn was necessary for 10-hour fuel specific gravity a separate specific gravity value was used for each growing season since burn (Table 3).

Table 3. Specific gravity values for size classes of fuel in Shenandoah National Park. Values are from a subsample of plots (N = 24) used in calculating fuel loads for all study sites. No significant ($p > 0.05$) differences were found between study sites. However a test for the relative efficiency of blocking showed it was necessary to use a separate value for the 10-hr fuels at each site.

		# of growing seasons since burn							
Size Class	All	1	2	3	4	14	17	18	23
	……g/cm^3……								
1-Hour	0.51								
10-Hour		0.48	0.46	0.54	0.36	0.42	0.54	0.44	0.35
100-Hour	0.37								
1000-Hour Sound	0.54								
1000-Hour Rotten	0.28								

Separate ANOVAs were used for each of the 1-hr, 10-hr, and 100-hr fuel size classes to test the null hypothesis that there is no difference in quadratic mean diameter relative to wildfire intensity or age class. Tests showed for all 3 size classes that there was no significant association ($p > 0.05$) between the quadratic mean diameter and age class, intensity or the interaction of age class and intensity (Appendix C, Tables 8-10). The null hypothesis was not rejected and one quadratic mean diameter was used for each size class on all sites as follows, 0.136 in, 0.478 in, and 1.778 in, for 1-hour, 10-hour, and 100-hr fuels respectively.

Overall Analysis

Normality was visually assessed and a natural log transformation was applied as necessary to meet the assumptions of the ANOVA (Little and Hills 1978). Natural log

transformations were used on numerous variables including: species richness, percent cover of invasive shrub species, percent cover of invasive herbaceous strata species, duff depth, 1-hr fuel loading, 10-hr fuel loading, 1000-hr sound fuel loading, 1000-hr rotten fuel loading, 1000-hr total fuel loading, and total fuel loading. General linear model univariate analysis was used with age class as a fixed factor and intensity and vector distance, both as random factors to look at all 216 plots and test for an association between the dependent and independent variables. If there was an association between an interaction variable and the dependent variable then an ANOVA was used followed by Tukey's test to indicate which means were statistically ($p < 0.05$) different. The independent variable vector distance was assessed by comparing the means for each level of vector distance within an intensity level, because if they were examined between sites as well, that would only leave 3 plots at each vector distance, in each intensity level, in each age class. For the purpose of explaining differences, the sample size would then be too small. Slope and aspect were also assessed using a One way ANOVA for each of the dependent variables. Slope and aspect were classified by rank following the Forest Site Quality Index (FSQI) rankings (Table 4). If slope or aspect were significant, a regression was run to determine the amount of variation explained. For dependent variables which had a significant relationship with slope or aspect in both high intensity areas and control areas, a test for the homogeneity of regression coefficients was performed to determine if the slope and intercept coefficients were significantly different, between wildfire intensity levels.

Table 4. The Forest Site Quality Index (FSQI) used for ranking slope and aspect into moisture availability categories, on potential productivity categories in Shenandoah National Park (Smith and Burkhart 1976).

Rank	Aspect	Slope
	…….(°)……	….(%)….
1	196 - 260	60 +
2	166-195; 261-280	45-59
3	146-165; 281-340	30-44
4	0-20; 341-360	15-29
5	81-145	0-14
6	21-80	

Two variables were analyzed using methods for categorical variables. 100-hr fuel loading had a distribution resembling a categorical variable. The data were analyzed using the number of pieces intersected in the 100-hr fuel size class as a surrogate for the 100-hr fuel loading. All other fuel size classes were analyzed using fuel loading in tons/ac. The other variable assessed this way was the invasive species presence. This 0,1 variable was used to indicate the presence of at least one invasive on the plot. The Chi-Square Goodness of Fit Test was used to determine if the proportions of values in each category cell were random or if they were associated with the dependent variable. The Chi-Square Goodness of Fit Test is calculated as (Ott and Longnecker 2001):

$$\chi^2 = \Sigma[(n_i - E_i)^2/E_i]$$

where n = observed value

E = expected value

The test shows whether the association is positive or negative. The Chi-Square Goodness of Fit Test requires independent samples and constant cell probabilities. The null hypothesis that there is no difference in invasive species presence relative to roads/trails was also tested in this manner (Table 5).

Table 5. Sampling locations and proximity to a road or trail, that may serve as an invasive species vector in Shenandoah National Park. A 0,1 variable for roads/trails was created to indicate whether plots in a given intensity/growing season burn combination were located near a road or a trail.

Wildfire	Growing Seasons Since Burn	Intensity Level		
		High	Low	Control
	…….(yrs)……...…….			
Rocky Top	1	Road	Road	Road
Fultz Run	2	Road	Road	Road
Shen Complex	3	Road	Road	Road
Bootens Gap	4	Road	Trail	Road
Madison Run	14	Trail	Trail	Trail
Neighbor Mountain	17	Trail	Trail	Trail
Big Run	18	Road	Road	Trail
Piney River	23	Trail	Trail	Trail

Additional tests were conducted for the variable invasive presence to determine if an association existed with other measured variables such as basal area, species richness, Shannon's Index of Community Diversity, Pielou's Evenness Index, and fuel loading. One way ANOVA were used to separately test the null hypothesis that there is no difference in each of the dependent variables relative to invasive presence.

Covariate analysis was used with the fuel loading data to remove the influence of site differences within an intensity level. The adjacent unburned areas were treated as a covariate by subtracting the fuel load in the unburned area from both the high and low intensity areas on a given site. These site differences between ≤ 4 and ≥ 14 growing season adjacent unburned areas will be discussed further in the fuel loading discussion. Significant differences associated with age class for the unburned areas were not included in tables, because the actual differences were not due to age class but site differences between the two age classes.

Results and Discussion

Species Richness

The data showed a significant main effects association between wildfire intensity and total species richness on a per plot basis (Appendix C, Table 11). Mean total species richness was significantly lower in high intensity wildfires, where 13 species were inventoried, compared to either the low intensity or adjacent unburned (control) areas which each had 16 species per plot on average (Appendix C, Table 12). There was no significant main effects association between age class (≤ 4 or ≥ 14 growing seasons) and species richness; nor was the interaction between age class and intensity significant in explaining differences in species richness. These results vary a bit from Elliot et al. (1999), who found 1 growing season after a prescribed wildfire on the Nantahala National Forest that tree and shrub species richness decreased in high intensity areas, while herb species richness increased. Elliot et al. (1999) found shrub species richness also decreased in low intensity areas, while tree species were unaffected in the low intensity area. In contrast, Abrahamson (1984) found no statistical difference in total species richness 2 years after a wildfire of varying intensity in a swale and gallberry flatwood. Regelbrugge (1988) found results similar to Abrahamson (1984) for the shrub strata (1-5 m) and regeneration strata (< 1 m) 2 years after the Big Run wildfire in Shenandoah National Park.

According to Wilson (1987) a prescribed burn in Shenandoah National Park resulted in increased species richness for the shrub and herb strata. Two growing seasons after a wildfire in the Sydney Region of Australia, Morrison (2002) found overall species richness increased with increasing wildfire intensity. Brockway and Lewis (1997), studying the longleaf pine wiregrass ecosystem, found low intensity winter burning led to higher herbaceous species richness than in unburned areas. Keeley et al. (2003) reported the highest overall species richness on high

severity burn sites. Contrary to our findings, Keeley et al. (2003) found there was a significant interaction between wildfire severity and time since burn for overall species richness in high elevation coniferous forests.

It was clear that wildfire intensity affected species richness, but statistically there was not an influence of time since burn on species richness. Total species richness would be expected not to change significantly as herbaceous richness would be decreasing by age 20 or so, while tree and shrub richness would still be increasing. Increasing the sample size to include more growing seasons since burn, especially between 5 and 13 growing seasons may show a relevant trend. No significant association between vector distance, from a road or trail, and species richness was found (See Appendix C, Table 13). The lack of association between vector distance and species richness was not surprising, as the spacing between plot centers did not allow for any gap between plots. Essentially the overstory plots touched one another, however there was spacing between the shrub and herb plots.

Significantly higher mean species richness values were recorded on plots where at least one invasive species was recorded (Table 6; Appendix C, Table 13), regardless of how intensely the area burned. This was unexpected as invasive species are suspected to negatively impact existing vegetation. This finding concurs with Dale and Adams (2003), who tracked species richness over a 15 year period following a debris avalanche from Mount Saint Helens, where it was found that areas with invasive species had higher native species richness, compared to areas where invasive species were absent. In contrast, Mack et al. (2000) suggests that native species richness will decrease with the presence of invasive species due to increased competition for resources.

Table 6. Descriptive statistics for species richness for all strata relative to the presence of invasive species in Shenandoah National Park.

Invasive Species	Number of Plots	Mean Species Richness	Std. Error
		……..…(#/plot)……..……..	
Absent	149	13.4	0.4
Present	67	18.7	0.6

The expected relationship of higher species richness with higher site quality, using surrogates slope and aspect (Appendix C, Tables 14 & 15), was not significantly affected by increasing wildfire intensity. The highest species richness values in terms of aspect were found on southwest aspects, for both high and low intensity burn areas. More research is needed to draw a link between invasive species presence, wildfire intensity, and growing seasons since burn, to allow managers to predict if invasive species will be present after wildfire.

Tree Stratum

The interaction between age class and wildfire intensity was significant in explaining differences in the mean Shannon's Index of Community Diversity for the tree stratum (Table 7; Appendix C, Table 16). High intensity burns occurring ≤ 4 growing seasons ago, had significantly lower mean Shannon's Index of Community Diversity values than the more recent low intensity burn areas. There was also significantly lower tree diversity in burned areas compared to the adjacent unburned areas (Appendix C, Table 17). Contrary to these findings, Wilson (1987) found after one growing season the Shannon's Index of Community Diversity for the tree strata increased following a low intensity burn.

Table 7. Shannon's Index of Community Diversity within growing seasons since burn of the overstory of hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number in a row do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	Shannon's Diversity Index[Z] for Trees							
	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
≤ 4	0.51[a]	0.09	1.05[b]	0.08	0.78	0.07	1.04	0.07
≥ 14	0.74[a,b]	0.09	0.80[a,b]	0.10	0.77	0.06	0.85	0.09
Average	0.62	0.06	0.92	0.06	0.77[1]	0.05	0.95[2]	0.06

[Z] $H' = \Sigma(pi*\ln(pi))$
(Pielou 1966)

Brockway and Lewis (1997) conducted periodic low intensity burns in a long leaf pine wiregrass ecosystem and found the tree diversity to be the same as the no burn area. In a situation similar to a high intensity burn, Brashears et al. (2004) determined that tree diversity decreased 2 – 26 years after clear cutting. Low intensity wildfires did not affect diversity in the tree strata, which supports a similar finding by Regelbrugge (1988), who 2 years after a low intensity burn in Shenandoah National Park, found no difference in diversity between low intensity and unburned areas. While an interaction between wildfire intensity and age class existed with tree diversity, it is still unclear how tree diversity changes over time since burning. The results showed differences in intensity, but the difference between age classes was not significant. A larger sample size with more burn areas in the 5 – 13 growing seasons since burn range may show more conclusive results in terms of change in diversity over time.

The presence of invasive species was associated with diversity in the tree stratum, as areas having invasive species had significantly higher tree stratum diversity (Table 8; Appendix C, Table 18). Only 1 invasive tree species, *Ailanthus altissima* (tree-of-heaven) was encountered

during the inventory of plots, while the majority of invasive species were herbs, vines, or shrubs. It had been hypothesized that invasive species presence would decrease diversity because of invasive species ability to spread rapidly and out compete native species. However, where invasive tree species were encountered on plots, only a few stems were inventoried and they were not dominating the vegetation on the site. This does not refute the ability of invasive species to dominate a site, but this study did not find that situation to exist in the tree stratum.

Table 8. Descriptive statistics for Shannon's Index of Community Diversity (H') for the tree stratum relative to the presence of invasive species in Shenandoah National Park.

Invasive Species	Number of Plots	Mean (H')	Std. Error
		……...(Index/plot)……...	
Absent	149	0.72	0.04
Present	67	1.07	0.05

There was no statistically significant association between vector distance, from a road or trail and tree diversity (Appendix C, Table 18). Using surrogates, slope and aspect, for site quality, the relationship with tree diversity did not significantly change as wildfire intensity increased (Appendix C, Tables 19 & 20). However, Keeley et al. (2003), in a mixed coniferous forest in the southern Sierra Nevada, found diversity decreased with increasing elevation. These relationships make sense, because lower slope positions tend to have better site quality and thus support more diverse vegetation, possibly including invasive species. The results showed no significant difference in tree diversity attributable to age class alone and tree diversity increased with the presence of invasive species, indicating it may not be necessary to take management steps to control invasive species in all locations. However, the National Parks see invasive species as their biggest threat and thus are targeted for removal, but in other management jurisdictions benefits of control may not outweigh the cost of such control. It may however be suitable to manage for the control of invasive tree species where they occur in high frequency.

The interaction between age class and wildfire intensity was significantly associated with the mean Pielou's Evenness Index for the tree strata (Table 9; Appendix C, Table 21). Pielou's Evenness Index for the tree strata was significantly lower in the more recent high intensity burn areas than the recent low intensity burn areas. Recent high intensity burns had the lowest evenness of any of the sites, as there was a significant age class and wildfire intensity interaction associated with tree evenness. There was not a significant difference between tree evenness in burned areas and the adjacent unburned areas (Appendix C, Table 22). Brashears et al. (2004) found that tree evenness decreased 2 – 26 years after a clearcut, which created conditions similar to a high intensity burn; however there were no significant differences associated with the time since clearcut, which contradicts the element of age class interaction. In a long-leaf pine-wiregrass ecosystem, Brockway and Lewis (1997) found that tree evenness was unaffected by periodic low intensity burns, which agrees with my findings in Shenandoah National Park.

Table 9. Pielou's Evenness Index within growing seasons since burn for the overstory of hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	Evenness Index[Z] for Overstory							
	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
≤ 4	0.48[a]	0.07	0.73[b]	0.04	0.61	0.05	0.77	0.03
≥ 14	0.71[a,b]	0.07	0.65[a,b]	0.06	0.68	0.04	0.70	0.05
Average	0.60	0.05	0.69	0.04	0.64	0.03	0.73	0.03

[Z] J' = (H'/Ln(S)) (Pielou 1966)

No association existed between vector distance from a road or trail and Pielou's Evenness Index for the tree stratum (Appendix C, Table 23). The expected relationship of Pielou's Evenness Index for the tree stratum, being positively correlated with site quality, as measured by surrogates slope and aspect(Appendix C, Tables 24 & 25), was not significantly altered by

increasing wildfire intensity. However, inconsistent with my hypothesis, significantly higher evenness values were associated with the presence of invasive species (Table 10; Appendix C, Table 23). Record of this relationship in the literature is rare, as most studies simply report that the presence of invasive species reduces diversity, rather than speaking specifically about evenness. In terms of concern for management, recent high intensity burns had the lowest tree evenness and it was not significantly different from any of the older wildfire intensities, but it was different from recent low intensity and adjacent unburned areas. On the other hand, the presence of invasive species increased tree evenness, so it may be best to look at high intensity burns and assess the situation on a case by case basis, as there was no relationship between the presence of invasive species and wildfire intensity.

Table 10. Descriptive statistics for Pielou's Evenness Index for the tree stratum relative to invasive species presence in Shenandoah National Park.

Invasive Species	Number of Plots	Mean Evenness Index	Std. Error
		……..….(Index/plot)…..…….	
Absent	149	0.61	0.03
Present	67	0.82	0.03

A significant interaction effect for age class and wildfire intensity existed in terms of mean tree basal area (Table 11; Appendix C, Table 26). Mean basal area was significantly lower at 56.3 sq.ft/ac (12.9 m^2/ha) in the recent high intensity burn areas compared to either age class in the low intensity areas. The older high intensity burn areas also had significantly lower basal area than the more recent low intensity burn areas (Table 11). The goal for installation of plots was to locate areas of at least 70% pre-wildfire canopy removal for high intensity plots, and less than 30% pre-wildfire canopy removal for low intensity plots. The data showed recent high intensity burn areas had significantly lower basal area, than either of the low intensity burned areas. The high intensity areas which burned ≥ 14 growing seasons ago were statistically

different from the older low intensity burns. In a comparison of tree basal area between burned and adjacent unburned areas there were significant main effect associations due to age class and burning (Appendix C, Table 27). Unburned areas had significantly higher basal area. In agreement with our findings, Regelbrugge (1988) and Elliot et al. (1999) reported significantly higher reductions in tree basal area associated with high intensity wildfires compared to low intensity wildfires. Contrary to our findings, and using different methodology, Brockway and Lewis (1997) found after repeated low intensity winter burning, that basal area in the burned areas was lower than in unburned areas of the longleaf pine wiregrass ecosystem.

Table 11. Differences in tree basal area (sq.ft/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number in a row do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……....(Sq.ft/ac)…….……..							
≤ 4	56.3[a]	8.7	118.8[c]	6.7	87.5	6.6	117.6	4.8
≥14	62.7[a]	6.2	89.1[b]	5.8	75.9	4.5	102.0	7.0
Average	59.5	5.3	103.9	4.7	81.7[1]	4.0	109.8[2]	4.3

Basal area was significantly higher in areas were invasive species were present compared to areas where invasive species were absent (Table 12; Appendix C, Table 28). In contrast, Mack et al. (2000) showed that invasive species decreased native species abundance. Perhaps because many invasive species can achieve very fast growth rates by monopolizing site resources, which would cause a reduction in basal area. A reduction was not observed in our study and this likely occurred because where invasive species were found only a few individuals were present. This was likely not enough to impact the growth of native tree species; however if invasive species had been present in greater numbers then a reduction in basal area may have been evident.

Table 12. Descriptive statistics for basal area (sq.ft/ac) relative to the presence of invasive species in Shenandoah National Park.

Invasive Species	Number of Plots	Mean Basal Area	Std. Error
		……..…(sq.ft/ac)…..…….	
Absent	149	85.7	3.8
Present	67	103.1	5.4

Vector distance from a road or trail was not statistically associated with any differences in basal area (Appendix C, Table 28). The relationship of site quality, using slope as a surrogate (Appendix C, Table 29), with tree basal area, was significantly different for each level of wildfire intensity, however this was not the case in terms of aspect (Appendix C, Table 30). The slope (Appendix C, Table 31) and intercept (Appendix C, Table 32) coefficients for regression equations in each intensity level were significantly different from each other (Figures 12 – 14). For a given slope, high intensity areas had significantly lower basal area than either the low intensity or adjacent unburned areas. Similarly in low intensity areas, slope was highly correlated with basal area, where basal area was highest on low slope locations. These sites are likely to be those with higher moisture and better soils, providing more site resources to trees than would be found on very steep slopes.

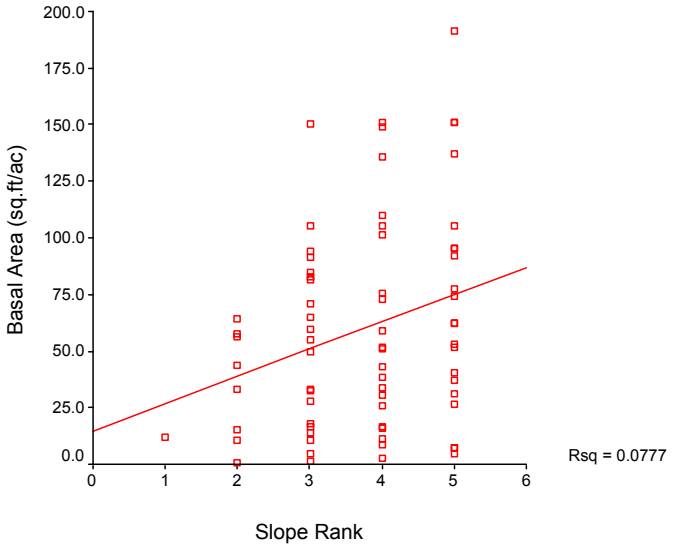

Figure 12. Scatter plot of basal area (sq. ft/ac) relative to slope rank in the high intensity burn areas of Shenandoah National Park, Virginia. A regression line is shown : y = 12.018x + 14.609. P = 0.018. Slope Rank: 1 = 60% +, 2 = 45 – 59%, 3 = 30 – 44%, 4 = 15 – 29%, 5 = 0 – 14%.

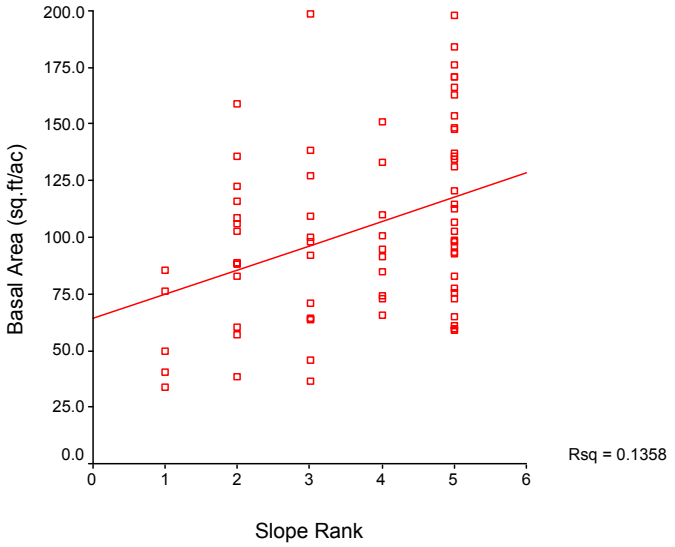

Figure 13. Scatter plot of basal area (sq.ft/ac) relative to slope rank in the low intensity burn areas of Shenandoah National Park, Virginia. A regression line is shown : y = 10.766x + 63.98. P = 0.001. Slope Rank: 1 = 60% +, 2 = 45 – 59%, 3 = 30 – 44%, 4 = 15 – 29%, 5 = 0 – 14%.

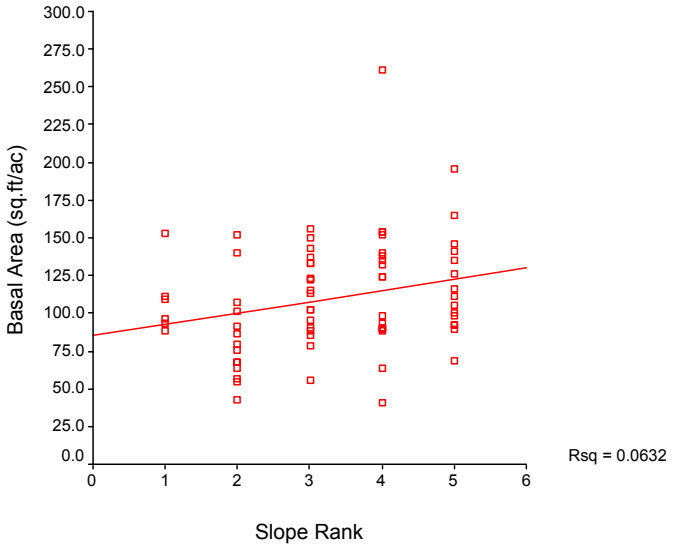

Figure 14. Scatter plot of basal area (sq.ft/ac) relative to slope rank in the unburned areas of Shenandoah National Park, Virginia. A regression line is shown : y = 7.393x + 85.283. P = 0.033. Slope Rank: 1 = 60% +, 2 = 45 – 59%, 3 = 30 – 44%, 4 = 15 – 29%, 5 = 0 – 14%.

A total of 37 tree species were observed in the overstory 1/10th acre (0.04 ha) plots in Shenandoah National Park. The relative basal area of each species for each of the 24 age/wildfire intensity strata is presented in Appendix B (Table 1). Chestnut oak, northern red oak, and scarlet oak accounted for over 50% of the basal area on average, contributing 34%, 9%, and 8% respectively. On average, 13.9 overstory species were encountered in control areas, while the high intensity areas and low intensity areas, had only 11.3 and 13.1 overstory tree species, respectively. The Piney River study site, which has had 23 growing seasons since the burn had the most overstory species in the high and low intensity areas, at 16 and 18 species respectively.

Shrub Stratum

A significant interaction of age class and wildfire intensity existed in relation to mean Shannon's Index of Community Diversity for the shrub stratum (Table 13; Appendix C, Table 33). Shrub diversity in the more recent high intensity burn areas was significantly lower than in

the older low intensity burn areas. A significant correlation existed within the interaction with growing seasons since burn, which is expressed by the linear regression equation: shrub diversity = 0.906 + 0.0215*(growing seasons since burn), $R^2 = 0.146$, P = 0.001. The data showed an interaction effect of wildfire and age class with shrub diversity when comparing burned areas to adjacent unburned areas (Appendix C, Table 34). Recent burned areas had significantly lower shrub diversity than older burned areas or the unburned areas adjacent to older burns. In contrast to this study, Regelbrugge (1988) found wildfire intensity to be unrelated to differences in shrub stratum diversity two years after a wildfire in Shenandoah National Park. However, Wilson (1987) and Elliot et al. (1999) both found wildfire intensity to be influential in terms of shrub diversity, even though their results were contradictory. Wilson (1987) found that shrub diversity was increased one year after a low intensity burn, while Elliot et al. (1999) reported that a low intensity burn decreased shrub diversity. Elliot et al. (1999) also found that shrub diversity was increased following a high intensity burn. Site differences between the Coweeta Basin in western North Carolina and Shenandoah National Park in Virginia may be a reason why opposing results were identified by Elliot et al. (1999) and Wilson (1987).

Table 13. Shannon's Index of Community Diversity within growing seasons since burn for the shrub stratum of hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	Shannon's Index of Community Diversity[Z] for Shrubs							
	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
≤ 4	1.01[a,b]	0.07	0.82[a]	0.08	0.91[1]	0.05	1.03[1,2]	0.07
≥ 14	1.23[b,c]	0.06	1.47[c]	0.05	1.35[3]	0.04	1.17[2,3]	0.06
Average	1.12	0.05	1.14	0.06	1.13	0.04	1.10	0.05

[Z] $H' = \Sigma(pi*\ln(pi))$ (Pielou 1966)

Vector distance was not associated with the Shannon's Index of Community Diversity in the shrub layer (Appendix C, Table 35). Shrub diversity was significantly higher on plots where invasive species were found compared to areas void of invasive species (Table 14; Appendix C, Table 35). These two findings are linked in that it was hypothesized that invasive species would be present in greater abundance at the closer vector distances to a road or trail, and would negatively impact shrub strata diversity. However, since invasive species occurred in low abundances and at all vector distances, the shrub diversity was not negatively impacted. It should be noted that prior to making management decisions regarding control of invasive species, more research is needed in areas where invasive species in the shrub strata occur in large abundance. It was anticipated that invasive species would out compete existing native vegetation and that may have been seen more readily if more abundant invasive species were encountered.

Table 14. Descriptive statistics for Shannon's Index of Community Diversity (H') for the shrub stratum relative to the presence of invasive species in Shenandoah National Park.

Invasive Species	Number of Plots	Mean (H')	Std. Error
		……...(Index/plot)……	
Absent	149	1.05	0.04
Present	67	1.27	0.05

Slope and aspect (Appendix C, Tables 36 & 37), surrogates for site quality, did not significantly explain any variation in shrub diversity as wildfire intensity increased. Shrub stratum diversity followed the expected ecological pattern in terms of site quality, regardless of the presence or intensity of wildfire.

Differences in the mean Pielou's Evenness Index for the shrub stratum were significantly associated with the interaction of age class and wildfire intensity (Table 15; Appendix C, Table 38). Evenness Index values in more recent low intensity burns were significantly lower than in either older high intensity areas or older low intensity areas. The main effects were not

significant. The low intensity area, was the only study area where a significant difference occurred between those wildfires occurring ≤ 4 growing seasons ago and those occurring ≥ 14 growing seasons ago. There was also a significant interaction effect of wildfire and age class in terms of shrub evenness when comparing burned areas and adjacent unburned areas (Appendix C, Table 39). More research is necessary to determine the impact of growing seasons since burn on shrub evenness. If more age classes were included in a study a significant trend may become evident, which would aid managers in understanding how shrub stratum evenness changes after a wildfire.

Table 15. Pielou's Evenness Index within growing seasons since burn for the shrub stratum of hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

	Pielou's Evenness Index[Z] for Shrubs							
Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
≤ 4	0.58[a,b]	0.03	0.48[a]	0.04	0.53[1]	0.02	0.59[1,2]	0.04
≥ 14	0.64[b]	0.03	0.70[b]	0.02	0.67[2]	0.02	0.59[1,2]	0.02
Average	0.61	0.02	0.59	0.02	0.60	0.02	0.59	0.02

[Z] J' = (H'/Ln(S)) (Pielou 1966)

Pielou's Evenness Index for the shrub stratum was not statistically associated with the presence of invasive species (Appendix C, Table 40). Shrub diversity and species richness did increase with the presence of invasive species, however the increase in species was evenly distributed in that they were all consistent in their relative frequency. As a result, the evenness of the shrub stratum remained unaffected. Neither vector distance from a road or trail (Appendix C, Table 40), nor aspect rank (Appendix C, Table 41) was significantly associated with shrub evenness. There was no significant association between slope rank, a surrogate for site quality,

and Pielou's Evenness Index for the shrub stratum as wildfire intensity increased (Appendix C, Table 42). This was not surprising as there was no association for shrub diversity either.

Relative frequency of species in the shrub stratum for each of the age/wildfire intensity combinations is found in Appendix B (Table 2). Low intensity areas had the most species present on average with 11 species, followed by the control area with 9 species and the high intensity area with 8 species per 1/100th ac (0.004 ha) plot. Low bush blueberry, Virginia creeper, and mountain laurel comprised over 50% of the species present in the shrub strata overall, with each contributing on average 31%, 15%, and 10% of the species composition. These same three species also accounted for almost 50% of the relative dominance of the shrub species based on percent cover class (Appendix B, Table 3). Low bush blueberry was the most dominant at 25%, followed by mountain laurel at 14%, and Virginia creeper at 10%. A total of 30 species, either shrubs or vines, were encountered in the shrub stratum overall.

Only 4 of the 8 study areas had invasive shrub species present, including porcelain berry (*Amelopsis brevipedunculata*) and tartarian honeysuckle (*Lonicera tatarica*). These invasive species were found on the following sites, including Fultz Run (2 growing seasons since burn), Booten's Gap (4 growing seasons since burn), Madison Run (14 growing seasons since burn), and Big Run (18 growing seasons since burn). A significant interaction between age class and intensity existed in explaining differences in the mean percent cover of invasive shrub species (Table 16; Appendix C, Table 43). This average was calculated based only on those plots where invasive species were present, rather than on all plots. The data showed significantly higher invasive shrub percent cover in the younger age class of the control, compared to the older age class of the control. This difference does not make any type of ecological sense, since both areas were free from disturbance. A site difference may be an explanation, as the Madison Run

wildfire unburned area was located at a much lower elevation, than the three other unburned study sites. The difference was significant because invasive species were present in the ≤ 4 growing seasons adjacent unburned areas but not in the ≥ 14 growing seasons unburned areas.

There was less variation amongst the high intensity burn areas in terms of invasive shrub species dominance compared to the low intensity areas (Table 16). An interaction effect of age class and burning was significant when comparing burned and unburned areas (Appendix C, Table 44). While the data showed a statistically significant difference between age classes in the unburned areas, it is believed to not be valid. The more likely relationship would be more cover of invasive species in disturbed areas than undisturbed areas. There was no association between vector distance from a road or trail and invasive shrub species dominance, nor was there an association between proximity to a major road (< 1 mile or > 1 mile) and invasive species presence (Appendix C, Table 45). This lack of association does not assist managers in their planning of where to expect invasive species to occur, and as such more research will be necessary to examine possible associations, which may exist to aide managers. Slope (Appendix C, Table 46) and aspect (Appendix C, Table 47) were not significant in explaining variation of invasive shrub percent cover as wildfire intensity increased. This finding was expected since slope and aspect were not significant in any of the other shrub stratum parameters.

Table 16. Differences in percent cover of invasive shrubs associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Age classes 1, 3, 17, and 23 did not have any invasive shrubs present. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……..(%)……							
≤ 4	0.09[a]	0.04	0.32[a,b]	0.11	0.21[1]	0.06	1.41[2]	0.68
≥ 14	0.09[a]	0.04	0.76[b]	0.47	0.43[1,2]	0.24	0.00[1]	0.00
Average	0.09	0.04	0.54	0.28	0.32	0.17	0.70	0.24

Herbaceous Stratum

There was a significant main effects association between wildfire intensity and mean Shannon's Index of Community Diversity for the late spring herbaceous stratum (Appendix C, Table 48). High intensity areas had significantly lower diversity, compared to either the low intensity or unburned areas, with index values of 1.21, 1.43, and 1.43 respectively (Appendix C, Table 49). No significant association existed between age class and herbaceous strata diversity, nor did an association exist for the interaction of age class and wildfire intensity with herbaceous strata diversity.

The majority of studies related to wildfire intensity and herbaceous stratum diversity involved low intensity wildfires. For example, Vandermast et al. (2004) found that 3 months following a low intensity burn there was no difference in herbaceous diversity compared to the unburned site. In contrast, Price and Weltzin (2003) determined that 5 years after a low intensity burn, there was significantly greater herbaceous diversity compared to the unburned areas. In contrast to our findings, Regelbrugge (1988) reported no effect on regeneration stratum diversity due to wildfire intensity, 2 years after a wildfire. Our sampling scheme did not detect a significant association with growing seasons since burn for herbaceous stratum diversity. This may have resulted because of decreasing herbaceous species diversity while the regeneration diversity was still increasing, and the combination of the two in the herbaceous stratum negated any differences.

There was a significant association between the presence of invasive species and herbaceous strata species diversity in the late spring (Table 17; Appendix C, Table 50). On average, plots where invasive species were present, had higher herbaceous strata diversity than

areas where invasive species were absent. It had originally been hypothesized that more invasive species would be present at the closer vector distances to a road or trail, and it had been expected that those invasive species would have negatively impacted herbaceous strata species diversity (Appendix C, Table 50). However, this was not the case, as invasive species existed at all vector distances and herbaceous strata species diversity was actually higher where invasive species were present. As mentioned previously with the tree stratum and shrub stratum, this condition of higher herbaceous strata species diversity with the presence of invasive species, likely exists only because the invasive species were not present in great abundance. Had the invasive species dominated the area, they would likely have out competed some native species and reduced herbaceous strata species diversity. Results may vary if repeated sampling for invasive species were to be done throughout the growing season.

Table 17. Descriptive statistics for Shannon's Index of Community Diversity (H') for the herbaceous layer relative to invasive species presence in Shenandoah National Park.

Invasive Species	Number of Plots	Mean (H')	Std. Error
		……..(Index/plot)…….	
Absent	149	1.24	0.04
Present	67	1.62	0.05

As site quality, based on slope (Appendix C, Table 51) and aspect (Appendix C, Table 52), increased, herbaceous strata species diversity increased as expected (Huebner et al. 1995), regardless of wildfire intensity level. In contrast to Vandermast et al. (2004), who found herbaceous strata diversity to remain constant across slopes in the unburned areas, our results showed herbaceous stratum diversity to be associated with slope. No association existed between vector distance from a road or trail and Shannon's Index of Community Diversity for herbaceous strata species. It would be beneficial to further examine invasive species presence and its impact on herbaceous strata diversity before making any management recommendations.

This study only inventoried minor incidents of invasive species, so areas where large infestations of invasive species occurred should also be evaluated for impacts on diversity as the results would likely be different.

Overall the mean Pielou's Evenness Index for the herbaceous stratum had a value of 0.72 ± 0.01. No significant associations existed between herbaceous strata evenness and the main effects or interactions of age class and wildfire intensity (Appendix C, Tables 53 & 54). Though herbaceous strata diversity was associated with wildfire intensity, herbaceous strata evenness was not impacted by wildfire intensity, age class, or an interaction. This likely occurred because species richness increased in areas, causing the diversity to increase while the evenness was unaffected because any new species were not encountered in large frequencies. These results vary from Brockway and Lewis (1997) who reported higher evenness for the herbaceous strata after periodic low intensity winter burning compared to unburned areas in the long leaf pine wiregrass ecosystem. Morrison (2002) also reported higher evenness in low intensity burn areas compared to unburned areas.

The presence of invasive species also did not affect the evenness index for late spring herbaceous strata species (Appendix C, Table 55). Again diversity and richness likely increased without any one species being present in greater frequency to upset the evenness index. No association existed between vector distance and Pielou's Evenness Index (Appendix C, Table 55). The relationship of herbaceous strata evenness and site quality (Huebner et al. 1995), using slope (Appendix C, Table 56) and aspect (Appendix C, Table 57) as surrogates, was not significantly affected by wildfire intensity level. Slope was not found to be associated with herbaceous strata evenness in this study, which agrees with findings reported by Vandermast et al. (2004). From an ecological standpoint it would make sense for there to be a trend towards

more diversity and more evenness as sites become of higher quality. If an association with any other variables existed, this sampling scheme did not capture them, but as mentioned before, slope and aspect were chosen to be analyzed after the fact. As herbaceous strata evenness was not impacted by any of the variables it may suggest that the herbaceous strata was fairly stable and not easily imbalanced. It also suggests that no change in management is necessary if there was no impact being caused to the herbaceous stratum evenness alone. However, it is important to look at the bigger picture of the herbaceous stratum as a whole, as well as the shrub and tree strata when making management recommendations. These findings are not representative of areas where large scale invasions of non-native species have occurred.

Overall 138 species were encountered in the herbaceous stratum over the entire study area. The herbaceous stratum included both herbaceous plants and tree species less than 4.5 ft (1.3 m) tall. Three species comprised 25% of the individuals encountered on herbaceous strata plots. Sassafras, red maple, and Viola spp. comprised 10%, 9%, and 6% of the species composition respectively over all plots. On average 37.8 species were encountered in the low intensity burn area at a study site, followed by 24.6 species in the unburned area and 19.9 species in the high intensity area. Relative frequency of plants in the herbaceous stratum for each age and intensity combination is shown in Appendix B (Table 4).

Three study sites, the Rocky Top wildfire (1 growing season since burn), the Fultz Run wildfire (2 growing seasons since burn), and the Neighbor Mountain wildfire (17 growing seasons since burn) did not have any invasive species present in the herbaceous stratum. In the remaining study sites there was no statistical association between wildfire intensity and percent cover of invasive herbaceous strata species. Age class and the interaction of age class and wildfire intensity were also non-significant in explaining differences in invasive herbaceous

strata species percent cover (Appendix C, Tables 58 & 59). On plots where invasive species were present the mean invasive herbaceous strata species percent cover was 1.44% ± 0.46%. The invasive species encountered in the herbaceous stratum of the study areas included garlic mustard, white bedstraw, common mullein, and tree-of-heaven. No statistical association existed between vector distance and percent cover of invasive herbaceous strata species (Appendix C, Table 60). Wildfire intensity did not significantly alter the expected relationship between invasive herbaceous strata species percent cover and site quality (Appendix C, Tables 61 & 62).

When looking at the presence of invasive species, there were no main effects or interactions amongst wildfire intensity, age class or vector distance, significant in explaining differences (Appendix C, Tables 63 & 64). Slope and aspect were not significant in explaining variation in the presence of invasive species (Appendix C, Table 65). In addition, proximity to a road or trail was not significant in explaining differences in invasive species presence. There was no association between the presence of invasive species and proximity to a major road (<1 mile (1.6 km) or > 1 mile (1.6 km)) (Appendix C, Table 66). The abundance of invasive herbaceous strata species appears to be completely random in its distribution. In contrast to our findings, Keeley et al. (2003) found wildfire severity and time since burn to be significantly associated with invasive species abundance in the southern Sierra Nevada.

This does not aid managers in deciding where to concentrate efforts to control invasive species. Invasive species were found at all vector distances evenly, however that does not necessarily mean when an invasive species was present at 250 ft (75 m) on a transect that there were also invasive species at 50 ft (15 m) and 150 ft (45 m) as well. Since invasive species occurred at all vector distances, the question arises to whether changing the plot spacing along transects would reveal any patterns in invasive species abundance.

Fuel Loading

No significant associations between age class, wildfire intensity, or their interaction existed with duff depth (Appendix C, Tables 67 & 68). The overall mean duff depth was 0.87 ± 0.04 in (2.21 ± 0.1 cm). It was not expected that there would be a difference between low intensity areas and no burn areas, but it had been anticipated that a high intensity wildfire would reduce the duff depth. No literature was found to discuss duff depth as it relates to high intensity wildfires, while three studies (Clinton et al. 1998; Waldrop et al. 2004; Elliot et al. 2002) found low intensity wildfires did not affect duff depth. These studies compared burned areas to unburned areas immediately after the wildfire, however, no studies on the accumulation of duff over time were found.

Neither vector distance nor the presence of invasive species had any statistical association with duff depth or any of the other fuel loading variables measured, except 1000-hr sound fuels. (Appendix C, Tables 69-76). The 1000-hr sound fuel loadings were significantly higher where invasive species were present, however there was no difference for 1000-hr rotten or the combination of sound and rotten 1000-hr fuels. Using surrogates, slope (Appendix C, Table 77) and aspect (Appendix C, Table 78), for site quality there was no difference in the relationship between site quality and duff depth as wildfire intensity level increased. The duff depth overall is related to landscape position (Waldrop et al. 2004). Ecologically, the landscape position is associated with moisture and species composition. Moisture level, whether the site is xeric or mesic, in addition to the amount of heating a site receives determines decomposition rates on a given site. More research is needed to determine how wildfire intensity affects duff depth and then over time how quickly duff depth returns to pre-wildfire levels.

No significant association existed between intensity, or age class, and litter depth (Appendix C, Table 79). The standard error was almost equal to the mean for litter depth, however a larger sample size may or may not have shown a statistical association. Due to the sample size and variation in the data no further statistical analyses were conducted in relation to litter depth. The weight of the litter layer was incorporated into the 1-hr fuel loading.

There was an association between the interaction of age class and wildfire intensity and the 1-hr fuel loading (Table 18; Appendix C, Table 80). The mean 1-hr fuel loading was significantly lower in the older low intensity burn areas compared to the other burn areas. Such a difference makes sense, as the high intensity wildfires killed a large proportion of the canopy and the fine twigs from those trees accumulated on the sites. In addition, the removal of the canopy greatly increased light in the understory, which prompted more herbaceous plants and tree seedlings to grow. Both of which would have added considerably to the litter and 1-hr fuel accumulation on the site. This however would not have occurred on the unburned areas. So while the high intensity wildfires likely consumed all the litter and 1-hr fuels, they quickly accumulated on the sites since the wildfires. The low intensity areas also had the litter and 1-hr fuels consumed but did not have as much input from the dying canopy into the fuel loading. In comparing burned areas to unburned areas, there was a significant interaction effect of burning and age class (Appendix C, Table 81). The recent adjacent unburned areas had significantly lower 1-hr fuel loading than the burned areas or the older unburned areas. The 1-hr fuel loadings encountered on my study sites in Shenandoah National Park were similar to Clinton et al. (1998), who found 4.5 tons/ac of litter in a southern Appalachian pine-hardwood stand.

Table 18. Covariate Analysis showed differences in 1-hr fuel loading (tons/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……(tons/ac)……							
≤ 4	4.6[a]	0.4	4.7[a]	0.4	4.6[1]	0.3	3.4[2]	0.4
≥ 14	6.9[a]	0.7	4.1[b]	0.4	5.5[1]	0.5	5.8[1]	0.5
Average	5.7	0.4	4.4	0.3	5.1	0.3	4.6	0.4

The available literature (Brown et al. 2003; Clinton et al. 1998; Waldrop et al. 2004; Elliot et al. 2002) supports the reduction of fuel loading for the 1-hr, 10-hr, and 100-hr size classes, after low intensity prescribed wildfires, but does not make reference to how the fuels react over time since the wildfire, nor were studies referencing high intensity burns located. However, none of these studies were conducted in mixed hardwood stands in Shenandoah National Park, Virginia. The consumption of fuels may differ within a given intensity level depending on the moisture content of the fuels (Stanturf et al. 2002). If a burn is conducted 1 day after a rain, while the fuels are still very wet, then less will be consumed than if the burn occurred a week after the last rain. The expected relationship between 1-hr fuel loading and site quality was not significantly altered as wildfire intensity level increased (Appendix C, Tables 82 and 83).

Significantly higher 1-hr fuel loadings were encountered in the older adjacent unburned areas compared to the younger adjacent unburned areas. This difference manifests itself through a distinction in age class, but actually resulted from site differences between the younger and older adjacent unburned areas. Slopes were significantly steeper on the older unburned areas, where higher fuel loadings were present (data not shown). Steeper slopes in the Appalachians of

Virginia are hypothesized to be associated with cooler and drier sites, leading to slower decomposition of litter and fuels. In addition, it is presumed that steeper slopes have been forested longer than those sites on lesser slopes, meaning the stands are older and have greater inputs to the fuel load than younger stands. While actual ages of the stands were not determined, from the species composition and their successional sere we can hypothesize that the stands with higher fuel loading are older. The ≤ 4 growing seasons adjacent unburned areas, which had lower fuel loadings, were dominated by species including oak, maple, birch, locust, cherry, and ash, whereas the ≥ 14 growing season adjacent unburned areas were dominated by oak, hickory, poplar, and basswood. Decomposition rates for the individual species were not assessed as previous studies suggest there is great variation in decomposition rates between sites (Mattson et al. 1987).

1-hr fuel loadings encountered in Shenandoah National Park were higher than all three of the applicable standard fuel models (Table 19). The 1-hr fuel loadings were closest to the Fuel Model 10 (Timber), but in all wildfire intensity levels, fuel loadings were 1.4 – 2.7 tons/ac higher than the Fuel Model 10 (Albini 1976 and Rothermel 1972). This model is applicable where stands have been damaged by wind, ice or insects and larger fuels have accumulated on the forest floor. In terms of planning for fire behavior it may be necessary to develop a custom fuel model, which could be more applicable to the forested areas of Shenandoah National Park.

Table 19. Differences in 1-hr fuel loads between the standard fuel models and the fuel loads present in Shenandoah National Park after varying levels of wildfire intensity.

	Wildfire Intensity			
	High	Low	Average	None
Fuel model	……(tons/ac)……			
8	4.2	2.9	3.6	3.1
9	2.8	1.5	2.2	1.7
10	2.7	1.4	2.1	1.6

The interaction of age class and wildfire intensity was significantly associated with differences in mean 10-hr fuel loading (Table 20; Appendix C Table 84). The 10-hr fuel loadings encountered in this study were slightly higher than those suggested in the fuel model 9 (Albini 1976 and Rothermel 1972), but were lower than the 5.4 tons/ac reported for 10-hr and 100-hr fuels combined in a southern Appalachian pine-hardwood stand (Clinton et al. 1998). The 10-hr fuel loading on high intensity older burned areas was significantly higher, compared to either more recent high intensity burn areas or older low intensity burn areas. The main effects of age class and wildfire intensity were not significant in explaining any variation in mean 10-hr fuel loading. However, there was no significant difference in 10-hr fuels between burned areas and adjacent unburned areas (Appendix C, Table 85). The 10-hr fuels responded basically the same as the 1-hr fuels for the same reasons mentioned previously. The dying canopy contributed to the buildup of fuels after the burn in the high intensity area. In addition, as the new seedlings grew to sapling stage and eventually began to self thin, more fuels were added to the forest floor. The older low intensity areas had significantly less 10-hr fuels than the older high intensity areas, which could possibly be due to different decomposition rates at the various sites; since the low intensity areas ≤ 4 growing seasons old, actually had about the same fuel loading as the older high intensity burn areas. Slope (Appendix C, Table 86) and aspect (Appendix C, Table 87) were not significant in explaining variation in 10-hr fuel loading in varying levels of wildfire intensity.

Table 20. Covariate Analysis showed differences in 10-hr fuel loading (tons/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……(tons/ac)……							
≤ 4	1.3[a]	0.2	2.0[a,b]	0.2	1.6	0.2	1.9	0.3
≥ 14	2.2[b]	0.2	1.2[a]	0.2	1.7	0.1	1.5	0.2
Average	1.8	0.1	1.6	0.2	1.7	0.2	1.7	0.2

The 10-hr fuel loadings encountered in Shenandoah National Park were higher than Fuel Models 8 and 9, but were lower than those in Fuel Model 10 (Albini 1976 and Rothermel 1972). 10-hr fuel loadings present on sites where varying levels of wildfire intensity occurred range from 0.6 – 1.4 tons/ac higher than Fuel Models 8 and 9, while 10-hr fuel loadings were only 0.2 – 0.4 tons/ac lower than Fuel Model 10 (Table 21). Fuel Models 8 and 9 are applicable on mixed oak-hickory and pine stands which have not had major disturbances, while Fuel Model 10 is more applicable to stands which have experienced some form of disturbance such as a wildfire.

Table 21. Differences in 10-hr fuel loads between the standard fuel models and the fuel loads present in Shenandoah National Park after varying levels of wildfire intensity.

	Wildfire Intensity			
	High	Low	Average	None
Fuel model		……(tons/ac)…….		
8	0.8	0.6	0.7	0.7
9	1.4	1.2	1.3	1.3
10	-0.2	-0.4	-0.3	-0.3

The data for 100-hr fuel loading had a categorical distribution and therefore were analyzed using Chi-square analyses. No main effect or interaction associations were found between age class or wildfire intensity and 100-hr fuel loading (Appendix C, Table 88). The overall mean 100-hr fuel loading was 1.3 ± 0.1 tons/ac. No significant difference in 100-hr fuel loading were found in relation to slope and aspect (Appendix C, Table 89). While, no

association was found in terms of 100-hr fuels in relation to wildfire intensity or age class, relationships were found with both the 1-hr and 10-hr fuels. A low intensity wildfire would consume less 100-hr fuels than it would of 10-hr and 1-hr fuels, and as such would likely not be statistically different from the unburned areas. A possible explanation for the lack of difference between wildfire intensities is that the high intensity wildfire consumed more 100-hr fuels, but that the 100-hr fuels accumulated much quicker after the wildfire in the high intensity areas than in the low intensity areas. This likely occurred because of the large proportion of canopy removal which took place during the high intensity wildfires.

100-hr fuel loadings were 1.2 tons/ac lower than Fuel Model 8 as well as being 0.7 tons/ac lower than Fuel Model 10. However, fuel loadings were 1.2 tons/ac higher than Fuel Model 9. As there was no significant differences associated with fire intensity for the 100-hr fuel loadings, an average 100-hr fuel loading was compared to the standard fuel models. These differences between existing conditions in Shenandoah National Park and the fuel models for all three size classes, support the notion that a custom fuel model may be more accurate for fire behavior prediction than the standard fuel models. No literature supporting or refuting these fuel loadings in Shenandoah National Park was found.

There was a significant association between wildfire intensity and 1000-hr sound fuel loading (Appendix C, Table 90). The 1000-hr sound fuel loading in the high intensity areas was significantly higher at 11.7 ± 2.7 tons/ac compared to 4.1 ± 1.0 tons/ac in the low intensity areas. However, the adjacent unburned areas, with a 1000-hr sound fuel loading of 6.1 ± 1.5 tons/ac was not significantly different from the burn areas (Appendix C, Table 91). As wildfire intensity increased there was no significant change in the relationship between 1000-hr sound fuel loading and site quality, using slope (Appendix C, Table 92) and aspect (Appendix C, Table

93) as surrogates. The available literature discussed the consumption of 1-hr, 10-hr, and 100-hr fuels but no mention was made of 1000-hr fuels. This was probably because the literature focused on low intensity wildfires, which would not have consumed 1000-hr fuels. The 1000-hr fuels have a much slower decomposition rate than any of the smaller size classes of fuel. A comparison of 1000-hr fuels to the fuel model 9 for the area was not possible as 1000-hr fuel loadings are not included in the model because they do not contribute much to the behavior of a wildfire, whereas the 1-hr, 10-hr, and 100-hr fuels do. The relationship between the low intensity area and control areas was expected. The high intensity area had higher fuel loading, though not significantly higher than the unburned areas, probably because of the same reasons mentioned for the 100-hr fuels. The high intensity wildfire consumed the 1000-hr fuels and then there was a quick initial increase in 1000-hr fuels as the dying canopy became part of the forest floor. This was then followed by little increase in fuel loading over time and a slow decomposition rate that would have been fairly consistent across all the intensity areas.

There were significant main effects associations for both age class and wildfire intensity with 1000-hr rotten fuel loading (Table 22; Appendix C, Table 94). The high intensity areas had on average significantly higher 1000-hr rotten fuel loadings, with 4.3 ± 0.7 tons/ac, compared to the low intensity area with 2.0 ± 0.4 tons/ac. On average, the more recently burned areas had lower 1000-hr rotten fuel loads than older sites. However, within an age class there was not a significant difference between burned areas and adjacent unburned areas (Appendix C, Table 95). Slope (Appendix C, Table 96) and aspect (Appendix C, Table 97) were not significantly associated with 1000-hr rotten fuel loads. Over time fuels will decompose, and as such it would be expected that the older wildfires would have a greater accumulation of 1000-hr rotten fuels. The significantly higher 1000-hr rotten fuel loads in the high intensity areas, make sense since

the high intensity areas had higher overall 1000-hr fuel loads. It was unclear as to why there was not an interaction effect for age class and wildfire intensity.

Table 22. Covariate Analysis showed differences in rotten 1000-hr fuel loading (tons/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same capital letter in a row do not differ at p = 0.05. Values followed by the same number do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……..……..…..(tons/ac)….....……..							
≤ 4	2.2	0.6	1.4	0.4	1.8[1]	0.4	2.0[1]	0.5
≥ 14	6.4	1.2	2.6	0.6	4.5[2]	0.7	5.7[2]	2.1
Average	4.3[A]	0.7	2.0[B]	0.4	3.1	0.4	3.9	1.1

Combining both sound and rotten fuels for the 1000-hr fuel class showed differences were significantly associated with the main effects for both age class and wildfire intensity, but not the interaction of the two (Table 23; Appendix C, Table 98). The average 1000-hr fuel loading was significantly lower in more recent burns compared to older burns. The 1000-hr fuel loading was significantly higher in the high intensity areas compared to the low intensity areas, but there was no significant difference between the burned areas and the adjacent unburned areas (Appendix C, Table 99). There was no significant difference in 1000-hr fuel loading related to slope (Appendix C, Table 100) or aspect (Appendix C, Table 101). These relationships existed for the reasons previously discussed under the 1000-hr sound and 1000-hr rotten sections.

Table 23. Covariate Analysis showed differences in total 1000-hr fuel loading (tons/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same number do not differ at p = 0.05. Values followed by the same capital letter do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……(tons/ac)……..							
≤ 4	10.3	2.6	5.3	1.3	7.8[1]	1.5	5.0[1]	0.9
≥ 14	21.9	5.0	6.9	1.8	14.4[2]	2.8	14.9[2]	3.3
Average	16.1[A]	2.9	6.1[B]	1.1	11.1	1.6	10.0	1.8

There was a significant association between the interaction of age class and wildfire intensity, with total fuel loading, which was comprised of all fuel size classes and litter (Table 24; Appendix C, Table 102). Mean total fuel loading was significantly higher in older high intensity burned areas, than either of the low intensity areas. Total fuel loading as it relates to site quality, was not significantly impacted differently by wildfire intensity level (Appendix C, Tables 103 & 104). Total fuel loading as was discussed in the literature only referred to the 1-hr, 10-hr, and 100-hr fuel loading and therefore was not applicable for comparisons in this situation. There was a significant difference associated with age class when comparing burned areas and the adjacent unburned areas, but there was not a significant difference between burned areas and unburned areas within an age class (Appendix C, Table 105).

Table 24. Covariate Analysis showed differences in total fuel loading (tons/ac) associated with intensity level within growing seasons since burn for hardwood stands in Shenandoah National Park, Virginia. Values followed by the same lower case letter do not differ at p = 0.05. Values followed by the same number in a column do not differ at p = 0.05. Adjacent unburned area values are included for both growing season classes to account for site differences.

Growing Seasons Since Burn	High Intensity Burned Area		Low Intensity Burned Area		Average Burned Area		Control (Unburned)	
	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error	Mean	Std. Error
	……(tons/ac)……..							
≤ 4	17.1[a]	2.5	13.1[a]	1.5	15.1[1]	1.5	11.7[1]	1.0
≥ 14	32.8[b]	5.2	13.5[a]	1.9	23.1[2]	3.0	23.3[2]	3.6
Average	25.0	3.0	13.3	1.2	19.1	1.7	17.5	2.0

The data suggest that the larger component of 1000-hr fuels in the high intensity area accounted for the higher fuel loading. It was not surprising that the total fuel loading in the older high intensity areas was not significantly different from the older control areas, since for each of the individual size classes of fuel there was no significant difference except in the 1000-hr fuels. While further research regarding the accumulation of fuels over time is needed, the results suggest that fuel loadings following a wildfire return to pre-burn levels quite rapidly, since neither of the burned intensity levels for either age class was significantly different from the unburned areas.

Conclusions

High intensity wildfires had the greatest impact on vegetation in mixed hardwood forests of Shenandoah National Park, Virginia. Similar to previous studies (Brashears et al. 2004 and Regelbrugge 1988), recent high intensity wildfires caused significantly lower evenness and diversity as well as lower basal area in the tree stratum. However, after 14 growing seasons, high intensity burn areas were no less diverse than areas impacted by low intensity wildfires or no wildfire at all. This suggests that the tree stratum may be able to recover quickly from disturbances such as high intensity wildfire. In addition, high intensity wildfires significantly reduced the diversity within the herbaceous stratum, which is contrary to Regelbrugge (1988) who reported no difference in diversity due to wildfire intensity. In the herbaceous stratum there was no indication of improving diversity over time. Finally, similar to Elliot et al. (1999), high intensity wildfires reduced the overall species richness on the impacted areas compared to either low intensity wildfires or no wildfire at all.

Low intensity wildfires resulted in much lower impacts on the mixed hardwood forest. Low intensity wildfires did not significantly alter the tree stratum, herbaceous stratum, or the fuel loadings. Previous studies found similar results for the tree stratum (Regelbrugge 1988) and the herbaceous stratum (Vandermast et al. 2004). Shrub diversity and evenness were initially reduced, more so than by high intensity wildfires, but after 14 growing seasons, the low intensity areas were the most diverse and even, in the shrub stratum, compared to either the high intensity areas or unburned areas. While the low intensity wildfires initially set back the shrub strata, it quickly recovered to pre-burn conditions.

Contrary to our initial hypotheses, the presence of invasive species seemed to be associated with higher diversity and evenness in all three strata. This finding could be deceiving

to land managers if taken out of context and applied elsewhere. The study sites involved in this research had fairly sporadic, low density, occurrences of invasive species, and as a result the full negative effect of the invasive species was not captured. If these sites were monitored over the next 5 or 10 years, one may notice the spread of these invasive species and the negative impacts of invasive species may become evident. Preliminary results from another invasive species survey being conducted by Shenandoah National Park, suggest that many invasive species were found in many areas but in low concentrations (Arsenault et al. 2004). It was hypothesized that the presence of invasive species would have reduced diversity, evenness, and richness on the sites, however it was also anticipated that invasive species would have been found in large abundance. Overall the presence of invasive species was seemingly random and not associated with wildfire intensity, growing seasons since burn, or vector distance from a road or trail as had been hypothesized. Contrary to our findings and using different methodology, invasive species were found to be present more frequently along the park boundaries and roads (Arsenault et al. 2004).

Additional research would be beneficial to monitor the presence of invasive species continually throughout the growing season, on a series of sites subjected to different intensities of wildfire, over time to determine if the populations of invasive species will increase or negatively impact the vegetation. The methods of this research could be easily applied to a long-term study.

Total fuel loading was associated with the interaction of wildfire intensity and age class. However, fuel loadings returned to levels found on unburned sites after 14 growing seasons. Overall fuel loadings were comparable to previous studies (Clinton et al. 1998). So, while there was an initial decrease in fuel loading as a result of the wildfires, the mortality associated with

the burn and the growth of new vegetation quickly added to the fuels on site, to the point fuel loading was not significantly different in the future. Though the fuel loadings did not significantly differ from the unburned areas as time since disturbance increased, the fuel loadings were different from the standard fuel models.

From a management standpoint, the high intensity wildfires cause more negative impacts than low intensity wildfires in Shenandoah National Park, Virginia. Steps could be taken to utilize low intensity prescribed fires, to create of mosaic of fuel loadings with the goal of reducing the risk for high intensity wildfires. Such actions would involve the implementation of a fire regime characteristic of that used by Native Americans.

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Appendix A – Site Characteristics

Table 1. Site characteristics for the Rocky Top wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
1-L-15-716	N 38° 18.599'	W 78° 43.693'	1240	4	W
1-L-45-717	N 38° 18.595'	W 78° 43.676'	1250	4	NE
1-L-75-718	N 38° 18.598'	W 78° 43.653'	1240	11	NE
1-L-15-719	N 38° 18.58'	W 78° 43.708'	1240	8	NW
1-L-45-720	N 38° 18.586'	W 78° 43.677'	1250	5	NE
1-L-75-721	N 38° 18.583'	W 78° 43.648'	1240	15	NE
1-N-15-722	N 38° 18.599'	W 78° 43.730'	1230	10	NW
1-N-45-723	N 38° 18.599'	W 78° 43.735'	1240	6	NE
1-N-75-724	N 38° 18.601'	W 78° 43.752'	1250	6	SE
1-N-15-725	N 38° 18.584'	W 78° 43.710'	1230	5	N
1-N-45-726	N 38° 18.580'	W 78° 43.750'	1240	4	NE
1-N-75-727	N 38° 18.582'	W 78° 43.744'	1250	5	E
1-L-15-728	N 38° 18.548'	W 78° 43.687'	1270	9	N
1-L-45-729	N 38° 18.535'	W 78° 43.677'	1280	5	N
1-L-75-730	N 38° 18.525'	W 78° 43.666'	1270	13	E
1-N-15-731	N 38° 18.547'	W 78° 43.722'	1270	12	N
1-N-45-732	N 38° 18.554'	W 78° 43.745'	1270	6	NE
1-N-75-733	N 38° 18.571'	W 78° 43.767'	1270	5	E
1-H-15-734	N 38° 18.105'	W 78° 43.968'	1360	15	SW
1-H-45-735	N 38° 18.111'	W 78° 43.953'	1370	21	W
1-H-75-736	N 38° 18.116'	W 78° 43.932'	1380	6	W
1-H-15-737	N 38° 18.127'	W 78° 43.969'	1360	16	SW
1-H-45-738	N 38° 18.131'	W 78° 43.943'	1370	14	W
1-H-75-739	N 38° 18.143'	W 78° 43.933'	1380	1	W
1-H-15-740	N 38° 18.146'	W 78° 43.972'	1360	15	SW
1-H-45-741	N 38° 18.156'	W 78° 43.954'	1370	1	SW
1-H-75-742	N 38° 18.165'	W 78° 43.937'	1380	4	NW

Table 2. Site characteristics for the Fultz Run wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
5-N-15-852	N 38° 31.607'	W 78° 34.267'	1100	27	NE
5-N-45-853	N 38° 31.582'	W 78° 34.279'	1120	40	NE
5-N-75-854	N 38° 31.586'	W 78° 34.300'	1140	21	NE
5-N-75-855	N 38° 31.602'	W 78° 34.302'	1140	22	E
5-N-45-856	N 38° 31.614'	W 78° 34.292'	1120	22	N
5-N-15-857	N 38° 31.618'	W 78° 34.269'	1100	45	E
5-N-15-858	N 38° 31.646'	W 78° 34.283'	1100	32	NE
5-N-45-859	N 38° 31.626'	W 78° 34.306'	1120	29	NE
5-N-75-860	N 38° 31.618'	W 78° 34.321'	1140	32	NE
5-H-15-864	N 38° 31.215'	W 78° 33.783'	1450	48	SW
5-H-45-865	N 38° 31.225'	W 78° 33.776'	1500	58	SW
5-H-75-866	N 38° 31.230'	W 78° 33.764'	1540	53	S
5-H-75-867	N 38° 31.246'	W 78° 33.783'	1530	35	W
5-H-75-868	N 38° 31.264'	W 78° 33.795'	1520	35	W
5-H-45-869	N 38° 31.260'	W 78° 33.815'	1460	43	W
5-H-45-870	N 38° 31.241'	W 78° 33.793'	1500	44	SW
5-H-15-871	N 38° 31.233'	W 78° 33.809'	1450	61	SW
5-H-15-872	N 38° 31.246'	W 78° 33.826'	1450	55	SW
5-L-15-861	N 38° 31.558'	W 78° 34.198'	1100	39	NE
5-L-45-862	N 38° 31.574'	W 78° 34.160'	1050	7	NW
5-L-75-863	N 38° 31.571'	W 78° 34.168'	1040	9	NW
5-L-15-873	N 38° 31.542'	W 78° 34.167'	1100	35	NE
5-L-45-874	N 38° 31.559'	W 78° 34.146'	1040	12	NW
5-L-75-875	N 38° 31.557'	W 78° 34.148'	1040	9	NW
5-L-75-876	N 38° 31.534'	W 78° 34.128'	1060	9	N
5-L-45-877	N 38° 31.527'	W 78° 34.138'	1080	10	NW
5-L-15-878	N 38° 31.522'	W 78° 34.156'	1100	44	NE

Table 3. Site characteristics for the Shenandoah Complex wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
3-L-15-761	N 38° 37.410'	W 78° 19.655'	3440	4	NE
3-L-45-762	N 38° 37.406'	W 78° 19.643'	3430	10	E
3-L-75-763	N 38° 37.397'	W 78° 19.628'	3420	17	E
3-L-15-838	N 38° 37.392'	W 78° 19.668'	3440	5	SW
3-L-45-839	N 38° 37.385'	W 78° 19.649'	3430	20	SE
3-L-75-840	N 38° 37.377'	W 78° 19.631'	3420	22	SE
3-L-15-764	N 38° 37.367'	W 78° 19.699'	3440	13	SE
3-L-45-765	N 38° 37.340'	W 78° 19.686'	3430	13	SE
3-L-75-766	N 38° 37.344'	W 78° 19.671'	3420	13	SE
3-H-15-767	N 38° 38.242'	W 78° 18.782'	2840	4	SE
3-H-45-768	N 38° 38.297'	W 78° 18.766'	2820	4	SE
3-H-75-769	N 38° 38.311'	W 78° 18.766'	2800	4	SE
3-H-15-841	N 38° 38.342'	W 78° 18.787'	2840	8	SE
3-H-45-842	N 38° 38.333'	W 78° 18.774'	2820	7	SE
3-H-75-843	N 38° 38.328'	W 78° 18.743'	2800	3	E
3-H-15-844	N 38° 38.362'	W 78° 18.762'	2840	8	E
3-H-45-770	N 38° 38.356'	W 78° 18.749'	2820	9	SE
3-H-75-771	N 38° 38.350'	W 78° 18.728'	2800	4	SE
3-N-15-772	N 38° 36.577'	W 78° 21.842'	3160	16	SE
3-N-45-773	N 38° 36.566'	W 78° 21.843'	3120	33	SE
3-N-75-774	N 38° 36.556'	W 78° 21.827'	3080	45	SE
3-N-15-845	N 38° 36.560'	W 78° 21.869'	3160	14	SE
3-N-45-846	N 38° 36.552'	W 78° 21.859'	3120	23	SE
3-N-75-847	N 38° 36.545'	W 78° 21.841'	3080	37	E
3-N-15-848	N 38° 36.551'	W 78° 21.889'	3160	13	E
3-N-45-775	N 38° 36.529'	W 78° 21.878'	3120	25	E
3-N-75-776	N 38° 36.521'	W 78° 21.863'	3080	35	SE

Table 4. Site characteristics for the Booten's Gap wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
7-L-15-93	N 38° 27.961'	W 78° 26.652'	2520	10	S
7-L-45-94	N 38° 27.942'	W 78° 26.651'	2520	10	SW
7-L-75-95	N 38° 27.948'	W 78° 26.656'	2520	10	S
7-L-75-96	N 38° 27.960'	W 78° 26.673'	2520	8	SE
7-L-45-97	N 38° 27.963'	W 78° 26.678'	2530	8	SE
7-L-15-98	N 38° 27.966'	W 78° 26.666'	2530	10	SW
7-L-15-99	N 38° 27.993'	W 78° 26.664'	2540	11	S
7-L-45-100	N 38° 27.989'	W 78° 26.674'	2530	12	S
7-L-75-862	N 38° 27.959'	W 78° 26.682'	2530	12	SE
7-H-15-863	N 38° 28.116'	W 78° 26.729'	2720	27	SW
7-H-45-864	N 38° 28.718'	W 78° 26.652'	2760	37	SW
7-H-75-865	N 38° 28.143'	W 78° 26.708'	2800	23	SW
7-H-15-866	N 38° 28.101'	W 78° 26.720'	2720	34	S
7-H-45-867	N 38° 28.121'	W 78° 26.718'	2760	35	S
7-H-75-868	N 38° 28.132'	W 78° 26.691'	2800	23	S
7-H-75-869	N 38° 28.121'	W 78° 26.669'	2800	24	S
7-H-45-870	N 38° 28.109'	W 78° 26.670'	2760	23	S
7-H-15-871	N 38° 28.095'	W 78° 26.682'	2720	23	S
7-N-15-872	N 38° 29.837'	W 78° 26.539'	3320	16	N
7-N-45-873	N 38° 29.857'	W 78° 26.520'	3300	17	N
7-N-75-874	N 38° 29.869'	W 78° 26.513'	3280	17	NW
7-N-75-875	N 38° 29.861'	W 78° 26.503'	3300	15	NW
7-N-75-876	N 38° 29.847'	W 78° 26.461'	3320	14	N
7-N-45-877	N 38° 29.836'	W 78° 26.472'	3340	13	N
7-N-45-878	N 38° 29.841'	W 78° 26.499'	3320	13	NW
7-N-15-879	N 38° 29.833'	W 78° 26.507'	3340	17	N
7-N-15-880	N 38° 29.827'	W 78° 26.480'	3350	14	N

Table 5. Site characteristics for the Madison Run wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
4-N-15-777	N 38° 15.255'	W 78° 45.282'	1500	30	S
4-N-45-778	N 38° 15.268'	W 78° 45.277'	1560	50	S
4-N-75-779	N 38° 15.284'	W 78° 45.279'	1600	53	SW
4-N-15-780	N 38° 15.251'	W 78° 45.259'	1500	45	S
4-N-45-781	N 38° 15.263'	W 78° 45.247'	1560	45	S
4-N-75-782	N 38° 15.272'	W 78° 45.238'	1600	52	S
4-N-15-783	N 38° 15.227'	W 78° 45.218'	1500	45	S
4-N-45-784	N 38° 15.256'	W 78° 45.218'	1520	61	S
4-N-75-785	N 38° 15.272'	W 78° 45.210'	1560	70	S
4-H-15-786	N 38° 15.385'	W 78° 45.917'	1400	8	S
4-H-45-787	N 38° 15.405'	W 78° 45.880'	1440	52	S
4-H-75-788	N 38° 15.412'	W 78° 45.874'	1480	45	S
4-H-15-789	N 38° 15.392'	W 78° 45.898'	1400	14	S
4-H-45-790	N 38° 15.403'	W 78° 45.904'	1440	52	SW
4-H-75-791	N 38° 15.417'	W 78° 45.900'	1480	43	S
4-H-75-792	N 38° 15.422'	W 78° 45.928'	1480	49	SW
4-H-45-793	N 38° 15.413'	W 78° 45.923'	1440	40	S
4-H-15-794	N 38° 15.391'	W 78° 45.929'	1400	44	S
4-L-15-795	N 38° 15.401'	W 78° 45.370'	1550	39	S
4-L-45-796	N 38° 15.391'	W 78° 45.365'	1570	43	S
4-L-75-797	N 38° 15.425'	W 78° 45.375'	1620	53	S
4-L-75-798	N 38° 15.423'	W 78° 45.400'	1640	41	S
4-L-45-799	N 38° 15.410'	W 78° 45.398'	1620	40	S
4-L-15-800	N 38° 15.397'	W 78° 45.391'	1580	46	S
4-L-15-849	N 38° 15.376'	W 78° 45.402'	1540	38	SW
4-L-45-850	N 38° 15.388'	W 78° 45.409'	1580	52	SE
4-L-75-851	N 38° 15.404'	W 78° 45.411'	1620	52	SE

Table 6. Site characteristics for the Big Run wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
6-N-15-879	N 38° 17.698'	W 78° 39.520'	2600	33	E
6-N-45-880	N 38° 17.700'	W 78° 39.507'	2580	52	NE
6-N-75-881	N 38° 17.712'	W 78° 39.484'	2560	60	E
6-N-75-882	N 38° 17.732'	W 78° 39.488'	2560	63	E
6-N-75-883	N 38° 17.752'	W 78° 39.480'	2560	60	E
6-N-45-884	N 38° 17.746'	W 78° 39.495'	2580	55	E
6-N-45-885	N 38° 17.727'	W 78° 39.491'	2580	55	E
6-N-15-886	N 38° 17.729'	W 78° 39.500'	2600	48	E
6-N-15-887	N 38° 17.751'	W 78° 39.518'	2600	32	NE
6-H-15-888	N 38° 17.783'	W 78° 39.553'	2640	8	NE
6-H-45-889	N 38° 17.783'	W 78° 39.568'	2640	12	SW
6-H-75-890	N 38° 17.776'	W 78° 39.585'	2600	10	W
6-H-15-891	N 38° 17.803'	W 78° 39.557'	2640	19	N
6-H-45-892	N 38° 17.796'	W 78° 39.578'	2640	5	N
6-H-75-893	N 38° 17.796'	W 78° 39.596'	2600	38	W
6-H-15-894	N 38° 17.182'	W 78° 39.580'	2640	13	NE
6-H-45-895	N 38° 17.818'	W 78° 39.579'	2620	33	W
6-H-75-896	N 38° 17.812'	W 78° 39.618'	2600	33	W
6-L-15-897	N 38° 16.135'	W 78° 40.573'	2400	40	NE
6-L-45-898	N 38° 16.151'	W 78° 40.572'	2320	60	N
6-L-75-899	N 38° 16.163'	W 78° 40.564'	2260	60	N
6-L-75-900	N 38° 16.179'	W 78° 40.588'	2280	43	NE
6-L-75-88	N 38° 16.174'	W 78° 40.626'	2280	50	NE
6-L-45-89	N 38° 16.164'	W 78° 40.611'	2320	47	NE
6-L-45-90	N 38° 16.150'	W 78° 40.597'	2320	62	N
6-L-15-91	N 38° 16.156'	W 78° 40.599'	2400	74	NE
6-L-15-92	N 38° 16.146'	W 78° 40.620'	2400	63	N

Table 7. Site characteristics for the Neighbor Mountain wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
8-L-15-881	N 38° 41.985'	W 78° 22.163'	2020	48	NW
8-L-45-882	N 38° 42.002'	W 78° 22.177'	1980	50	NW
8-L-75-883	N 38° 42.014'	W 78° 22.192'	1960	47	NW
8-L-75-884	N 38° 42.022'	W 78° 22.164'	1960	44	NW
8-L-45-885	N 38° 42.020'	W 78° 22.153'	1980	43	W
8-L-15-886	N 38° 42.002'	W 78° 22.150'	2020	45	NW
8-L-15-887	N 38° 41.980'	W 78° 22.191'	2030	36	NW
8-L-45-888	N 38° 41.981'	W 78° 22.192'	200	50	NW
8-L-75-889	N 38° 42.010'	W 78° 22.203'	1990	57	NW
8-H-15-890	N 38° 42.056'	W 78° 22.219'	1850	30	W
8-H-45-891	N 38° 42.062'	W 78° 22.191'	1920	25	W
8-H-75-892	N 38° 42.063'	W 78° 22.171'	1960	35	W
8-H-75-893	N 38° 42.088'	W 78° 22.180'	1950	25	W
8-H-75-894	N 38° 42.106'	W 78° 22.187'	1950	18	W
8-H-45-895	N 38° 42.101'	W 78° 22.210'	1910	20	SW
8-H-45-896	N 38° 42.082'	W 78° 22.197'	1920	18	W
8-H-15-897	N 38° 42.074'	W 78° 22.214'	1880	30	W
8-H-15-898	N 38° 42.089'	W 78° 22.232'	1860	35	W
8-N-15-899	N 38° 42.551'	W 78° 22.151'	1350	35	N
8-N-45-900	N 38° 42.535'	W 78° 22.178'	1390	39	N
8-N-75-71	N 38° 42.528'	W 78° 22.199'	1400	61	NE
8-N-75-72	N 38° 42.516'	W 78° 22.199'	1430	27	N
8-N-75-73	N 38° 42.499'	W 78° 22.192'	1480	11	N
8-N-45-74	N 38° 42.496'	W 78° 22.168'	1480	36	NE
8-N-45-75	N 38° 42.521'	W 78° 22.173'	1420	47	N
8-N-15-76	N 38° 42.521'	W 78° 22.146'	1400	46	NE
8-N-15-77	N 38° 42.497'	W 78° 22.151'	1450	35	E

Table 8. Site characteristics for the Piney River wildfire. Plot labeling: Wildfire-Intensity Level-Vector Distance-Plot Number.

Plot	Latitude	Longitude	Elevation	Slope (%)	Aspect
2-N-15-743	N 38° 44.155'	W 78° 16.499'	2160	25	E
2-N-45-744	N 38° 44.160'	W 78° 16.484'	2120	40	SE
2-N-75-745	N 38° 44.142'	W 78° 16.459'	2080	44	SE
2-N-15-829	N 38° 44.174'	W 78° 16.488'	2160	25	SE
2-N-45-830	N 38° 44.170'	W 78° 16.479'	2120	30	SE
2-N-75-831	N 38° 44.171'	W 78° 16.453'	2080	15	SE
2-N-15-746	N 38° 44.194'	W 78° 16.493'	2170	29	E
2-N-45-747	N 38° 44.180'	W 78° 16.456'	2130	20	E
2-N-75-748	N 38° 44.191'	W 78° 16.456'	2100	30	E
2-H-15-749	N 38° 44.135'	W 78° 16.533'	2180	25	SE
2-H-45-750	N 38° 44.147'	W 78° 16.528'	2200	35	SE
2-H-75-751	N 38° 44.163'	W 78° 16.567'	2220	20	SE
2-H-15-832	N 38° 44.158'	W 78° 16.511'	2180	35	SE
2-H-45-833	N 38° 44.162'	W 78° 16.537'	2200	29	E
2-H-75-834	N 38° 44.169'	W 78° 16.552'	2230	35	E
2-H-15-752	N 38° 44.193'	W 78° 16.503'	2180	28	SE
2-H-45-753	N 38° 44.193'	W 78° 16.534'	2210	25	SE
2-H-75-754	N 38° 44.195'	W 78° 16.547'	2240	35	SE
2-L-15-755	N 38° 44.587'	W 78° 16.119'	2450	21	E
2-L-45-756	N 38° 44.584'	W 78° 16.143'	2470	13	SE
2-L-75-757	N 38° 44.585'	W 78° 16.161'	2490	24	SE
2-L-15-835	N 38° 44.562'	W 78° 16.116'	2440	21	SE
2-L-45-836	N 38° 44.568'	W 78° 16.140'	2450	15	S
2-L-75-837	N 38° 44.567'	W 78° 16.156'	2470	13	SE
2-L-15-758	N 38° 44.542'	W 78° 16.114'	2410	20	SE
2-L-45-759	N 38° 44.555'	W 78° 16.130'	2420	12	S
2-L-75-760	N 38° 44.555'	W 78° 16.182'	2470	23	SE

Appendix B – Plot Data

Table 1. Relative basal area by species for each of the study sites. Each growing seasons since burn and intensity combination are represented. H = high, L = low, N = none.

	1H	1L	1N	2H	2L	2N	3H	3L	3N	4H	4L	4N
Acer pensylvanicum	-	-	-	0.006	0.006	0.019	0.009	-	0.061	-	-	-
Acer rubrum	0.061	0.003	0.030	0.014	0.044	0.032	0.062	-	0.007	0.010	0.005	0.023
Acer saccharum	-	-	-	0.010	-	0.031	-	-	-	-	-	-
Amelanchier arborea	-	-	0.013	-	0.006	-	-	-	-	-	-	0.004
Betula lenta	-	-	-	0.062	-	0.090	0.022	0.001	-	-	-	0.001
Carya glabra	-	0.006	0.005	0.074	0.147	0.107	-	-	0.081	0.017	0.126	0.200
Carya ovata	-	-	-	-	-	0.023	-	-	-	-	0.004	-
Carya tomentosa	-	-	-	0.075	0.050	0.067	-	-	0.007	-	0.003	-
Castanea dentata	-	-	0.001	-	-	-	0.004	0.010	0.001	0.006	-	-
Cornus alternifolia	-	-	-	-	-	-	-	-	0.018	-	-	-
Cornus florida	-	-	0.027	0.006	0.004	0.011	-	-	-	0.005	0.037	-
Fraxinus pennsylvanica	-	-	-	0.016	0.049	0.105	0.015	-	0.012	-	-	0.009
Hamamelis virginiana	-	-	-	-	0.003	-	-	0.010	0.004	-	0.001	0.009
Ilex verticulata	-	-	-	-	-	-	-	-	-	-	-	-
Liriodendron tulipifera	-	-	-	0.256	0.000	0.195	0.037	-	-	-	-	-
Malus spp.	-	-	-	-	-	-	-	0.011	-	-	-	-
Nyssa sylvatica	0.112	0.017	0.082	-	-	0.011	0.042	-	-	0.136	0.009	0.004
Oxydendrum arboreum	-	-	-	0.063	-	-	-	-	-	-	-	-
Pinus echinata	-	-	0.091	-	-	-	-	-	-	-	-	-
Pinus pungens	-	-	-	-	-	-	-	-	-	-	-	-
Pinus rigida	0.044	0.008	-	-	-	-	-	-	-	-	-	-
Pinus strobus	-	-	-	-	-	-	-	-	0.037	-	-	0.007
Pinus taeda	-	-	-	-	-	-	-	-	-	-	-	-
Pinus virginiana	-	-	-	-	-	-	-	-	-	0.276	0.025	0.045
Platanus occidentalis	-	-	-	-	-	-	-	-	-	-	0.001	-
Prunus avium	-	-	-	-	-	-	-	-	0.008	-	-	-
Prunus serotina	-	-	-	0.001	0.031	-	-	0.001	0.029	-	-	-
Quercus alba	-	0.084	0.089	-	0.089	0.004	-	0.042	0.141	0.069	-	0.055
Quercus coccinea	0.221	0.161	0.522	-	0.069	-	0.038	0.584	0.072	-	0.018	0.033
Quercus falcata	-	0.022	-	-	-	-	-	-	-	-	-	-
Quercus ilicifolia	-	-	-	-	-	-	-	-	-	0.082	0.005	-
Quercus prinus	0.533	0.444	0.062	0.079	0.206	0.170	0.249	-	0.065	0.161	0.522	0.589
Quercus rubra	-	0.069	0.021	-	0.072	-	0.504	0.299	0.129	0.078	0.127	0.018
Quercus velutina	0.029	0.140	0.040	0.001	0.074	0.063	-	0.019	0.167	0.137	0.084	0.004
Robinia pseudoacacia	-	-	-	0.015	0.027	0.005	0.012	0.014	0.058	-	0.003	-
Sassafras albidum	-	-	0.002	0.255	0.013	0.014	0.005	0.009	0.004	0.022	0.030	0.001
Tilia americana	-	-	-	0.067	0.110	0.115	-	-	0.061	-	-	-

Table 1 continued. Relative basal area by species for each of the study sites. Each growing seasons since burn and intensity combination are represented. H = high, L = low, N = none.

	5H	5L	5N	6H	6L	6N	7H	7L	7N	8H	8L	8N
Acer pensylvanicum	-	-	-	-	0.026	-	-	0.005	0.003	-	0.001	-
Acer rubrum	-	0.035	0.021	0.290	0.097	0.101	-	0.068	0.139	-	0.010	0.004
Acer saccharum	-	-	0.006	-	-	-	-	-	-	-	-	-
Amelanchier arborea	-	-	0.005	0.002	0.006	-	-	-	-	0.026	-	-
Betula lenta	-	-	-	0.121	0.086	0.013	0.006	0.043	0.030	-	0.003	-
Carya glabra	-	0.026	0.056	0.012	0.009	-	-	-	-	-	-	0.001
Carya ovata	-	0.003	-	-	-	-	0.049	0.002	-	-	-	-
Carya tomentosa	-	0.090	0.077	-	-	-	-	-	0.004	-	-	0.011
Castanea dentata	-	-	-	-	-	0.005	0.087	-	-	0.037	-	-
Cornus alternifolia	-	-	-	-	-	-	-	-	-	-	-	-
Cornus florida	-	0.001	0.050	-	0.002	-	0.001	-	-	-	-	0.010
Fraxinus pennsylvanica	-	-	-	-	-	-	-	0.251	0.004	-	-	-
Hamamelis virginiana	-	0.001	-	0.022	-	0.019	-	-	-	-	-	-
Ilex verticulata	-	-	-	-	-	-	-	-	-	-	-	-
Liriodendron tulipifera	-	0.025	0.016	-	-	-	0.005	0.474	-	-	-	-
Malus spp.	-	-	-	-	-	-	-	-	0.001	-	-	-
Nyssa sylvatica	0.063	0.013	0.017	0.035	-	0.039	0.007	0.020	-	0.186	0.030	0.043
Oxydendrum arboreum	-	-	-	-	-	-	-	-	-	-	-	-
Pinus echinata	-	-	-	-	-	-	-	-	-	-	-	-
Pinus pungens	0.080	-	-	-	0.026	-	-	-	-	0.210	0.004	-
Pinus rigida	-	-	-	-	-	-	-	-	-	-	-	-
Pinus strobus	-	-	-	0.004	0.043	0.001	-	0.036	-	-	-	-
Pinus taeda	0.060	0.024	0.009	0.032	0.001	-	-	-	-	-	0.007	0.050
Pinus virginiana	0.003	-	-	-	0.009	-	-	-	-	-	-	-
Platanus occidentalis	-	-	-	-	-	-	-	-	-	-	-	-
Prunus avium	-	-	-	-	-	-	-	-	-	-	-	-
Prunus serotina	-	-	0.001	0.040	-	-	0.001	-	0.738	-	-	-
Quercus alba	-	0.140	0.263	-	-	-	-	-	0.017	-	-	-
Quercus coccinea	0.060	0.019	-	0.004	-	-	-	-	-	0.051	-	0.081
Quercus falcata	-	-	-	-	-	-	-	-	-	0.218	-	-
Quercus ilicifolia	0.024	-	-	-	-	-	-	-	-	0.089	-	-
Quercus prinus	0.380	0.442	0.242	0.291	0.511	0.682	0.733	0.056	-	0.145	0.930	0.766
Quercus rubra	0.255	0.064	0.011	0.019	0.136	0.129	0.058	0.002	0.051	-	0.013	0.031
Quercus velutina	0.057	0.117	0.223	-	0.007	0.005	-	-	-	0.020	0.002	-
Robinia pseudoacacia	-	-	-	-	0.001	0.002	0.013	0.033	0.002	-	-	-
Sassafras albidum	0.017	-	0.005	0.106	0.034	0.004	0.040	0.001	-	0.017	-	0.001
Tilia americana	-	-	-	-	-	-	-	-	0.005	-	-	-

Table 2. Relative frequency of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	1H	1L	1N	2H	2L	2N	3H	3L	3N
*Ampelopsis brevipedunculata**	-	-	-	-	-	-	-	-	-
Azalea spp.	0.068	0.027	0.220	-	-	-	-	0.337	0.252
Ceanothus americanus	-	-	-	-	-	-	-	0.005	-
Clematis virginiana	-	-	-	-	-	-	-	-	-
Cornus Canadensis	-	-	-	-	-	-	-	-	-
Crataegus spp.	-	-	-	-	-	-	-	-	-
Euphorbia lathyris	-	-	-	-	-	-	-	-	-
Fragaria virginiana	-	-	-	-	0.030	-	-	0.048	-
Gaylussacia baccata	-	-	-	-	-	0.005	-	-	-
Ilex Montana	-	-	-	-	-	-	-	-	-
Ipomea pandurata	-	-	-	-	-	-	-	-	-
Kalmia latifolia	0.144	0.002	0.001	-	0.004	-	0.259	0.179	0.087
Lindera benzoin	-	-	-	0.022	0.002	0.016	0.003	-	-
*Lonicera tatarica**	-	-	-	-	-	-	-	-	-
Lyonia ligustrina	-	-	-	-	-	-	-	-	-
Parthenocissus quinquefolia	-	-	-	0.473	0.365	0.689	0.029	0.107	0.351
Ribes americanum	-	-	-	-	0.022	-	-	-	-
Rosa Carolina	-	0.006	-	-	-	-	-	-	-
Rubus allegheniensis	-	-	-	0.153	0.194	0.116	0.433	0.014	0.012
Rubus occidentalis	-	-	-	0.066	0.064	0.058	0.071	0.008	0.005
Rubus phoenicolasius	-	-	-	-	-	-	0.173	-	-
Sambucus Canadensis	-	-	-	-	-	-	-	-	-
Smilax glauca	0.005	-	-	-	-	-	-	-	-
Smilax rotundifolia	0.003	0.003	0.024	0.023	-	0.007	-	-	0.001
Solanum dulcamara	-	-	-	-	-	-	-	-	-
Toxicodendron radicans	-	-	-	0.188	0.139	0.025	-	-	0.204
Vaccinium angustifolium	0.578	0.923	0.714	-	0.043	-	0.015	0.169	0.052
Vaccinium stamineum	0.201	0.035	0.031	-	0.088	-	-	-	-
Viburnum acerifolium	-	0.003	0.009	0.008	0.049	0.075	0.018	0.003	0.016
Vitas aestivalis	-	-	-	0.066	0.002	0.009	-	-	0.021

Table 2 continued. Relative frequency of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	4H	4L	4N	5H	5L	5N	6H	6L	6N
*Ampelopsis brevipedunculata**	0.009	0.017	0.001	0.001	0.021	0.081	0.011	0.006	-
Azalea spp.	0.075	0.019	-	-	0.225	0.177	0.009	0.074	0.010
Ceanothus americanus	-	0.038	-	-	-	-	-	0.130	-
Clematis virginiana	-	-	-	-	-	-	-	-	-
Cornus Canadensis	-	-	0.054	-	-	-	-	-	-
Crataegus spp.	-	-	-	-	0.010	0.013	-	-	0.016
Euphorbia lathyris	-	0.055	-	-	-	-	-	0.002	-
Fragaria virginiana	0.006	-	0.094	-	-	-	-	-	-
Gaylussacia baccata	-	0.007	-	0.102	-	-	-	0.062	-
Ilex Montana	-	-	-	-	-	-	-	0.010	-
Ipomea pandurata	0.004	0.004	-	-	0.013	-	-	-	-
Kalmia latifolia	0.217	-	-	0.171	0.102	-	0.107	0.126	0.372
Lindera benzoin	-	-	-	-	-	-	-	-	-
*Lonicera tatarica**	-	-	-	-	-	-	0.009	-	-
Lyonia ligustrina	-	-	-	-	-	0.068	-	-	-
Parthenocissus quinquefolia	-	0.034	0.136	-	0.003	0.251	0.269	0.018	-
Ribes americanum	-	-	-	-	-	-	-	-	-
Rosa Carolina	-	0.006	0.003	-	0.085	0.018	0.003	0.030	-
Rubus allegheniensis	-	0.082	-	-	0.029	0.018	0.003	0.002	-
Rubus occidentalis	-	-	0.003	-	-	-	-	-	-
Rubus phoenicolasius	-	-	-	-	-	-	-	-	-
Sambucus canadensis	-	-	-	-	-	-	0.005	-	-
Smilax glauca	0.020	0.069	0.033	0.106	0.013	0.015	-	-	-
Smilax rotundifolia	0.053	-	0.056	-	0.004	0.015	-	-	-
Solanum dulcamara	-	-	0.005	-	-	-	-	-	-
Toxicodendron radicans	-	0.004	-	-	0.007	-	-	-	-
Vaccinium angustifolium	0.519	0.419	0.473	0.621	0.376	0.061	0.312	0.330	0.422
Vaccinium stamineum	0.097	0.205	0.112	-	0.015	0.025	0.014	0.073	0.180
Viburnum acerifolium	-	0.012	0.028	-	0.065	0.013	0.239	0.133	-
Vitas aestivalis	0.001	0.029	0.003	-	0.034	0.246	0.017	0.002	-

Table 2 continued. Relative frequency of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	7H	7L	7N	8H	8L	8N
*Ampelopsis brevipedunculata**	0.002	0.003	-	-	-	-
Azalea spp.	0.419	-	-	-	0.083	0.219
Ceanothus americanus	-	-	-	0.005	-	0.031
Clematis virginiana	-	-	0.171	-	-	-
Cornus canadensis	-	-	-	-	-	-
Crataegus spp.	-	-	0.259	-	-	-
Euphorbia lathyris	-	-	-	-	-	-
Fragaria virginiana	-	-	-	-	-	-
Gaylussacia baccata	-	-	-	0.497	0.133	0.124
Ilex montana	0.001	-	-	-	-	-
Ipomea pandurata	-	-	-	-	-	-
Kalmia latifolia	0.080	-	-	0.317	0.168	0.039
Lindera benzoin	-	-	-	-	-	-
*Lonicera tatarica**	-	-	-	-	-	-
Lyonia ligustrina	-	-	-	0.002	-	0.000
Parthenocissus quinquefolia	-	0.855	-	-	-	0.001
Ribes americanum	-	-	-	-	-	-
Rosa Carolina	-	0.001	-	-	-	0.005
Rubus allegheniensis	0.094	0.073	0.571	-	-	0.009
Rubus occidentalis	-	-	-	-	-	-
Rubus phoenicolasius	-	-	-	-	-	-
Sambucus canadensis	-	-	-	-	-	-
Smilax glauca	-	-	-	0.001	-	-
Smilax rotundifolia	-	0.031	0.014	-	-	-
Solanum dulcamara	-	-	-	-	-	-
Toxicodendron radicans	-	0.007	-	-	-	-
Vaccinium angustifolium	0.281	-	-	0.145	0.513	0.463
Vaccinium stamineum	0.071	0.002	-	0.002	0.066	0.083
Viburnum acerifolium	0.051	0.026	-	-	0.037	0.016
Vitas aestivalis	0.001	0.005	-	-	-	0.008

Table 3. Relative dominance of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	1H	1L	1N	2H	2L	2N	3H	3L	3N
*Ampelopsis brevipedunculata**	-	-	-	-	-	-	-	-	-
Azalea spp.	0.076	0.069	0.200	-	-	-	-	0.177	0.204
Ceanothus americanus	-	-	-	-	-	-	-	0.018	-
Clematis virginiana	-	-	-	-	-	-	-	-	-
Cornus canadensis	-	-	-	-	-	-	-	-	-
Crataegus spp.	-	-	-	-	-	-	-	-	-
Euphorbia lathyris	-	-	-	-	-	-	-	-	-
Fragaria virginiana	-	-	-	-	0.010	-	-	0.062	-
Gaylussacia baccata	-	-	-	-	-	0.004	-	-	-
Ilex montana	-	-	-	-	-	-	-	-	-
Ipomea pandurata	-	-	-	-	-	-	-	-	-
Kalmia latifolia	0.106	0.004	0.015	-	0.010	-	0.282	0.194	0.273
Lindera benzoin	-	-	-	0.027	0.002	0.068	0.007	-	-
*Lonicera tatarica**	-	-	-	-	-	-	-	-	-
Lyonia ligustrina	-	-	-	-	-	-	-	-	-
Parthenocissus quinquefolia	-	-	-	0.279	0.167	0.409	0.007	0.072	0.217
Ribes americanum	-	-	-	-	0.023	-	-	-	-
Rosa carolina	-	0.025	0.003	-	-	-	-	-	-
Rubus allegheniensis	-	-	-	0.127	0.336	0.161	0.381	0.051	0.054
Rubus occidentalis	-	-	-	0.170	0.126	0.107	0.117	0.016	0.012
Rubus phoenicolasius	-	-	-	-	-	-	0.150	-	-
Sambucus canadensis	-	-	-	-	-	-	-	-	-
Smilax glauca	0.011	-	-	-	-	-	-	-	-
Smilax rotundifolia	0.006	0.011	0.042	0.049	-	0.015	-	-	0.004
Solanum dulcamara	-	-	-	-	-	-	-	-	-
Toxicodendron radicans	-	-	-	0.135	0.090	0.041	-	-	0.110
Vaccinium angustifolium	0.560	0.770	0.586	-	0.069	-	0.007	0.377	0.032
Vaccinium stamineum	0.241	0.112	0.068	-	0.130	-	-	-	-
Viburnum acerifolium	-	0.011	0.071	0.031	0.031	0.146	0.049	0.033	0.074
Vitas aestivalis	-	-	0.015	0.130	0.004	0.049	-	-	0.020

Table 3 continued. Relative dominance of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	4H	4L	4N	5H	5L	5N	6H	6L	6N
*Ampelopsis brevipedunculata**	0.020	0.050	-	0.005	0.076	0.318	0.013	0.078	-
Azalea spp.	0.064	0.019	-	-	0.164	0.071	0.013	0.058	0.016
Ceanothus americanus	-	0.075	-	-	-	-	-	0.096	-
Clematis virginiana	-	-	-	-	-	-	-	-	-
Cornus canadensis	-	-	0.094	-	-	-	-	-	-
Crataegus spp.	-	-	-	-	0.027	0.066	-	-	0.005
Euphorbia lathyris	-	0.038	-	-	-	-	-	0.006	-
Fragaria virginiana	0.002	-	0.034	-	-	-	-	-	-
Gaylussacia baccata	-	0.034	-	0.124	-	-	-	0.090	-
Ilex montana	-	-	-	-	-	-	-	0.015	-
Ipomea pandurata	0.004	0.023	0.003	-	0.031	-	-	-	-
Kalmia latifolia	0.241	-	-	0.201	0.111	-	0.130	0.156	0.589
Lindera benzoin	-	-	-	-	-	-	-	-	-
*Lonicera tatarica**	-	-	-	-	-	-	0.013	-	-
Lyonia ligustrina	-	-	-	-	-	0.045	-	-	-
Parthenocissus quinquefolia	-	0.031	0.124	-	0.009	0.131	0.200	0.031	-
Ribes americanum	-	-	-	-	-	-	-	-	-
Rosa carolina	-	0.023	0.006	-	0.101	0.012	0.013	0.015	-
Rubus allegheniensis	-	0.057	-	-	0.018	0.020	0.013	0.006	-
Rubus occidentalis	-	-	0.003	-	-	-	-	-	-
Rubus phoenicolasius	-	-	-	-	-	-	-	-	-
Sambucus canadensis	-	-	-	-	-	-	0.013	-	-
Smilax glauca	0.016	0.065	0.055	0.123	0.027	0.020	-	-	-
Smilax rotundifolia	0.145	-	0.086	0.005	0.009	0.035	-	-	-
Solanum dulcamara	-	-	0.003	-	-	-	-	-	-
Toxicodendron radicans	-	0.008	-	-	0.009	-	-	-	-
Vaccinium angustifolium	0.409	0.273	0.396	0.542	0.258	0.050	0.229	0.238	0.251
Vaccinium stamineum	0.097	0.228	0.164	-	0.040	0.025	0.038	0.091	0.139
Viburnum acerifolium	-	0.015	0.028	-	0.084	0.030	0.276	0.114	-
Vitas aestivalis	0.002	0.061	0.006	-	0.036	0.175	0.051	0.006	-

Table 3 continued. Relative dominance of individuals in the shrub plots by species for each of the study sites. Each growing season since burn and intensity combination are represented. H = high, L = low, N = none.

	7H	7L	7N	8H	8L	8N
*Ampelopsis brevipedunculata**	0.013	0.023	-	-	-	-
Azalea spp.	0.357	-	-	-	0.065	0.098
Ceanothus americanus	-	-	-	0.004	-	0.025
Clematis virginiana	-	-	0.209	-	-	-
Cornus canadensis	-	-	-	-	-	-
Crataegus spp.	-	-	0.419	-	-	-
Euphorbia lathyris	-	-	-	-	-	-
Fragaria virginiana	-	-	-	-	-	-
Gaylussacia baccata	-	-	-	0.447	0.167	0.104
Ilex montana	0.004	-	-	-	-	-
Ipomea pandurata	-	-	-	-	-	-
Kalmia latifolia	0.136	-	-	0.399	0.269	0.147
Lindera benzoin	-	-	-	-	-	-
*Lonicera tatarica**	-	-	-	-	-	-
Lyonia ligustrina	-	-	-	0.004	-	-
Parthenocissus quinquefolia	-	0.712	-	-	-	0.006
Ribes americanum	-	-	-	-	-	-
Rosa carolina	-	0.006	-	-	0.006	0.011
Rubus allegheniensis	0.081	0.092	0.337	-	-	0.011
Rubus occidentalis	-	-	-	-	-	-
Rubus phoenicolasius	-	-	-	-	-	-
Sambucus canadensis	-	-	-	-	-	-
Smilax glauca	-	-	-	0.004	-	-
Smilax rotundifolia	-	0.094	0.035	-	-	-
Solanum dulcamara	-	-	-	-	-	-
Toxicodendron radicans	-	0.011	-	-	-	-
Vaccinium angustifolium	0.217	-	-	0.136	0.326	0.251
Vaccinium stamineum	0.149	0.006	-	0.004	0.116	0.097
Viburnum acerifolium	0.039	0.045	-	-	0.052	0.025
Vitas aestivalis	0.004	0.011	-	-	-	0.225

Table 4. Relative frequency of individuals in the herbaceous strata plots by species for each of the study sites. Each growing seasons since burn and intensity combination are represented. H = high, L = low, N = none.

	1H	1L	1N	2H	2L	2N	3H	3L	3N	4H	4L	4N
Acer pensylvanicum	-	-	-	-	-	0.001	-	-	0.001	-	-	-
Acer rubrum	0.019	0.033	0.387	-	0.002	0.005	0.010	-	0.005	0.096	0.031	0.050
Achillea millefolium	-	-	-	-	-	-	-	-	-	-	-	-
Agrostis spp.	0.002	-	-	-	-	-	-	0.006	-	-	-	-
*Ailanthus altissima**	-	-	-	0.001	-	-	-	-	-	-	-	-
*Alliaria officinalis**	-	-	-	0.065	0.092	0.017	-	-	0.065	-	-	0.015
Amelanchier arborea	-	-	0.079	-	0.001	-	-	-	-	-	0.004	0.004
Amphicarpa bracteata	-	-	-	0.019	0.020	0.287	-	-	-	0.003	0.002	0.007
Anemone quinquefolia	-	-	-	-	-	0.007	-	-	-	-	-	-
Antennaria plantaginiflolia	-	-	-	-	0.003	-	-	-	-	-	0.002	-
Arisaema triphyllum	-	-	-	-	-	-	-	-	-	-	-	-
Aristolochia macrophylla	-	0.004	-	-	-	-	-	-	-	-	-	-
Asplenium platyneuron	-	-	-	-	-	-	-	-	-	-	-	0.035
Aster 2 spp.	-	-	-	-	-	-	0.020	-	-	-	-	-
Aster divaricatus	-	-	-	-	-	-	-	-	-	-	-	-
Aster spp.	-	-	-	0.001	0.025	0.022	0.112	0.005	0.024	-	0.033	-
Athyrium angustum	-	-	-	-	-	-	-	-	-	-	-	-
Betula lenta	-	-	-	-	0.002	-	0.009	-	-	-	-	-
Boehmeria cylindica	-	-	-	-	-	0.006	-	0.010	-	-	-	-
Botrychium dissectum	-	-	-	-	0.003	-	-	-	-	-	-	-
Botrychium matricariaefolium	-	-	-	0.005	-	0.004	-	-	-	-	-	-
Botrychium virginianum	-	-	-	-	-	-	0.388	0.283	0.073	0.024	-	-
Campanula rotundifolia	-	-	-	-	0.005	-	-	-	-	-	0.023	-
Carex spp.	0.017	-	-	0.024	0.259	0.028	0.040	0.346	0.089	0.069	0.424	0.353
Carya glabra	-	0.004	-	-	0.002	0.002	-	0.001	0.001	-	0.002	-
Carya tomentosa	-	-	-	0.001	-	0.001	-	-	-	-	0.002	-
Castanea dentata	0.002	0.082	0.026	-	-	-	-	0.011	-	-	-	-
Centaurea cyanus	-	-	-	-	-	-	-	-	-	-	-	0.024
Cercis canadensis	-	-	-	-	-	-	-	-	-	-	-	-
Chimaphila maculata	-	-	0.021	-	-	-	-	-	-	0.119	0.027	0.028
Chimaphila umbellata	-	-	-	-	-	-	-	-	-	-	0.010	-
Cimicifuga racemosa	-	-	-	-	-	0.010	-	-	0.008	-	-	-
Convallaria montana	-	-	-	-	-	-	-	-	-	0.036	-	-
Cornus florida	-	-	0.084	0.001	0.003	-	-	-	-	0.003	0.006	0.015
Dactylis glomerata	-	-	-	-	-	-	-	-	-	-	0.002	-
Danthonia spicata	-	-	-	-	-	-	-	-	-	-	-	-
Dennstaedtia punctilobula	-	-	-	-	-	0.007	-	-	-	-	-	-
Dentaria heterophylla	-	-	-	-	-	-	-	-	-	-	-	-
Desmodium rotundifolium	-	-	-	-	-	-	-	-	-	-	0.040	-
Dioscorea villosa	-	-	-	0.001	0.003	-	0.004	0.002	0.007	-	0.004	-
Dryopteris campyloptera	-	-	-	-	-	-	-	-	-	-	-	-
Epigaea repens	-	-	-	-	-	-	-	-	-	-	-	-
Epilobium angustifolium	0.685	-	-	-	0.007	-	0.106	0.008	0.036	-	-	0.026

	1H	1L	1N	2H	2L	2N	3H	3L	3N	4H	4L	4N
Euonymus americanus	-	-	-	0.004	0.007	0.015	-	0.001	0.001	-	-	0.004
Eupatorium rugosum	-	-	-	-	-	-	-	-	-	-	-	-
Fragaria vesca	-	-	-	-	-	-	-	-	-	-	-	-
Fraxinus americana	-	-	-	0.008	0.016	0.015	-	-	0.001	-	-	-
Galium lanceolatum	-	-	-	-	0.008	-	-	-	-	-	-	-
*Galium mollugo**	-	-	-	0.059	0.040	0.086	-	0.013	0.016	-	-	0.031
Galium spp.	-	-	-	-	-	-	-	-	-	-	0.008	-
Geranium spp.	-	-	-	-	-	-	-	-	0.016	-	-	-
Gerardia pedicularia	-	-	0.021	-	-	-	-	-	-	-	-	-
Gnaphalium obtusifolium	-	-	-	-	-	-	-	-	-	-	-	-
Hamamelis virginiana	-	-	-	-	0.003	-	-	0.003	0.001	-	-	0.007
Helianthus divaricatus	-	-	-	-	-	-	-	-	-	-	-	-
Helianthus microcephelus	-	-	-	-	-	-	-	-	-	-	-	-
Heuchera americana	-	-	-	-	-	-	-	-	-	-	-	-
Hieracium paniculatum	-	-	-	-	-	-	-	-	-	-	0.002	-
Hieracium venosum	-	-	-	-	-	-	-	-	-	-	0.008	-
Houstonia longifolia	-	-	-	-	-	-	-	-	-	-	-	-
Houstonia tenuifoila	-	-	-	-	-	-	-	-	-	-	0.006	-
Hypericum perforatum	-	-	-	-	-	-	-	-	-	-	-	-
Ilex verticulata	-	-	-	-	-	-	-	-	-	-	-	-
Impatiens capensis	-	-	-	-	-	-	-	-	0.020	-	-	-
Iris cristata	-	-	-	-	-	-	-	-	-	-	-	-
Lactuca canadensis	-	-	-	0.018	0.024	-	-	-	-	-	-	0.002
Liriodendron tulipifera	0.010	-	-	0.001	-	0.002	0.024	-	-	-	-	-
Lycopodium spp.	-	-	-	-	-	-	-	-	-	-	-	-
Lysimachia quadrifolia	0.002	-	-	-	-	-	0.118	0.167	0.007	0.003	-	0.002
Malus pumila	-	-	-	-	-	-	-	0.001	-	-	-	-
Medeola virginiana	-	-	-	-	-	-	-	0.009	0.001	-	-	-
Meehania cordata	-	-	-	0.001	-	-	-	-	-	-	-	-
Mentha spp.	-	-	-	-	-	-	-	-	-	-	0.002	-
Mimulus ringens	-	-	-	0.044	0.024	0.037	0.052	0.017	0.031	-	-	-
Nyssa sylvatica	0.104	0.108	0.047	0.002	-	0.003	0.018	-	-	0.054	0.004	-
Osmorhiza clatonii	-	-	-	0.038	0.019	0.160	-	0.011	0.035	-	-	-
Osmundaceae cinnamomea	-	-	-	-	-	-	-	-	-	-	-	-
Panax quinquefolius	-	-	-	-	-	-	-	-	-	-	-	-
Panicum boscii	0.026	-	-	-	-	-	-	-	-	-	0.048	-
Panicum clandestinum	-	-	-	-	-	-	-	-	-	-	0.071	-
Pastinaca sativa	-	-	-	-	-	-	-	-	0.092	-	-	-
Pedicularis canadensis	-	-	-	-	-	-	-	-	-	-	-	-
Penstemon digitalis	-	-	-	-	-	-	-	-	-	-	-	-
Phaseolus polystachios	-	-	-	0.121	-	0.015	-	-	-	-	-	-
Phytolaccaceae americana	0.003	-	-	-	-	-	-	-	-	-	-	-
Pinus pungens	-	-	-	-	-	-	-	-	-	-	-	-
Pinus strobus	-	-	-	-	-	0.001	-	-	-	-	-	-
Pinus taeda	-	-	-	-	-	-	-	-	-	-	-	-
Pinus virginiana	-	-	-	-	-	-	-	-	-	0.009	0.002	0.007
Polemonium reptans	-	-	-	-	-	-	-	-	-	0.021	-	-

	1H	1L	1N	2H	2L	2N	3H	3L	3N	4H	4L	4N
Polygonatum biflorum	0.004	0.004	0.037	0.015	0.012	0.032	-	0.006	0.008	0.024	0.004	0.015
Polystichum acrostichoides	-	-	-	-	-	0.001	-	-	-	-	-	-
Potentilla simplex	-	-	-	0.002	-	0.003	-	-	-	-	-	-
Prenanthes alba	-	-	-	0.013	0.002	-	-	0.014	-	-	-	-
Prunella vulgaris	-	-	-	-	-	-	-	-	-	-	0.002	-
Prunus serotina	-	-	0.011	0.002	0.002	-	-	0.001	0.001	-	-	-
Pterdium aquilinum	0.006	-	-	-	-	-	-	-	-	-	-	-
Quercus alba	-	0.007	0.005	-	-	0.001	-	0.003	0.001	0.006	-	0.007
Quercus coccinea	0.001	0.004	0.068	-	-	-	-	0.018	-	-	0.004	-
Quercus falcata	-	-	-	-	-	-	-	-	-	-	-	-
Quercus ilicifolia	-	-	-	-	-	-	-	-	-	0.003	-	-
Quercus prinus	0.004	0.164	0.016	0.001	0.005	0.030	-	-	0.001	0.042	0.031	0.031
Quercus rubra	0.001	0.071	0.037	-	0.003	-	0.007	0.018	-	0.036	0.013	-
Quercus velutina	0.002	0.019	0.042	0.003	0.002	0.001	-	0.004	0.001	0.027	0.006	0.020
Ranunculus bulbosus	-	-	-	-	-	-	-	-	-	-	-	-
Ranunculus spp.	-	-	-	-	-	-	-	-	-	-	-	-
Rhus copallina	-	-	-	-	-	-	-	-	-	-	-	0.002
Robinia pseudoacacia	0.001	-	-	0.001	0.001	-	0.017	0.002	0.001	-	-	-
Rumex acetosella	-	-	-	-	0.001	0.008	-	-	-	-	-	-
Ruta graveolens	-	-	-	-	-	-	-	-	-	-	-	-
Sassafras albidum	0.117	0.491	0.121	0.016	0.006	0.007	0.009	0.011	0.006	0.418	0.09	0.017
Smilacina racemosa	-	-	-	0.001	0.008	-	-	-	0.001	-	-	-
Smilax herbacea	-	-	-	0.239	0.007	0.045	-	-	-	-	0.006	0.002
Solidago 2 spp.	-	-	-	-	-	-	-	-	-	-	-	-
Solidago spp.	-	-	-	-	0.003	-	-	0.016	0.001	-	0.008	0.011
Stellaria media	-	-	-	-	-	0.045	-	-	-	-	-	-
Taenidia integerrima	-	-	-	-	-	-	-	-	-	0.009	-	0.007
Taraxacum officinale	-	-	-	-	-	-	0.001	-	-	-	-	-
Thalictrum spp.	-	-	-	-	0.002	-	-	-	0.008	-	-	-
Tilia americana	-	-	-	0.001	0.001	0.011	-	-	-	-	-	-
Tovara virginiana	-	-	-	-	-	-	-	-	-	-	-	-
Trifolium arvense	-	-	-	-	-	-	-	-	-	-	-	0.002
Ulmus americana	-	0.007	0.000	-	0.011	-	-	-	-	-	-	-
Uniola latifolia	-	-	-	-	-	-	-	-	-	-	-	0.216
Unknown 18	-	-	-	-	-	0.003	-	-	-	-	-	-
Unknown 19	-	-	-	-	-	0.002	-	-	0.013	-	-	-
Unknown 52	-	-	-	-	-	0.005	-	-	-	-	-	-
Unknown 99	-	-	-	-	-	-	-	0.002	-	-	-	-
Unknown cotyledons	0.001	-	-	-	-	-	-	-	-	-	0.006	-
Urtica diocia	-	-	-	0.006	-	-	-	-	-	-	-	-
Uvalaria perfoliata	-	-	-	0.004	-	-	-	-	-	-	-	-
*Verbascum thaspus**	-	-	-	-	-	-	-	-	-	-	-	0.002
Vicia caroliniana	-	-	-	-	-	-	-	-	-	-	0.025	-
Vicia spp.	-	-	-	-	-	-	-	-	-	-	0.004	-
Viola 2 spp.	-	-	-	-	-	-	-	-	-	-	-	0.009
Viola pubescens	-	-	-	0.005	-	0.026	-	-	-	-	-	-
Viola spp.	-	-	-	0.257	0.366	0.059	0.065	0.013	0.428	-	0.029	0.050

	1H	1L	1N	2H	2L	2N	3H	3L	3N	4H	4L	4N
Viola triloba	-	-	-	-	0.003	0.017	-	-	-	-	-	-

Table 4. Relative frequency of individuals in the herbaceous strata plots by species for each of the study sites. Each growing seasons since burn and intensity combination are represented. H = high, L = low, N = none.

	5H	5L	5N	6H	6L	6N	7H	7L	7N	8H	8L	8N
Acer pensylvanicum	-	-	0.004	-	-	-	-	-	0.001	-	0.010	0.004
Acer rubrum	-	0.356	0.314	0.338	0.093	0.100	0.003	0.043	0.002	0.015	0.030	0.165
Achillea millefolium	-	-	-	-	0.010	-	-	-	-	-	-	-
Agrostis spp.	-	-	-	-	-	-	-	-	-	-	-	-
*Ailanthus altissima**	-	-	-	-	-	-	-	-	-	-	-	-
*Alliaria officinalis**	-	-	-	-	0.005	-	-	0.004	-	-	-	-
Amelanchier arborea	-	0.003	-	0.023	0.002	0.021	-	0.001	-	-	0.010	-
Amphicarpa bracteata	-	-	-	-	-	-	-	-	-	-	-	-
Anemone quinquefolia	-	-	-	-	-	-	-	-	-	-	-	-
Antennaria plantaginiflolia	-	-	-	-	-	-	-	-	-	-	-	-
Arisaema triphyllum	-	-	-	-	-	-	-	-	0.011	-	-	-
Aristolochia macrophylla	-	-	-	-	-	-	-	-	-	-	-	-
Asplenium platyneuron	-	-	-	-	-	-	-	-	-	-	-	-
Aster 2 spp.	-	-	-	-	-	-	-	-	-	-	-	-
Aster divaricatus	-	-	-	0.060	0.170	-	0.089	0.033	0.002	-	-	-
Aster spp.	0.003	-	-	0.015	-	-	-	0.001	-	-	-	-
Athyrium angustum	-	-	-	-	-	-	-	-	-	-	-	-
Betula lenta	-	-	-	0.165	0.022	0.012	0.005	0.001	0.007	-	-	-
Boehmeria cylindica	-	-	-	-	-	-	-	-	-	-	-	-
Botrychium dissectum	-	-	-	-	-	-	-	-	-	-	-	-
Botrychium matricariaefolium	-	-	-	-	-	-	-	-	-	-	-	-
Botrychium virginianum	-	-	-	-	-	-	-	-	-	-	-	-
Campanula rotundifolia	-	-	-	-	-	-	0.016	0.001	-	-	-	-
Carex spp.	0.044	0.015	0.013	-	-	-	0.137	0.037	0.015	-	-	-
Carya glabra	-	0.045	0.054	0.007	0.012	0.003	0.003	0.001	-	-	-	0.016
Carya tomentosa	-	0.021	0.009	-	-	-	-	-	-	-	-	-
Castanea dentata	-	0.006	-	-	-	0.009	0.003	-	-	-	-	-
Centaurea cyanus	-	-	-	-	-	-	-	-	-	-	-	-
Cercis canadensis	-	-	0.009	-	-	-	-	-	-	-	-	-
Chimaphila maculata	-	-	-	-	0.002	0.497	-	-	-	-	0.335	0.094
Chimaphila umbellata	-	0.006	-	-	-	-	-	-	-	-	-	-
Cimicifuga racemosa	-	-	0.004	-	-	-	-	-	-	-	-	-
Convallaria montana	-	-	-	-	-	-	-	-	-	-	-	-
Cornus florida	-	-	0.004	-	0.010	-	-	-	-	-	-	-
Dactylis glomerata	0.320	-	-	-	-	-	-	-	-	-	-	-
Danthonia spicata	-	-	-	-	-	-	-	-	0.219	-	-	-
Dennstaedtia punctilobula	-	-	-	-	0.012	0.015	0.026	0.021	0.113	-	-	-
Dentaria heterophylla	-	-	0.013	-	-	-	-	-	-	-	-	-
Desmodium rotundifolium	-	0.047	0.233	-	-	-	-	0.001	-	-	-	-
Dioscorea villosa	-	-	-	0.105	0.005	0.003	0.008	-	-	-	0.041	0.024
Dryopteris campyloptera	-	-	0.004	-	-	-	-	0.010	-	-	0.046	-
Epigaea repens	-	0.036	-	-	-	0.053	-	-	-	-	-	-
Epilobium angustifolium	0.083	-	-	-	-	-	-	-	-	-	-	-

	5H	5L	5N	6H	6L	6N	7H	7L	7N	8H	8L	8N
Euonymus americanus	-	-	-	-	-	-	-	-	-	-	-	-
Eupatorium rugosum	-	-	-	0.038	0.012	-	-	0.013	-	-	-	-
Fragaria vesca	-	-	-	-	-	-	-	0.004	0.016	-	-	-
Fraxinus americana	-	-	-	-	-	-	-	0.005	-	-	-	-
Galium lanceolatum	-	-	-	-	-	-	-	-	-	-	-	-
*Galium mollugo**	-	-	-	-	-	-	-	-	-	-	-	-
Galium spp.	-	0.098	0.009	-	0.012	-	0.016	0.009	0.006	-	-	-
Geranium spp.	-	-	-	-	-	-	-	-	-	-	-	-
Gerardia pedicularia	-	-	-	-	-	-	-	-	-	-	-	-
Gnaphalium obtusifolium	0.001	-	-	-	-	-	-	-	-	-	-	-
Hamamelis virginiana	-	0.018	-	0.007	-	0.050	-	-	-	-	-	-
Helianthus divaricatus	-	-	-	-	0.030	-	-	-	-	-	-	-
Helianthus microcephelus	-	-	-	-	0.037	-	-	-	-	-	-	-
Heuchera americana	-	-	-	-	0.047	-	-	-	-	-	-	-
Hieracium paniculatum	-	-	-	-	-	-	-	-	-	-	-	-
Hieracium venosum	-	-	-	-	-	-	-	-	-	-	-	-
Houstonia longifolia	-	-	-	-	-	-	0.058	0.005	-	-	-	-
Houstonia tenuifoila	-	-	-	-	-	-	-	-	-	-	-	-
Hypericum perforatum	-	-	-	-	-	-	-	0.001	-	-	-	-
Ilex verticulata	-	-	-	-	-	-	-	-	0.002	-	-	-
Impatiens capensis	-	-	-	-	-	-	-	-	-	-	-	-
Iris cristata	0.036	-	-	-	-	-	-	-	-	-	-	-
Lactuca canadensis	-	-	-	-	0.015	-	-	-	0.003	-	-	-
Liriodendron tulipifera	0.004	0.036	0.031	-	-	-	0.005	0.006	-	-	-	-
Lycopodium spp.	-	-	-	-	-	-	-	-	-	0.250	-	-
Lysimachia quadrifolia	-	-	0.022	-	0.044	-	0.074	0.014	-	-	-	-
Malus pumila	-	-	-	-	-	-	-	-	-	-	-	-
Medeola virginiana	-	-	-	-	-	-	-	0.104	0.010	-	-	0.004
Meehania cordata	-	-	-	-	-	-	-	-	-	-	-	-
Mentha spp.	-	-	-	-	-	-	-	-	-	-	-	-
Mimulus ringens	-	-	-	-	0.015	-	0.029	-	0.002	-	-	0.004
Nyssa sylvatica	0.020	0.038	0.004	-	-	0.012	0.005	0.001	-	0.030	0.020	0.039
Osmorhiza clatonii	-	-	-	-	-	-	-	0.008	0.001	-	-	-
Osmundaceae cinnamomea	-	-	-	0.023	0.081	-	-	-	0.044	-	-	0.024
Panax quinquefolius	-	-	-	0.015	-	-	-	-	-	-	-	-
Panicum boscii	0.108	0.003	-	-	0.054	-	-	-	-	-	-	-
Panicum clandestinum	-	-	-	-	0.123	-	-	0.047	-	-	-	-
Pastinaca sativa	-	-	-	-	-	-	-	-	-	-	-	-
Pedicularis canadensis	-	-	-	-	-	-	0.039	-	0.001	-	-	-
Penstemon digitalis	0.008	-	-	-	-	-	-	-	-	-	-	-
Phaseolus polystachios	-	-	-	-	-	-	-	-	-	-	-	-
Phytolaccaceae americana	-	-	-	-	-	-	-	-	-	-	-	-
Pinus pungens	0.009	-	-	-	-	-	-	-	-	-	-	-
Pinus strobus	-	-	-	-	-	-	-	-	-	-	-	-
Pinus taeda	0.005	-	-	-	-	-	-	-	-	-	-	-
Pinus virginiana	0.001	-	-	-	-	-	-	-	-	-	-	-
Polemonium reptans	-	-	-	-	-	-	-	-	-	-	-	-

	5H	5L	5N	6H	6L	6N	7H	7L	7N	8H	8L	8N
Polygonatum biflorum	0.005	0.006	-	-	-	-	-	0.001	-	-	-	0.004
Polystichum acrostichoides	-	-	0.009	-	-	-	-	-	-	-	-	-
Potentilla simplex	-	-	-	-	0.116	-	0.013	0.360	0.021	-	-	-
Prenanthes alba	-	-	-	-	-	-	-	-	0.004	-	-	-
Prunella vulgaris	-	-	-	-	-	-	-	-	-	-	-	-
Prunus serotina	-	0.003	0.013	-	-	-	0.003	0.015	0.005	-	-	-
Pterdium aquilinum	0.208	-	-	-	-	0.021	-	-	-	0.118	-	-
Quercus alba	-	0.003	-	-	-	-	-	0.006	-	-	-	0.004
Quercus coccinea	-	-	-	-	-	-	-	-	-	0.191	-	0.027
Quercus falcata	-	-	-	-	-	-	-	-	-	0.117	-	-
Quercus ilicifolia	0.007	-	-	-	-	-	-	-	-	-	-	-
Quercus prinus	0.001	0.107	0.058	0.007	0.025	0.071	0.150	0.004	-	0.059	0.452	0.271
Quercus rubra	0.001	0.021	-	0.030	0.005	0.012	0.016	0.013	-	-	0.005	0.031
Quercus velutina	-	0.042	0.049	0.007	0.002	0.050	-	-	-	0.015	0.030	-
Ranunculus bulbosus	-	-	-	-	-	-	-	0.001	-	-	-	-
Ranunculus spp.	-	-	-	-	-	-	-	0.001	-	-	-	-
Rhus copallina	-	-	-	-	-	-	-	-	-	-	-	-
Robinia pseudoacacia	-	-	-	-	-	-	-	0.001	-	-	-	-
Rumex acetosella	-	-	-	-	-	-	-	0.023	0.014	-	-	-
Ruta graveolens	-	-	-	-	-	-	-	-	-	-	-	-
Sassafras albidum	0.097	0.08	0.108	0.151	0.005	0.068	0.089	0.015	-	0.191	-	0.282
Smilacina racemosa	-	0.006	0.004	-	-	-	-	-	-	-	0.020	0.008
Smilax herbacea	-	-	-	-	-	-	-	-	-	-	-	-
Solidago 2 spp.	-	-	-	-	-	-	0.134	0.003	-	0.015	-	-
Solidago spp.	0.037	-	-	-	0.034	-	0.047	0.038	-	-	-	-
Stellaria media	-	-	-	-	-	-	-	-	-	-	-	-
Taenidia integerrima	-	-	-	-	-	-	-	-	-	-	-	-
Taraxacum officinale	-	-	-	-	-	-	-	-	-	-	-	-
Thalictrum spp.	-	-	-	-	-	-	-	0.003	-	-	-	-
Tilia americana	-	-	-	-	-	-	-	-	-	-	-	-
Tovara virginiana	-	-	-	-	-	-	-	-	0.446	-	-	-
Trifolium arvense	-	-	-	-	-	-	-	-	-	-	-	-
Ulmus americana	-	-	-	-	-	-	-	-	-	-	-	-
Uniola latifolia	-	-	-	-	-	-	-	-	-	-	-	-
Unknown 18	-	-	-	-	-	-	-	-	-	-	-	-
Unknown 19	-	-	-	-	-	-	-	-	-	-	-	-
Unknown 52	-	-	-	-	-	-	-	-	-	-	-	-
Unknown 99	-	-	-	-	-	-	-	-	-	-	-	-
Unknown cotyledons	-	-	-	0.007	-	-	-	-	-	-	-	-
Urtica diocia	-	-	-	-	-	-	-	-	0.007	-	-	-
Uvalaria perfoliata	-	-	-	-	0.002	0.006	-	0.009	-	-	-	-
*Verbascum thaspus**	-	-	-	-	-	-	-	-	-	-	-	-
Vicia caroliniana	-	-	-	-	0.020	-	-	-	-	-	-	-
Vicia spp.	-	-	-	-	-	-	-	-	-	-	-	-
Viola 2 spp.	-	-	-	-	-	-	-	0.016	-	-	-	-
Viola pubescens	-	-	-	-	-	-	-	-	-	-	-	-
Viola spp.	-	-	0.004	-	-	-	0.032	0.098	0.046	-	-	-

	5H	5L	5N	6H	6L	6N	7H	7L	7N	8H	8L	8N
Viola triloba	-	-	-	-	-	-	-	0.042	-	-	-	-

Appendix C - Analyses

Table 1. ANOVA for testing interaction of age class and wildfire intensity on duff bulk density* (N = 24)

Source	df	Type III SS	Mean Square	F Value	p
age class	1	1.532	1.532	1.620	0.219
intensity	2	0.028	0.014	0.015	0.985
a*i interaction	2	1.563	0.782	0.827	0.453

* Natural Log of duff bulk density values were used.
Relative efficiency of blocking test showed blocking was not beneficial. Value = 0.93.

Table 2. ANOVA for testing interaction of age class and wildfire intensity on litter bulk density*
(N = 24).

Source	df	Type III SS	Mean Square	F Value	P
age class	1	4.108	4.108	4.268	0.060
intensity	2	0.535	0.267	0.278	0.761
a*i interaction	2	0.700	0.350	0.363	0.700

* Natural Log of litter bulk density values were used.
Relative efficiency of blocking test showed blocking was not beneficial. Value = 1.01.

Table 3. ANOVA for testing interaction of age class and wildfire intensity on specific gravity of the 1-hr fuel size class (N = 24).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.045	0.045	3.636	0.073
intensity	2	0.027	0.014	1.093	0.356
a*i interaction	2	0.008	0.004	0.328	0.725

Relative efficiency of blocking test showed blocking was not beneficial. Value = 1.15.

Table 4. ANOVA for testing interaction of age class and wildfire intensity on specific gravity of the 10-hr fuel size class* (N = 24).

Source	df	Type III SS	Mean Square	F Value	P
age class	1	0.006	0.006	0.085	0.775
intensity	2	0.074	0.037	0.512	0.608
a*i interaction	2	0.269	0.134	1.85	0.188

* Natural Log of specific gravity of the 10-hr fuel size class values were used.
Relative efficiency of blocking test showed blocking was beneficial. Value = 1.37.

Table 5. ANOVA for testing interaction of age class and wildfire intensity on specific gravity of the 100-hr fuel size class (N = 18).

Source	Df	Type III SS	Mean Square	F Value	P
age class	1	0.004	0.004	0.096	0.763
intensity	2	0.155	0.078	1.919	0.193
a*i interaction	2	0.078	0.039	0.97	0.409

Relative efficiency of blocking test showed blocking was not beneficial. Value = incomplete.

Table 6. ANOVA for testing interaction of age class and wildfire intensity on specific gravity of the 1000-hr sound fuel size class* (N = 17).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.001	0.001	0.028	0.870
intensity	2	0.160	0.080	1.609	0.244
a*i interaction	2	0.145	0.073	1.457	0.275

* Natural Log of specific gravity for the 1000-hr sound fuel size class values were used.
Relative efficiency of blocking test showed blocking was not beneficial. Value = incomplete.

Table 7. ANOVA for testing interaction of age class and wildfire intensity on specific gravity of the 1000-hr rotten fuel size class (N = 13).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.007	0.007	0.607	0.462
intensity	2	0.045	0.023	2.014	0.204
a*i interaction	2	0.002	0.001	0.093	0.912

Relative efficiency of blocking test showed blocking was not beneficial. Value = incomplete.

Table 8. ANOVA for testing interaction of age class and wildfire intensity on quadratic mean diameter of the 1-hr fuel size class (N = 24).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.00047	0.00047	0.946	0.344
intensity	2	0.00002	0.00001	0.022	0.978
a*i interaction	2	0.00072	0.00036	0.730	0.496

Relative efficiency of blocking test showed blocking was not beneficial. Value = 1.10.

Table 9. ANOVA for testing interaction of age class and wildfire intensity on quadratic mean diameter of the 10-hr fuel size class (N = 24).

Source	df	Type III SS	Mean Square	F Value	P
age class	1	0.037	0.037	2.898	0.107
intensity	2	0.028	0.014	1.104	0.354
a*i interaction	2	0.058	0.029	2.237	0.134

Relative efficiency of blocking test showed blocking was not beneficial. Value = 1.04.

Table 10. ANOVA for testing interaction of age class and wildfire intensity on quadratic mean diameter of the 100-hr fuel size class (N = 18).

Source	df	Type III SS	Mean Square	F Value	P
age class	1	0.150	0.150	0.728	0.412
intensity	2	0.530	0.265	1.284	0.315
a*i interaction	2	0.240	0.120	0.581	0.576

Relative efficiency of blocking test showed blocking was not beneficial. Value = incomplete.

Table 11. ANOVA for testing interaction of age class and wildfire intensity on species richness* for burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.079	0.079	0.570	0.451
intensity	2	2.138	1.069	7.740	0.001
a*i interaction	2	0.157	0.079	0.569	0.567

* Natural Log of species richness values were used.

Table 12. ANOVA for testing interaction of age class and wildfire on species richness* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.129	0.129	0.919	0.339
fire	1	0.506	0.506	3.615	0.059
a*f interaction	1	0.080	0.080	0.571	0.451

* Natural Log of species richness values were used.

Table 13. ANOVA for testing association of vector distance and invasive species presence on species richness* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	0.158	0.079	0.570	0.566
invasive presence	1	5.928	5.928	51.949	<0.001

Table 14. Regressions for testing change in expected relationship of species richness* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.0324	2.379	0.01	0.431
Low	72	1	0.0460	2.543	0.02	0.190
Control	72	1	0.0630	2.501	0.06	0.039

* Natural Log of species richness values were used.

Table 15. Regressions for testing change in expected relationship of species richness* with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.0580	2.351	0.08	0.019
Low	72	1	0.0515	2.910	0.04	0.099
Control	72	1	0.0028	2.722	0.00	0.913

* Natural Log of species richness values were used.

Table 16. ANOVA for testing interaction of age class and wildfire intensity on Shannon's Index of Community Diversity for the tree stratum for burned areas (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.467	0.467	1.683	0.196
fire	1	1.488	1.488	5.367	0.021
a*f interaction	1	0.391	0.391	1.412	0.236

Table 17. ANOVA for testing interaction of age class and wildfire on Shannon's Index of Community Diversity for the tree stratum (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.253	0.253	0.994	0.320
intensity	1	4.715	4.715	9.250	<0.001
a*i interaction	1	2.411	2.411	4.730	0.010

Table 18. ANOVA for testing association of vector distance and invasive species presence on Shannon's Index of Community Diversity for the tree stratum (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
vector distance	2	0.071	0.036	0.125	0.883
invasive presence	1	5.429	5.429	20.948	<0.001

Table 19. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity for the tree stratum with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.1320	0.128	0.07	0.026
Low	72	1	0.1470	0.377	0.14	0.001
Control	72	1	0.0904	0.648	0.06	0.042

Table 20. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity for the tree stratum with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.0854	0.403	0.08	0.019
Low	72	1	0.0305	0.806	0.01	0.470
Control	72	1	0.0251	0.835	0.01	0.495

Table 21. ANOVA for testing interaction of age class and fire intensity on Pielou's Evenness Index for the tree stratum for burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.032	0.032	0.288	0.592
intensity	1	0.725	0.725	3.263	0.040
a*i interaction	1	1.092	1.092	4.912	0.008

Table 22. ANOVA for testing interaction of age class and wildfire on Pielou's Evenness Index for the tree stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.000	0.000	0.000	0.987
fire	1	0.423	0.423	3.664	0.570
a*f interaction	1	0.239	0.239	2.006	0.152

Table 23. ANOVA for testing association of vector distance and invasive species presence on Pielou's Evenness Index for the tree stratum (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
vector distance	2	0.014	0.007	0.061	0.941
invasives presence	1	2.042	2.042	18.883	<0.001

Table 24. Regressions for testing change in expected relationship of Pielou's Evenness Index for the tree stratum with site quality using slope as a surrogate.

	N	df	Slope	B	r^2	p
High	72	1	0.102	0.213	0.06	0.037
Low	72	1	0.057	0.475	0.06	0.037
Control	72	1	0.038	0.609	0.04	0.085

Table 25. Regressions for testing change in expected relationship of Pielou's Evenness Index for the tree stratum with site quality using aspect as a surrogate.

	N	df	Slope	B	r^2	p
High	72	1	0.080	0.389	0.099	0.007
Low	72	1	0.023	0.600	0.012	0.359
Control	72	1	0.021	0.640	0.020	0.242

Table 26. ANOVA for testing interaction of age class and wildfire intensity on tree basal area for burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	4882.184	4882.184	2.808	0.096
intensity	1	70953.148	70953.148	40.808	<0.001
a*i interaction	1	11733.902	11733.902	6.749	0.010

Table 27. ANOVA for testing interaction of age class and wildfire on tree basal area (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	8896.412	8896.412	4.528	0.035
fire	1	37945.562	37945.560	19.312	<0.001
a*f interaction	1	186.018	186.018	0.095	0.759

Table 28. ANOVA for testing association of vector distance and invasive species presence on tree basal area (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	2380.088	1190.044	0.549	0.578
invasives presence	1	13918.534	13918.534	6.622	0.011

Table 29. Regressions for testing change in expected relationship of tree basal area with site quality using slope as a surrogate.

Slope	N	df	slope	B	r^2	p
High	72	1	12.018	14.609	0.078	0.018
Low	72	1	10.766	63.980	0.136	0.001
Control	72	1	7.393	85.283	0.063	0.033

Table 30. Regressions for testing change in expected relationship of tree basal area with site quality using aspect as a surrogate.

aspect	N	df	slope	B	r^2	p
High	72	1	14.336	22.676	0.295	0.001
Low	72	1	-1.463	109.495	0.003	0.640
Control	72	1	4.193	90.954	0.030	0.143

Table 31. Test for homogeneity of regression slope coefficients associated with wildfire intensity level (N = 216).

Source	df1, df2	Type III SS	Mean Square	F Value	p
slope	2, 210	354845.92	1689.74	500.16	<0.001

Table 32. Test for homogeneity of regression intercept coefficients associated with wildfire intensity level (N = 216).

Source	df1, df2	Type III SS	Mean Square	F Value	P
intercept	2, 210	56.18	0.27	322.517	<0.001

Table 33. ANOVA for testing interaction of age class and wildfire intensity on Shannon's Index of Community Diversity for the shrub stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	6.965	6.965	44.202	<0.001
intensity	1	0.018	0.018	0.111	0.739
a*i interaction	1	1.724	1.724	10.938	0.001

Table 34. ANOVA for testing interaction of age class and wildfire on Shannon's Index of Community Diversity for the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	4.029	4.029	23.877	<0.001
Fire	1	0.066	0.066	0.391	0.533
a*f interaction	1	1.082	1.082	6.411	0.012

Table 35. ANOVA for testing association of vector distance and invasive species presence on Shannon's Index of Community Diversity for the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	0.375	0.187	0.933	0.395
Invasives presence	1	2.200	2.200	11.494	0.001

Table 36. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.031	1.008	0.006	0.521
Low	72	1	-0.197	1.875	0.287	0.001
Control	72	1	-0.099	1.427	0.088	0.012

Table 37. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.051	0.993	0.043	0.082
Low	72	1	0.009	1.178	0.001	0.828
Control	72	1	0.012	1.041	0.002	0.705

Table 38. ANOVA for testing interaction of age class and wildfire intensity on Pielou's Evenness Index for the shrub stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.662	0.662	20.429	<0.001
intensity	1	0.012	0.012	0.380	0.539
a*i interaction	1	0.250	0.250	7.708	0.006

Table 39. ANOVA for testing interaction of age class and wildfire on Pielou's Evenness Index for the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.214	0.214	6.468	0.012
Fire	1	0.010	0.010	0.306	0.581
a*f interaction	1	0.227	0.227	6.852	0.009

Table 40. ANOVA for testing association of vector distance and invasive species presence on Pielou's Evenness Index for the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	0.057	0.029	0.797	0.452
Invasives presence	1	0.060	0.060	1.669	0.198

Table 41. Regressions for testing change in expected relationship of Pielou's Evenness Index with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.027	0.542	0.065	0.030
Low	72	1	0.009	0.559	0.004	0.599
Control	72	1	-0.006	0.614	0.003	0.666

Table 42. Regressions for testing change in expected relationship of Pielou's Evenness Index with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.029	0.503	0.028	0.158
Low	72	1	-0.082	0.895	0.283	0.001
Control	72	1	-0.022	0.659	0.024	0.197

Table 43. ANOVA for testing interaction of age class and wildfire intensity on percent cover of invasive species* in the shrub stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	P
age class	1	5.611	5.611	1.329	0.250
intensity	1	14.471	14.471	1.713	0.183
a*i interaction	1	33.528	33.528	3.969	0.020

* Natural Log of percent cover of invasive species values used.

Table 44. ANOVA for testing interaction of age class and wildfire on percent cover of invasive species* in the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	P
age class	1	16.912	16.912	4.002	0.047
Fire	1	7.150	7.150	1.692	0.195
a*f interaction	1	31.783	31.783	7.520	0.007

* Natural Log of percent cover of invasive species values used.

Table 45. ANOVA for testing association of vector distance on percent cover of invasive species* in the shrub stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	P
vector distance	2	6.262	3.131	0.714	0.491

* Natural Log of percent cover of invasive species values used.

Table 46. Regressions for testing change in expected relationship of percent cover of invasive species* in the shrub stratum with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	-0.009	1.160	0.015	0.305
Low	72	1	-0.026	1.303	0.017	0.275
Control	72	1	0.001	1.195	0.000	0.982

* Natural Log of percent cover of invasive species values used.

Table 47. Regressions for testing change in expected relationship of percent cover of invasive species* in the shrub stratum with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	-0.004	1.136	0.009	0.439
Low	72	1	0.010	1.168	0.003	0.630
Control	72	1	0.639	0.910	0.070	0.024

* Natural Log of percent cover of invasive species values used.

Table 48. ANOVA for testing interaction of age class and wildfire intensity on Shannon's Index of Community Diversity for the herbaceous stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.285	0.285	0.966	0.327
intensity	1	1.708	1.708	5.782	0.018
a*i interaction	1	0.064	0.064	0.218	0.642

Table 49. ANOVA for testing interaction of age class and wildfire on Shannon's Index of Community Diversity for the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.437	0.437	1.586	0.209
Fire	1	0.627	0.627	2.278	0.133
a*f interaction	1	0.002	0.002	0.007	0.933

Table 50. ANOVA for testing association of vector distance and invasive species presence on Shannon's Index of Community Diversity for the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	1.240	0.620	2.269	0.106
Invasives presence	1	6.437	6.437	25.993	<0.001

Table 51. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity for the herbaceous stratum with site quality using slope as a surrogate.

Slope	N	df	slope	B	r^2	p
High	72	1	0.047	1.036	0.008	0.447
Low	72	1	0.067	1.182	0.028	0.157
Control	72	1	0.106	1.081	0.081	0.001

Table 52. Regressions for testing change in expected relationship of Shannon's Index of Community Diversity for the herbaceous stratum with site quality using aspect as a surrogate.

aspect	N	df	slope	B	r^2	p
High	72	1	0.062	1.052	0.038	0.099
Low	72	1	0.065	1.676	0.034	0.123
Control	72	1	0.059	1.170	0.037	0.107

Table 53. ANOVA for testing interaction of age class and wildfire intensity on Pielou's Evenness Index for the herbaceous stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.034	0.034	0.682	0.410
intensity	1	0.041	0.041	0.807	0.371
a*i interaction	1	0.047	0.047	0.929	0.337

Table 54. ANOVA for testing interaction of age class and wildfire on Pielou's Evenness Index for the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.032	0.032	0.718	0.398
Fire	1	0.060	0.060	1.367	0.244
a*f interaction	1	0.001	0.001	0.029	0.865

Table 55. ANOVA for testing association of vector distance and invasive species presence on Pielou's Evenness Index for the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	0.180	0.090	2.064	0.129
Invasives presence	1	0.065	0.065	1.480	0.225

Table 56. Regressions for testing change in expected relationship of Pielou's Evenness Index for the herbaceous stratum with site quality using slope as a surrogate.

	N	df	slope	B	R^2	p
High	72	1	0.016	0.632	0.004	0.592
Low	72	1	0.009	0.758	0.005	0.574
Control	72	1	0.002	0.736	0.000	0.895

Table 57. Regressions for testing change in expected relationship of Pielou's Evenness Index for the herbaceous stratum with site quality using aspect as a surrogate.

	N	df	slope	B	R^2	p
High	72	1	0.008	0.670	0.003	0.643
Low	72	1	-0.004	0.742	0.001	0.755
Control	72	1	0.035	0.587	0.088	0.011

Table 58. ANOVA for testing interaction of age class and wildfire intensity on invasive species percent cover* in the herbaceous stratum in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.574	0.574	8.289	0.005
intensity	1	0.000	0.000	0.006	0.938
a*i interaction	1	0.031	0.031	0.444	0.506

* Natural Log of invasive species percent cover in the herbaceous stratum were used.

Table 59. ANOVA for testing interaction of age class and wildfire on invasive species percent cover* in the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	P
age class	1	0.117	0.117	1.550	0.214
Fire	1	0.010	0.010	0.132	0.717
a*f interaction	1	0.284	0.284	3.777	0.053

* Natural Log of invasive species percent cover in the herbaceous stratum were used.

Table 60. ANOVA for testing association of vector distance on invasive species percent cover* in the herbaceous stratum (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
vector distance	2	4.438	2.219	0.493	0.612

* Natural Log of invasive species percent cover in the herbaceous stratum were used.

Table 61. Regressions for testing change in expected relationship of invasive species percent cover* for the herbaceous stratum with site quality using slope as a surrogate.

	N	df	slope	B	R^2	p
High	72	1	-0.015	1.231	0.003	0.636
Low	72	1	0.019	1.101	0.010	0.409
Control	72	1	0.039	1.058	0.027	0.165

* Natural Log of invasive species percent cover in the herbaceous stratum were used.

Table 62. Regressions for testing change in expected relationship of invasive species percent cover for the herbaceous stratum with site quality using aspect as a surrogate.

	N	df	slope	B	R^2	p
High	72	1	0.066	1.008	0.174	<0.001
Low	72	1	0.007	1.148	0.001	0.757
Control	72	1	0.017	1.266	0.008	0.461

* Natural Log of invasive species percent cover in the herbaceous stratum were used.

Table 63. Chi-Square for testing association of age class, wildfire intensity, and vector distance on invasive species presence in burned areas.

Source	Value	n	df	p
age class	1.06	216	1	0.303
intensity	2.12	216	2	0.346
vector distance	1.601	216	2	0.449

Table 64. Chi-Square for testing association of age class and wildfire on invasive species presence.

Source	Value	n	df	p
age class	1.06	216	1	0.303
Fire	0.043	216	1	0.835

Table 65. Chi-Square for testing association of slope and aspect on invasive species presence.

Source	Value	n	df	p
slope	8.372	216	4	0.079
aspect	16.601	216	5	0.064

Table 66. Chi-Square for testing association of proximity to roads or trails on invasive species presence.

Source	Value	n	df	p
road distance	1.366	216	1	0.243
roads/trails	1.932	216	1	0.165

Table 67. ANOVA for testing interaction of age class and wildfire intensity on duff depth* in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.014	0.014	0.556	0.457
intensity	1	0.087	0.087	1.714	0.183
a*i interaction	1	0.052	0.052	1.017	0.363

* Natural Log of duff depth values were used.

Table 68. ANOVA for testing interaction of age class and wildfire on duff depth* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.077	0.077	0.186	0.667
Fire	1	0.007	0.007	0.016	0.899
a*f interaction	1	0.215	0.215	0.517	0.473

* Natural Log of duff depth values were used.

Table 69. ANOVA for testing association of invasive species presence and vector distance on duff depth* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.121	0.061	0.146	0.864
invasives presence	1	0.182	0.182	0.440	0.508

* Natural Log of duff depth values were used.

Table 70. ANOVA for testing association of invasive species presence and vector distance on 1-hr fuel loading (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.749	0.375	0.023	0.978
invasives presence	1	16.543	16.543	1.015	0.315

Table 71. ANOVA for testing association of invasive species presence and vector distance on 10-hr fuel loading (n = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.291	0.146	2.171	0.117
invasives presence	1	0.048	0.048	0.703	0.403

Table 72. Chi-Square for testing association of invasive species presence and vector distance on 100-hr fuel loading.

Source	N	df	value	p
Vector distance	216	16	18.048	0.321
invasives presence	216	8	9.196	0.326

Table 73. ANOVA for testing association of invasive species presence and vector distance on 1000-hr sound fuel loading* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.330	0.165	0.216	0.806
invasives presence	1	0.057	0.057	6.611	0.011

* Natural Log of 1000-hr sound fuel loading values were used.

Table 74. ANOVA for testing association of invasive species presence and vector distance on 1000-hr rotten fuel loading* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.172	0.086	0.234	0.792
invasives presence	1	0.015	0.015	1.189	0.277

* Natural Log of 1000-hr rotten fuel loading values were used.

Table 75. ANOVA for testing association of invasive species presence and vector distance on 1000-hr total fuel loading* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.012	0.006	0.309	0.734
invasives presence	1	0.000	0.000	0.004	0.949

* Natural Log of 1000-hr fuel loading values were used.

Table 76. ANOVA for testing association of invasive species presence and vector distance on total fuel loading* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Vector distance	2	0.017	0.008	0.352	0.704
invasives presence	1	0.000	0.000	0.002	0.960

* Natural Log of total fuel loading values were used.

Table 77. Regressions for testing change in expected relationship on duff depth with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.035	1.231	0.037	0.105
Low	72	1	-0.004	1.328	0.002	0.728
Control	72	1	-0.015	1.394	0.016	0.290

* Natural Log of duff depth values were used.

Table 78. Regressions for testing change in expected relationship on duff depth with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	-0.003	1.370	0.001	0.808
Low	72	1	0.001	1.310	0.000	0.931
Control	72	1	0.023	1.240	0.055	0.047

* Natural Log of duff depth values were used.

Table 79. ANOVA for testing association of age class and wildfire intensity on litter depth (N = 24).

Source	Df	Type III SS	Mean Square	F Value	p
age class	1	0.023	0.023	4.238	0.064
intensity	2	0.027	0.000	0.025	0.975
a*I interaction	2	0.002	0.001	0.173	0.843

Table 80. ANOVA for testing interaction of age class and wildfire intensity on 1-hr fuel loading* in burned areas (N = 144).

Source	Df	Type III SS	Mean Square	F Value	p
intensity	1	61.871	61.871	4.100	0.025
age class	1	77.984	77.984	5.167	0.045
a*i interaction	1	77.690	77.690	5.148	0.025

* Natural Log of 1-hr fuel loading values were used.

Table 81. ANOVA for testing interaction of age class and wildfire on 1-hr fuel loading* (N = 216).

Source	Df	Type III SS	Mean Square	F Value	p
Fire	1	0.529	0.529	1.259	0.263
age class	1	5.578	5.578	13.279	<0.001
f*a interaction	1	3.863	3.863	9.197	0.003

* Natural Log of 1-hr fuel loading values were used.

Table 82. Regressions for testing change in expected relationship on 1-hr fuel loading* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	-0.323	2.742	0.259	0.001
Low	72	1	0.095	0.944	0.040	0.092
Control	72	1	-0.042	1.451	0.006	0.517

* Natural Log of 1-hr fuel loading values were used.

Table 83. Regressions for testing change in expected relationship on 1-hr fuel loading* with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.019	1.486	0.002	0.679
Low	72	1	0.004	1.281	0.000	0.936
Control	72	1	0.003	1.299	0.000	0.960

* Natural Log of 1-hr fuel loading values were used.

Table 84. ANOVA for testing interaction of age class and wildfire intensity on 10-hr fuel loading* in burned areas (N = 144).

Source	df	Type III SS	Mean Square	F Value	p
intensity	1	1.410	1.410	0.757	0.386
age class	1	7.873	7.873	4.229	0.042
a*i interaction	1	26.755	26.755	14.372	<0.001

* Natural Log of 10-hr fuel loading values were used.

Table 85. ANOVA for testing interaction of age class and wildfire on 10-hr fuel loading* (N = 216).

Source	df	Type III SS	Mean Square	F Value	P
Fire	1	0.005	0.005	0.003	0.956
age class	1	1.529	1.529	0.942	0.333
f*a interaction	1	2.621	2.621	1.615	0.205

* Natural Log of 10-hr fuel loading values were used.

Table 86. Regressions for testing change in expected relationship on 10-hr fuel loading* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.001	1.525	0.000	0.959
Low	72	1	0.037	1.346	0.038	0.099
Control	72	1	0.046	1.355	0.047	0.067

* Natural Log of 10-hr fuel loading values were used.

Table 87. Regressions for testing change in expected relationship on 10-hr fuel loading* with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.005	1.518	0.001	0.782
Low	72	1	0.018	1.414	0.012	0.366
Control	72	1	0.020	1.417	0.014	0.329

* Natural Log of 10-hr fuel loading values were used.

Table 88. Chi-Square for testing association of age class and wildfire intensity on 100-hr fuel loading.

Source	n	df	value	p
age class	216	8	6.664	0.573
intensity	216	16	11.166	0.799

Table 89. Chi-Square for testing association of slope and aspect on 100-hr fuel loading.

Source	n	df	value	p
aspect	216	40	48.316	0.172
slope	216	32	30.693	0.533

Table 90. ANOVA for testing interaction of age class and wildfire intensity on 1000-hr sound fuel loading* in burned areas (N = 144).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.031	0.031	2.437	0.121
intensity	1	0.060	0.060	4.723	0.031
a*i interaction	1	0.026	0.026	2.045	0.155

* Natural Log of 1000-hr sound fuel loading values were used.

Table 91. ANOVA for testing interaction of age class and wildfire on 1000-hr sound fuel loading* (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	1.603	1.603	2.114	0.147
Fire	1	0.283	0.283	0.374	0.542
a*f interaction	1	0.548	0.548	0.723	0.396

* Natural Log of 1000-hr sound fuel loading values were used.

Table 92. Regressions for testing change in expected relationship on 1000-hr sound fuel loading* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.144	1.539	0.023	0.205
Low	72	1	0.068	1.358	0.017	0.280
Control	72	1	-0.055	1.948	0.007	0.485

* Natural Log of 1000-hr sound fuel loading values were used.

Table 93. Regressions for testing change in expected relationship on 1000-hr sound fuel loading* with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.274	1.373	0.220	<0.001
Low	72	1	0.125	1.131	0.071	0.023
Control	72	1	0.029	1.636	0.003	0.651

* Natural Log of 1000-hr sound fuel loading values were used.

Table 94. ANOVA for testing interaction of age class and wildfire intensity on 1000-hr rotten fuel loading* in burned areas (N = 144).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.040	0.040	2.253	0.136
intensity	1	0.146	0.146	8.218	0.005
a*i interaction	1	0.026	0.026	1.474	0.227

* Natural Log of 1000-hr rotten fuel loading values were used.

Table 95. ANOVA for testing interaction of age class and wildfire on 1000-hr rotten fuel loading* (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	4.040	4.040	11.822	0.001
Fire	1	0.013	0.013	0.038	0.845
a*f interaction	1	0.407	0.407	1.190	0.276

* Natural Log of 1000-hr rotten fuel loading values were used.

Table 96. Regressions for testing change in expected relationship on 1000-hr rotten fuel loading* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.040	1.613	0.004	0.580
Low	72	1	-0.006	1.502	0.000	0.883
Control	72	1	0.001	1.601	0.000	0.989

* Natural Log of 1000-hr rotten fuel loading values were used.

Table 97. Regressions for testing change in expected relationship on 1000-hr rotten fuel loading* with site quality using aspect as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.093	1.523	0.064	0.033
Low	72	1	0.009	1.444	0.001	0.806
Control	72	1	0.034	1.451	0.006	0.517

* Natural Log of 1000-hr rotten fuel loading values were used.

Table 98. ANOVA for testing interaction of age class and wildfire intensity on 1000-hr total fuel loading* in burned areas (N = 144).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	0.087	0.087	3.443	0.066
intensity	1	0.273	0.273	10.765	0.001
a*i interaction	1	0.067	0.067	2.642	0.106

* Natural Log of 1000-hr total fuel loading values were used.

Table 99. ANOVA for testing interaction of age class and wildfire on 1000-hr total fuel loading* (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	8.253	8.253	11.029	0.001
Fire	1	0.054	0.054	0.072	0.788
a*f interaction	1	0.196	0.196	0.261	0.610

* Natural Log of 1000-hr total fuel loading values were used.

Table 100. Regressions for testing change in expected relationship on 1000-hr total fuel loading* with site quality using slope as a surrogate.

	N	df	slope	B	r^2	p
High	72	1	0.126	1.985	0.020	0.238
Low	72	1	0.046	1.713	0.007	0.472
Control	72	1	-0.068	2.362	0.009	0.419

* Natural Log of 1000-hr total fuel loading values were used.

Table 101. Regressions for testing change in expected relationship on 1000-hr total fuel loading* with site quality using aspect as a surrogate.

	N	df	Slope	B	r^2	p
High	72	1	0.286	1.723	0.270	<0.001
Low	72	1	0.112	1.458	0.054	0.050
Control	72	1	0.046	1.931	0.006	0.503

* Natural Log of 1000-hr total fuel loading values were used.

Table 102. ANOVA for testing interaction of age class and wildfire intensity on total fuel loading* in burned areas (N = 144).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	462.107	462.017	1.377	0.243
intensity	1	4885.078	4885.078	14.562	<0.001
a*i interaction	1	2116.000	2116.000	6.308	0.013

* Natural Log of total fuel loading values were used.

Table 103. Regressions for testing change in expected relationship on total fuel loading* with site quality using slope as a surrogate.

	N	df	Slope	B	r^2	p
High	72	1	-0.030	3.005	0.002	0.734
Low	72	1	0.074	2.118	0.030	0.147
Control	72	1	-0.026	2.656	0.002	0.709

* Natural Log of total fuel loading values were used.

Table 104. Regressions for testing change in expected relationship on total fuel loading* with site quality using aspect as a surrogate.

	N	df	Slope	B	r^2	p
High	72	1	0.217	2.336	0.238	<0.001
Low	72	1	0.088	2.057	0.053	0.053
Control	72	1	0.051	2.340	0.012	0.367

* Natural Log of total fuel loading values were used.

Table 105. ANOVA for testing interaction of age class and wildfire on total fuel loading* (N = 216).

Source	df	Type III SS	Mean Square	F Value	p
age class	1	6.358	6.358	12.774	<0.001
Fire	1	0.262	0.262	0.527	0.469
a*f interaction	1	0.535	0.535	1.076	0.301

* Natural Log of total fuel loading values were used.

Vita

Jeff Michael Matthews was born on September 21, 1980, in Manassas, Virginia, to Michael and Janeen Matthews. He was raised with three sisters. His fathers' love of the outdoors and work for the U.S.D.A. Forest Service instilled in him a great appreciation for the forests and need to manage them. In 1997, he started a lawn service while attending Northern Virginia Community College. In 1998, he transferred to the College of Natural Resources at Virginia Tech. He spent summers working for the Virginia Department of Forestry and the U.S.D.A. Forest Service. He graduated in May 2002 with a Bachelor of Science degree in Forest Resource Management. While at Virginia Tech, he completed the Cooperative Education Program, by working for a year with MeadWestvaco and the U.S.D.A. Forest Service in Chillicothe, Ohio. This program sparked his interest in fire science and its use in forest management. After a year of graduate level courses at the University of Montana, he returned to Virginia Tech in May 2003, to pursue a Master of Science degree in Forestry. He successfully completed all degree requirements in December 2004. In January 2005, he began working for the U.S.D.A. Forest Service on the Oconee National Forest.