Wildland Fire in the Central Appalachian Mountains: Impacts on Above- and Belowground Resources

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ABSTRACT

Prescribed fire use in Virginia and West Virginia has increased over the past ten years as forest managers on public lands have increasingly used prescribed fire to meet management goals. These goals include hazardous fuel reduction, wildlife habitat restoration and management, and control of less desired vegetation. Research is needed to better understand the effects of wildland fire on forest ecosystems. In this study, we addressed wildland fire's effects on water, vegetation, and soil resources in the central Appalachian Region. Moreover, the long-term efficacy of various types of timber harvests on forest fuel reduction was analyzed.

Over fifty peer-reviewed articles were evaluated to characterize the effects of prescribed fire on physical, chemical, and biological water quality parameters throughout the eastern United States. It was determined that fires of low to moderate intensity and severity may cause short-term sediment and nutrient increases in nearby waterbodies, but these effects often dissipate within 2-3 years. Effects on biological organisms are more transient, frequently lasting from a few weeks to a few months. Regeneration following wildfires at three sites in Virginia and West Virginia varied due to fire behavior and time since fire. Preferred and undesired species responded differently at each site. Follow-up treatments and continued monitoring are needed to obtain desired vegetative compositions post-fire. Two dormant season prescribed fires on the Fishburn Forest near Blacksburg, Virginia were studied for mineral soil chemistry effects. Both treatment and time affected macronutrient levels, but no differences were present 6 and 14 months post-fire between burned and unburned locations. Forest fuels were quantified approximately 20 years following different silvicultural harvests on the George Washington-Jefferson National Forest. Fuels of different size classes responded differently to different harvests as fine fuels were reduced by the high-leave shelterwood treatment, and coarse woody fuels were reduced by the clearcut and low-leave shelterwood treatments. Overall, low intensity and low severity fires induce minimal, potentially negative changes in water and soil quality. In contrast, wildfires of high intensity and severity may potentially contribute to changes in species composition and forest floor properties. Furthermore, varying levels of overstory removal may reduce extreme wildfire risk for decades. The findings of this study reinforce the need for continued research and monitoring of both wildfire effects and prescribed fire use in the central Appalachian Region.

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GENERAL AUDIENCE ABSTRACT

It is well-documented that fire has occurred in forested ecosystems for millennia. In addition to natural ignitions, indigenous peoples used fire for various reasons, such as understory reduction, hunting, and crop cultivation. As European settlers arrived and advanced across North America, they continued to use fire as a tool to shape the landscape to fit their societal needs. The use of fire by humans in North America all but ceased in the early 20th century. Large fire events in the western United States motivated the newly created United States Forest Service to restrict fire from the landscape. The fire exclusion policy of the early 20th century had unintended consequences, such as increased fire risk due to fuel accumulation and a shift from fire-tolerant species, such as oaks and pines, to fire-intolerant species. More recently, the perception of wildland fire has been re-examined due to ecological and societal issues. Although federal and state agencies are burning more acres, the public's wariness towards wildland fire is prevalent.

As attitudes about wildland fire have changed, so have the research needs. Information regarding the effects of both wild and prescribed fires on forest ecosystems is needed throughout the United States, including the eastern United States, and more specifically, within the central and southern Appalachian Mountains. This dissertation discusses the effects of both wild and prescribed fires on various forests processes within these regions. In this dissertation, 1) the impacts of prescribed fire on water quality, 2) the responses of forest vegetation to wildfire, and 3) and the effects of prescribed fire on soil nutrients were investigated. Additionally, different timber harvests were studied to determine

their long-term effects on potentially hazardous fuel loads. The results indicated that water quality is generally not impacted by low intensity and severity prescribed fires in the eastern United States. It was determined that vegetation often responds vigorously to wildfires, and subsequent species composition varies based on factors such as fire severity, site conditions, time since fire, and overstory species composition. When examining soil nutrients for 14 months following prescribed fires, nutrient changes occurred in both unburned and burned locations. When fuel loads were compared between timber harvests of varying intensities, woody fuels were reduced in the long-term. This reduction may minimize potential wildfire behavior and effects.

While both wild and prescribed fires impact forest processes, they generally do so in different ways. This is mainly due to differences in fire behavior between these fire types. Effects of wildfires on water quality, soil chemistry, and vegetation tend to last longer than prescribed fire. Additionally, prescribed fire, when used in conjunction with other forest management activities, may reduce potentially negative wildfire impacts. Monitoring post-fire effects is critical to understanding the best way to use prescribed fire as a forest management tool.

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THE WILD GEESE
by Wendell Berry

Horseback on Sunday morning,
harvest over, we taste persimmon
and wild grape, sharp sweet
of summer's end. In time's maze
over fall fields, we name names
that went west from here,
names that rest on graves. We open
a persimmon seed to find the tree
that stands in promise,
pale, in the seed's marrow.
Geese appear high over us,
pass, and the sky closes. Abandon,
as in love or sleep, holds
them to their way, clear,
in the ancient faith: what we need
is here. And we pray, not
for new earth or heaven, but to be
quiet in heart, and in eye
clear. What we need is here.

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Chapter 1. Introduction

Wildland fire, which consists of both wild and prescribed fires, has impacted global forest ecosystems for millennia (Dey & Guyette, 2000, McEwan et al. 2007, Lafon et al. 2017, Klimaszewski-Patterson & Mensing 2020). Within the last 100-120 years, wildland fire regimes have been altered due to climate change and anthropogenic activities (Brose et al. 2001, Guyette et al. 2002, Keeley & Syphard 2016, Liu & Wimberly 2016). Historical evidence of wildland fire frequency in the eastern United States varies by region and forest type (Abrams 1992, Delcourt & Delcourt 1997, Cleland et al. 2004, Stambaugh et al. 2018). In the Ridge and Valley physiographic province, fire scar studies indicate a composite mean fire interval (MFI) ranging from 2.5-19.5 years (Schuler & McClain 2003, DeWeese 2007, Hoss et al. 2008, Aldrich et al. 2010, Hessl et al. 2011, Aldrich et al. 2014). These studies include data from the late 18th century to the early 21st century, thereby providing insight before and during the era of wildfire suppression.

More recently, the beneficial role of fire on forest processes has been acknowledged (Gleim et al. 2019, Pausas & Keeley 2019, Uzun et al. 2020). As attitudes towards wildland fire's role in forested ecosystems have changed, so have the research needs. A better understanding of wildland fire effects on forest resources such as water, vegetation, and soil, will aid researchers and forest managers as they promote appropriate occurrences of fire on the landscape.

Maintaining water quality during forest management activities is a goal of many forest managers. Forested watersheds improve water quality through sediment and nutrient filtration, flood regulation, and groundwater replenishment (Anderson et al. 1976, Jackson et al. 2004, Lockaby et al. 2011, Caldwell et al. 2014). Research investigating the effects of wildland fire on water quality has primarily been focused on western US wildfires, thereby leaving a knowledge gap regarding the effects of prescribed fire on physical, chemical, and biological water quality components in eastern US forests.

Regeneration response to wildland fire is of increasing importance to forest managers in the central Appalachian Mountains, with a special focus on oak (*Quercus sp.*) and upland pine (*Pinus rigida* and *Pinus pungens*) restoration (Welch et al. 2000, Marschall et al. 2016, Keyser et al. 2019). Physiological and morphological traits of these species suggest they are fire tolerant (Blankenship & Arthur 1999, Arthur et al. 2012). Prescribed fire research in the central Appalachian Mountains has attempted to quantify the response of oaks and upland pines to dormant-season prescribed fire (Waldrop & Brose 1999, Van Lear et al. 2000, Brose et al. 2013), but little is known regarding their response to growing-season wildfires.

Generalizing the response of mineral soil nutrients to prescribed fire presents a challenge due to the high variability of soils and fire behavior (Knoepp et al. 2018, Hiers et al. 2020). Soil nutrients influence above-ground productivity, such as forest growth and yield (Fox et al. 2007). Additional studies in the Central Appalachian Mountains examining the response of mineral soil nutrients are needed to develop an improved understanding of prescribed fire's effects on mineral soil nutrients.

In addition to the response of forest resources to wildland fire, determining the most effective forest fuel reduction technique is a common management question. Silvicultural treatments, such as mastication and chemical treatments, have been studied alone and in conjunction with prescribed fire to determine the most effective methods to mitigate extreme wildfire behavior (Schwilk et al. 2009, Waldrop et al. 2010). While these studies provide insight into effective forest fuel reduction practices, long-term studies that examine overstory tree removal commonly implemented in commercial timber harvests are lacking.

To address these regional knowledge gaps, this dissertation will address questions regarding wildland fire's effects on water, vegetation, and soil resources in the Ridge and

Valley Province of Virginia and the Central Appalachian Mountains. Additionally, the long-term fuel response to timber harvests of varying intensities will be examined so as to provide a comprehensive analysis of wildland fire's role in the Central Appalachian Mountains. Chapter 2 is a brief literature review of current research in relation to regeneration dynamics following wildfire, soil nutrient response to dormant-season prescribed fire, and fuels management. Chapter 3 summarizes current literature concerning the impact of prescribed fire on freshwater ecosystems in the eastern United States. Chapter 4 examines the response of woody regeneration to wildfires in three forests (including portions of the Monongahela National Forest in West Virginia, George Washington & Jefferson National Forest and Fishburn Forest in Virginia). Chapter 5 summarizes the effects of a single, dormant season prescribed burn on mineral soil properties up to one-year post-burn on the Fishburn Forest in Virginia. Chapter 6 examines the long-term response of woody fuels two decades after various overstory silvicultural treatments were implemented on the George Washington & Jefferson National Forest in Virginia. Finally, a synthesis of the effects noted in Chapters 3-6 is presented in Chapter 7.

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Chapter 2. Literature Review

2.1 Wildfire Regeneration Dynamics

The fire exclusion policy of the early 20th century had unforeseen consequences on Appalachian Mountain forests such as hazardous fuel accumulation and a shift in composition to more fire intolerant species (Nowacki & Abrams 2008, Waldrop et al. 2016). The failure to regenerate certain species, such as oaks (*Quercus sp*.) and Table Mountain pine (*Pinus pungens*), has led scientists and managers to use prescribed fire as a management tool to restore these species. The application of prescribed fire to promote oak and Table Mountain pine (TMP) regeneration has produced mixed results (Welch & Waldrop 2000, Brose et al. 2013, Keyser et al. 2019). Several reasons have been posited to explain these results, including season and frequency of burning, fire behavior, and the differences between wildfires versus prescribed fires. Generally, wildfires are of higher intensity and consume more duff than prescribed fires (Hahn et al. 2019). Also, wildfires may occur either in the spring or fall, whereas prescribed fires are conventionally ignited in the dormant season or early spring (Keyser et al. 2019). These differences have led researchers to question the effects of wildfire on regeneration dynamics. Although post-wildfire regeneration studies have been conducted in other regions, central Appalachian Mountain research is lacking.

Research from other regions highlights a mixture of wildfire-related tree regeneration impacts. A peatland wildfire at the Red Lake Wildlife Management Area in Minnesota provided an opportunity for Rowe et al. (2017) to compare tree and shrub regeneration following a wildfire and timber harvest. Two sites were examined: a black spruce bog (BSB) and a rich tamarack swamp (RTS). Fire severity was classified into low, moderate, and high classes using field observations of canopy tree mortality and peat consumption. In the BSB, disturbance type was significantly related to black spruce (*Picea mariana* Mill.), quaking aspen (*Populus tremuloides* Michx.), and tamarack (*Larix laricina* Du Roi.) regeneration. Black spruce seedling densities followed this trend: low severity fire plots (13.6 stems m^{-2}) > moderate severity fire plots (3.5 stems m^{-2}) > harvested plots (1.6 stems m^{-2}) > control (0.7 stems m^{-2}). Aspen seedling response showed similar trends: moderate severity fire plots (23.6 stems m^{-2}) > low severity fire plots (10.6 stems m^{-2}) > harvested plots (1.1 stems m^{-2}) > control plots (0 stems m^{-2}). In the RTS, tamarack seedlings showed significantly higher densities in harvested (2.1 stems m^{-2}) plots compared to all other treatments (low severity burn=0.2 stems m^{-2}, moderate severity burn=0.5 stems m^{-2}, high severity burn=0.0 stems m^{-2}, control=0.0 stems m^{-2}). Aspen seedlings showed a positive response to moderate severity fire (6.6 stems m^{-2}) as opposed to harvest or control treatments (0 stems m^{-2}). Historically, Minnesota peatland fires were rare events, but the authors suggested that fires in this ecosystem, such as site preparation burns, were becoming a common necessary practice to achieve management goals.

Hagan et al. (2015) investigated vegetative structure and density changes following two separate, severe wildfires at the Linville Gorge Wilderness Area in North Carolina. Vegetation was compared before and after the wildfire. The authors found that midstory and overstory mortality was substantial following wildfires. Significant results were found regarding oak and TMP regeneration. The number of oak seedlings increased following both wildfires (+143 stems ha^{-1} following first fire, +118 stems ha^{-1} following second fire). A comparison of pre-burn and post-burn TMP numbers revealed a significant increase on xeric sites following the first burn (+2,643 stems ha^{-1}). However, TMP seedlings decreased following the second wildfire, most likely as a result of direct fire mortality. Tree-of-heaven (*Ailanthus altissima* Mill.) was also found in wildfire affected areas.

Following a 65 ha "very intense wildfire," Gnehm & Hadley (2007) quantified shortleaf pine (*Pinus echinata*) density in a section of the Ozark Highlands in Arkansas. The authors compared densities between unburned, wildfire-affected, and prescribed burn units. Efforts were made to select reference sites with similar physical characteristics as the wildfire unit. Although the wildfire unit had more pine seedlings (142 stems ha^{-1}) in comparison to the prescribed burn unit (104 stems ha^{-1}) and unburned units (91 stems ha^{-1}), these differences were not statistically significant. Because salvage-logging was enacted following the fire, the creation of open overstory conditions and exposure of mineral soil may have led to an increase in shortleaf pine seed germination.

Certain trends emerged when examining research regarding the effects of wildfire on tree regeneration in the eastern US. First, regeneration responded positively to fire as an increase of stem density was found following fires of various intensity and severity. Second, site quality was essential to the regeneration response as xeric sites supported higher densities of pyrophytic regeneration compared to mesic sites. Finally, non-native species behave opportunistically and may inhabit sites that have been recently burned.

2.2 Soil nutrient response following prescribed fire

Soils are essential to the productivity of forest vegetation. At a primary level, soil acts as a medium to support root systems and anchor above ground vegetation (Neary et al. 2009). Secondarily, numerous chemical and physical properties such as cation exchange capacity (CEC), and texture, determine nutrient and moisture availability (Schoenholtz, et al. 2000). Landscape-scale variation in soil properties may affect the spatial composition of vegetative species (Stewart & Edwards 2006). Because of its importance, soil nutrient availability is a major consideration of forest managers.

Disturbances such as fire have been known to alter soil nutrient composition (Neary et al. 2005). Wildfires, often exhibit both high intensity and severity. As a result, this frequently leads to nutrient loss. Nutrient loss can occur directly through volatilization, or indirectly through increased runoff (Neary et al. 2009, Bladon et al. 2014, Bixby et al. 2015). Alternatively, prescribed fires are typically both low intensity and severity. Because of this, fluctuations in nutrient levels pre- and post-fire are short-term or non-existent (Coates et al. 2018). While many studies have investigated the effects of prescribed fire on mineral soil chemistry in the Piedmont and Coastal Plain, studies in the central Appalachian Mountains are lacking.

A comparison of long-term burning in the South Carolina coastal plain was conducted by Coates et al. (2019) to examine mineral soil characteristics. Burned and unburned portions of the Tom Yawkey Center were compared. Twenty burns were conducted from 1978-2015. From 2004-2015, within the burned plots, short-term fire frequency was as follows: (a) two compartments had been burned four times, (b) three compartments were burned six times, (c) three compartments were burned eight times, and (d) one compartment remained unburned during that time period. A suite of nutrients was compared: aluminum, boron, carbon, calcium, copper, iron, magnesium, manganese, nitrogen, phosphorous, potassium, zinc, and sulphur. Additionally, organic matter, pH, and CEC were compared. Soil properties were examined at two soil depths: 0-10 cm and 10-20 cm. At the 0-10 cm depth, iron, aluminum, manganese, sulphur and pH in burned units had significantly different values than in unburned compartments. Calcium, manganese, and pH had the highest values in compartments burned eight times, but only the values for calcium were significantly greater than unburned compartments. Although aluminum was highest at locations burned three times, there was only a significant difference from compartments burned six times. Sulphur and iron had highest values in the unburned compartments. However, only iron was significantly higher in all burn treatments. Calcium and manganese showed a weak, positive relationship with burn frequency ($r^2 \leq 0.23$). At the 10-20 cm depth, iron, potassium, manganese, phosphorus, and pH had significantly different values due to treatment. Iron and potassium were greatest in the unburned compartments. Phosphorus was greatest in the compartment burned six times, but was only significantly different from the compartment burned eight times. Manganese was highest in the compartment burned eight times, but only significantly different from compartment burned six times. Soil pH had the highest value in the compartment burned eight times, but was only significantly different from the compartments burned four times and six times. Soil pH also displayed a weak, positive relationship with fire frequency ($r^2 \leq 0.16$), while phosphorus showed a negative relationship with fire frequency. Iron and potassium were the only nutrients that showed a significant reduction due to fire. Based on these results, minimal differences in soil nutrient levels were found as a result of frequent prescribed fire use.

A comprehensive approach to studying the effects of prescribed fire was attempted by Taylor and Midgley (2018). This study was done in East Woods section of the Morton Arboretum in Illinois. This oak-dominated site is unique in that it has been under an annual prescribed fire regime beginning in 1985. The authors were interested in comparing soil properties such as pH and amount of organic matter along with total and available nutrients (carbon, nitrogen, and phosphorus) to an unburned control section of the forest. Additionally, differences in microbial biomass and C-, N-, and P- degrading enzyme abundance was quantified. Burned area concentrations were significantly higher for organic matter ($p=0.032$), total nitrogen ($p=0.007$) and carbon ($p=0.001$) compared to unburned areas. Even though there was more total carbon in the burned areas, the amount of available carbon was not significantly different in the burned areas compared to unburned control. However, available nitrogen was significantly greater in burned areas ($p=0.009$). The main contributors to the increase of nitrogen were nitrate (54% increase) and organic nitrogen (27% increase). The ratio of organic nitrogen and inorganic nitrogen was insignificant between burned and unburned plots (p=0.312). Changes in total available phosphorus was insignificant between burned and unburned plots (p=0.108). There was a significant increase in microbial nitrogen biomass (p=0.04). There was no significant difference in microbial phosphorus or carbon microbial biomass. In regards to the enzymes, there was no significant effect from burning in abundance of enzymes, but there was in terms of enzyme ratios with more C:N- enzyme ratio (p=0.044), and N-:P- enzyme ratio (p<0.001). A significant increase in pH due to burning was also found (p<0.001). The authors postulated that the increase in available nitrogen, helped create a positive feedback loop which increased the presence of nitrogen degrading enzymes and microbes.

A study was done at three sites (Alarka Branch, Robin Branch, and Roach Mill Branch) in western North Carolina to determine the effects prescribed fire had on forest floor mass, soil and soil solution properties (Knoepp et al. 2009). Specifically, the authors were interested in how forest floor mass, carbon, and nitrogen changed following treatments of low to moderate intensity prescribed fires. Soil and water solution concentrations of NO_3-N and NH_4-N were also tested for significant changes at various depths (0-5 cm, 5-15 cm, 30-60 cm, and >60 cm). Oi layer mass loss ranged from 82.5% to 91.1% and was greatest at the Roach Mill Branch site (91.1%). Oe+Oa layer mass loss ranged from 25.7% to 46.4% and was highest at the Roach Mill Branch site. There was a significant decrease in forest floor mass between burned and unburned plots (p<0.0001), but not between sites (p=0.2002). At the Alarka Laurel Branch site, the only significant difference was an increase in nitrogen (1.145% pre-burn, 1.504% post-burn) and carbon:nitrogen ratio (42.5 pre-burn, and 25.2 post-burn) in the Oi horizon. At the Robin Branch site, there was a significant gain (0.888% pre-burn, 1.362% post-burn) in nitrogen in the Oi horizon. At Roach Mill Branch, a significant loss in the carbon:nitrogen ratio (65.4 pre-burn, 51.53 post-burn) in the Oi layer was found. In terms of soil solutions for NO_3-N and NH_4-N, a trend was found across all sites in which immediately following the prescribed burns there was an increase of NO_3-N and NH_4-N. These pulses were short-lived however and returned to pre-burn levels by the following summer. Significant increases in NO_3-N solution concentrations were found at the 0-5 cm depth based on treatment (p=0.0540), and at the 30-60 cm depth based on site * time interaction (p=0.0065). Significant increases were also found for NH_4-N at the 0-5 cm depth based on the time * treatment interaction (p=0.0001), at the 30-60 cm depth based on time (p=0.0001), and site (p=0.0044). All other depths and interactions for NO_3-N and NH_4-N soil solution concentrations were found to be insignificant. Water solution concentrations for NO_3-N and NH_4-N after burning were found to be insignificant as well. These findings of high forest floor fuel consumption and ephemeral, pulses of nutrient concentrations following low to moderate severity prescribed fires were consistent with similar studies in the region.

Coates et al. (2008) contrasted the effects of prescribed fire, mechanical thinning, and combination treatments of prescribed fire and mechanical thinnings on total inorganic nitrogen, net nitrogen mineralization, net nitrogen mineralization per unit of soil organic carbon, net nitrification, and proportional nitrification. Two sites were studied as part of the National Fire and Fire Surrogate Study (FFS), one site being the Ohio Hills site in Ohio and the other being the Green River site in North Carolina. Sites were measured pre-treatment, one year following treatments, and again three or four years following treatments in North Carolina and Ohio, respectively. All significant differences were found one year following treatment. Between the two sites, only one significant change in nitrogen cycling was found at the Green River site. There was a significant increase (79%) in proportional nitrification in the mechanical+burn plots compared to the control, mechanical-only, and burn-only plots. The Ohio Hills site had significant findings in all parameters measured. There was a significant increase in total inorganic nitrogen in the mechanical plots compared to the control, mechanical+burn, and burn-only plots. Regarding net nitrogen mineralization, the burn-only showed a significant decrease when compared to the control (42% less) and mechanical-only treatments (71%). Both mechanical-only and mechanical+burn treatments displayed a significant increase in net nitrogen mineralization, while both the control and mechanical-only treatments showed a significant increase in net nitrification. Regarding proportional nitrification, mechanical-only and mechanical+burn plots were significantly lower than the control or burn-only plots. The deficiency of significant findings in North Carolina was attributed to the abundance of ericaceous shrubs such as mountain laurel (*Kalmia latifolia* L.) and rhododendron (*Rhododendron maximum* L.). The litter chemical composition of these species inhibits nitrogen mineralization (Waterman and Mole 1994, Coates et al. 2008). The Ohio Hills site lacked the ericaceous shrub layer and therefore, significant responses were found.

Burning frequency and seasonality were studied in the Lombard/Paradise Hollow Research Area of the Cape Cod National Seashore in Massachusetts (Neil et al. 2007). Frequent burning in this area is utilized to restore and maintain oak-pine forest conditions. The authors were interested in how spring (March/April) and summer (July/August) burns at different frequencies (annual burning, every two years, every three years, and every four years) affected soil bulk density, pH, and acidity. Total extractable cations, soil carbon, and nitrogen were also compared. Plots containing three replicates of spring, summer, and unburned treatments were sampled and analyzed. Samples were divided into two strata: organic layer and mineral layer. Analysis results showed that in the organic layer horizon thickness (decrease), bulk density (increase), pH (increase), and exchangeable acidity (decrease) were significantly altered by burning. In the mineral layer, only pH (increase) was significantly affected by burning. No significant differences were found concerning total extractable cations, soil carbon or soil nitrogen. The authors analyzed the data furthermore by computing a two-way ANOVA with frequency, season, and frequency * season interaction.

With this analysis, only organic layer thickness (p=0.0012) and bulk density (p=0.012) were significantly affected by burning season with summer burn plots being significantly different that control plots. Fire frequency only had an effect on bulk density in the organic layer ($p<0.0001$). The season * frequency interaction had an effect on bulk density (p=0.0007) and pH (p=0.005) in the organic layer, and on pH (p=0.0003) in the mineral layer. In both analyses, annual burning was the only frequency that led to significant changes.

The effects of the fell and burn technique on long-term soil chemistry were investigated on the Nantahala National Forest in North Carolina (Knoepp et al. 2004). Three sites (Jacob Branch East, Jacob Branch West, and Devil Den) were clearcut in the summer of 1990 with none of the felled stems removed from the sites. Burns were implemented in September 1990. Researchers were focused on quantifying the changes in soil nutrient availability and potential nutrient loss after a timber harvest and site preparation burn. Cation concentrations in the A (depth of 4 cm) and B (depth of 11 cm) horizons were quantified as well as total carbon and nitrogen percentages. Additionally, inorganic soil nitrogen, nitrogen transformations, soil solutions, and nitrate responses in an adjacent stream were measured. Post-burn measurements continued up to three years following the burns. In the A horizon, calcium showed a significant increase on four of the six post-burn measurement dates (range: ~100 mg kg^{-1} - ~175 mg kg^{-1}). Potassium (range: ~80 mg kg^{-1} - ~125 mg kg^{-1}) and magnesium (range: ~28 mg kg^{-}1 - ~32 mg kg^{-}1) showed a significant increase on three of the six post-burn dates. In the B horizon, calcium (range: ~12 mg kg^{-1} - ~37 mg kg^{-1}) was significantly greater on five of the six post-burn measurement dates and Potassium (range: ~75 mg kg^{-1} - ~905 mg kg^{-1}) was significantly greater on four of the six measurement dates with the four being immediately after the burns. There was no significant change in magnesium following the burns (range: ~10 mg kg^{-1} – 18 mg kg^{-1}). The burns had no significant effect on total carbon (range: 4.2% to 5.7%), and only an effect on total nitrogen (range: 0.12-0.18%) for one sampling date following the burns. Soil NH_4-N concentrations increased significantly following the burns and gradually decreased, however this decrease was not enough to be significantly different than the control area. Nine out of thirteen measurement dates had significantly higher concentrations of NH_4-N (range: 0.5-5 mg kg^{-1}). Nitrogen mineralization rates were more variable following the burns as five of thirteen measurement dates had a significant increase (range: 0-4 mg kg^{-1}). Soil solution analysis at 30 cm revealed only one post-burn measurement had a significant increase in potassium (range: 1-1.5 mg L^{-1}) and nitrate (range: 0.0-15 mg L^{-1}) while calcium (range: 0.2-0.38 mg L^{-1}) showed no significant changes compared to the unburned plots. Soil solution at 60 cm showed more significant changes as both calcium (range: 0.05-0.08 mg L^{-1}) and potassium (range: 0.9-1.25 mg L^{-1}) had increases in five of ten post-burn measurement dates and nitrate (range: 0-0.1 mg L^{-1}) showed increases in only four of ten post-burn dates. Stream solution of nitrate showed a seasonally predictable pattern of an increase immediately following the burns and in the succeeding winter months, followed by a decrease during the summer months.

Overall, the effects of prescribed fire on soil nutrients and properties are dependent upon variables such as aspect, slope, above ground species composition, and inherent soil properties. These variables are multiplied by the inconsistency in fire behavior during every prescribed burn. However, certain trends were exhibited among the studies in this review. First, pH consistently increased following fire, and when fire was applied multiple times regardless of season. Second, most changes in nutrient composition and mineralization rates occurred in the top 10 cm of the soil profile. The influx of nitrogen and carbon in the top 10 cm of soil is most likely due to the combustion of leaf litter and organic matter present on the forest floor at the time of ignition. Additionally, these changes were short-lived, in some cases only lasting two weeks. Changes to the mineral layer of the soil profile were uncommon and transient as well. This is most likely due to the fact that prescribed fires in the eastern hardwood forest region are low intensity, low severity surface fires. Currently, research regarding the effects of prescribed fire on soil chemistry are largely focused on pine ecosystems within the Piedmont and Coastal Plain. More studies are needed in the Appalachian Mountains to elucidate the response of nutrients in hardwood-dominated systems, particularly as a consequence of time since fire.

2.3 Efficacy of wildland fuel reduction treatments

The fire triangle consists of heat, oxygen and fuel (Neary et al. 1999). All three components are needed for fire to occur. Fuel influences fire behavior as the quantity and arrangement of fuels in forested settings affect fire intensity and severity (Cochrane & Ryan 2010). Spatially, contiguous surface fuels provide a means for fire to easily move throughout the stand, while a patchy arrangement often times limit the spread of fire. Furthermore, the presence of fuels arranged horizontally, often referred to as “ladder fuels”, can cause a fire to reach the canopy, thereby creating a crown fire.

Recent high intensity and severity wildfires in the southeastern United States can be partially attributed to the accumulation of hazardous fuels due to the fire suppression policy implemented in the early 20th century (Waldrop et al. 2016). As a result, fuels management and fire resilience have become major topics in the conversation regarding forest management policy. In addition to the use of prescribed fire in Appalachian hardwood stands with the intended purpose of promoting oak regeneration (Keyser et al. 2017), it has been suggested that prescribed fire may reduce hazardous fuel loads (Greenberg & Waldrop 2008). However, narrow burn windows and a lack of available personnel limit its use (Welch & Waldrop 2001) thereby leading to the use of other treatments, such as mechanical felling, as a possible, alternate, fuel reduction method (Stephens et al. 2012).

A study conducted by Vander Yacht et al. (2019) focused on fuel responses during woodland and savanna restoration at three different sites in North Carolina and Tennessee. The sites were located at the Catoosa Wildlife Management Area (CWMA; Tennessee), Green River Game Lands (GRGL; North Carolina), and the Land Between the Lakes National Recreation Area (LBL; Tennessee). Because the site characteristics varied greatly, each site was treated as an independent experiment. Numerous treatments were implemented including: thinning to woodland basal area (14 m^2 ha^{-1}) and spring prescribed burning (SpW), thinning to woodland basal area and fall prescribed burning (FaW), thinning to savanna basal area (7m^2 ha^{-1}) and spring prescribed burning (SpS), thinning to savanna basal area and fall prescribed burning. Additionally, spring-only and fall-only prescribed burning was implemented on GRGL and LBL. Fuels were classified into five categories: (a) litter/1-hour (0-0.6 cm), (b) 10-hour (0.6-2.5 cm), (c) 100-hour (2.5-7.6 cm), (d) 1000 hour (≥7.6 cm), and (e) herbaceous fuels which mainly consisted of C_4 grasses. As expected, conflicting results were found due to site variability, prescribed burning characteristics, and thinning operations. Most treatments led to total woody fuel increases except for the burn-only treatments. Also, thinning regimes consistently added 1000-hr fuels that were not consumed by the prescribed burns. One-hour fuels in the thinned+burned areas decreased on average by 62%, but herbaceous fuels increased. The authors stated that increases in grass cover and 1000-hr fuels was not surprising as both woodlands and grasslands are inherently more flammable and have more frequent fire return intervals than closed-canopy forests.

The response of fuels to the shelterwood-burn technique was examined by Brose (2016) on the Horsepen Wildlife Management Area (HWMA) in the Virginia Piedmont. In this study, three treatments were compared: (a) shelterwood-only, (b) shelterwood+burn combination, and (c) control. Fuels were measured pre-treatment in 1994, immediately post-treatment in 1995, and again in 1997. Fuels were classified into small (<1.0 in in diameter), medium (1-3 in in diameter) and large (>3 in in diameter) size classes. Pre-treatment data showed that the burn+shelterwood area had greater small sized woody fuels (5.1 tons ac^{-1}) followed by shelterwood-only (4.3 tons ac^{-1}) and control (4.1 tons ac^{-1}). The shelterwood-only area had more medium sized woody fuels pre-treatment when compared to the burn+shelterwood (6.8 tons ac^{-1}) and control (1.5 tons ac^{-1}). Large woody fuels were dominant in the burn+shelterwood area (11.1 tons ac^{-1}) followed by shelterwood-only (10.8 tons ac^{-1}) and control (4.9 tons ac^{-1}). One year following treatments, the control and shelterwood-only areas had 4.3 tons ac^{-1} while the burn+shelterwood had only 0.8 tons ac^{-1} of small woody fuels. The shelterwood-only area had 7.9 tons ac^{-1} of medium woody fuels while the burn-shelterwood had 2.1 tons ac^{-1} and the control had 1.5 tons ac^{-1}. The shelterwood-only area contained relatively more large woody fuels (12.2 tons ac^{-1}) while the shelterwood+burn and control less (6.1 tons ac^{-1} and 4.9 tons ac^{-1}, respectively). Similar trends were found in 1997 as in 1995. Small fuels were more prevalent in the control (4.3 tons ac^{-1}) and shelterwood-only (4.3 tons ac^{-1}), while the shelterwood-only contained more medium (7.8 tons ac^{-1}) and large (12.1 tons ac^{-1}) woody fuels than the burn+shelterwood or control. The shelterwood-only treatment had no significant effect on fuel loadings immediately or two years post-treatment. The burn+shelterwood treatment initially reduced all woody fuels, but fuel levels returned to pre-treatment levels within the sampling period.

Phillips and Waldrop (2013) examined the interaction between fuel reduction treatments and natural disturbances on the Green River Game Lands (GRGL) in North Carolina. Fuel reduction treatments included mechanical-only, mechanical and burn, and burn-only. Mechanical treatments at GRGL consisted of all stems >1.8 m tall and <10.2 cm in diameter at breast (dbh) being felled. Fuels were classified into two classes: (a) fine fuels, which consisted of 1-hr, 10-hr, and 100-hr, and (b) 1000-hr fuels. Following treatments, the authors observed that the mechanical treatments increased fine fuels while the burn-only treatment decreased fuels by almost half. An ice storm that occurred four years post-treatment caused a significant fine fuel increase in in the mechanical-only and control areas. Eight years post-treatment, the mechanical-only treated areas had the greatest fine fuel levels (23 tons ha^{-1}). Overstory mortality from the prescribed burning coupled with mortality from the ice storm led to an increase in 1000-hr fuels in all treatment areas. In addition to woody fuels, litter and duff weights were compared. Litter and duff weights were greater in the mechanical-only area than in the control area throughout the study except five years post-treatment. At that time, duff weight was virtually the same in the mechanical (19.8 mg ha^{-1}) and control (19.0 mg ha^{-1}) areas.

A study was conducted in the southern Appalachian Mountains by Waldrop et al. (2007). In this study, fuel loading was compared on disturbed and undisturbed sites in four states across three national forests (Nantahala National Forest in North Carolina, Chattahoochee National Forest in Georgia, Sumter National Forest in South Carolina, and one national park (Great Smoky Mountains National Park in Tennessee). Within each site a 26 km^2 area was sampled. Plots were stratified in each area by slope position and aspect. The following slope position-aspect combinations were created: Northeastern Upper, Northeastern Lower, Ridge, Southwestern Upper, Southwestern Lower. Disturbance history was also taken into consideration with and "disturbed" plots were identified as those being impacted by either a natural or anthropogenic disturbance within the previous ten years. Five disturbance classifications were used: undisturbed, wind, fire, harvest, and southern pine beetle (*Dendroctonus frontalis* [SPB]). Although the authors discuss the differences in wildfires and prescribed fires they did not differentiate between the two for analysis. Likewise, harvest types were not differentiated. When looking at fuel size classes, harvesting disturbances resulted in the greatest amount of litter fuels with 4.3 tons ha^{-1} followed by SPB outbreak (4.1 tons ha^{-1}), undisturbed (4.0 tons ha^{-1}), windthrow (3.9 tons ha^{-1}), and fire (3.5

tons ha^{-1}). Southern Pine Beetle damage resulted in the largest amount of 10-hr, and 100-hr fuels. Wind disturbance resulted in the greatest amount of 1000-hr fuels (15.5 tons ha^{-1}), followed by fire (13.7 tons ha^{-1}), wind (13.0 tons ha^{-1}), harvest (11.4 tons ha^{-1}), and undisturbed (10.7 tons ha^{-1}). The authors found that harvesting disturbances led to the greatest amount of litter and, 1-hr fuels on lower northeastern slopes. There was no difference in 10-hr fuels due to slope position or aspect. The authors suggested this study could help fire managers predict fuel information based on disturbance history and site characteristics.

A study by Graham and McCarthy (2006) was conducted in Ohio to determine the difference in fuel responses to thin-only, burn-only, and burn+thin combination. The thinning treatment was a thinning from below which retained approximately 14 m^2 ha^{-1} of basal area. Prescribed burns were conducted in late March and into early April of 2001. In addition to treatment effects, interaction effects with time were also analyzed. Overall, the thin-only treatment led to an increase in 1-hr, 10-hr, 100-hr, 1000-hr solid fuels, coarse woody debris (CWD), and litter. Rotten1000-hr fuels decreased due to treatment. Three years post thinning, only the increase in solid 1000-hr fuels and CWD density effects remained. One year following the burn-only treatment, there was an increase in duff, 1000-hr rotten fuels, and litter. Three years post-burn, all fuel types recovered to pre-treatment levels. The thin+burn combination treatment increased 1-hr, 10-hr, 100-hr, 1000-hr solid, CWD, and duff fuel types and a decrease in rotten 1000-hr fuel types after one year. After three years, only 1000-hr soft and CWD density fuels were significantly increased. The thin-only treatment was found to be significant for 1000-hr solid fuels, CWD mass and density. Additionally, the thinning * year interaction was found to be significant for 10-hr, 1000-hr solid, and CWD density.

The use of mechanical and fire treatments to reduce woody fuels has shown to have mixed results. These results are linked to the size class and time frame of the fuels being monitored. Prescribed fire alone tends to reduce surface fuels in the short term (~3 years), while mechanical treatments such as pre-commercial and commercial thinnings often increase woody fuel loadings, especially of the larger size classes (100-hr and 1000-hr). Other types of disturbance that mimic mechanical treatments such as pest outbreaks have similar effects on large fuels. Site productivity has little to no effect on fuel loads.

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Chapter 3. Prescribed fire effects on water quality and freshwater ecosystems in moist-temperate eastern North America

3.1 Introduction

Forests occupy approximately 31% (4 billion hectares) of Earth's land surface (Bladon et al. 2014). Filtration of water is one of the most important ecosystem services these forests provide (Brooks et al. 2012). Nearly two-thirds of municipalities in the United States and approximately one third of the world's largest cities obtain the majority of their consumable water from forested watersheds (Bladon et al. 2014). Globally, natural filtration services have been estimated to save approximately 4.1 trillion dollars annually in water treatment costs (Bladon et al. 2014). Increasingly, forest management practices and natural disturbances are monitored for potential impacts on forests and any subsequent impacts on water quality (Brooks et al. 2012).

The extreme wildfire events that occurred in the western United States in 2017 and the southern United States in 2016 (National Interagency Fire Center 2018) provide clear and dramatic examples of the tremendous hazards of wildfires to forest resources, human property, and lives. Wildfires have the potential to cause problems in watersheds due to the widespread consumption of soil organic matter (Neary et al. 2009, Pereira et al. 2012, Bladon et al. 2014, Bixby et al. 2015), which exposes mineral soils to erosive precipitation following fire (Brooks et al. 2012). Wildfires may also cause stem mortality, destabilize roots (Callaham et al. 2012), and favor the spread and growth of invasive species (Martin & Hamman 2016).

Long-term fire exclusion has resulted in hazardous fuel accumulation in many forests throughout the United States (Keifer et al. 2006). Therefore, forest managers commonly need feasible fuel reduction strategies. In some situations, prescribed fire is a viable tool for reducing hazardous fuel loads (Waldrop and Goodrick 2012). Additionally, prescribed fire may be used to improve wildlife habitat, reduce undesired species competition, combat invasive species, enhance specific environmental attributes, and prepare a stand for future forest management (Brender & Cooper 1968, Whelan 1997, Stanturf et al. 2000, Fairchilds & Trettin 2006, Waldrop & Goodrick 2012). In 2017, over 2 million hectares were managed with prescribed fire in the eastern United States. The area burned by prescribed fire in this region has grown steadily in recent years (Figure 3.1; Melvin 2015).

The scientific literature documents high risk posed by wildfires to water quality and the viability of prescribed fire as a tool for minimizing wildfire occurrence and impact. Despite the common and widespread implementation of prescribed fire as a land management tool in the eastern United States, little research has been conducted to understand the potential impacts of prescribed fire on water quality and freshwater ecosystems (Lafayette et al. 2012). In this review we compile evidence and summarize known effects of prescribed fire on water resources in the eastern United States, identify urgent research needs, and explore implications for policy and land stewardship practice.

Prescribed fires differ from wildfires in many ways. The primary differences often center on the distinction between fire intensity and fire severity. While the two may correlate, they often do not. Fire intensity refers to the total energetic output of a fire (Keeley 2009). A fire of high intensity may or may not be high in severity and vice versa, depending on local factors such as the type of vegetation burned—whether dominated by fire-tolerant or fire-sensitive species, concomitant fire behavior, and moisture levels in soil organic matter.

Fire severity refers to the ecosystem effects of fire, for example the degree to which forest soil organic resources are consumed and vegetation is killed in a given fire event (Whelan 1997, Keeley 2009). Fire severity is the result of the interaction between fire intensity and the burned environment. When intact and decomposed plant litter in the soil (known variously as soil organic horizons, duff, and litter) is fully consumed, mineral soil is exposed to heating and subsequent precipitation (Callaham et al. 2012). Following high-severity fires, crusting of mineral soil can occur as the result of heating; this condition is known as hydrophobicity (Brooks et al. 2013). Hydrophobic soils reduce infiltration and increase runoff volume and energy, which can accelerate soil erosion, particularly following large precipitation events (Brooks et al. 2013). Fire-induced vegetative mortality may also contribute to increased soil erosion due to reduced canopy interception and reduced litter production. High fire severity typically results in either immediate or delayed plant mortality (Goforth and Minnich 2008). Large quantities of soil may be dislodged and displaced as a result of root death, further exacerbating soil erosion (Fairchilds & Trettin 2006). Eroded soil materials may be trapped on site, but removal of soil organic layers favors their transport to streams. Deposition of eroded material in streams, known as sedimentation, may cause myriad problems ranging from increased water temperatures to a variety of mineralization outcomes affecting overall water quality, treatability, and aquatic life (Van Lear and Danielovich 1988, Minshall 2003, Grace et al. 2006, Malison and Baxter 2010, Clapcott et al. 2012).

Four combinations of fire severity and fire intensity are possible if the ranges of fire severity and fire intensity are divided into simple categories of “low” and “high” (Figure 3.2). Within this simplistic construct, wildfires can exhibit behavior that falls into low and high intensity and severity while prescribed fires are typically (but not always) low in both intensity and severity. In general, wildfires consume more organic matter, induce more vegetative mortality, expose more mineral soil, and thus lead to more-severe water resource impacts (Robinne et al. 2018) when compared to prescribed fires. Prescribed fires typically consume little or no soil organic matter, induce little overstory mortality, expose little mineral soil, and thus have been assumed to be of minor water resource concern (Boerner et al. 2005, Fairchilds & Trettin 2006). Prescribed fires are implemented under a prescription or plan that emphasizes safety of life and property, thus they are most often low in intensity and severity. For safety concerns and to achieve specific goals and objectives, prescribed fires are carefully planned to occur under specific conditions of wind, temperature, humidity, and ground moisture that minimize the risk of escape. The likelihood and magnitude of water impacts are also assumed to be low for many prescribed fire scenarios in eastern forests because they disproportionately occur on relatively flat terrain, including the Atlantic and Gulf coastal plains and Midwestern tallgrass prairies, where gentle topography favors slower and lower-volume surface drainage into streams.

The simplistic, four-cell scenario of fire intensity and severity defined above limits fire intensity and severity to two categories: low and high. These categories do not fully encompass the heterogeneity of intensity and severity across the landscape in a given fire event (Keeley 2009) and do not address the effects of fires of moderate intensity or severity. In some cases, prescriptions require moderate, rather than low, fire intensity and severity to achieve specific management objectives. They might include enhancement or restoration of fire-adapted species, reduction of shrubland vegetation to reduce public safety risks (J. Stowe, pers. comm.), or site preparation burns. Moderate- to high-intensity site preparation burns are often implemented following a timber harvest to create a more conducive environment for seed germination.

3.2 Current Literature

Others have compiled and synthesized evidence of the effects of fire on water resources in the eastern United States, including Fulton & West (2002), Elliott & Vose (2006), and Lafayette et al. (2012). We have summarized their findings in a model linking water impacts to fire behavior (Figure 3.3). Our review highlights more recent research not included in those syntheses.

3.2.1 Chemical Properties

Alterations to water acidity or alkalinity have important water chemistry effects (Brooks et al. 2013). Changes in soil pH can directly affect the presence of biota (Ågren et al. 2010), soil chemical transformations and losses, and the water solubility of chemicals and nutrients (Beyers et al. 2005). In a laboratory experiment, Battle and Golloday (2003) examined how burned longleaf pine (*Pinus palustris* Mill.) litter, wiregrass (*Aristida beyrichiana* Trin. & Rupr.), and soil organic matter (SOM) affected water chemistry in Georgia wetlands. The authors found that pH increased in wetlands following fire due to the consumption and translocation of SOM during the fire. SOM consumption was also a major driving force behind significant increases in nutrients, such as dissolved organic carbon (DOC), soluble reactive phosphorus (SRP), and ammonium (NH4 +).

Other water chemistry factors investigated in relation to fire include dissolved oxygen and potentially harmful metals such as mercury. Mercury naturally occurs in aquatic life and bioaccumulates throughout the food web, but high concentrations of mercury can cause infertility in wildlife, among other health effects (Hopkins et al. 2013). Similarly, high mercury concentrations in humans can lead to poor fetal development and death (Liu et al. 2012). Riggs et al. (2017) found that although yellow perch (*Perca flavescens* [Mitchill, 1814]) mercury levels increased after a low-severity prescribed fire and moderate-severity wildfire in a Minnesota watershed, mercury levels also increased in an adjacent, unburned watershed. The authors found no link between low- and moderate-severity fires and mercury accumulation in the perch. Hagerthey et al. (2014) found that prescribed fire in the Florida Everglades increased dissolved oxygen and led to higher diversity of aquatic flora and fauna, thereby facilitating a more complex food web.

Increases in nutrient concentrations, such as nitrogen and phosphorus, can lead to harmful algal blooms. These blooms may adversely affect aquatic life through a reduction of dissolved oxygen due to bacterial decomposition of dead algae, which can cause fish die-offs, and through shading, which can reduce aquatic plant biomass and diversity (Anderson et al. 2008, Brooks et al. 2016, Konopacky 2017). Increases in sediment can also adversely affect water quality. Bedload sediments can fill spaces between gravel and rocks where fish and other aquatic biota lay eggs and forage for food. Suspended sediment can increase turbidity and thereby decrease the amount of light available for aquatic vegetation (DeBano et al. 1998, Brooks et al. 2012). The majority of research concerning the effects of prescribed fire on nutrient and sediment loads was conducted between 20 and 40 y ago and is synthesized in Lafayette et al. (2012). Because the majority of prescribed fires in the eastern United States are low-intensity, low-severity surface fires, the effects on sediment and nutrient transport into water bodies were either not significant or were of low magnitude and returned to baseline or control levels within 1–3 years post-fire (see also Wendel and Smith 1986, Shumway et al. 2001, Guyette & Spetich 2003, Smith & Sutherland 2006).

Post-harvest, high-intensity, site preparation burns to reduce logging slash, reduce competing vegetation, and prepare the seedbed for regeneration (Waldrop & Goodrick 2012) have produced mixed results in terms of water quality impacts. Van Lear & Danielovich (1988) conducted high-intensity site preparation burns following a timber harvest in western South Carolina. As a result of this practice, they found that soil nutrients increased. Vegetative regeneration also increased during the following growing season. Emerging vegetation assimilated the available nutrients, thereby offsetting potential assart effects (accelerated nutrient mineralization following a disturbance creating a pulse of available nutrients) and minimizing nutrient input into local streams. Knoepp & Swank (1993) conducted a high-intensity site preparation burn in western North Carolina following a clearcut and found that available nitrogen in soil and water temporarily increased, but the increases were within the historically sampled range without fire. Similar short-term pulses were found by Kolka (2012), and again those pulses were within the range of samples collected prior to burning. Both high- and low-severity site preparation burns were conducted in a mixed hardwood–pine forest in upland South Carolina by Robichaud and Waldrop (1994). Sediment yields were approximately 40 times greater in high-severity burn plots than in low-severity burn plots.

3.2.2 Water Treatability

Drinking water demand, especially in the more densely populated coastal areas of the eastern United States, has increased significantly in recent years (Milesi et al. 2003, Bladon et al. 2014). Watershed disturbances, such as wildfires, can amplify the challenge of meeting rising demands for clean water by altering source water quality and quantity. Such disturbances can subsequently increase costs and chemical usage at water treatment facilities (Emelko et al. 2011, Smith et al. 2011). Wildfires can increase surface runoff, which results in increased erosion, elevating sediment (Moody et al. 2008, Emelko et al. 2011), ions, and metals in streams (Crouch et al. 2006). Increased sediments, turbidity, and metals, such as iron and manganese, increase chemical treatment needs and can produce a larger volume of sludge at water treatment facilities (Moody and Martin 2009, Bladon et al. 2014). The impacts on source water quality from a severe wildfire can last from a few years to decades, whereas impacts from low-intensity prescribed fires are seldom pronounced or long-lasting. A watershed-scale study at the Santee Experimental Forest in South Carolina compared flow and nutrients at paired first-order watersheds (one burned and one control). Although prescribed burning initially increased the water yield by 72%, outflow differences disappeared after 2 years (Amatya et al. 2007). Furthermore, no significant differences in nutrient levels were observed between the two watersheds after 2 years.

Severe wildfires can alter the quantity and chemical composition of terrestrial dissolved organic matter (DOM; Wang et al. 2016, Tsai et al. 2017). DOM plays a significant role in the transport of pollutants and in water treatment processes, especially coagulant dosing (Smith et al. 2011, Chow et al. 2013, Majidzadeh et al. 2017). At water treatment facilities, DOM reacts with chlorine or other oxidants forming carcinogenic disinfection byproducts (DBPs), such as chloroform (Sharifi et al. 2013, Writer et al. 2014, Wang et al. 2015). DBP ingestion or inhalation can have negative impacts on human health, including bladder cancer, rectal cancer, and adverse birth outcomes (Chow et al. 2009, 2011, Liu et al. 2012). The Environmental Protection Agency (EPA) regulates maximum contamination levels for two major classes of DBPs: trihalomethanes (THMs: 80 μg L–1) and haloacetic acids (HAAs: 60 μg L–1). Recent studies have documented that unregulated nitrogenous (N-) DBPs, such as haloacetonitriles (HANs) and N-nitrosodimethylamine (NDMA), can have even more genotoxic effects than regulated carbonaceous (C-) DBPs, THMs, and HAAs (Plewa et al. 2002, Zeng et al. 2016).

A severe wildfire can result in a significant increase in DOM concentration, especially during storm events, for years after the disturbance (Emelko et al. 2011). Besides DOM concentration, increases in DOM aromaticity (polycondensed aromatic structures such as polycyclic aromatic hydrocarbons, indicated by specific UV absorbance at 254 nm) and the abundance of hydrophobic compounds have also been observed after wildfire (Wang et al. 2015). Increases in DOM aromaticity can increase DOM reactivity in the formation of nitrogenous DBPs (Tsai et al. 2015, Wang et al. 2015). Increases in DOM aromaticity after wildfire may be due to white ash formation during intense wildfires (>510 °C) whereas black ash, typical of low-intensity prescribed fires (200–500 °C), can decrease DOM aromaticity (Wang et al. 2015). In contrast to wildfire, formation of white ash in low-intensity (including prescribed) fires is often very limited; thus, changes in DOM export and DBP formation can be minimal. Numerous laboratory and field studies have shown a significant reduction of C-DBP formation potential following prescribed fire (Tsai et al. 2015, Wang et al. 2015).

However, Majidzadeh et al. (2015) showed in a laboratory study that post-fire DOM, even after a low-intensity fire, can favor formation of N-DBPs. Further studies are necessary to quantify the formation of N-DBPs at field scales after prescribed burns.

3.2.3 Physical Properties

Water yield is a commonly monitored physical component of freshwater streams (Brooks et al. 2013). Studies of prescribed fire effects on water yield in the eastern United States are limited. Elliott et al. (2017) explored 80 y of water flow and vegetation records at Coweeta Hydrologic Laboratory and found that historically, due in part to prescribed fire, ring-porous species (oaks and hickories) were dominant and consequently water yields were higher than present-day conditions in which fire exclusion has caused a shift in species composition to dominance by diffuse-porous species such as red maple (*Acer rubrum* L.) and yellow-poplar (*Liriodendron tulipifera* L.). Hallema et al. (2017) studied the effects of repeated prescribed fire on water yield in a South Carolina watershed. They found water yield decreased by 39%; however, there was no experimental control and the decrease was more likely attributable to a decrease in precipitation during the sample period than burning. Buma & Livneh (2017) examined the influences of different disturbances such as insect outbreak, timber harvesting, and fire (both wildfire and prescribed fire) on water yield. The authors suggested that prescribed fire can alter streamflow in Georgia and at other sites across the country; however, they failed to separate effects attributable to different disturbances.

3.2.4 Biological Properties

Biological components are often considered the most comprehensive and sensitive indicators of water quality (Clapcott et al. 2012, Woznicki et al. 2015). The Index of Biotic Integrity (IBI), which is used to determine the presence and quantity of certain benthic macroinvertebrates, is the primary model used to quantify biological diversity and the health of a waterbody (Brooks et al. 2013). Although no studies in the eastern United States were identified that directly related prescribed fire to IBI, some studies have examined prescribed fire timing and frequency effects on the presence of particular biota. Venne et al. (2016) found that prescribed fire treatments in the Florida Everglades led to short-term increases of periphyton, which in turn increased fish populations. Hagerthey et al. (2014) determined that prescribed fire and the application of herbicides have the potential to assist in eutrophic wetland rehabilitation. Both studies attributed temporary changes in nutrient composition and increased light as the driving forces that enhanced habitat for periphyton. Robertson et al. (2017) concluded that frequent prescribed burning did not limit the genetic diversity or restrain the connectivity between breeding ponds of the endemic pine woods tree frog (*Hyla femoralis* Bosc, 1800) in Florida.

3.3 Research Gaps

Counter to the simplified fire intensity and severity matrix in Figure 3.2, real-world prescribed fires in the East are complex phenomena (Loudermilk et al. 2017, Yedinak et al. 2018). The ecological effects of fire are dictated by numerous factors whose combination is unique to each individual fire, varying greatly from one ecosystem to another and often among patches within a single fire (Whelan 1997), and depend heavily on the season and weather conditions pre-, during, and post-fire. As topography, fuel arrangement and composition, and weather interact in a specific location on a given day, fire effects are variable across the landscape (Coates et al. 2018). Not all eastern prescribed fires are ignited in flat terrain; they are increasingly being used as a potential restoration tool in the southern Appalachian Mountains (Yaussy & Waldrop 2010), the Ozarks (Knapp et al. 2017), and other steep sites.

Climate change and projections of extended growing seasons offer potential for increased fuel loads, increased incidence and severity of pests and disease, and more frequent, longer, and more severe droughts (Dale et al. 2001). Therefore, the need for fuel reduction, including the use of prescribed fire, is expected to increase. Furthermore, increasing human population increases the need for fuels management as more people move into fire-prone areas. Nowacki and Abrams (2008) concluded that up to a century of fire exclusion in parts of the eastern United States has initiated a positive feedback cycle whereby microenvironmental conditions have become cooler, damper, and more shaded and fuel beds less flammable. This process, referred to as mesophication, improves conditions for shade-tolerant, mesophytic species and degrades them for shade-intolerant, fire-adapted species, including oaks and pines. One line of evidence for this process on the landscape is the widespread decline of oak regeneration and the gradual replacement of oak forests with types lacking a historical antecedent; for instance, the switch in upland forests to dominance by red maple or yellow-poplar (Van Lear 2000). Some stands affected by long-term fire exclusion have proven resistant to restoration (Van Lear 2000, Kreye et al. 2018) and appear to have altered decomposition rates, which affect soil nutrients (Alexander & Arthur 2014). Prescribed fire in the dormant season alone does not necessarily enhance oak regeneration sufficiently for oaks to outcompete red maple, yellow-poplar, and other mesophytic, fire-intolerant species that are replacing historically oak-dominated forests in many parts of the East (Oakman 2018). These challenges to prescribed fire implementation, difficulties in effectively predicting fire behavior, and a limited understanding of fire effects on water quality and quantity are significant research gaps. Improved understanding of burning in stands with altered fuels and flora is needed to support stewardship decision-making.

Evidence from Coweeta Hydrologic Laboratory and other locations is consistent with the mesophication hypothesis, confirming that long-term fire exclusion shifts forest species dynamics to more mesophytic, fire-intolerant species (Elliott & Vose 2011, Ryan et al. 2013, Elliott et al. 2017). These species channel more water into evapotranspiration, resulting in less groundwater and surface water yield at the watershed scale (Caldwell et al. 2016).

Increased use of prescribed fire appears to lead to greater water yields in watersheds in the historic range of oak– hickory forests, such as at Price Mountain near Blacksburg, Virginia (Silver et al. 2013), in south-central Illinois (Singh et al. 2017), and in coastal pine–hardwood forests of the southeastern Coastal Plain on the Santee Experimental Forest, South Carolina (Amatya et al. 2006, 2007).

The water quality results on the Santee Experimental Forest complement the findings on water yield. After 40 y of comparison between burned and unburned watersheds, water quality has been either unaffected or temporarily enhanced immediately postfire by repeated prescribed fires (Richter 1982, Amatya et al. 2007). These results may be related to the low intensity and low severity of prescribed surface fires at Santee and in many fire-maintained forests of the eastern United States. Evidence suggests that fires with this prescription minimally alter forest floor chemistry, leaving behind a mixture of slightly burned or partially charred material post-fire that minimizes water quality effects even if post-fire erosion occurs (Coates et al. 2017). This provides a stark contrast to studies suggesting substantial yields of polycyclic aromatic hydrocarbons (PAHs), which are known carcinogens (Abdel-Shafy & Mansour 2016), following wildfires (Olivella et al. 2005). Similar studies in large watersheds are needed to understand the unique dynamics of many landscapes of the eastern United States where forested areas provide substantial quantities of water treated for human use and consumption.

Currently, prediction of prescribed fire's ecological effects at specific sites in the eastern United States is constrained by incomplete information regarding the nuances of fire behavior as affected by local fuel conditions and terrain (Loudermilk et al. 2017). Enhanced technology is needed to parse the subtleties of fire dynamics, such as levels of intensity and severity, by improving our ability to evaluate them in the field accurately and at a fine spatial scale to hone predictive models based on combinations of key site factors (Bova & Dickinson

2008). With increasingly greater areas being included in fire prescription plans, better understanding of fire effects will become ever more critical. An expanded understanding of prescribed fire will provide managers and scientists with more and better opportunities to predict and then test the effects of specific practices and their outcomes. This will further enhance prescribed fire professionals' abilities to protect watersheds and freshwater ecosystem integrity.

3.4 Conclusions

Research conducted to date suggests that prescribed fires of low intensity and severity in the eastern United States have minimal negative effects on the chemical, physical, and biological properties of surface waters. In several cases, it appears that prescribed fire may alter forest floor chemistry and overstory composition in ways that may improve both water quality and yield in forested watersheds. Because most prescribed fires are implemented under prescriptions that leave riparian buffer zones unburned, (Lorber et al. 2018), overall effects on water are typically either negligible, slightly adverse but short-lived, or slightly beneficial. In almost every instance, prescribed fire effects on water are inconsequential compared to the effects of wildfires. Indeed, prescribed fires are often implemented to reduce fuels and decrease the probability of an uncontrolled wildfire.

Our review indicates considerable need for additional research regarding the impacts of prescribed fire on water quality in the eastern United States for sites and circumstances where moderate-severity fire will be applied to complex terrain. Additionally, managers need information regarding the use of more intense and severe prescribed burns to achieve certain management objectives. Novel fire effects might occur in many situations where forest stands are burned after long periods of fire exclusion, including high levels of duff consumption, immediate and delayed mortality, and undesired changes in species composition or stand density. New methods and models that define heat release in the conductive, convective, and radiative phases are being developed to better define fire behavior and subsequent fire effects resulting from deliberate burning on the landscape (Yedinak et al. 2018). Such methods could enhance our ability to measure fire intensity and severity and predict fire effects, which should lead to improved prescriptions designed to produce specific short- and long-term fire effects and minimize adverse impacts. Given our great dependence upon forests, shrublands, and grasslands for a broad array of ecosystem services, the potential impacts on water resources of all facets of land stewardship, including prescribed fire, warrant greater scrutiny.

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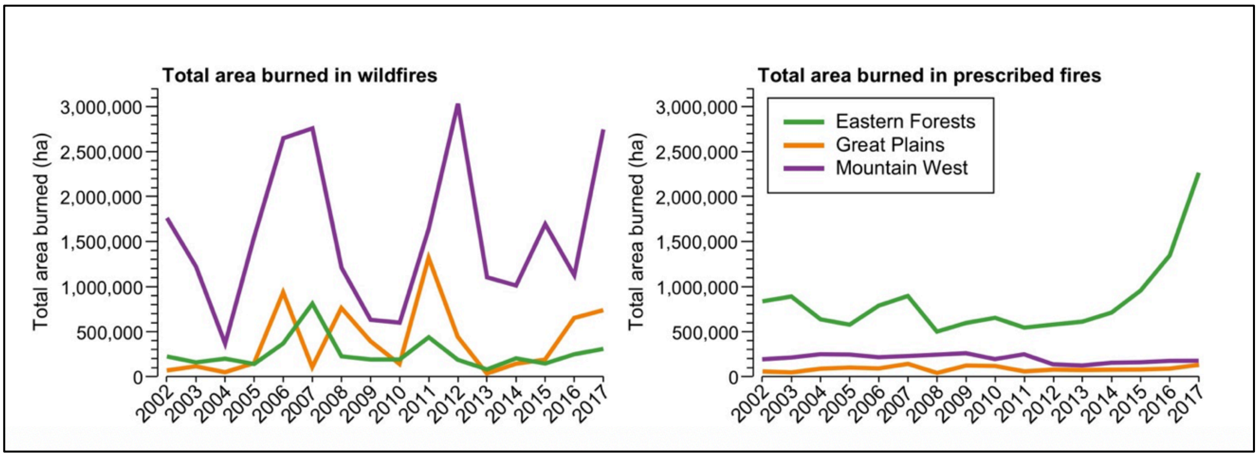

Figure 3.1. Trends in area burned in wildfires and prescribed fires by ecoregions in the continental U.S.Data are from National Interagency Fire Center (2018); ecoregions from U.S. Environmental Protection Agency (2016). Eastern Forests are 30 states representing Level I ecoregions eastern Temperate Forests, Northern Forests, and Tropical Wet Forests (AL, AR, CT, DE, FL, GA, KY, LA, MA, MD, ME, MI, MN, MO, MS, NC, NH, NJ, NY, OH, PA, RI, SC, TN, VA , VT, WI, WV); Great Plains are 7 states representing Level I ecoregion Great Plains (IA, KS, ND, NE, OK, SD, TX); Mountain West is 11 states representing Level I ecoregions Northwestern Forested Mountains, Marine West Coast Forest, Mediterranean California, Temperate Sierras, Southern Semi-arid Highlands, and North American Deserts (AZ, CA, CO, ID, MT, NM, NV, OR, UT, WA, WY).

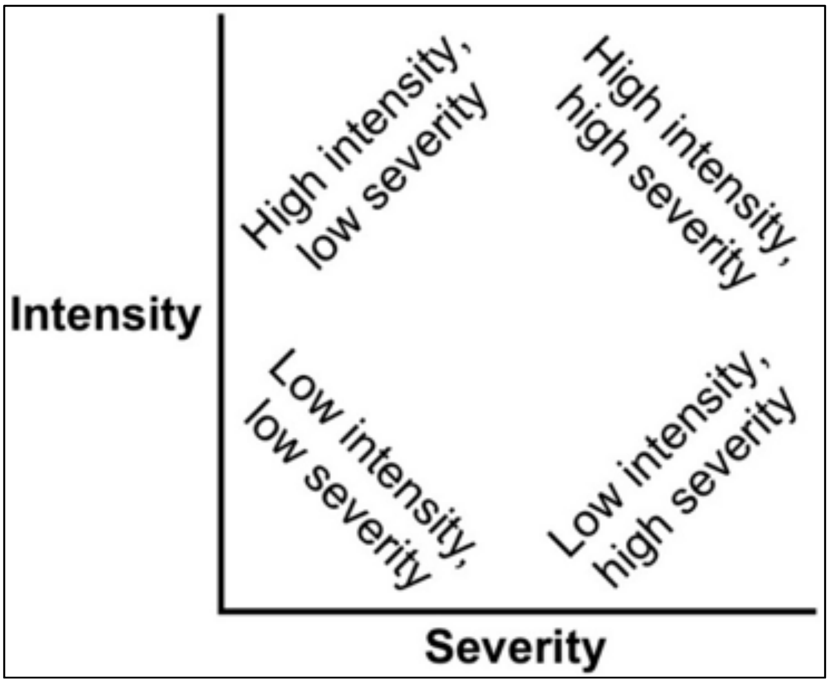

Figure 3.2. Simplified classification of fires by intensity and severity

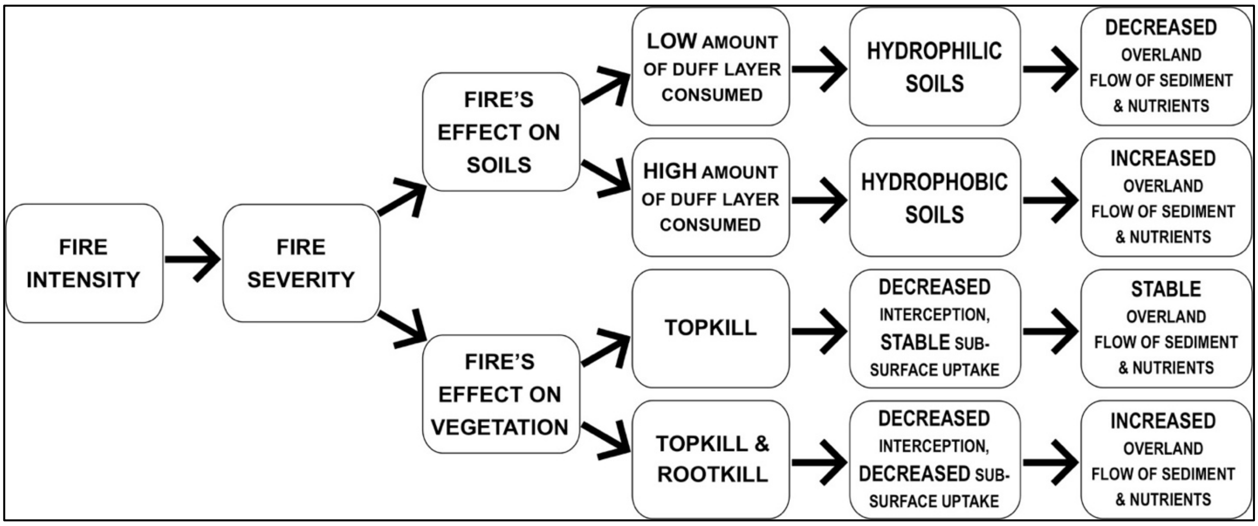

Figure 3.3. Flow chart depicting fire's effect on water quality, based on Keeley (2009). Each succeeding box is a function of the preceding box.

Chapter 4. Tree regeneration and forest floor dynamics following wildfires in the central Appalachian Mountains, USA

4.1 Introduction

Historically, wildland fires were important, formative agents in the development of deciduous forests in the eastern United States (Abrams 1992, McEwan et al. 2007, Arthur et al. 2012, Keyser et al. 2017). Native Americans used fire as a management tool to improve hunting grounds and travel corridors, control pests and disease, and prepare agricultural fields (Sauer 1950, Delcourt & Delcourt 1998, Flatley et al. 2013). As European settlement expanded, Native American populations declined. In some locations, fire frequency decreased as Native American villages were depopulated (Lafon et al. 2017). In other locations, fire frequency remained relatively unchanged or increased well into the 19th century as Europeans adopted many Native American agricultural and land management practices (Prunty, 1965, Guyette & Spitech 2003, Flatley et al. 2013).

The extensive fire exclusion policy enacted across the United Sates in the early 20th century resulted in species composition changes and hazardous fuel accumulation (Abrams 1992, Welch et al. 2000, Lafon & Kutac 2003). The cumulative impact of fire exclusion was obvious in 2016, when numerous wildfires followed an extreme drought in the southern Appalachian Mountains and resulted in 54,731 ha of burned forestland (James et al. 2020). This wildfire event demonstrated a regional research need to quantify the impacts of wildfire on tree regeneration and forest floor dynamics. Litter contributes to ecosystem function by altering soil temperature, protecting seeds and seedlings, and facilitating nutrient exchange (Facelli & Pickett 1991, Guinta & Shaw 2018). Duff regulates soil moisture, provides habitat for symbiotic ectomycorrhizal fungi, and prevents soil erosion (Fitter & Garbaye 1994, Bonan 2002, Smith et al. 2005, Royse et al. 2010, Woodall et al. 2012).

The purpose of this study was to document wildfire effects on tree regeneration and forest floor properties. Specific objectives were to: (1) compare tree seedling and sapling regeneration in areas burned by wildfires with unburned areas, and (2) compare litter and duff depths in areas burned by wildfires with unburned areas of the same forests.

4.2 Methods

4.2.1 Site and fire information

This observational study took advantage of wildfires that occurred at three separate locations within the central Appalachian Mountains: (1) the Castle Fire in Grant County, West Virginia, (2) the Brushy Fire in Bland County, Virginia, and (3) the Price Mountain Fire in Montgomery County, Virginia (Figure 4.1). Weather information for each fire is described in Table 4.1. Fuel temperature ranged from a low 14.7˚C at the Price Mountain site to a high of 22.2˚C at the Brushy site. Days since last rain ranged from 3 at the Castle site to 6 at the Brushy site.

On April 14, 2018, the 173 ha Castle Fire was detected on the Monongahela National Forest near Petersburg, West Virginia. It was extinguished on May 8, 2018. This wildfire exhibited mixed intensity and severity, evidenced by 7.7 m char heights on some trees in portions of the burned area, but not throughout its entirety (Thompson et al. 2019). At this site, overstory species are: northern red oak (*Quercus rubra* L.), chestnut oak (*Quercus montana* Willd.), red maple (*Acer rubrum* L.), striped maple (*Acer pensylvanicum* L.), blackgum (*Nyssa sylvatica* Marshall), eastern white pine (*Pinus strobus* L.), American beech (*Fagus grandifolia* Ehrh.), and various hickories (*Carya* spp.). Mean annual temperature ranges from 0˚C (Winter) to 2˚C (Summer [Soil Survey Staff Natural Resources Conservation Service, 1989]). Mean annual precipitation is 84 cm.

The Brushy Fire occurred on May 3, 2018 in Bland County, Virginia on the George Washington and Jefferson National Forest. The fire consumed 58 ha on a western slope and exhibited both high fire intensity and severity, evidenced by 10 m char heights in some locations (Thompson et al. 2019). The site is dominated by Table Mountain pine (*Pinus pungens* Lamb.), pitch pine (*Pinus rigida* Mill.), and chestnut oak. Mountain laurel (*Kalmia latifolia* L.) and lowbush blueberry (*Vaccinium angustifolium*) are common in the understory. Mean temperatures range from -5˚C (Winter) to 27˚C (Summer). Mean annual precipitation is 138 cm (Thompson et al. 2019).

A 43-ha wildfire (Price Mountain Fire) occurred on an eastern slope of the Fishburn Forest on November 9, 2010 near Blacksburg, Virginia. This fire exhibited low to moderate intensity and severity, as evidenced by a lack of significant char heights on tree boles (Thompson et al. 2019) and low post-fire erosion rates (Christie et al. 2013). Overstory species are common eastern deciduous trees, such as chestnut oak, scarlet oak (*Quercus coccinea* Münchh), blackgum, and sourwood (*Oxydendrum arboreum* L.). The current midstory is dominated by blackgum, red maple, sassafras (*Sassafras albidum* Nutt.), and flowering dogwood (*Cornus florida* L.). Mean annual temperature ranges from -6˚C (Winter) to 27˚C (Summer). Mean annual precipitation is 102 cm.

4.2.2 Study Design and Data Collection

During the summer of 2019, 0.008 ha plots were established at each location (n=136). Price Mountain Fire plots (n=36) were located in a systematic grid while the plots on the Brushy (n=60) and Castle (n=40) Fire sites were located along linear transects previously established by Thompson et al. (2019). Stems ≤ 5.1 cm diameter at breast height (dbh) were tallied as regeneration at each site, in both wildfire-affected and similar, adjacent unburned areas. Individual species were sorted into two compositional groups: pyrophytic and pyrophobic (Table 4.2). Species was recorded and stems were assigned to height classes:

seedling (< 1 m tall) or sapling (≥ 1 m tall). Regeneration was also assigned a form class: single- or multi-stemmed. If more than one stem was protruding from below the soil surface, the stem was classified as multi-stemmed, but tallied as one individual stem. Litter (Oi horizon) and duff (Oe+Oa horizons) depths (cm) were measured in the same locations.

4.2.3 Data Analysis

Data were non-normally distributed with unequal variance, therefore, a Kolmogorov-Smirnov test (Conover 1971) was used to determine significant differences in regeneration densities by height and form class. Additionally, litter and duff depths were tested between the burned and unburned areas. Each site was analyzed separately due to differences in time since burn (1.5 years for Castle and Bland sites; 9 years for Fishburn Forest site) and aspect (western aspect for Castle and Bland sites; eastern aspect for Fishburn Forest site). Differences were declared statistically significant at $\alpha = 0.10$.

4.3 Results

4.3.1 Castle Fire effects on tree regeneration forest floor properties

While there were more pyrophytic seedlings on the Castle Fire site (Table 4.3) in the unburned area compared to the burned area (9 stems ha^{-1} unburned, 1 stem ha^{-1}; $p = 0.3291$), this difference was insignificant. Significantly more pyrophobic seedlings (416 stems ha^{-1} unburned, 265 stems ha^{-1} burned; $p = 0.0348$) and saplings (186 stems ha^{-1} unburned, 6 stems ha^{-1} burned; $p < 0.0001$) were found in the unburned area compared to the burned area. The only significant difference in form class was among the single-stemmed pyrophobic species group (597 stems ha^{-1} unburned, 266 stems ha^{-1} burned; $p = 0.0015$). While not an objective of this study, and not statistically analyzed due to limitations of the Kolmogorov-Smirnov test, a substantial number of tree-of-heaven (*Ailanthus altissima* Mill.) stems (416 stems ha^{-1}) were found in the burned area while no tree-of-heaven stems were observed in the unburned area.

Both litter and duff (Table 4.6) depths were significantly less ($p < 0.0001$) in the burned (litter 1.11 cm, duff 1.05 cm) area when compared to the unburned area (litter 5.08 cm, duff 4.63 cm).

4.3.2 Brushy Fire effects on tree regeneration and forest floor properties

At the Brushy Fire site (Table 4.4) significantly more pyrophytic seedlings were found in the unburned area compared to the burned area (937 stems ha^{-1}, unburned, 444 stems ha^{-1} burned; $p = 0.0049$), while significantly more pyrophytic saplings were in the burned area relative to the unburned area (9 stems ha^{-1} unburned, 87 stems ha^{-1} burned; $p < 0.0001$). There was no significant difference in pyrophobic saplings between the burned and unburned areas (38 stems ha^{-1} unburned, 31 stems ha^{-1} burned; $p = 0.9251$). In terms of form class, single-stemmed regeneration was greater in the unburned area for both pyrophytic (939 stems ha^{-1} unburned, 273 stems ha^{-1} burned; $p <0.0001$) and pyrophobic (1,166 stems ha^{-1} unburned, 155 stems ha^{-1} burned; $p < 0.0001$) species. Alternatively, the burned area contained significantly more multi-stemmed regeneration for both species groups (pyrophytic: unburned 7 stems ha^{-1} burned, 257 stems ha^{-1}; $p < 0.0001$; pyrophobic: unburned 26 stems ha^{-1}, burned 134 stems ha^{-1}; $p < 0.0001$).

Litter depth (Table 4.6) was significantly less ($p < 0.0001$) in the burned area (2.42 cm) compared to the unburned area, while duff depth (Table 4.6) did not differ significantly between the unburned and burned areas (4.11 cm unburned, 4.39 cm burned; $p = 0.9853$).

4.3.3 Price Mountain Fire effects on tree regeneration and forest floor properties

Nine years following the Price Mountain Fire (Table 4.5), there were more pyrophytic seedlings in the burned area compared to the unburned area (944 stems ha^{-1} unburned, 5,070 stems ha^{-1} burned; $p < 0.0001$). Additionally, more single-stemmed pyrophytic regeneration was found in the burned area, compared to the unburned area (948 stems ha^{-1} unburned, 4,889 stems ha^{-1} burned; $p < 0.0001$) while differences in pyrophytic multi-stemmed regeneration were not significant (7 stems ha^{-1} unburned, 181 stems ha^{-1} burned; $p < 0.9999$). Pyrophobic seedlings were more abundant in the burned area compared to the unburned area (1,206 stems ha^{-1} unburned, 6,800 stems ha^{-1} burned; $p = 0.0002$). Furthermore, significantly more pyrophobic saplings were found in the burned area compared to the unburned area (75 stems ha^{-1} burned, 61 stems ha^{-1} unburned, $p = 0.0222$). Although there was no difference in multi-stemmed pyrophobic regeneration (13 stems ha^{-1} unburned, 24 stems ha^{-1} burned; $p = 0.9999$), significantly more single-stem pyrophobic regeneration was found in the burned area compared to the unburned area (1,253 stems ha^{-1} unburned, 6,852 stems ha^{-1} burned; $p = 0.0007$).

Litter depth (Table 4.6) did not differ between the burned (3.81 cm) and unburned areas (5.17 cm) of the Fishburn Forest ($p = 0.2125$), but duff depth was significantly different (1.33 cm burned, 4.20 cm unburned; $p < 0.0001$).

4.4 Discussion

4.4.1 Tree regeneration

Less regeneration following the Castle Fire may be a function of the site itself, as the unburned portion of the site had relatively low densities. The portion of the burn area sampled lay on a ridge and may have influenced productivity, as well. While the abundance of tree-of-heaven in the unburned and burned areas was not statistically tested, the density (416 stems ha^{-1}) is greater than the density of the pyrophytic (1 stem ha^{-1}) and pyrophobic (271 stems ha^{-1}) groups combined. The spread of tree-of-heaven post-fire at the Castle Fire site is likely related to increased light resulting from overstory mortality (Kuppinger et al. 2010, Guthrie 2016). The Castle Fire was a high intensity fire and likely created light conditions favorable for tree-of-heaven establishment, which may have limited native species regeneration.

While promoting pyrophytic species regeneration is an important first step in creating future oak-hickory stands, seedling recruitment into the overstory is just as critical (Dey 2014). Our observations of more pyrophytic saplings following the Brushy Fire agreed with similar studies which showed oak stems outnumbering non-oak stems on central and southern Appalachian Mountain sites that were recently burned by wildfires (Signell 2005, Hagan et al. 2015, Thomas-Van Gundy et al. 2015). It has been posited that growing season fires may benefit oak more than its mesic competitors due to each group's physiological traits (Brose & Van Lear 1998, Brose et al. 1999, Brose et al. 2007). Generally speaking, pyrophobic species, such as maples and birches, display indeterminate growth (Kays & Canham 1991), while oaks display determinate or semi-determinate growth (Cardon & Czaja 2002). By using stored carbohydrates in the early growing season, little reserves are left to replenish shoots topkilled by growing season fires. On the other hand, species that regulate their stored carbohydrate use may be more likely to recover following topkill.

Increased light availability resulting from high overstory mortality post-fire may have influenced the height of re-sprouting oak and hickory stems at the Brushy Fire site. Vickers et al. (2014) reported an increase in periodic annual height of oak and hickory saplings (1.25 m increase) as overstory basal area was reduced to 0-5 m^2 ha^{-1}. Additional treatments may be necessary to maintain dominant oak and hickory regeneration long-term and to encourage this regeneration to progress into the overstory.

The inability of oak and hickory seedlings to remain competitive following prescribed fires of low to moderate intensity has perplexed forest managers in the southern and central Appalachian Mountains for some time (Nowacki & Abrams 2008, Keyser et al. 2019). Nine years following the Price Mountain Fire, significantly more pyrophytic stems were observed in the burned area when compared to the unburned area, however, the same was true for pyrophobic species. Furthermore, there were significantly more pyrophobic saplings in the burned area, but no pyrophytic saplings. This may be related to the low intensity and severity noted for this dormant season fire. Single-entry, low intensity prescribed fires have shown mixed results in terms of oak regeneration (Barnes & Van Lear 1998, Franklin et al. 2003, Dolan & Parker 2004, Elliott et al. 2004). In some instances, regeneration of mesic competitors has outnumbered oak and hickory regeneration (Wendel & Smith 1986, Collins & Carson 2003). If oak regeneration is the management goal, other silvicultural treatments may be necessary to release oak and hickory stems from mesic competition long-term (Oakman et al. 2019).

4.4.2 Forest floor properties

Studies suggest litter re-accumulation post-fire occurs rapidly when the overstory is minimally impacted by fire (Stambaugh et al. 2006, Arthur et al. 2017). Litter re-accumulation following a high intensity, high severity wildfire would presumably be slower if a large proportion of the overstory was killed during the fire, which may explain the significant difference in litter depth between the burned and unburned areas following the Castle and Brushy Fires. Conversely, no difference in litter depths between the burned and unburned areas of the Price Mountain site is likely a result of time since fire and minimal overstory mortality induced by the fire. Because duff development occurs over a relatively long time period (Kreye et al. 2014), it is not surprising that duff depth post-fire at the Price Mountain site differed between the burned and unburned areas. In contrast, the Castle Fire site was sampled only two growing seasons post-fire, which likely explains the difference in duff depth between the burned and unburned areas. Additionally, the duration of the Castle Fire ($\geq$ 3 weeks) suggests that the fire may have smoldered, leaving more time for duff to be consumed. A lack of significant difference in duff depths at the Brushy Fire site may suggest the fire was an active crown fire, inhabiting the litter layer and forest canopy simultaneously with low residence time.

Pronounced and prolonged forest floor disturbances are largely documented in high intensity, high severity wildfires (Bladon et al. 2014). This fire behavior often leads to significant vegetative mortality. Coupled with the loss of living vegetation, significant loss of forest floor materials may lead to pronounced erosion post-fire (Rhoades et al. 2011). Eroded soil and sediments may enter water bodies, increase nutrient loads, increase water temperatures, and affect aquatic life (Hahn et al. 2019). Negative consequences on water quality can affect human health, as well. Increased dissolved organic matter (DOM) may react with oxidants during the water treatment process (Wang et al. 2015), creating disinfection by-products (DPB's). These by-products have been linked to various cancers and birth defects (Chow et al. 2011, Liu et al. 2012). Careful pre-burn planning is imperative to mitigate any potential, negative effects associated with high intensity, high severity fires.

It has been suggested that a lack of mineral soil exposure resulting from low intensity, low severity, dormant season prescribed fires may impede xeric pine and oak regeneration in the eastern US (Waldrop et al. 2009). This framework suggests that fires of high intensity are required to promote Table Mountain pine (TMP) regeneration (Waldrop et al 2006). It has been hypothesized that conditions created by high intensity, high severity fires may best promote successful TMP germination. Fires of high intensity would generate sufficient heat needed to release seeds from TMP's serotinous cones and high severity fires would consume duff and expose bare mineral soil to promote successful seed germination. However, these fires are difficult for prescribed burn managers to implement due to safety concerns associated with extreme fire behavior (Waldrop et al. 2003). Observations following the Brushy Fire suggest that mineral soil exposure may not be necessary for successful TMP regeneration, as duff depth averaged 4.39 cm. Waldrop et al. (1999) also reported 80% of sampled TMP seedlings successfully penetrated duff depths up to 7.5 cm and Mohr et al. (2002) found TMP seedlings were capable of penetrating duff depths up to 10 cm. Repeated low intensity, low severity prescribed fires conducted after successful TMP regeneration establishment may reduce competition from less desired species (Waldrop et al. 2013).

Adding further to potential changes in forest floor depth are potential changes in forest floor chemistry that were not measured in this study. More significant changes in forest floor and mineral soil chemistry may be expected if overstory species composition changes following repeated, higher intensity fires (Caldwell et al. 2020). It is hypothesized that pre-colonial vegetative compositions may have been supported by more nitrogen-limiting soil conditions that may have preferred species with mycorrhizal associations, such as oaks, pines, and hickories (Boerner et al. 2008, Dukes et al. 2020). A recent study of two southeastern Coastal Plain forested watersheds suggested that a combination of prescribed fire, salvage logging following hurricanes, harvesting, and understory mastication over 50 years led to alterations in overstory species composition. Those physical changes in overstory dominance were linked to differences in litter, duff, and mineral soil chemistry for calcium, carbon, magnesium, nitrogen, phosphorus, and potassium (Coates et al. 2020). In this regard, forest floor properties may respond to potential disturbances and support growth requirements for one species as opposed to another. Therefore, forest floor properties must continually be considered and monitored during active forest management.

4.4.3 Management implications

While each site exhibited different trends in regeneration species composition, the observed differences were consistent with post-fire research findings throughout the Appalachian Mountains. Documenting the observations at these sites will add to the limited knowledge regarding post-fire regeneration dynamics in this region. If forest managers seek to use prescribed fire to restore pyrophytic species, growing season burns may likely be a viable option when regeneration dynamics are considered. Treatments other than fire may be needed to control competition. Eventually, a fire-free period is warranted to allow saplings to advance into the overstory (Dey & Fan 2009). As with any disturbance, the threat of non-native species colonization exists. However, post-fire monitoring may enhance the ability of forest managers to limit non-native species invasions. Furthermore, these biological considerations do not speak of site-specific, physical constraints, such as weather and personnel, that may limit prescribed fire use to the dormant season.

Several study limitations should be noted. Two of the wildfires evaluated were in remote locations and had limited access. Therefore, the number of sampling locations at those sites were limited. Trends found at the sampling points investigated may or may not have been sustained across the entire burn areas. The number of wildfires evaluated as part of this study was also another potential limitation of these results. Time since burn and aspect differed for the Price Mountain site relative to the Castle and Brushy sites, thereby complicating observational interpretations. As noted earlier, overstory composition and basal area in both the burned and adjacent, unburned areas was not measured, therefore inclusion of these comparisons at each study site are based upon ocular estimations and records of previous disturbance. Regardless of these potential constraints, these results provided additional information to support regional fire managers and practitioners as they anticipate and predict specific wildland fire tree regeneration and forest floor effects. Based upon these observations, it appears that growing season fires or fires of higher intensity may help forest managers achieve pyrophytic restoration goals and long-term monitoring of tree regeneration is needed following wildland fire to ensure that desired species compositions are achieved.

4.5 Conclusions

Natural and anthropogenic wildfire ignitions have been known to influence vegetative composition and forest floor properties. A lack of research examining both of these properties in the central Appalachian Region prompted this observational study. While not an ideal comparison between sites due to differences in season of burn, aspect, and time since burn, several trends were observed. First, the role that site characteristics, fire behavior, and existing vegetation may contribute to post-burn effects was reinforced. While pyrophobic stems still dominated two of the three burned sites in comparison to pyrophytic stems, an increase in pyrophytic regeneration was observed on the Brushy Fire site (high intensity, growing season, wildfire). Additionally, stems ≥ 1 m tall consisted mostly of pyrophytic stems on the Brushy Fire site. These results may be advantageous, displaying the potential benefits high intensity and severity fires may promote for desired species composition. Conversely, the presence of non-native species at the Castle Fire site (growing season wildfire) is a concern for managers. The presence of non-native competitors to desired species may warrant future stand entries to perform mechanical or chemical treatments.

Overstory mortality following fires in the central Appalachian Region may result from fires displaying: 1) high intensity, 2) high severity, or 3) both high intensity and severity. While the changes in litter and duff were statistically significant on some sites, these results may not necessarily be biologically significant. Post-fire litter depth reduction is common and post-fire litter accumulation may occur rapidly as surviving overstory trees and regenerating stems provide litterfall in subsequent autumn months. Litterfall decomposition will in turn contribute to future duff formation. Monitoring of the recently burned sites included in this study may be warranted to document how species composition shifts over time with or without future management actions or natural disturbance.

4.6 References

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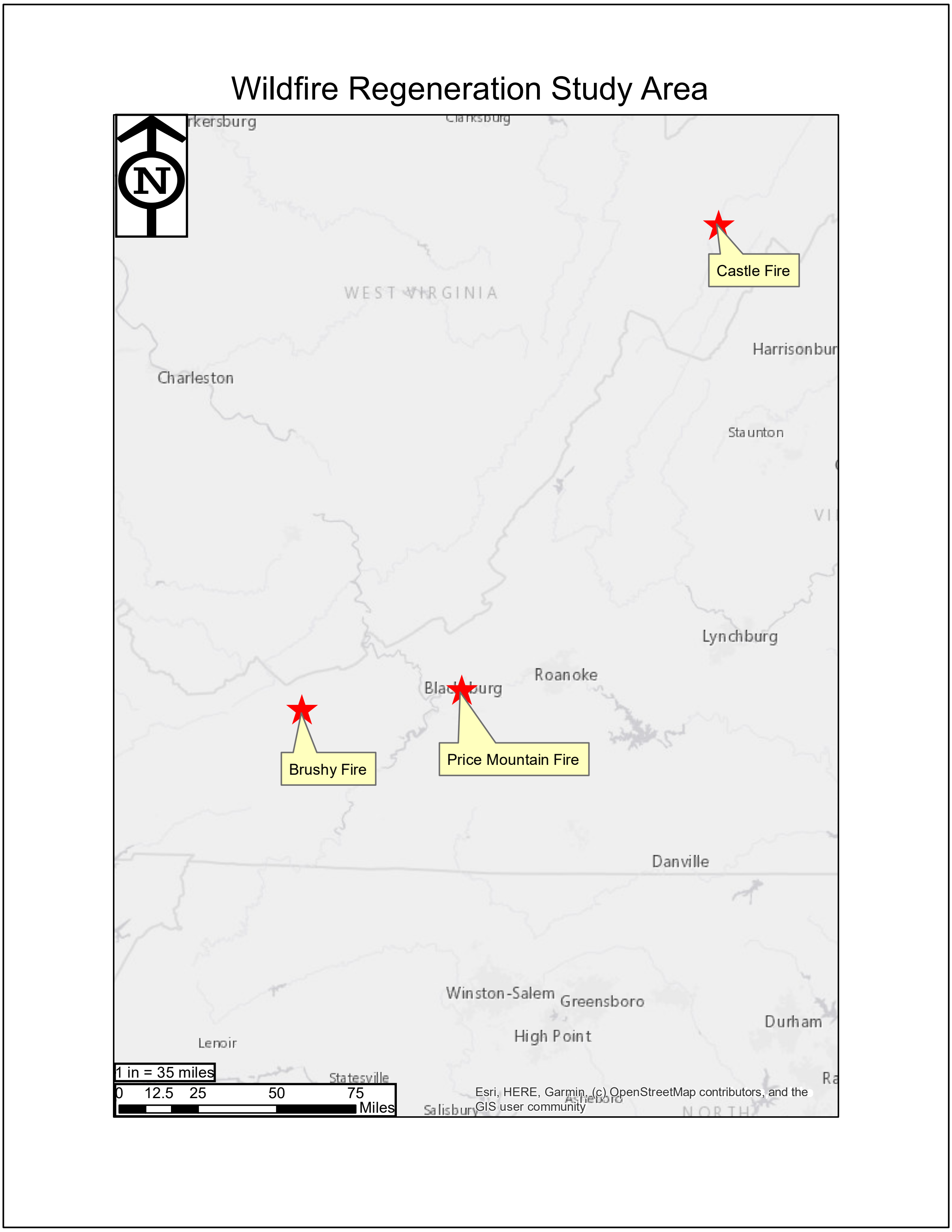

Figure 4.1. Location of the Castle (West Virginia), Brushy (Virginia), and Price Mountain (Virginia) Fire sites.

Table 4.1. Weather conditions for the Castle, Brushy, and Price Mountain Fires.

Weather Day of Fire	**Castle Fire**	**Brushy Fire**	**Price Mountain Fire**
Air Temperature (˚C) [mean, min, max]	19.2, 26.1, 12.2	19.7, 8.9, 30.0	14.7, 3.9, 23.3
Fuel Temperature (˚C) [mean, min, max]	19.8, 8.9, 35.0	22.2, 7.2, 42.8	14.7, 2.2, 31.7
RH (%)	34.8	49.4	34.8
Days Since Last Rain (amount in cm)	3 (0.4)	6 (0.1)	4 (2.1)
Wind Speed (kph) [mean, max]	6.5, 32.0	4.3, 24.1	4.3, 17.6

Table 4.2. Individual species sorted by compositional groups used to differentiate fire effects for the Castle, Brushy, and Price Mountain Fires.

Compositional Group	**Species Common Name**	**Species Scientific Name**
Pyrophytic	Chestnut oak, black oak, scarlet oak, hickory, Table Mountain pine, pitch pine, shortleaf pine, Virginia pine	*Quercus montana, Quercus velutina, Quercus coccinea, Carya spp., Pinus pungens, Pinus rigida, Pinus echinata, Pinus virginiana*
Pyrophobic	blackgum, red maple, American beech, sassafras, witch hazel, yellow-poplar, striped maple, cucumber tree, sourwood, black locust, serviceberry, white pine	*Nyssa sylvatica, Acer rubrum, Fagus grandifolia, Sassafras albidum, Hamamelis virginiana, Liriodendron tulipifera, Acer pensylvanicum, Magnolia acuminata, Oxydendrum arboreum, Robinia pseudoacacia, Amalenchier arborea, Pinus strobus*

Table 4.3. Regeneration densities by species, height, and form class for wildfire and unburned areas at Castle Fire site (p-values with asterisks indicate significance at α = 0.10).

Castle Fire Regeneration Densities											
Species Group	**Condition**	**Seedling stems ha^{-1} ±(SE)**	**p-value**	**Sapling stems ha^{-1} ±(SE)**	**p-value**	**Single Stemmed stems ha^{-1} ±(SE)**	**p-value**	**Multi-stemmed stems ha^{-1} ±(SE)**	**p-value**	**Total stems ha^{-1} ±(SE)**	**p-value**
Pyrophytic	Unburned	9.11 ±(3.74)	0.3291	2.02 ±(1.39)	—	11.13 ±(4.52)	0.1725	—	—	11.13 ±(4.52)	0.1725
	Burned	1.01 ±(1.01)		0.00 ±(0.00)		1.01 ±(1.01)		—		1.01 ±(1.01)	
Pyrophobic	Unburned	415.82 ±(59.46)	0.0348*	186.16 ±(24.50)	<0.0001*	597.93 ±(71.55)	0.0015*	4.05 ±(3.15)	1.000	601.98 ±(72.11)	0.0015*
	Burned	265.08 ±(40.07)		6.07 ±(3.63)		266.09 ±(38.54)		5.06 ±(3.56)		271.15 ±(40.16)	

Table 4.4. Regeneration densities by species, height, and form class for wildfire and unburned areas at Brushy Fire site (p-values with asterisks indicate significance at α = 0.10).

Brushy Fire Regeneration Densities											
Species Group	**Condition**	**Seedling stems ha^{-1} ±(SE)**	**p-value**	**Sapling stems ha^{-1} ±(SE)**	**p-value**	**Single Stemmed stems ha^{-1} ±(SE)**	**p-value**	**Multi-stemmed stems ha^{-1} ±(SE)**	**p-value**	**Total stems ha^{-1} ±(SE)**	**p-value**
Pyrophytic	Unburned	936.87 ±(199.60)	0.0049*	9.10 ±(3.74)	<0.0001*	938.89 ±(119.01)	<0.0001*	7.08 ±(3.96)	<0.0001*	945.97 ±(118.36)	0.0281*
	Burned	443.64 ±(35.72)		86.52 ±(10.72)		273.18 ±(29.22)		256.98 ±(26.77)		530.16 ±(40.57)	
Pyrophobic	Unburned	1,153.38 ±(161.52)	<0.0001*	38.45 ±(9.95)	0.9251	1,165.52 ±(159.57)	<0.0001*	26.31 ±(7.06)	<0.0001*	1,191.83 ±(159.50)	<0.0001*
	Burned	257.49 ±(27.49)		31.36 ±(6.99)		155.30 ±(21.74)		133.55 ±(13.12)		288.85 ±(27.32)	

Table 4.5. Regeneration densities by species, height, and form class for wildfire and unburned areas at Price Mountain Fire site (p-values with asterisks indicate significance at α = 0.10).

Price Mountain Fire Regeneration Densities											
Species Group	**Condition**	**Seedling stems ha^{-1} ±(SE)**	**p-value**	**Sapling stems ha^{-1} ±(SE)**	**p-value**	**Single Stemmed stems ha^{-1} ±(SE)**	**p-value**	**Multi-Stemmed stems ha^{-1} ±(SE)**	**p-value**	**Total stems ha^{-1} ±(SE)**	**p-value**
Pyrophytic	Unburned	944.27 ±(1,58.93)	<0.0001*	10.13 ±(4.09)	—	947.66 ±(159.99)	<0.0001*	6.74 ±(93.66)	0.9999	954.40 ±(158.91)	<0.0001*
	Burned	5,069.92 ±(1,146.12)		0 ±(0.00)		4,888.93 ±(1,146.62)		180.99 ±(179.80)		5,069.92 ±(1,146.12)	
Pyrophobic	Unburned	1,206.21 ±(234.11)	0.0002*	60.74 ±(11.91)	0.0222*	1,253.46 ±(233.22)	0.0007*	13.49 ±(4.63)	0.9999	1,266.95 ±(232.36)	0.0007*
	Burned	6,799.99 ±(1,129.56)		75.32 ±(29.84)		6,851.70 ±(1,122.02)		23.61 ±(7.36)		6,875.31 ±(1126.22)	

Table 4.6. Litter and duff depths for wildfire and unburned areas of the Castle, Brushy, and Price Mountain Fires (p-values with asterisks indicate significance at α = 0.10).

Site	**Litter Unburned (cm)**	**Litter Burned (cm)**	**p-value**	**Duff Unburned (cm)**	**Duff Burned (cm)**	**p-value**
Castle	5.08	1.11	<0.0001*	4.63	1.05	<0.0001*
Brushy	2.42	0.402	<0.0001*	4.11	4.39	0.9853
Price Mountain	5.17	3.81	0.2125	4.20	1.33	<0.0001*

Chapter 5. Soil chemistry following single-entry, dormant season prescribed fires in the Ridge and Valley Province of Virginia, USA

5.1 Introduction

Soils play a vital role in forest productivity (Grier et al. 1989, Powers et al. 2005, Voldseth et al. 2011). At a primary level, soil acts as a medium to support root systems and anchor aboveground vegetation (Neary et al. 2009). Additionally, chemical and physical properties, such as cation exchange capacity (CEC) and soil texture, influence nutrient and moisture availability (Schoenholtz et al. 2000). Variations in soil properties across a given landscape may affect vegetative species composition (Stewart & Edwards 2006). Because of its recognized importance, soil nutrient availability is often a major consideration in forest management regimes. Components frequently monitored include, but are not limited to: soil pH, calcium (Ca), carbon (C), magnesium (Mg), nitrogen (N), phosphorus (P), and potassium (K).

Disturbances, such as wildfires, have been known to alter post-fire soil nutrient composition and abundance. These changes are often measured both immediately following and many years after wildfire occurrence (Neary et al. 2005, Rhoades et al. 2019). Wildfires, in a general sense, exhibit high intensity and severity, often leading to a loss of soil nutrients either directly through volatilization or indirectly through increased runoff following fire extinction (Neary et al. 2009, Bladon et al. 2014, Bixby et al. 2015). Alternatively, prescribed fires generally exhibit low intensity and severity. In many cases, fuels on the forest floor may only be partially charred by surface fires exhibiting short residence times (Arthur et al. 2017, Coates et al. 2017). Soil nutrient fluctuations are generally short-lived under these conditions (Coates et al. 2018). Nutrient pulses lasting 6 months to 1 year following single, dormant season ignitions have been measured in the southeastern Coastal Plain of the United States

(McKee 1982). Following these pulses, soil chemistry generally adjusts to pre-fire conditions or conditions found in similar, unburned, adjacent areas. The frequent and repeated use of prescribed fire long-term in some Coastal Plain pine stands (*Pinus* spp.) has led to increases in some soil chemical properties (Schoch & Binkley 1986, Butnor et al. 2020). This directly contradicts results presented in some studies suggesting that increased global fire frequency may cause deleterious and detrimental impacts to soil C and N (Carter & Foster 2004, Pellegrini et al. 2017).

Fewer research studies have investigated prescribed fire effects on soil properties in the southern and central Appalachian Mountains with many being focused on wildfires (Elliott et al. 2013), high intensity site preparation burns (Knoepp et al. 2004), or organic matter properties (Knoepp et al. 2009). In a study of fuel reduction and ecosystem restoration treatments in the southern Appalachian Mountains, Coates et al. (2010) determined that changes in soil chemistry 1-2 years following the mechanical felling of ericaceous shrubs (great rhododendron: *Rhododendron maximum* L.; mountain laurel: *Kalmia latifolia* L.), prescribed fire, and the combination of felling and fire did not exist 4 years post-fire. Following additional treatments at this site (2 total cuts, 4 total burns), Dukes et al. (2020) reported changes in soil chemistry that most aligned with restoration priorities, primarily reductions in mineral soil N that might favor upland oak (*Quercus* spp.) and pine (*Pinus* spp.) community restoration.

Additional information regarding soil chemical properties in this region is needed because many entities are currently using prescribed fire and other restoration practices as part of their long-term management regimes. The achievement of some management objectives, such as the restoration of upland pine-hardwood forests, may be aided by the reduction of soil N to favor species that possess adaptations allowing them to capitalize on less N-rich soils (Boerner et al. 2008, Taylor & Midgley 2018, Dukes et al. 2020). To address this research priority, 2 prescribed fires were ignited on western slopes in the Ridge and Valley Province of Virginia. The objective of this study was to quantify and compare mineral soil chemistry (Ca, K, Mg, P, C, N, C:N, pH) at different time periods before and after (6 and 14 months post-fire) prescribed fire to determine potential fluctuations in soil chemistry resulting from prescribed fire use. The hypotheses for these analyses were that the mean soil chemistry values would not be altered by a single prescribed fire.

5.2 Methods

5.2.1 Study Area

The Fishburn Forest, located near Blacksburg, Virginia, is the property of Virginia Polytechnic Institute and State University (Figure 5.1). Mean annual temperatures range from 0°C (winter) to 27°C (summer). Annual precipitation is 102 cm with an average annual snowfall of 71 cm. Current overstory species consist of chestnut oak (*Quercus montana* Willd.), scarlet oak (*Quercus coccinea* Münchh), blackgum (*Nyssa sylvatica* Marshall), and sourwood (*Oxydendrum arboretum* L.). The mid-story is dominated by blackgum, red maple (*Acer rubrum* L.), sassafras (*Sassafras albidum* Nutt.), and flowering dogwood (*Cornus florida* L.). While ericaceous shrubs such as mountain laurel (*Kalmia latifolia* L.), lowbush blueberry (*Vaccinium angustifolium* Aiton) and wild azalea (*Rhododendron canescens* Michx.) dominate the understory.

Soils of the study area are Inceptisols (loamy-skeletal, mixed, active, mesic Typic Dystrudepts) of the Berks-Weikert and Berks-Clymer soil complexes. Parent material consists of shale, siltstone, and sandstone residuum. Bulk density averages 1.3 g cm^{-3} and pH averages 5.1. The soils are on average 31% sand, 53% silt, and 16% clay (Soil Survey Staff, Natural Resources Conservation Service, November 14, 2020).

5.2.2 Data Collection and Treatment Descriptions

This experiment was established as a completely randomized design. Three, 0.02-ha units were established on west-facing slopes with 2 units identified as treatment units and 1 unit as an unburned control. Within each unit, 9 sampling plots were systematically located approximately 15 m apart. Soil sampling locations were marked with a 20 cm spike. Two mineral soil samples per plot were obtained using a shovel to a depth of 10 cm. Individual samples were composited to form one mineral soil sample per plot for a total of 9 samples per unit. Samples were collected pre-fire and at 6 and 14 months post-fire (n = 81 total; 27 control and 54 burned). Following each collection, soil samples were oven-dried at 65˚C for at least 72 hours and sieved to 2 mm. Chemical analyses were contracted to Brookside Analytical Laboratories in New Bremen, OH, USA. Mineral soil C and N were determined by dry combustion and the resulting measurements were derived with the Perkin-Elmer 2400 Series II CHNS/O Analyzer (Nelson et al. 1996). Mineral soil Ca, Mg, P, and K concentrations were derived using Mehlich III methodology (Mehlich 1984) and resulting analyses for each element were conducted using ICP-Optical Emission Spectrometry (Boss & Freeden 2004). Soil pH was determined using a 1:1 soil to water solution (McLean 1982).

Treatment units were burned with the assistance of the Virginia Department of Forestry on February 5, 2019 and February 25, 2019. On February 5, ignitions began mid-morning and mop-up activities concluded in the early evening. Air temperatures ranged from 9–17˚C with an average temperature of 13˚C. The relative humidity ranged from 57–89% (mean 75%). Wind speed range was 6–15 km h^{-1}. On February 25, ignitions began mid-morning and mop-up activities concluded in the early evening. Air temperatures ranged from -2–7˚C with an average temperature of 7˚C. Relative humidity range was 12–37% (mean 26%). Wind speed range was 10–15 km h^{-1}. Both burns were ignited with drip torches. A backing fire was ignited along each ridge to establish a fire perimeter, then strip headfires were ignited downslope at approximately 10 m intervals until the entire unit was ignited. Fire intensity and severity were low to moderate in both fires, with a mean flame lengths of 0.3 m. In some locations, however, flame length approximated 1 m. One-thousand hour woody fuel consumption was generally negligible, but consumption of 1-, 10-, and 100-hr fuels did occur in some locations throughout the burn units.

5.2.3 Statistical Analysis

Data were not normally distributed, therefore non-parametric data analyses were conducted using JMP Pro 15 (SAS Institute, Cary, NC, USA). The following designations were assigned to the treatment and time combinations: 1) control, pre-fire; 2) control, 6 months post-fire; 3) control, 14 months post-fire; 4) burned, pre-fire; 5) burned, 6 months post-fire; 6) burned, 14 months post-fire. A Kruskal-Wallis test was performed to determine potential soil chemistry differences between and within the treatment and time combinations. When treatment and time were significantly related to soil chemical responses, a Steel-Dwass mean comparison test was performed to determine specific differences between the treatment combinations. Significance of differences was established with $\alpha = 0.05$.

5.3 Results and Discussion

5.3.1 Differences between treatment and time combinations

Differences between at least one treatment and time combination existed for all of the soil chemistry variables, except soil pH ($p = 0.0610$; Table 1). Some of these differences existed between a time period in one treatment and a separate, asynchronous time period in the other treatment (i.e. pre-fire Ca in the control was less than 6 months post-fire Ca in the burned units). Even though the lowest values for Mg, P, C, N, and C:N were found 14 months post-fire in the burned plots, no significant differences were noted between the control and burned locations for any of the soil chemistry variables when individual time periods were evaluated between treatments (1. Control, pre-fire vs. burned, pre-fire; 2.

Control, 6 months post-fire vs. burned, 6 months post-fire; 3. Control, 14 months post-fire vs. burned, 14 months post-fire). This agreed with other studies of prescribed fire effects on soil chemistry in the Appalachian Mountains that stated non-significant differences in some soil chemical properties between burned and unburned locations 1-2 years following single entry, low intensity and severity, prescribed, surface fires (Knoepp & Swank 1993, Coates et al. 2010). Additionally, soil responses may continue to change 2-4 years post-fire following single-entry prescribed fires and those changes may not align or agree with immediate post-fire results (Coates et al. 2010). Additional soil chemistry changes may occur as subsequent, repeated ignitions are added as part of a long-term management strategy (Dukes et al. 2020). Repeated ignitions are commonly used in the Appalachian Mountains and other regions to achieve long-term management goals, such as hazardous wildland fuel reduction, wildlife habitat restoration and maintenance, or vegetative species maintenance (Waldrop & Goodrick 2012).

Soil responses to wildland fire may be related to many factors, including but not limited to: 1) fire intensity and severity, 2) soil type, 3) dominant aboveground vegetative species, 4) time of sampling post-fire, and 5) fire frequency (McKee 1982, Kutiel & Shaviv 1992, Certini 2005, Fairchilds & Trettin 2006, Rau et al. 2008, Mataix-Solera et al. 2011, Coates et al. 2018, Coates et al. 2020). To frame this particular study with these criteria in mind, we investigated: 1) low intensity and severity, dormant season, prescribed, surface fires, 2) Inceptisols (0-10 cm depth), 3) upland hardwoods, 4) pre-fire, 6 months post-fire, 14 months post-fire, and 5) single-entry (following long-term fire exclusion). It is not uncommon for single-entry wildfire and prescribed fire impacts to differ within the same ecosystem (Thompson et al. 2019), nor is it uncommon for inherent soil fertility to influence post-fire soil chemistry (Alcañiz et al. 2018). Changes in dominant overstory species affect coarse woody debris and forest floor chemistry, subsequently impacting mineral soil chemistry in both unburned and burned mineral soils (Coates et al. 2020). Therefore, meaningful comparisons between this study's results and those of other soils studies related to wildland fire should be framed within this context.

5.3.2 Differences within treatments based upon time since fire

Separate, individual evaluations of the control and burned plots alone were noteworthy, despite no differences between treatments (i.e. control vs. burned) at 6 and 14 months. No statistically significant increases or decreases for any of the soil chemical variables were noted within the control plots alone over the course of this study (Table 1). However: Ca nearly doubled, K increased 41%, Mg decreased 8%, P decreased 20%, C decreased 52%, N decreased 33%, and C:N decreased 23%. Similar trends (increases or decreases) were noted for each variable within the burned plots, although the value of those trends were not necessarily equal to the control trends: Ca increased 52%, K increased 12%, Mg decreased 17%, P decreased 36%, C decreased 53%, N decreased 40%, and C:N decreased 13%.

For P, the 14 month post-fire values were significantly less than the pre-fire values in the burned plots (Table 1). For C, the 14 month post-fire values were significantly less than both the pre-fire and 6 month post-fire values within the burned plots (Table 1). Decreases in soil C (Waldrop et al. 2013) and P (Field et al. 2000) 1 to 2 years following prescribed fires have been noted in other studies of prescribed fire effects on soil chemistry. For N and Mg, the 6 month post-fire values were significantly greater than the 14 month post-fire values within the burned plots, suggesting that responses may differ even within a few months of each other during the first year to 14 months following a single-entry, dormant season fire.

5.3.3 Management Implications

For the long-term restoration and maintenance of upland pine-oak communities in the Appalachian Mountains, it has been suggested that reduced soil N might be desirable due to these species’ advantageous ectomycorrhizal associations (Boerner et al. 2008, Dukes et al. 2020). The continued use of prescribed fire every 3-8 years on the Fishburn Forest may be desirable to see if the N reduction noted between the pre-fire and 14 month post-fire values in the burned plots might be extended to include more pronounced and significant, long-term N differences between the unburned and burned locations.

Evaluating these results within treatment further suggested that soils should continue to be monitored when prescribed fire is utilized as a management tool. The continued use of low intensity and severity, dormant season, prescribed, surface fires may elicit different long-term responses for each variable (Dukes et al. 2020). Additionally, these results suggested that pre-fire soil conditions in treated units should be considered in conjunction with synchronous sampling in untreated, control units. Furthermore, the potential roles specific soil variables may play in the achievement of restoration and management objectives cannot be understated because the responses may be dependent upon the particular soil chemical in question.

As with any research study, additional samples (n = 81 total) and study locations (n = 2 burns, 1 control) may have enhanced these results. However, these results agreed with additional short-term, soils results following prescribed fires in the Appalachian Mountains: 1) short-term effects of single-entry, dormant season, prescribed fires on soil chemistry are largely negligible between adjacent, unburned and burned soils, and 2) soil chemistry is highly variable, even when unaffected by wildland fire.

5.4 Conclusions

Despite differences for some of the soil chemicals within the burned plots based upon time since fire when sampling occurred and asynchronous comparisons between treatments, no significant soil chemistry differences existed between the soil chemical variables when synchronous pre-fire, 6 months post-fire, or 14 months post-fire comparisons were made.

These results agreed with those of other studies in the region that found negligible soil responses to single-entry, low intensity and severity, prescribed, surface fires. Similar prescribed fires in other locations have resulted in incomplete combustion of forest floor fuels and therefore resulted in minimal to non-existent mineral soil responses 1-2 years post-fire. Different intensity and severity fires occurring in different vegetative systems may elicit different responses, therefore wildland fire effects on soil chemistry will continue to be highly variable. This variability should be considered by forest managers when using prescribed fire to meet specific management objectives.

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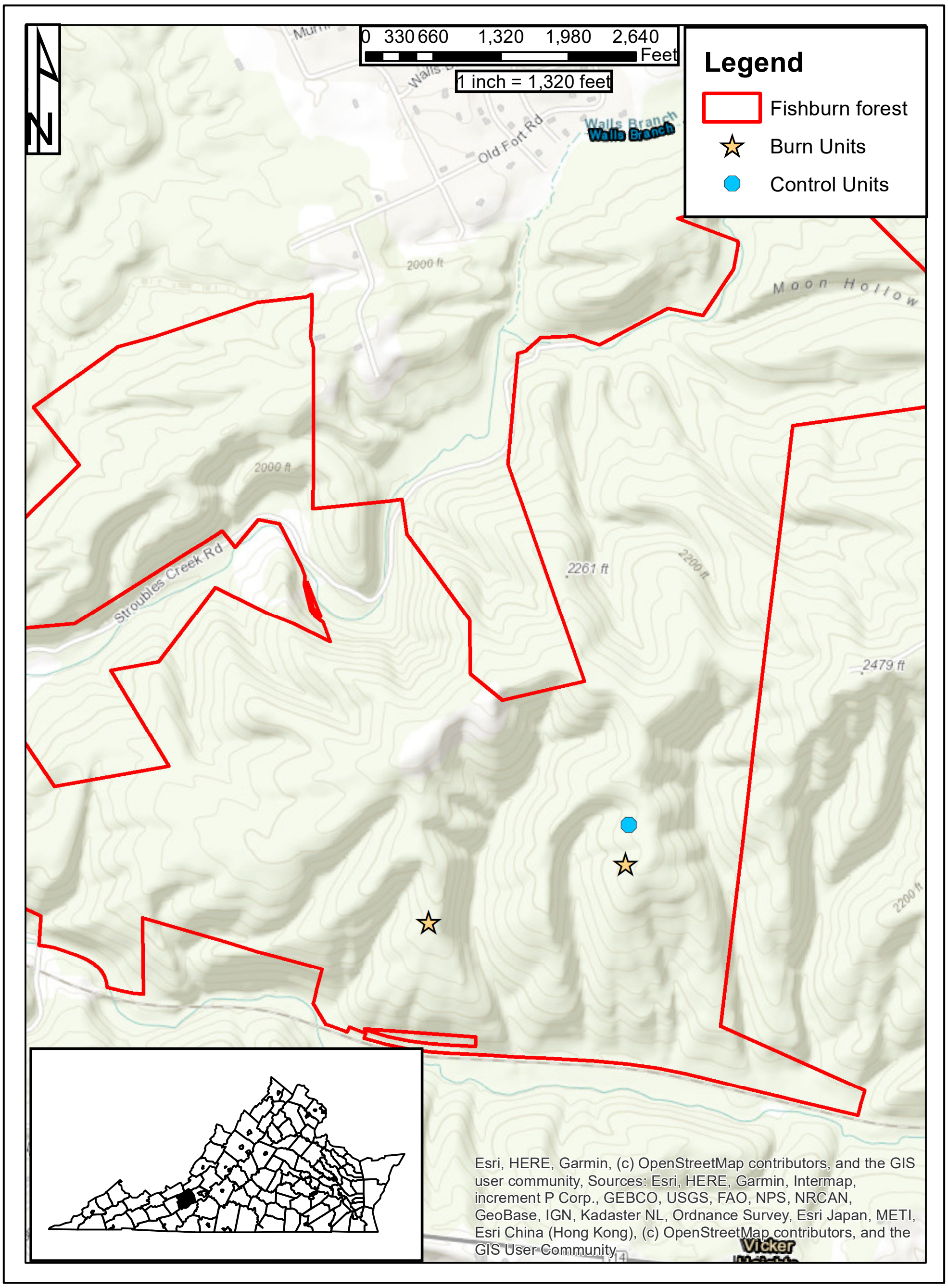

Figure 5.1. Study area of soil nutrient variation study within the Fishburn Forest (Montgomery County, Virginia, USA).

Table 5.1. Mineral soil (0-10 cm depth, Inceptisols) property mean (and Kruskal-Wallis score means) for western aspects of the Fishburn Forest (Montgomery County, Virginia, USA) sampled pre- and post-fire (6 and 14 months post-fire) in 2019 and 2020. Steel-Dwass distinctions (by row) are noted by letters in each cell ($\alpha = 0.05$).

Treatment	**Control (n = 9)**			**Burned (n = 18 for pH, Ca, K, Mg, P, C) (n = 17 for N, C:N)**			**p-value**
Time	**Pre-fire**	**6 months post-fire**	**14 months post-fire**	**Pre-fire**	**6 months post-fire**	**14 months post-fire**	
pH	3.94 (25.9) a	4.04 (36.4) a	4.00 (32.6) a	4.08 (38.6) a	4.18 (46.6) a	4.23 (51.8) a	0.0610
Ca ($mg\ kg^{-1}$)	79.44 (29.1) b	99.44 (30.2) ab	149.22 (37.2) ab	112.17 (35.3) ab	188.00 (54.8) a	170.22 (46.3) ab	0.0277
K ($mg\ kg^{-1}$)	29.44 (32.3) ab	41.33 (64.0) a	41.44 (54.4) ab	28.17 (29.2) b	31.11 (38.7) ab	31.67 (41.2) ab	0.0036
Mg ($mg\ kg^{-1}$)	17.56 (42.8) ab	18.11 (50.4) ab	16.22 (37.1) ab	17.94 (42.3) ab	19.22 (51.3) a	14.83 (25.9) b	0.0260
P ($mg\ kg^{-1}$)	8.44 (49.3) ab	6.78 (30.1) b	6.78 (31.6) b	10.00 (59.3) a	7.72 (41.2) ab	6.44 (28.5) b	0.0008
C (%)	5.91 (62.2) a	4.25 (54.8) a	2.88 (37.3) ab	3.86 (43.6) a	3.66 (46.7) a	1.82 (17.0) b	<0.0001
N (%)	0.18 (58.4) a	0.14 (48.8) a	0.12 (38.6) ab	0.15 (43.6) ab	0.13 (43.7) a	0.09 (21.0) b	0.0017
C:N	31.70 (57.7) a	30.32 (56.1) a	24.29 (34.2) ab	25.80 (34.9) ab	27.43 (47.2) ab	22.49 (25.3) b	0.0016

Chapter 6. Long-term impacts of silvicultural treatments on wildland fuels in the Ridge and Valley Province, Virginia (USA)

6.1 Introduction

Since 2013, over 50% of the United States Forest Service's discretionary funding has been allocated for wildfire suppression activities (Congressional Research Service 2018). Current wildfire projections through the year 2100 include increased wildfire size, suppression costs, and resource usage in highly developed areas and longer wildfire seasons (Collins & Knutti, 2013, Abatzoglou & Williams, 2016). Specifically, forests of the southeastern United States are projected to experience extended periods of drought, and increased wildfire incidence (Mitchell et al. 2014). For this reason, fuel reduction has increasingly been included in forest management plans for federal lands to reduce potential wildfire ignitions, mitigate extreme fire behavior, and reduce subsequent, negative wildfire effects, such as increased soil erosion (Robichaud et al. 2008, Moody & Martin 2009), air pollution (Liu et al. 2016, Goodrick et al. 2012, Cascio 2018) and residual tree damage or mortality (O'Brien et al. 2010, Varner et al. 2016, Schweitzer et al. 2019, Kreye et al. 2020).

Fuel loading as a stand-alone metric is not adequate in predicting wildland fire behavior. Spatial arrangement and continuity of forest fuels also influences wildland fire behavior (Rowell et al. 2016). For example, a continuous arrangement of forest fuels enhances the ability of a fire to spread across an area (Andreu et al. 2012). Additionally, fuel composition and vertical structure may influence fire behavior by increasing fire intensity (Ottmar et al. 2007, Mohr et al. 2010).

Research investigating fuel reduction treatments in the Ridge and Valley Province and Appalachian Mountains is limited in scope. This research has been primarily focused on the efficacy of prescribed fire, either alone, or in conjunction with mechanical treatments (Waldrop et al. 2016). Because applying prescribed fire to forests of the Appalachian

Mountains can be limited by short burn windows, lack of public support and qualified personnel (Ryan et al. 2013), mechanical treatments are often applied to reduce hazardous fuels. A common mechanical fuels reduction method is the mastication of shrubs and small diameter stems (McIver et al. 2009). These methods generally increase fuel loads in the short-term and can be costly to implement (Jernigan et al. 2016, Waldrop et al. 2016). Few studies in the eastern United States have quantified fuel loads following the removal of overstory stems either following intermediate treatments, such as thinnings (Waldrop et al. 2008, Vander Yacht et al. 2019) or regeneration harvests (Brose 2016). Of the current research, most studies compare fuel loads 3-5 years post-treatment (Graham and McCarthy 2006, Stephens et al. 2009, Philips and Waldrop 2013, Brose 2016, Waldrop et al 2016). Quantifying long-term effects of timber harvests on fuel loads will provide insight regarding the indirect effects of timber harvest operations on wildfire mitigation in the eastern United States and locations with similar land management goals around the globe.

The objectives of this study were to measure and compare 1) woody fuel mass and depth and 2) forest floor (i.e. litter and duff) fuel mass and depth across 3 different silvicultural treatments (clearcut, low-leave shelterwood, high-leave shelterwood) two decades after their implementation in the Ridge and Valley Province of Virginia. The initial harvests were designed to determine potential regeneration success. Due to the widespread use of these silvicultural systems within this region and others, we capitalized on this study design to investigate long-term changes in fuel loads resulting from these management practices. Hypotheses for this study were: (a) 1-hour (0-0.64 cm diameter), 10-hour (0.65-2.54 cm diameter), and 100-hour (2.55-7.62 cm diameter) woody fuel loads would be greatest following the low-leave shelterwood treatment compared to other treatments and the control, while 1,000-hour (> 7.62 cm diameter) fuel loads would be greatest in the high-leave shelterwood treatment compared to other treatments and the control; (b) total [woody + forest floor (or O Horizon)] fuel loads would be greatest in the control followed by the low-leave shelterwood, high-leave shelterwood, and clearcut; (c) fuelbed depths would be greater in treated units than in control units and forest floor depths (Oi and Oe+Oa) would be greater in control units than in treated units.

6.2 Methods

6.2.1 Study Areas

Three sites were utilized for this study: Blacksburg 1 (BB1), Blacksburg 2 (BB2), and New Castle (NC). All sites are in the Eastern Divide Ranger District of the George Washington and Jefferson National Forest (Table 6.1; Figure 6.1). The BB1 and BB2 sites are located in Montgomery County, Virginia and contain mixed-hardwood stands (approximately 110 years old). Common species present are chestnut oak (*Quercus montana* Willd.), white oak (*Quercus alba* L.), American beech (*Fagus grandifolia* Ehrh.), and red maple (*Acer rubrum* L.). Mean annual temperatures range from 2˚C (Winter) to 21˚C (Summer). Annual precipitation is 103 cm (Hammond 1997, Wender 2000, Sucre 2008, Atwood et al. 2009). Soils at the BB1 and BB2 sites consist mainly of the Jefferson series (Ultisols; Typic Hapludults). The silvicultural treatments described in section 2.2 were conducted at these locations in 1995 (BB1) and 1996 (BB2) (Atwood et al. 2009). The New Castle (NC) site is located in Craig County, Virginia (37.455216N, 80.382528W) and contains an approximately 70-year-old mixed-hardwood stand (Atwood et al. 2009). Common species listed for BB1 and BB2 are also present at the NC site. Mean annual temperatures range from 2˚C (Winter) to 21˚C (Summer). Annual precipitation is 95 cm (Hammond 1997, Wender 2000, Sucre 2008, Atwood et al. 2009). Soils at the NC site consist mainly of the Oriskany series (Ultisols; Typic Hapludults). Silvicultural treatments began in 1995 and were completed in 1996 (Atwood et al. 2009).

6.2.2 Silvicultural Treatments

Four, 2-ha stands were treated at each site. Silvicultural treatments were as follows: (a) clearcut (CC; in which all woody stems > 5 cm diameter at breast height (DBH) were felled, regardless of height - merchantable stems were removed and non-merchantable stems were left on-site); (b) high-leave shelterwood (HLS; in which overtopped and intermediate stems were removed, leaving 12-15 m^2 ha^{-1} in basal area of dominant or co-dominant trees); (c) low-leave shelterwood (LLS; in which dominant and co-dominant trees were removed, leaving 4-7 m^2 ha^{-1} in basal area of trees 5-25 cm dbh); and (d) control (C) with no treatment. No more than 10 stems ha^{-1} of mast, snag, or cull trees were permitted to be left on-site in the clearcut unit to maintain wildlife habitat (Hood 2001, [Figure 6.2]).

6.2.3 Study Design and Field Measurements

The silvicultural treatments at BB1, BB2, and NC have been studied previously by other investigators and were established in a randomized complete block design with each stand serving as a block (Hammond 1997, Wender 2000, Hood 2001, Sucre 2008, Atwood et al. 2009). Fuels within three replications of the clearcut, high-leave shelterwood, low-leave shelterwood, and control were quantified in our assessment, yielding a total of 12 stands. Within each of the 12 stands, 16 fuel inventory plots were established on a 30.5 m x 30.5 m grid. Down and dead woody fuels were tallied using a modified version of Brown's Planar Intercept Method (1974), similar to that implemented by Coates et al. (2019).

To summarize, this method used woody debris tallies along planar transects to approximate woody debris mass. Woody debris particles were distinguished based upon time-lag size classes (Cohen and Deeming 1985): 1-hour (0-0.64 cm), 10-hour (0.65-2.54 cm), 100-hour (2.55-7.62 cm), and 1,000-hour (>7.62 cm).

Each transect was 15.2 m in length with the first transect being oriented at 0, the second at 120, and the third at 240 (Figure 6.3). Along each transect, 1- and 10-hour fuels were tallied within the first 1.8 m and 100-hour fuels were tallied within the first 3.7 m. One thousand-hour fuels were tallied along the entire 15.2 m transect and were identified as either hardwood or softwood. Additionally, a decay class (sound or rotten) was assigned to each 1,000-hour fuel based upon that fuel's exterior physical properties (Maser et al. 1979, Lutes et al. 2006). If any portion of a 1,000-hour fuel's texture appeared soft when kicked, for example, that fuel was classified as rotten. Diameter was measured to the nearest 0.64 cm. Fuel counts were converted to masses using equations developed by Brown (1974) and modified by Coates et al. (2019):

For material with diameter $\leq$ 7.62 cm: $=2.24[(11.64)(n*d^2*s*a*c)/N*L]$

For material with diameter > 7.62 cm: $=2.24[(11.64)(d^2*s*a*c)/N*L]$

where:

2.24 = conversion factor of tons acre^{-1} to Mg ha^{-1}

11.64 = conversion factor of volume to tons acre^{-1}

n = the number of woody fuels tallied per timelag-size class

d = quadratic-mean-diameter of particles (in)

s = specific gravity of fuels (s=0.70, 0.58, 0.58, and 0.30 for 1- and 10-hour, 100-hour, 1000-hour sound, and 1000-hour rotten material, respectively [Anderson, 1982])

a = non-horizontal angle factor correction factor

c = slope correction factor

N = number of transects at each plot (N=3)

L = length (ft) of sampling plane (L=6 for 1- and 10-hour fuels; L=12 for 100-hour fuels; and L=50 for 1000-hour fuels)

Non-horizontal angle factor correction factor values from Brown (1974) were used in these equations. Slope percent is factored into this equation; and was measured using a Suunto clinometer along each transect (percent scale).

Litter, duff, and dead fuel height (defined as the length from the top of the Oi horizon to the top of a down and dead woody fuel particle lying along a planar transect) were measured at 3.7, 7.6, and 12 m along each linear transect. Additionally, litter and duff were destructively sampled within a 0.09 m^2 PVC plastic frame to determine forest floor mass.

Along each transect, one sample was collected at the end of the 120 and 240 transects and one sample was collected within 1.5 m of the origin for the 0 transect. This yielded three destructive samples of litter and duff per plot. Litter and duff samples were oven-dried for at least 72 hours at 65ºC then weighed to the nearest 0.1 g.

6.2.4 Data Analysis

Raw data failed to meet the equal variance assumption for one-way analysis of variance (ANOVA). Therefore, a square root transformation was applied to all woody fuel masses except sound and rotten 1,000-hour fuels which were transformed by square and cube root transformations, respectively. Additionally, a log(x+1) transformation was applied to the litter and duff masses, depths, and dead fuel height. An ANOVA was conducted for the following: woody fuel time-lag size class loads (1-, 10-, 100-, 1,000-hour sound, 1,000-hour rotten, and total 1,000-hour); fine fuel loads (1-hour + 10-hour + 100-hour); total woody fuel load (all down and dead woody fuels); litter mass; duff mass; forest floor mass (litter + duff); total fuel load (woody fuel + forest floor fuel mass); dead fuel height; litter depth; duff depth; forest floor depth (litter depth + duff depth). If differences were detected between treatments, a Tukey's Honestly Significant Difference (HSD) test was conducted to determine specific differences between treatments. All analyses were conducted using JMP Pro 14 (SAS Institute Inc., Cary, NC, USA). Differences were declared significant at $\alpha = 0.05$.

6.3 Results

Significant differences in fuel loads between treatments (Table 6.3) were found in all time-lag classes except for 100-hour ($p = 0.4679$) and 1,000-hour sound ($p = 0.3492$). The CC had the highest 1- ($p = 0.0044$) and 10-hour fuel loads ($p < 0.0001$), while the LLS had the most fine (1-, 10-, and 100-hour fuels) fuel loads ($p = 0.0253$). Rotten 1,000-hour fuel loads were greatest in the C ($p = 0.002$). Total 1,000-hour fuel loads were greatest in the C compared to all other treatments ($p < 0.001$). Total woody fuel load was greatest in the C ($p = 0.0018$), but was only significantly different from the CC and HLS.

Analysis of destructive samples of litter and duff revealed significant differences ($p < 0.0001$) based upon treatment, with the highest litter and duff loads in the C and CC units, respectively. Total fuel loads were greatest in the C treatment ($p < 0.0001$). Significantly greater duff depth ($p < 0.0001$) was found in the C relative to all other treatments (Table 6.4), while litter depth was not significantly different between treatments ($p = 0.2115$). Combined litter and duff depths were greater ($p < 0.0001$) in the C treatment relative to the treated units and fuelbed height was greater in the C and HLS relative to the CC ($p = 0.0081$). No difference was found in fuelbed height between the LLS and CC.

6.4 Discussion

6.4.1 Fuel Masses and Depths

Two decades following regeneration harvests of various intensities, C woody fuel loads were greater than treatment woody fuels. Overall, the HLS reduced fine fuels, litter, and duff loads more effectively, while the CC and LLS treatments reduced coarse woody fuels more effectively. Differences in litter depth were indistinguishable between treatments, but duff depths and dead fuel heights were lower in treated units compared to untreated controls. These responses highlight a secondary benefit, namely long-term reduced fuel levels, of commonly applied silvicultural treatments within the region.

Fuel loads for each size class and forest floor property in our study were noticeably lower than others reported in the southern Appalachian Mountains (Vander Yacht et al. 2019, Coates et al. 2019), however, our fuel loads were comparable to those reported by Brose (2009) in Pennsylvania. This emphasizes the spatial and temporal variability that may be present for fuel loads between similar ecosystems and ecoregions (Keane et al. 2013). Although, the CC treatment contained statistically more 1-hour fuels than the other treatments, this difference may not be significant from a fuel management perspective as the largest difference in 1-hour fuel loads between any two treatments was only 0.09 Mg ha^{-1}. The small difference in 1-hour fuels would most likely lead to minimal differences in fire behavior between the treated areas. Differences in 10-hour fuel loads between the LLS and CC treatments may be related to similarities in harvest intensity and the size of removed stems. Because larger trees were removed relative to the HLS treatment, damage during the LLS and CC felling and skidding processes may have facilitated greater 10-hour fuel loads as tree limbs were broken by equipment and falling trees. Clatterbuck (2006) found more damaged trees associated with harvesting operations in which a higher percentage of stems were retained. This increase in damaged stems could potentially increase post-harvest woody fuel accumulation. However, any increase in fuel loads resulting from harvesting alone may be less than additions resulting from the long-term use of prescribed fire because trees impacted by prescribed fire may be susceptible to delayed mortality, depending upon fire intensity and severity (Yaussy & Waldrop 2010). No significant differences in fine woody fuels were noted between the treated and control units, but this result is nuanced: 10-hour, litter, and duff masses were significantly less in the treated units than the control.

No significant differences in 100-hour fuel loads were found in our study. This result was similar to those reported in another study of fuel properties resulting from shelterwood harvests conducted in the Virginia Piedmont (Brose 2016). Conversely, rotten and total 1,000-hour fuels in the C were more than double those found in the treated units. This difference seems logical because larger stems were removed from the treated areas during the harvests.

The ability to quantify litter and duff characteristics is important for fire managers of the eastern United States for various reasons. These fuel sources influence not only fire behavior, but also contribute to secondary fire effects, such as soil erosion, overstory mortality, and nutrient loss (Fowler 2004, Johnston et al. 2004, Hiers et al. 2005, Ottmar & Andreu 2007). Litter and duff masses are often quantified by measuring litter and duff depth, and those depths are then converted to masses using bulk density equations. These equations vary by region and may generate unreliable mass estimates due to fluctuating correlations between mass and depth (Crosby and Loomis 1974, McNab et al. 1978, Ottmar et al. 2007, Stottlemyer et al. 2015). For this reason, destructive litter and duff samples were taken in addition to physical measurements of litter, duff, and woody fuelbed depth to obtain more site-specific assessments of potential forest floor mass differences.

Litter masses were significantly lower in two of the treated units (HLS, LLS) when compared to the C, however all values were comparable to litter masses found by Sharpe et al. (1980) in the southern Appalachian Mountains. Time since harvest was likely a factor as regeneration contributed to the litter layer following the harvests. Lower duff masses in the HLS and LLS stands compared to the C is consistent with Mushinski et al. (2017) as they found significantly lower duff masses twenty years after a whole-tree harvest in a loblolly pine (*Pinus taeda* L.) stand. Graham & McCarthy (2006) found that immediately following thinning in mixed-oak stands in Ohio, litter and duff masses increased, followed by a decrease in masses that reached equilibrium with the control within 3 years. The increase immediately following thinning was attributed to elevated decomposition rates due to changing site conditions. As vegetation regenerated, site conditions became comparable to the control units and decomposition rates decreased. No difference in litter depth coupled with differences in litter mass may suggest that litter was compacted. Compacted litter is less likely to burn due to a lack of oxygen needed for combustion and the likelihood that compacted fuel beds may require longer drying times to ignite (Waldrop & Goodrick 2012).

Ideally, fuel tallies would have been conducted prior to the implementation of the silvicultural treatments at these sites to provide a direct pre- and post-treatment comparison.

However, pre-treatment fuel loads were not measured. Another potential limitation of this study could be related to the use of Brown's Planar Intercept method for quantifying woody fuel loads. This method was developed in the western United States and although it is widely used in the eastern United States, there is no region-specific equivalent. Furthermore, Brown's Planar Intercept method has been documented to over-estimate the size and quantity of 1,000-hour fuels (Westfall and Woodall 2007).

6.4.2 Management Implications

Lower fuel loads were found in the treated stands 20+ years post timber harvest in Virginia. No additional treatments, such as prescribed fire or herbicides, were combined with harvesting at these locations. These findings indicate that active forest management may facilitate timber production and other societal benefits. Additionally, the types of harvests in this study often produce income for the landowner, thereby providing another incentive for their use. As with any biomass removal, the impacts of common silvicultural treatments on secondary site properties, such as nutrient cycling or wildlife habitat, should be considered.

6.5 Conclusions

While not the intended objective of the treatments, long-term reduction of forest fuel loads was achieved through the use of silvicultural harvests of varying intensities in the Ridge and Valley province of Virginia. Different fuel sizes and groups were reduced by different types of harvests. The removal of mostly overstory stems led to reductions in coarse woody debris while the removal of midstory stems facilitated reductions in fine woody fuels. Reductions in litter and duff masses were achieved, regardless of stem removal canopy position. Reduced forest fuel loads two decades post-treatment suggested that commonly applied silvicultural prescriptions may be effective at minimizing extreme wildfire behavior. When prescribed fire is unfeasible due to logistical constraints, these prescriptions might be used to reduce long-term fuel loads.

6.6 References

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Table 6.1. Site descriptions for the Blacksburg 1, Blacksburg 2, and New Castle sites.

Site	**Predominant Soil Series***	**Cover Type**[†]	**Average Slope**	**Aspect**	**Harvest Year**[††]
Blacksburg 1	Jefferson	Chestnut Oak	11%	Southeast	1995
Blacksburg 2	Jefferson	Chestnut Oak	16%	Southeast	1996
New Castle	Oriskany	Chestnut Oak	7%	Southeast	1996

*Web Soil Survey, Montgomery and Craig County, VA, [†]Eyre, 1980, [††]Hood, 2000

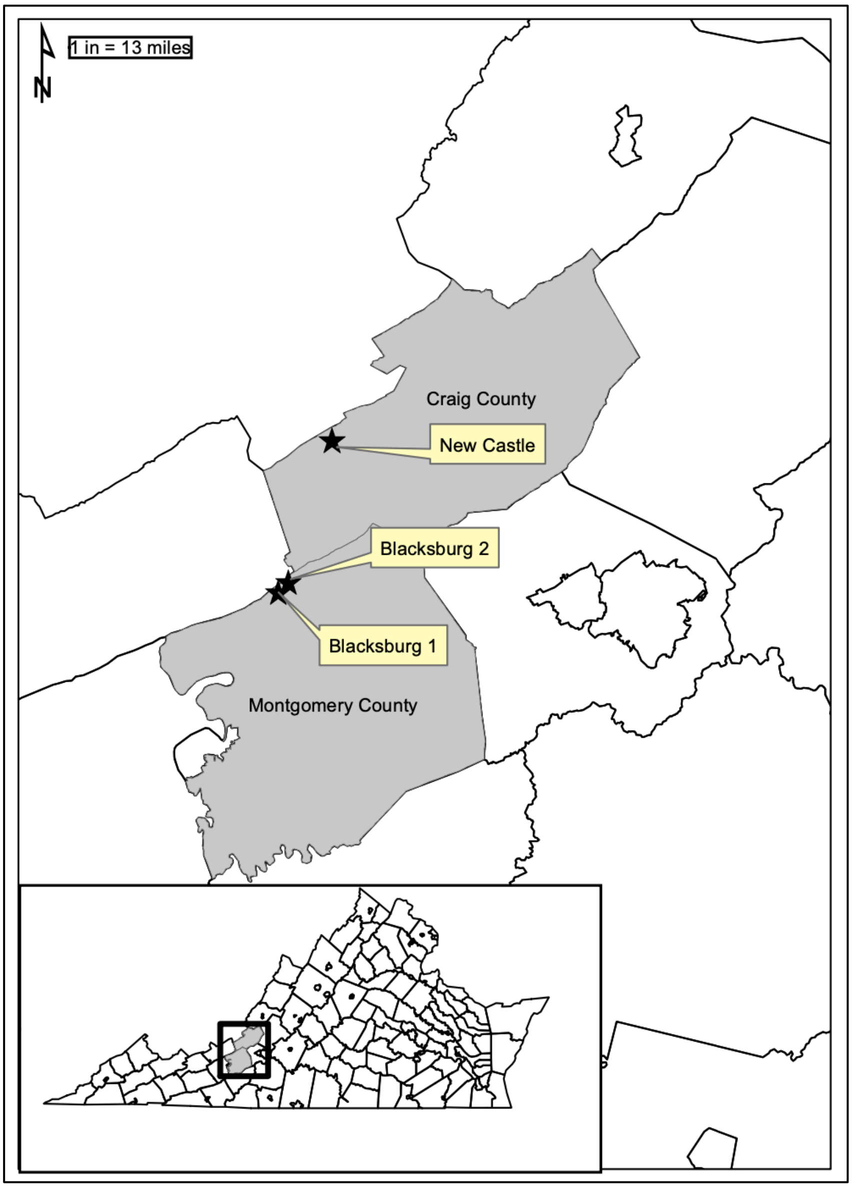

Figure 6.1. Study sites for long-term silvicultural fuel study, Ridge and Valley Province, Virginia, USA.

Table 6.2. Post-treatment basal area (m^2 ha^{-1}) for each site*.

Site	**Clearcut**	**High-leave shelterwood**	**Low-leave Shelterwood**	**Control**
Blacksburg 1	0.89	14.33	7.78	36.97
Blacksburg 2	2.78	17.66	5.40	35.24
New Castle	0.57	7.53	0.00	36.92

*Howell (In preparation).

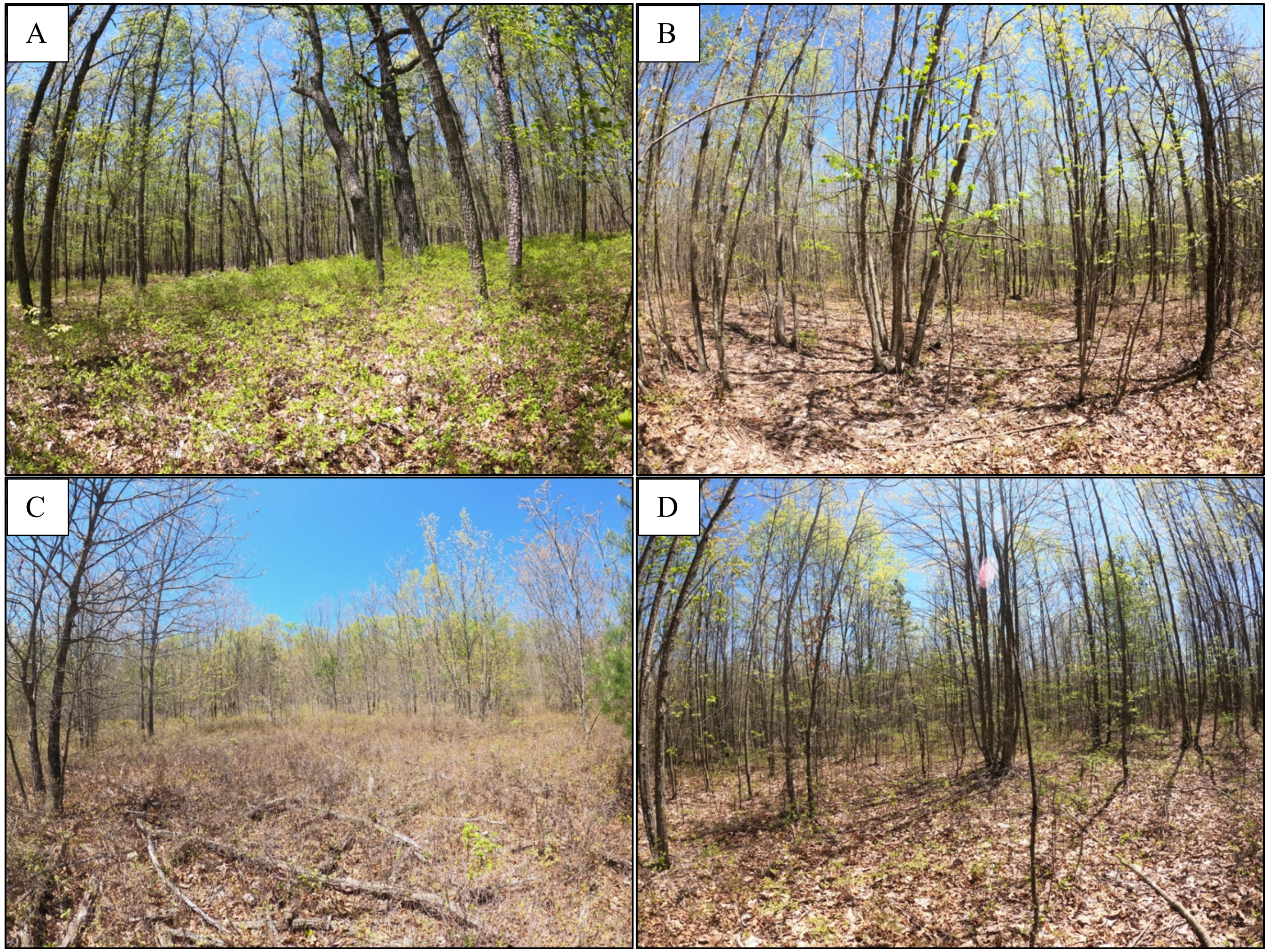

Figure 6.2. Photographs of silvicultural treatments at New Castle site. Control (A), low-leave shelterwood (B), high-leave shelterwood (C), and clearcut (D).

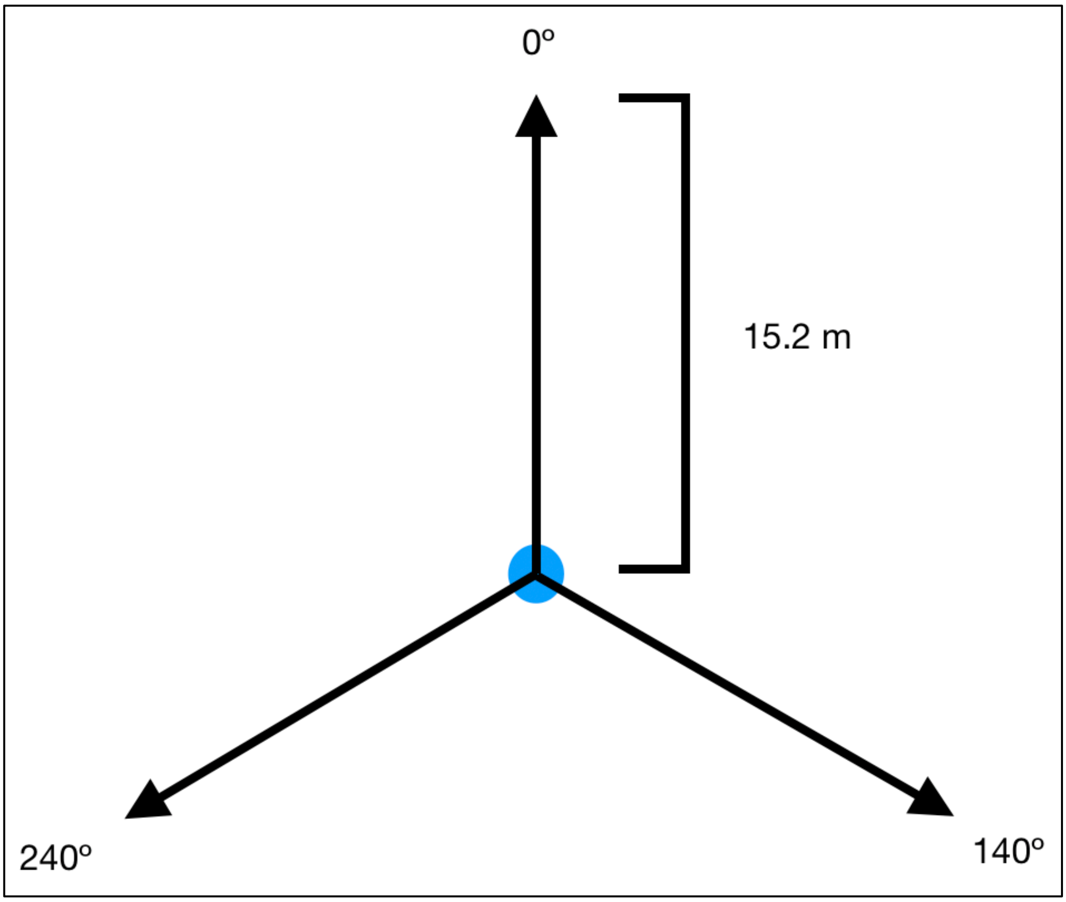

Figure 6.3. Brown's Planar intercept configuration utilized in this study. Three, 15.2 meter transects were established at 0º, 120º, and 240º.

Table 6.3. Woody fuel mass (Mg ha^{-1}) (standard error of the mean) comparisons between treatments. Different letters following mass and standard error are associated with significant differences within each row ($\alpha = 0.05$).

Fuel Size Class	**Mass (Mg ha^{-1})**				
	Clearcut	**High-leave Shelterwood**	**Low-leave Shelterwood**	**Control**	**p-value**
1-hr	0.36 ±(0.02)A	0.31 ±(0.03)B	0.29 ±(0.01)B	0.27 ±(0.02)B	0.0044*
10-hr	2.79 ±(0.18)A	1.68 ±(0.14)B	2.70 ±(0.18)A	1.69 ±(0.19)B	<0.0001*
100-hr	3.89 ±(0.36)A	3.50 ±(0.39)A	4.45 ±(0.45)A	3.85 ±(0.36)A	0.4679
1,000-hr Rotten	1.14 ±(0.23)B	2.18 ±(0.35)B	2.58 ±(0.60)B	4.16 ±(0.49)A	0.0002*
1,000-hr Solid	0.54 ±(0.48)A	1.02 ±(0.64)A	0.14 ±(0.9)A	2.47 ±(0.72)A	0.3492
Total 1,000-hr	1.69 ±(0.57)B	3.20 ±(0.75)B	2.72 ±(0.61)B	6.63 ±(0.86)A	<0.0001*
Fine Fuels (1-,10-,100-hr)	7.03 ±(0.44)AB	5.49 ±(0.41)B	7.43 ±(0.50)A	6.21 ±(0.51)AB	0.0253*
Woody Fuels (1-,10-,100-,1,000-hr)	8.72 ±(0.69)B	8.69 ±(0.98)B	10.15 ±(0.82)AB	12.85 ±(1.05)A	0.0018*
Litter Mass (Mg ha^{-1})	3.23 ±(0.13)AB	2.63 ±(0.09)C	3.00 ±(0.13)BC	3.47 ±(0.15)A	<0.0001*
Duff Mass (Mg ha^{-1})	4.86 ±(0.37)A	3.24 ±(0.23)B	3.57 ±(0.26)B	6.63 ±(0.61)A	<0.0001*
Litter and Duff (Mg ha^{-1})	8.09 ±(0.41)B	5.87 ±(0.26)C	6.57 ±(0.34)C	10.11 ±(0.62)A	<0.0001*
Total Fuel (woody, litter, duff)	16.81 ±(0.85)B	14.56 ±(1.00)B	16.72 ±(0.86)B	22.96 ±(1.29)A	<0.0001*

Table 6.4. Litter, duff, and fuelbed depth (standard error of the mean) comparisons between treatments. Different letters following depths and heights are associated with significant differences within each row ($\alpha = 0.05$).

Fuel Property	**Clearcut**	**High-leave shelterwood**	**Low-leave shelterwood**	**Control**	**p-value**
Litter depth (cm)	2.80 (0.10)A	3.02 (0.17)A	2.85 (0.13)A	3.16 (0.12)A	0.2115
Duff depth (cm)	2.29 (0.14)B	2.00 (0.15)B	2.05 (0.14)B	3.45 (0.18)A	<0.0001*
Forest floor depth (cm)	5.09 (0.22)B	5.02 (0.24)B	4.91 (0.22)B	6.61 (0.22)A	<0.0001*
Fuelbed depth (cm)	3.39 (0.30)B	5.15 (0.39)A	4.79 (0.51)AB	4.83 (0.32)A	0.0081*

Chapter 7. Conclusions and Management Implications

Wildland fire occurrence is on the rise in eastern US forests. Consequently, any land management agency, non-governmental organization (NGO), or industrial forest landowner with a fire management program needs information on fire and fire effects. In addition to firefighter safety and health considerations, fire management plans often contain elements related to: minimizing potential negative effects of fire on non-target forest resources, achieving specific goals and objectives, maintaining fuel composition and arrangement, and educating the public about wildland fire. This dissertation intended to provide insight into the effects of wildland fire on forest resources in the central Appalachian Mountains. In doing so, each of the aforementioned components of a fire management plan were addressed.

With regard to minimizing negative effects on non-target resources, our literature review indicates that in most cases negative impacts on water quality caused by low-intensity, low-severity prescribed fires in the eastern United States are short-lived. Furthermore, positive impacts, such as increased water yield and biological diversity and reduced disinfection by-products, have been noted. Because most prescribed fires in the eastern United States take place in Piedmont and Coastal Plain forests, additional research is needed to form a more complete understanding of how prescribed fire affects water quality in mountainous terrain dominated by hardwood species. Related to soil chemistry, changes to mineral soil chemistry following a single-entry, dormant season, prescribed fire were noted. However, these changes are difficult to generalize due to the inherent variation in both soil properties and prescribed fire behavior. Because of this, observed changes in soil nutrients post-fire should be interpreted carefully. Forest managers should also recognize that decreases in nutrient levels are not necessarily incompatible with their forest management goals.

A common objective for using prescribed fire in the central Appalachians is oak, hickory, and pine regeneration. Our observational study indicates that site factors, such as aspect and elevation, combined with fire behavior and time since burn influence post-fire regeneration response and forest floor properties. Additionally, long-term monitoring of post-fire effects is important due to the dynamic nature of forest ecosystem properties following disturbance. The ability to determine relationships between vegetative responses and other site factors will help forest managers meet their forest management objectives. Finally, the use of mechanical or chemical treatments is most likely necessary to promote advance oak and pine regeneration in the central and southern Appalachian Mountains.

Most research concerning fuels management is concentrated on using either prescribed fire, understory mechanical treatments, or some combination of both to reduce woody fuels. We assessed the long-term fuel reduction potential of traditional forestry practices, such as clearcutting and shelterwood harvesting, even though these treatments were not installed for that purpose. These treatments were shown to be effective at reducing forest fuels of various size classes long-term. Specifically, fuels of the 1- and 10-hour size class, in addition to litter and duff masses, were reduced in high-leave shelterwood stands. These fuels are a primary fuel source of both wild and prescribed fires in this region. As such, they are important for forest managers to consider when forest management practices are applied. Coarse woody fuels (1,000-hour) were reduced in the low-leave shelterwood and clearcut treatments, most likely due to the removal of larger diameter stems during harvesting. Fuels of this size class may lead to smoldering fire behavior and increased duff consumption. This is promising and can help forest managers reach silvicultural goals while simultaneously minimizing extreme fire behavior.

Traditionally, research results concerning wildland fire management are disseminated through scientific journal articles. While this format is appropriate for fellow researchers, and to some extent practitioners, it is not conducive for the circulation of knowledge among landowners and the general public. Cooperation between federal and state agencies, public universities, cooperative extension programs, and conservation groups may enhance information distribution to a broader audience. In doing so, educational programs should be developed and modeled to highlight the many facets and considerations related to wildland fire. Such programs will inform the public of the multiple benefits of active, forest management.

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