

**SHORT- AND LONG-TERM EFFECTS OF PRESCRIBED FIRE ON SOIL
PROPERTIES IN A PINUS RESINOSA FOREST IN NORTHERN MINNESOTA**

By

Joshua A. James

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ABSTRACT

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Prescribed fire is a widely used management tool, yet there are few studies investigating the short- and long-term effects of prescribed fire on soils within region to assess the effectiveness and compatibility of forest management objectives. Therefore, we leveraged a historical fire study (conducted 1959–70) with measurements in 2015 to evaluate the effects of prescribed fire season (dormant, summer), frequency (annual, biennial, periodic), and time (>45 years post-fire) since fire on soil properties in a red pine (*Pinus resinosa* Ait) forest in northern Minnesota, USA. We used a combination of statistical approaches including meta-analysis, non-metric multidimensional scaling (NMDS), analysis of variance (ANOVA), and linear regression to evaluate treatment effects and relationships of soil properties.

Prescribed fire treatments had legacy effects (>45 years post-fire) on many soil properties including N, P, K, Ca, pH, and forest floor depths but few persistent effects on C and PyC stocks and PyC concentrations. Short- and long-term soil properties appeared to differ by season of burning, and increased fire frequency within season magnified seasonal responses. In general, summer burns decreased nutrient stocks, whereas dormant season burns increased nutrient stocks. Our results suggest that summer burns may be a valuable approach to increase the variability in burn schedules more representative of historical regional fire regimes in red pine forests, and may help promote soil characteristics that maintain overall ecosystem health while supporting carbon sequestration objectives.

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CHAPTER 1

INTRODUCTION

1.1. Changing Fire Regimes and Ecological Legacies

Fire-adapted forest communities respond to changes in fire regime. Fire regimes, characterized by spatial and temporal patterns and effects on ecosystems have been altered by prolonged fire suppression policies as well as the contemporary use of prescribed fire (Brown and Smith, 2000; Krebs et al., 2010) that may have ecological legacies. Ecological legacies are long-lasting effects of disturbances, including fire, on soil and biota that are evident well after the disturbance (Foster et al., 2003). Natural and anthropogenic ecological legacies have been shown to influence forest resilience and alter ecosystem trajectories (Foster et al., 1998; Olga et al., 2012). For example, fires that occur outside the natural range of variability of historical fires may shift systems to novel alternative stable states and alter forest community structure and responses of ecosystems to future disturbances (Foster et al., 2003; Johnstone et al., 2016). These new conditions may not be reverted, or require significant alterations to recover previous stable state conditions (Johnstone et al., 2016). However, within-region use of prescribed fire and its potential legacy effects on soil and forest responses remain unclear. Recognizing the mechanisms that affect disturbance legacies will assist managers to anticipate when and how fire-adapted ecosystems respond to alterations in fire regimes and to guide current and future management decisions.

1.2. Red Pine Ecosystems of the Lake States Region

Red pine (*Pinus resinosa* Ait.) forests of the Lake States region are a fire-dependent ecosystem that have been impacted by changes in fire regimes. Historically, forests of the region consisted of mixed-pine, dominated by red and white pine (*Pinus strobus* L.) prior to European settlement (Anand et al., 2013). The distribution and abundance of these forest communities within the region was predominantly related to naturally occurring fire events (Leahy et al., 2003). Red pine ecosystems in the Lake States region co-evolved with a low to mixed severity surface fire regime (Drobyshev et al., 2008) with an irregular return frequency of ~ 30 years (Bergeron et al., 1990). The occurrence of high frequency, mixed-severity fires, encouraged red pine establishment and regeneration (Van Wagner, 1970; Heinselman, 1973; Dickmann, 1993). The dependence of red pine on these fire characteristics is due to several factors. First, red pine is shade-intolerant and fire increases accessibility of light with mortality of aboveground vegetation (Flannigan, 1993). Second, the species requires mineral soil for seedling establishment which is accomplished via fire to combust forest floor horizons (Alban, 1977). Third, fire often decreases soil nutrient stocks through volatilization of organic matter and nutrients, therefore promoting nutrient-poor conditions that favor red pine (Parker et al., 2006). Finally, fire decreases understory and fire-intolerant species that compete with red pine (Weyenberg and Pavlovic, 2014). Regeneration of red pine requires survivorship of mature trees as the cones are not serotinous and the species is incapable of reproducing vegetatively (Flannigan, 1993). Mature red pine bark is thick (2.5 cm) and able to withstand low-intensity fires but may succumb to injury or mortality from high-severity surface or crown fires (Dickmann, 1993; Van Wagner, 1970). The needles are extremely flammable and, in pure stands, this ecosystem may be one of the most flammable species in eastern North America (Flannigan, 1993). This characteristic of

red pine needles as well as the well-drained sandy soils typical of red pine forests, promotes frequent fire (Van Wagner, 1987). High severity fires have the potential to induce mortality of red pine, whereas minimally intense fires or infrequent fires will not provide suitable conditions for red pine establishment and growth (Dickmann, 1993; Flannigan, 1993; Van Wagner, 1970).

With the arrival of European occupants, an era of change to the Lake States forests began. Significant changes in forest ecosystems were a result of even-aged clear-cutting practices, active fire suppression, and catastrophic high severity fires (Bergeron and Brisson, 1990; Nyamai et al., 2014; Rist, 2008; Van Wagner, 1970). As a result, red pine ecosystems have been impacted by shifts in species composition, decreased regeneration, and excess accumulation of fuels that pose public safety concerns (Henning and Dickmann, 1996; Nyamai et al., 2014; Scherer et al., 2016; Tappeiner, 1971). Prolonged fire suppression has favored fire intolerant species including deciduous trees and understory species such as *Corylus* spp. that compete with red pine seedlings (Alban, 1977; Buckman, 1964; Tappeiner, 1971).

Fire-dependent ecosystems of the Lake States region have important ecological and economic value. Red pine forests provide a diverse number of ecological value including ecosystem services that are a result of naturally occurring processes that directly or indirectly benefit society (Jax et al., 2013). Examples of these ecosystem services include, but are not limited to, carbon sequestration (Makkonen et al., 2015), pollination reservoirs, recreation, cultural heritage, and wood products. Red pine forests of the Lake States region also have significant economic value that is often a function of ecological value. Financial products of red pine forests include recreational merchandise and numerous wood products including lumber and fuel (USDA, 2009). Because of its important economic value, the harvest, processing, distribution, and sale of red pine products supports local and national economies. Understanding

historical disturbance regimes and ecological functions of fire is important to secure these values and guide future management of red pine forests (Ryan and Noste, 1985; Knapp et al., 2009; Association for Fire Ecology, 2013).

1.3. Prescribed Fire in Red Pine Ecosystems

Prescribed fire is a management tool that may be used to mitigate the effects of undesirable changes in forest structure, excess fuel loading (Cassagne et al., 2011), and decreased natural regeneration (Switzer et al., 2012) as a result of fire suppression in red pine forests. However, contemporary implementation of prescribed fires nationwide (Knapp et al., 2009) and within in the Lakes States region often does not reflect historical regional wildland fire season, frequency, and intensity of fire. Historically, wildland fires in the Lakes States region occurred during dormant (i.e., spring or fall) and summer seasons, and were often associated with sporadic drought events during summer or late fall (Heinselman, 1973). Yet, dormant season prescribed fires are commonly implemented more frequency than summer burns due to logistical constraints of summer prescribed fires (Dickmann, 1993; Melvin, 2015; Quinn-Davidson and Varner, 2012). Summer prescribed fires have a higher risk of escape as they are generally conducted when relative humidity and fuel moisture content is lower (Knapp et al., 2009; Weyenberg and Pavlovic, 2014), relative to dormant season burns. These conditions often result in increased fire intensity (i.e. energy release) and behavior (i.e., flame length, rate of spread, etc.) in summer burns due to the interactions among weather, fuels, and topography. Fire suppression efforts also reduces the availability of local financial and personnel resources for summer prescribed fires (Quinn-Davidson and Varner, 2012). Burning is more economically feasible as the scale of prescribed fire increases but also poses greater containment risks. Public

perception and acceptance of burning is also a barrier to the use of prescribed fire (Melvin, 2015; Quinn-Davidson and Varner, 2012), including within the Lake States region. Protection of public safety as well as infrastructure and property is paramount when conducting prescribed fire which deters the use of summer prescribed fires that often pose a higher risk to these values. The use of prescribed fire carries with it many considerations, financial investments, and safety precautions. Despite these limitations, prescribed fire is gaining scientific and land manager acceptance and is being implemented at local and national scales to restore fire to fire-dependent ecosystems (Neary et al., 2005; Ryan et al., 2013). Efforts by local entities, such as the Lake States Fire Science Consortium, to increase fire ecology awareness and adoption of fire science may lead to the increased use of prescribed fire (Miesel et al., 2012) and influence management of wildland fires.

1.4. Fire Effects on Soil Properties

Fire can cause physical, chemical, and biological changes in soils (Certini, 2005; Knapp et al., 2009; Neary et al., 1999) (Table 1.1.). Deviations from historical fire regimes, including changes in fire season, frequency, and severity can influence post-fire vegetation recovery and ecosystem resiliency over the short- and long-term (Alban, 1977; Johnstone et al., 2016; Knapp et al., 2009; Tappeiner and Alm, 1975). Nutrient availability as well as losses and additions of nutrients to the soil as a result of fire are the most common processes (Neary et al., 2005) and are often related to soil temperature which is affected by fire intensity (i.e., energy released) (Figure 1.1.). Globally, soil functions as the largest terrestrial pool of carbon, storing more carbon than aboveground vegetation and the atmosphere combined (Jobbágy and Jackson, 2000), and has important implications in mitigating the impacts of climate change (Gonzalez-Perez et al., 2004).

Soil also represents a significant pool of fire-affected carbon in fire-dependent ecosystems that has been shown to influence soil nutrient availability and forest nutrient cycling (Pingree and DeLuca, 2017). Soil carbon as well as other soil properties are influenced by local differences (i.e., forest, fire, and soil type, and time since fire), and emphasize the need for region-specific estimates of soil responses (Alban, 1977; Johnson and Curtis, 2001; Knapp et al., 2009; Nave et al., 2011). Prescribed fire is a common management tool implemented to reduce the risk of high severity wildland fires and meet management objectives for silvicultural applications and ecosystem restoration, yet the short- and long-term effects of prescribed fire on soil properties remains unclear at local scales (Dickmann, 1993; Knapp et al., 2009; Ryan et al., 2013).

1.5. Prescribed Fire Soil Studies

One of the earliest attempts to understand the effects of fire on soils and its use as a management tool included a study on loblolly pine (*Pinus taeda* L.) in the Santee Experimental Forest, South Carolina. The study was initiated in 1946 and continued for 43 years through 1989 until Hurricane Hugo destroyed the overstory pines (White et al., 1990). This study established its regional significance and precedence for future long-term soil research.

Information on the effects of fire on soils of the Lakes States region is limited (Miesel et al., 2012). As a result, a second long-term study was established in 1959 in the Cutfoot Experimental Forest, Minnesota (Figure 1.2.), with the intention of replicating the study in the Santee Experimental Forest. The objective of the *Red Pine Prescribed Burning Experiment* was to provide information about the effects of contrasting prescribed fire treatments on red pine productivity, understory growth, and soil properties (Buckman, 1964; Alban, 1977). The study treatments, maintenance, and measurements were conducted through 1970 (Table 1.2.).

Two publications from the study were made available in 1964 and 1977 by Buckman and Alban, respectively. Buckman (1964) suggested that as few as two summer annual burns significantly decreased forest floor depths and understory shrubs (*Corylus* spp.) compared to less frequent summer burns or any frequency of dormant season burns. Alban (1977) concluded that ten years of prescribed fire decreased nutrients in the forest floor while increasing nutrients in the mineral soil without affecting site productivity. These two publications suggested that prescribed burning, when used appropriately, can reduce fuel loading, understory competition, and promote conditions for natural seedling regeneration without negatively affecting overstory production. Recent measurements at this study site in 2005 have shown that prescribed fire treatments have resulted in persistent decreases in understory shrub (*Corylus* spp.) stem densities >35 years post-fire. However, the long-term effects of prescribed fire and potential ecological legacies on soil properties since the last burning treatments remains unknown.

1.6. Research Objectives

We leveraged the *Red Pine Prescribed Burning Experiment* study site and initial raw datasets collected from 1959-1969 and also re-measured the original set of soil properties and measured additional soil properties in 2015 (> 45 years post-fire) to address the following research questions: how does prescribed fire season and frequency influence: (1) Short-term and intermediate trends over > 10 years (1959-1969) as well as cumulative effects of treatments on soil responses across years (1959-2015)? (2) Long-term soil responses and changes over time to treatments >45 years post-fire? (3) Total C and PyC concentrations and stocks and relationships with PyC and other soil properties >45 years post-fire?

My thesis addresses these questions in the two following research chapters. Chapter 2 addresses the first two questions in assessing short-term and intermediate trends (1959–1969) as well as cumulative effects of treatments (1959–2015) on soil responses using a meta-analysis approach, whereas we used non-metric multidimensional scaling (NMDS) ordination to investigate legacy (> 45 years post-fire) treatment effects. Chapter 3 addresses our third question and uses analysis of variance (ANOVA) to quantify treatment effects and time since fire (1970–2015) on soil C and PyC stocks and PyC concentrations and simple linear regression to assess the relationship of PyC with soil properties reported in our second chapter. Chapter 4 is intended to promote fire ecology science directed towards federal, state, local, and private stakeholders of the Lake States region. This chapter provides a brief summary of our research findings, limitations, and applied use of prescribed fire and its effects on soil properties to meet management objectives in red pine forests of the Lake States region.

APPENDIX

Table 1.1. Summary of common soil properties along with description and influence/function of properties affected by the direct and indirect effects of fire on soil physical, chemical, and biological variables. An asterisk [*] denotes an element's function in plant cellular activity and/or growth.

Properties	Description	Influence and Function	References
Nitrogen (N)	Most limiting nutrient; sources of N includes plant and animal residues (organic) and ammonium (NH ₄ ⁺), nitrate (NO ₃ ⁻), and atmospheric (N ₂) (inorganic)	Volatilization is most responsible for N loss and is directly proportional to organic matter loss; most N is not directly available to plants, N fixation by bacteria may increase post-fire with changes in soil pH and ash content; effects long-term productivity and encourages post-fire plant growth, [*] proteins (amino acids), energy-transfer (ATP), component of chlorophyll, and nucleic acids (DNA, RNA)	Neary et al., 2005
Phosphorous (P)	Second most limiting nutrient, anion, exists as Al/Fe precipitates in acidic soils	Does not exist in elemental form, organic matter is a source of organic P, highly available in surface ash of post-fire; plant absorption is affected by pH and mycorrhizae activity [*] energetic structures (ATP), cellular division, nucleic acids (DNA, RNA)	Alban, 1977; Neary et al., 2005
Potassium (K)	Base cation	Amount and composition determines base saturation [*] transport of sugars, stomata control, ion balance, disease resistance	Neary et al., 2005
Calcium (Ca)	Base cation	Amount and composition determines base saturation [*] building block of cell walls, cytoskeleton, disease resistance	Neary et al., 2005
Magnesium (Mg)	Base cation	Amount and composition determines base saturation [*] component of chlorophyll, DNA and RNA synthesis	Neary et al., 2005
pH	Hydrogen/hydroxyl ion concentration	Influenced by production of organic acids within organic matter, base/non-base cycling vegetation; influences CEC, plant nutrient availability, buffering capacity, microbial activity, and soil pedogenesis	Alban, 1977; Neary et al., 2005
Cation exchange capacity (CEC)	Exchangeable cations adsorbed to negatively charged particles (clay, organic matter)	Influences nutrient availability through adhesion of opposing charged particles (cations); pH buffering, soil pedogenesis; amount and type of clay and organic matter strongly influences CEC including cation availability and retention	Alban, 1977; Neary et al., 2005
Bulk density (BD)	Measure of compaction; mass per unit volume (g/cm ³)	Influences porosity, water infiltration, erosion, plant rooting, and microbial activity	Alban 1977; Neary et al., 2005
Forest Floor (depth)	Organic horizons; litter (Oi), fermentation (Oe), humus (Oa)	Source of organic matter and C; influences fire severity, moisture content, depth to mineral soil, and microbial activity	Neary et al., 2005

Table 1.2. Description of prescribed fire treatments implemented in the original *Red Pine Prescribed Burning Experiment* (1959-1970) in the Cutfoot Experimental Forest, Minnesota, USA. Prescribed fire treatments are shown by season, frequency, interaction of season and frequency, burn dates, and number of times burned, for n=4 replicates per treatment and control.

Treatment				
Season	Frequency	Trt	Burn dates (month/year)	Number of burns
Control	Control	CC	-	0
Dormant	Annual	DA	5/1960, 5/1961, 5/1962, 4/1963, 5/1964, 10/1964, 5/1966, 5/1967, 5/1969, 5/1970	10
	Biennial	DB	5/1960, 5/1962, 5/1964, 5/1966, 5/1969	5
	Periodic	DP	5/1960, 5/1969	2
Summer	Annual	SA	8/1960, 6/1961, 8/1962, 6/1963, 6/1964, 7/1965, 8/1966, 7/1967, 7/1968, 8/1969, 7/1970	11
	Biennial	SB	7/1960, 8/1962, 6/1964, 8/1966, 7/1968	5
	Periodic	SP	7/1960, 7/1967	2

Effects of prescribed fire on soil properties

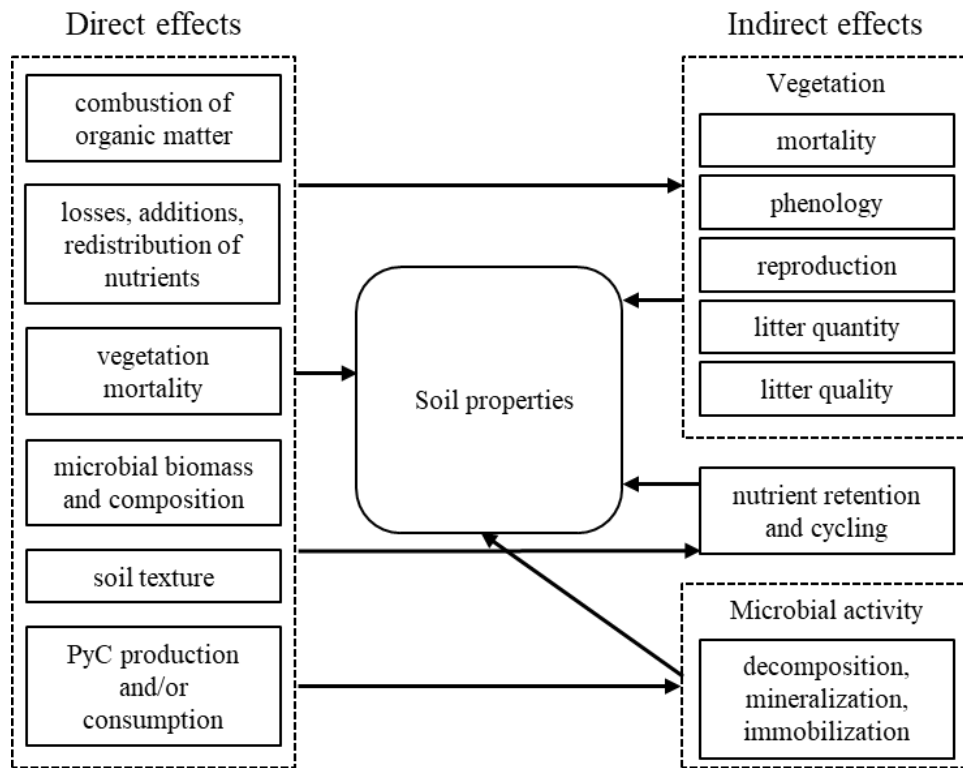


Figure 1.1. Conceptual diagram of the direct and indirect effects of prescribed fire on soil properties including pyrogenic carbon (PyC), vegetation, and microbial activity influencing short- and long-term soil responses.

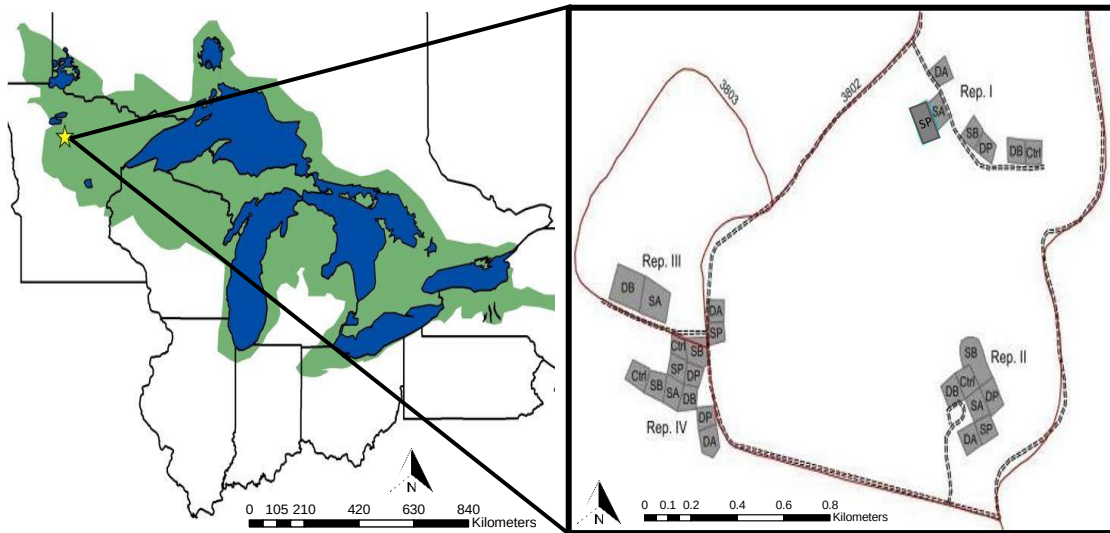


Figure 1.2. The *Red Pine Prescribed Burning Experiment* study site (left) is located in the Cutfoot Experimental Forest, Minnesota, USA (regional map from www.lakestatesfiresci.net). The historical study (1959-1970) established and used a randomized complete block (RCBD) design with four blocks (denoted Rep. I, II, III, IV on the inset panel), each of which contained one replicate of each of the seven prescribed fire treatments within block.

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CHAPTER 2

LEGACY EFFECTS OF PRESCRIBED FIRE SEASON AND FREQUENCY ON SOIL PROPERTIES IN A *PINUS RESINOSA* FOREST IN NORTHERN MINNESOTA

2.1. Abstract

Prescribed fire is a widely used ecosystem management approach and the vast majority of burns are conducted during the dormant season; however, these burning conditions (and therefore the type and persistence of fire effects) often differ from those of natural or historical fire regimes. Therefore, we leveraged a historical study (conducted 1959–70) with remeasurements in 2015 to evaluate effects of fire season (dormant, summer), frequency (annual, biennial, periodic), and their interaction on soil physical and chemical properties in a red pine (*Pinus resinosa* Ait.) forest in northern Minnesota, USA. To protect against across-year differences in sampling and analysis, we used a meta-analysis approach to evaluate treatment effects on soil properties. We also used non-metric multidimensional scaling (NMDS) ordination to investigate legacy (> 45 years post-fire) treatment effects.

The greatest effects of fire occurred in organic horizons, and decreased with depth. In the short-term, fire decreased organic horizon depths and nitrogen (N) and increased base cations (K, Ca, Mg) and pH in the mineral soil, whereas effects on phosphorus (P) were variable. Prescribed fire treatments had legacy effects on organic horizon and mineral soil properties >45 years post-fire. In general, summer burns decreased nutrient stocks, whereas dormant season burns increased nutrient stocks, and the majority of legacy effects occurred in annual burn treatments, in both seasons. Legacy effects of summer burns decreased organic horizon depths, organic matter, nutrient stocks (N, P, K), and pH, as well as lower (0–15 cm) mineral soil N; whereas, the dormant annual burn increased Ca in the total forest floor and N and P in the upper (15–91

cm) mineral soil. In contrast, the summer annual burn increased P, whereas the dormant annual burn decreased pH in the lower mineral soil. Trends in short- and long-term effect sizes appeared to differ by season of burning and further magnified by increased fire frequency within season. Relative to dormant season burns, summer burns resulted in immediate and long-lasting desirable effects for red pine ecosystems (e.g., decreased forest floor depths and nutrient stocks) without persistent undesirable effects (e.g., increased nutrient stocks or changes in cation exchange capacity, soil texture, and bulk density) in the mineral soil. Our results suggest that summer burns may be a valuable approach to increase the variability in burn schedules representative of historical regional fire regimes in red pine forests, and may help promote soil characteristics that support overall ecosystem health.

2.2. Introduction

Forest soils respond to changes in fire regime. Fire regimes, characterized by local spatial and temporal patterns and effects on ecosystems, have been altered by decades of prolonged fire suppression policies as well as contemporary use of prescribed fire that may have legacy effects on soil properties (Brown and Smith, 2000; Foster et al., 2003; Krebs et al., 2010). Historically, regional fire regimes were responsible for maintaining forest structure, species composition, and soil nutrient dynamics (Van Wagner, 1970; Ryan et al., 2013). Red pine (*Pinus resinosa* Ait.) forests of the Lake States region are an example of an ecosystem type that has developed on well-drained, nutrient-poor, sandy soils with a fire regime of low to mixed severity surface fires (Drobyshev et al., 2008) occurring with an irregular return frequency of approximately 30 years (Bergeron and Brisson, 1990). Historically, these fires occurred during dormant (i.e., spring or fall) and summer seasons, and were associated with localized drought events and human activity

(Heinselman, 1973; Guyette et al., 2016). Fires encouraged red pine establishment and regeneration by reducing overstory canopy density and understory competition as well as by preparing mineral seedbeds by combusting forest floor organic matter (Van Wagner, 1970). Similar to other fire-dependent ecosystems, red pine forests have experienced significant alterations in fire regimes that have resulted in shifts in species composition, mesophication (Nowacki and Abrams, 2008), structurally simplified stands, excess accumulation of fuels, and decreased natural regeneration (Cleland et al., 2004; Frelich, 1995). Prescribed fire is a management tool that may be used to mitigate the effects of prolonged wildland fire suppression and is being increasingly implemented at local and national levels to restore fire to fire-dependent ecosystems (Ryan et al., 2013). Dormant season prescribed fires are commonly implemented due to the weather, operational, and safety constraints associated with summer season prescribed fires (Quinn-Davidson and Varner, 2012; Melvin, 2015). Yet, contemporary implementation of infrequent dormant season prescribed fires in the Lakes States region may not reflect historical regional variability of wildland fire season, frequency, and intensity (Van Wagner, 1968; Heinselman, 1973; Dickmann, 1993). The effects of contrasting seasons and frequencies of prescribed fire on soils and ecosystem trajectories are poorly understood, yet are required to elucidate local responses of fire-adapted communities.

Fire influences physical, chemical, and biological properties of soils. Losses and additions of nutrients to the soil are a common effect of fire, and are closely associated with fire intensity (i.e., energy released) (Neary et al., 2005). For example, soil organic matter and nitrogen are volatilized at relatively low temperatures (200–450 °C), whereas phosphorous and base cations (potassium, calcium, magnesium) require much higher temperatures (770–1240 °C) to volatilize (Neary et al., 2005). Nutrients may be lost via volatilization into the atmosphere,

transported off-site by erosion, or remain *in situ* as post-fire ash deposits and immobilized by soil microorganisms and vegetation or translocated into the mineral soil profile (Certini, 2005; DeBano, 2000). Soil temperature during fire depends in part on fire intensity and may vary widely within and across season and frequency of prescribed fires (Keeley, 2009; Wittenberg, 2012). For example, seasonal dissimilarities may be driven by differences in fuel moisture, with summer fires often characterized by higher fire intensities than dormant season conditions (Govender et al., 2006), whereas increased frequency of fire within season may magnify seasonal effects of fire (Busse et al., 2014). Thus, the season as well as the number of burns conducted both have potential to influence ecosystem responses to fire.

The season and frequency of prescribed fire in red pine forests have direct and indirect effects on overstory and understory vegetation community composition and structure (Buckman, 1964; Henning and Dickmann, 1996; Weyenberg and Pavlovic, 2014; Scherer et al., 2016). Immediate and persistent responses of vegetation to fire can affect soil properties and nutrient dynamics by mitigating losses through erosion and leaching, accelerating nutrient recovery via litterfall inputs and atmospheric nitrogen fixation, and influencing belowground interactions among plants, microbes, and soil (Tappeiner and Alm, 1975; Staddon et al., 1997; Zeleznik and Dickmann, 2004).

Short-term (<10 years) responses of soil to fire are well-studied, and general trends include decreases in organic horizon depths, volatilization of nitrogen, and increases in pH and base cations, whereas phosphorous responses are variable (Certini, 2005; Neary et al., 2005). However, there is a particular lack of data on long-term effects of fire on soils in general, and in particular, in red pine forests of the Lake States region. For example, a review of fire effects on soils in the Lakes States region revealed that only 8% of the studies were conducted in mixed

pine or red pine forests and that 70% of the reported data from measurements were taken < 10 years after a fire event (Miesel et al., 2012).

Despite the ecological and economic value of red pine forests, there remains an absence of long-term studies regarding the use of prescribed fire to maintain regional fire-dependent ecosystems, and its influence on soil properties. An early study in a naturally-regenerated red pine forest in northern Minnesota investigated the effects of prescribed fire on site productivity, understory competition, and soil properties (Buckman, 1964; Alban, 1977). The *Red Pine Prescribed Burning Experiment* study began in 1959 with treatments and measurements through 1970. Alban (1977) concluded from a single year of measurements collected in 1969 that ten years of prescribed fire decreased understory competition and nutrients in the forest floor horizon, whereas nutrients in the mineral soil increased, without affecting site productivity. We leveraged the historical study site and initial raw datasets collected from 1959–1969, including the 1969 measurements previously reported by Alban (1977), with remeasurements in 2015 to: (1) evaluate short-term and intermediate trends over >10 years (1959–1969) as well as cumulative effects of prescribed fire treatments on soil responses across years (1959–2015) for which data were available; and (2) determine long-term soil responses and changes over time to prescribed fire treatments >45 years post-fire. We hypothesized that (1) summer prescribed fire treatments would result in the greatest magnitude in cumulative effect sizes on soil properties across years and (2) differences among fire treatments in organic and mineral soil properties would persist >45 years since the last prescribed fire. Our rationale for the first hypothesis was that summer burns are associated with lower fuel moistures and greater fire intensity; therefore greater combustion of soil organic horizons would result in greater losses and/or redistribution of nutrients in organic and mineral soil horizons relative to dormant season burns. Our second

hypothesis was based on the rationale that direct effects of fire on soil properties as well as the indirect effects of post-fire vegetation recovery and nutrient cycling over time would combine to influence persistent differences in soil properties among fire treatments.

2.3. Methods

2.3.1. Study Area

Our study site utilized the *Red Pine Prescribed Burning Experiment* located on the Cutfoot Experimental Forest (CEF) in the Chippewa National Forest, in Itasca County in northern Minnesota, USA (latitude 47°40'N, longitude 94°5'W) and is further described in Buckman (1964). The CEF is administered by the U.S. Forest Service Northern Research Station (Grand Rapids, MN). The study area is characterized by a continental climate with humid (80% relative humidity) summers exceeding temperatures of 32°C and winter minimum temperatures below -35°C (U.S. Forest Service, 2009). The growing season length is 100 to 120 days. Average annual precipitation ranges from 500–640 mm of rainfall with average winter snowfall depths between 1–2 meters, and summer droughts are common (U.S. Forest Service, 2009).

The forest community is dominated by red pine interspersed with jack pine (*Pinus banksiana* Lamb.), eastern white pine (*Pinus strobus* L.), paper birch (*Betula papyrifera* Marsh.), and quaking aspen (*Populus tremuloides* Michx.) (U.S. Forest Service, 2009). The forest at our study site originated naturally following a high severity fire in 1870, and fire scars indicate several major fires occurred in the mid to late 19th century (U.S. Forest Service, 2009). Measurements taken in 1959 prior to initiation of the original study indicated overstory trees were 90-year-old red pine with an average of 30.7 cm dbh (diameter at breast height, 1.37 m). The site index for red pine was 15.2 m at 50 years. The dominant understory species include

hazel (*Corylus* spp.) and alder (*Alnus* spp.). Fire suppression resulted in abundant hazel in the understory and several studies investigated the effects of prescribed fire to reduce hazel density and promote natural red pine regeneration (Alban, 1977; Buckman, 1964). Management history indicates few silvicultural treatments were applied on the site. The study site was thinned in the winter of 1959 to an overstory basal area of 27–29 m² ha⁻¹ to create a uniform tree density (Alban, 1977). The slash was removed from the burn treatment compartments to minimize fuel loading, site variability, and prescribed fire-induced tree mortality. No additional overstory management has been performed since the initial thinning.

The study area soil belongs to the Eagleview soil series, a mixed, frigid, Lamellic Udipsamment formed in glacial outwash parent material from the Late Wisconsin Age (NRCS, 2017). The soil is deep and well-drained with a medium to fine sand texture on 1–8% slopes typical of red pine forests of northern Minnesota. Prior to initiating the burning experiments in 1960, Alban (1977) described the soil as weakly developed with the forest floor approximately 8 cm thick and underlying mineral soil consisted of loamy sand including A (0–1 cm), E (1–11 cm), and B (11–47 cm) horizons. Stratified sands and gravels interspersed with thin lenses of very fine sandy loam were measured below the B horizon and calcium carbonate occurred intermittently below 127 cm.

2.3.2. Experimental Design and Treatments

Prescribed fire treatments representing contrasting fire seasons and frequencies were implemented within 0.4 ha compartments assigned using a randomized complete block design, with seven treatments replicated in each of four experimental blocks. The 28 compartments were each surrounded by a fire exclusion perimeter and contained a 0.08 ha circular plot. A total of

seven prescribed fire treatments were randomly assigned to compartments within blocks and were implemented from spring 1960 through the summer of 1970 to test the effects of fire season, frequency, and their interaction on soil physical and chemical properties. The seasonality of fire was categorized as either dormant or summer, with dormant season burns conducted in the absence of leaves on trees and shrubs (i.e., spring or fall), whereas summer burns were applied from late June through mid-August when vegetation assumed full physiological activity. The frequency of treatments was categorized as annual (every calendar year), biennial (every other calendar year), and periodic (every 6–9 years). The seven treatments administered included: dormant annual (DA), dormant biennial (DB), dormant periodic (DP), summer annual (SA), summer biennial (SB), summer periodic (SP), and an unburned control (CC) for reference conditions (Table 1).

Prescribed burns were conducted 5–15 days following a rain event (Buckman, 1964; Alban, 1977). This resulted in forest floor horizon moisture content averaging approximately 100% of dry weight in dormant season burns and 40% in summer season burns (Alban, 1977). Pre-burn preparation included constructing fire lines to mineral soil around each compartment, felling snags, and removing high risk dead and down woody fuels near fire lines. Backing fires were used to initiate burns within each compartment. Strip headfires followed varying from 3–6 meters in width. Fires were of low to moderate intensities with <1 m flame heights and resulted in minimal overstory tree damage.

Alban (1977) reported that burning led to the complete combustion of the litter horizon for all burn treatments and of the fermentation horizon for annual and biennial frequencies for both summer and dormant season burns in 1969. The summer annual burn decreased organic matter by approximately 50% and in some circumstances resulted in the complete combustion of

the forest floor horizon, exposing mineral soil in < 5% of the burned compartments (Alban, 1977). The last prescribed burn in 1970 resulted in a total of 10–11 burns in the annual treatments, five burns in the biennial treatments, and two burns in the periodic treatments (Table 1). No additional prescribed fire treatments or changes to the experimental units have been performed since the summer of 1970.

2.3.3. Field Methods

In June 2015 we re-sampled the original research plots and collected organic and mineral soil samples. The initial (1959–1969) soil samples were collected along a NE (45°) to SW (225°) transect bisecting the plot origin; however, all available sampling increments along these transects had been previously sampled. We therefore followed the original authors' instructions to establish a new sampling transect, which we established along an adjusted NE (22.5°) to SW (202.5°) azimuth.

We collected organic horizon and mineral soil samples at 3.05 meters from the plot origin along each corresponding azimuth within each of the 28 plots, for a total of 56 subsampling points. We placed a 30 cm diameter circular frame at each subsampling point to measure organic soil horizons (litter (O_i), fermentation (O_e), and humus (O_a)). Forest floor horizon depth was taken at each of three locations along the circumference of the circular frame. Four locations were used if any anomalies occurred (i.e., tree roots, rocks). We used a serrated gardening knife to cut around the inside circumference of the circular frame before collecting each of the three organic horizons from within the frame. Cones, bark, and woody debris was included as part of the organic horizons, whereas we omitted woody material >0.64 cm diameter. All organic

horizon samples were returned to the laboratory and dried at 60°C to constant mass prior to chemical analysis.

After we removed the organic horizons, we then collected mineral soil samples by depth within the circular frame. Two different sets of depth increments had previously been used for the study. We adopted the most recent set of depth increments: 0–10.16 cm, 10.16–50.80 cm, and 50.80–99.06 cm (Alban, 1977). The 0–10.16 cm increment was collected using a slide hammer with attached cup and sleeve, the 10.16–50.80 cm increment was collected using a t-handle soil probe, and the 50.80–99.06 cm increment was collected using a slide hammer with attached soil probe. Mineral soil samples were returned to the laboratory and dried at 60°C to constant mass prior to chemical analysis.

2.3.4. Laboratory Analysis

For our 2015 soil samples, we followed the soil chemical analysis methods used by Alban (1977) to the greatest extent possible. Organic soil horizon nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), magnesium (Mg), pH, depth, mass, organic matter (OM), ash content, and bulk density were measured along with mineral soil N, P, K, Ca, Mg, pH, cation exchange capacity (CEC), soil texture, and bulk density. We weighed each organic soil horizon after all living material (i.e., plants, roots, lichens, moss, insects, worms, etc.) as well as large rocks and scat were removed and discarded. Bulk density was calculated for each organic horizon as a mass per volume ratio (g cm^{-3}). The corresponding samples within plot were composited into one soil sample per plot prior to grinding. Each organic horizon was ground to pass a 1 mm screen. Organic horizons were analyzed using the following methods: N by Kjeldahl (Bremner, 1965), and P, K, Ca and Mg by ashing in a muffle furnace at 525°C for four

hours followed by uptake in 3N HCl. P was determined colorimetrically (Alban, 1972), whereas K, Ca, and Mg were determined by atomic absorption (PerkinElmer AAnalyst 400). We measured pH in a 4:1 water to volume ratio (LabFit AS-3000). Organic matter and ash content were determined from the loss on ignition at 525°C for four hours. In addition, we used elemental analysis (Costech, Italy, combustion temperature 1,000°C) to quantify total nitrogen and compare to Kjeldahl nitrogen.

We removed all visible organic material from mineral soil samples. We then sieved each mineral soil sample through a 2 mm screen and composited the fine fraction within each increment into one soil sample per plot for chemical analysis. Mineral soils were analyzed using the following methods: N by Kjeldahl, P was extracted using 0.01N HCl, whereas K, Ca, and Mg were extracted using 1N neutral ammonium acetate and determined as described above. We measured pH using a 1:1 water to volume ratio. CEC was calculated using pH buffer. Soil texture by particle size distribution was analyzed by hydrometer (Day, 1965). Bulk density was calculated as a mass per volume ratio (g cm^{-3}).

2.3.5. Statistical Analysis

We used a meta-analysis approach to estimate the effect size of prescribed fire treatments on soil properties across years using historical raw plot-level sample data collected from 1959–1969, including Alban's (1977) 1969 measurements, along with our new remeasurement data collected in 2015. For this approach, we considered individual years as similar to an individual study, and we calculated standardized treatment effects relative to the control treatment within year. This approach protects for across-year differences in soil sampling or analysis methods. For example, although we followed the original field and laboratory methods to the greatest extent

possible, identifying boundaries between organic soil horizons was somewhat subjective. Furthermore, the historical sampling events used two different sets of depth increments for sampling mineral soil (i.e., increments of 0–15.24 cm, and 15.24–91.44 cm were used in years prior to 1969, whereas increments of 0–10.16 cm, 10.16–50.80 cm, and 50.80–99.06 cm were used in 1969 and 2015). We therefore assigned the pre-1969 mineral soil increment depths across all years and calculated the weighted mean and weighted standard deviation for each upper (0–15.24 cm) and lower (15.24–91.44 cm) mineral soil increments. Differences in laboratory procedures and conditions between the historical and 2015 measurements may also affect measured responses. Lastly, the historical data measured between 1959–1969 included reports of some nutrients in parts per million (ppm) with insufficient information to determine whether the ppm was reported on a solution basis or soil mass basis. We performed the meta-analysis of soil responses to prescribed fire treatments using MetaWin 2.0 (Rosenberg et al., 2000). The natural log-transformed response ratio was used to estimate treatment effect size (ES) (Hedges et al., 1999):

$$ES = \ln(R) = \ln(\bar{X}_T) - \ln(\bar{X}_C) \quad (1)$$

where $\bar{X}_T$ is the mean soil response of the prescribed fire treatment within soil horizon and year and $\bar{X}_C$ is the mean soil response of the control within soil horizon and year ($n=4$). The effect size is a standardized unitless metric that allows comparison among soil response variables reported in different units across years. The variance, v , of the effect size was calculated as:

$$v_{ES} = \frac{(SD_T)^2}{n_T \bar{X}_T^2} + \frac{(SD_C)^2}{n_C \bar{X}_C^2} \quad (2)$$

where SD_T and SD_C is the standard deviation and n_t and n_c is the number of replicates of the prescribed fire treatments and control, respectively, within soil horizon and year. The 90%

confidence interval (CI), $100(1 - \alpha/2)\%$ around ES was used due to the high variability in studies of soils and calculated as:

$$CI = ES \pm Z_{\alpha/2} \sqrt{v} \quad (3)$$

where Z is the Z -score and α is the Type 1 error (0.10). The cumulative effect size ($\overline{ES}$) of prescribed fire treatments was determined for each soil variable within soil horizon and year across all years (1959–1969, and 2015) as:

$$\overline{ES} = \frac{\sum_{i=1}^n w_i E_i}{\sum_{i=1}^n w_i} \quad (4)$$

where the weight of the i^{th} study is the reciprocal of the sampling variance $w_i = 1/v_{ES}$, n was the number of years for which measurements existed, and E_i is the effect size for the i^{th} study. The cumulative effect size variance was calculated as:

$$s_{\overline{ES}}^2 = \frac{1}{\sum_{i=1}^n w_i} \quad (5)$$

The 90% confidence interval of the cumulative effect size was determined as:

$$CI = \overline{ES} \pm t_{\alpha/2, (n-1)} \times s_{\overline{ES}} \quad (6)$$

where t is the value from the Student's t -distribution and α is the Type 1 error (0.10). We were unable to calculate a robust analysis of standardized effect sizes for upper and lower mineral soil CEC, texture, and bulk density using the historical study (1959–1969) due to insufficient data. In addition, we used non-metric multidimensional scaling (NMDS) to investigate treatment effects on overall soil properties and changes over time (1969–2015) using the standardized effect sizes (ES) calculated in the meta-analysis approach as inputs for the ordination. We performed the NMDS using PC-ORD Version 7 (McCune and Mefford, 2015) with Euclidean distance measure in the slow and thorough mode with a maximum of 500 iterations. Kendall rank correlation

coefficients (τ) were calculated for correlations between individual soil response variables and NMDS axes, with statistical significance determined at the $\alpha = 0.10$ level.

Remeasurements in 2015 indicated there were no effects of fire treatments or time since the last prescribed fire on mineral soil CEC, texture, and bulk density. Therefore, these data are not included in the results presented here; however, we report supplementary data that provides descriptive statistics for all soil properties measured in 2015 (> 45 years post-fire) on a mass per unit area basis, along with results of analysis of variance (ANOVA) used to evaluate the effects of fire season and frequency on soil properties (Tables 2.2., 2.3., and 2.4.).

2.4. Results

2.4.1. Individual and Cumulative Soil Responses to Prescribed Fire

Prescribed fire treatments affected some, but not all, soil properties. In particular, litter and fermentation depths decreased across all treatments during the time period of the study that involved active burning (1960–1970) and returned to near original depths by 2015 (>45 years post-fire) (Figure 2.1.a, 2.1.b). However, the summer annual burn resulted in a persistent decrease in litter and humus horizon depths measured >45 years after the last fire treatments (Figure 2.1.a, 2.1.c). Total forest floor depth and organic matter (OM) content decreased during active burning years for annual and biennial frequencies, regardless of season (Figure 2.1.d, 2.1.e). The summer annual burn was the only treatment for which depth and OM decrease persisted to 2015 (Figure 2.1.d, 2.1.e), whereas a decrease in ash content persisted in both dormant annual and summer periodic treatments (Figure 2.1.f). Across all years (1959–2015), the cumulative effect size in the litter layer showed the most pronounced loss of depth, and the magnitude of effect increased with increased fire frequency (Figure 2.1.g); these patterns were

also evident in the total forest floor depth (Figure 2.1.j). We observed a trend toward increased OM content with increased fire frequency in dormant season treatments, but a decrease in OM with increased fire frequency in summer treatments; a similar inverse trend between seasons was observed for ash content (Figure 2.1.k, 2.1.l).

For the active burning years (1960–1970), summer annual and biennial burn treatments decreased total forest floor N and K, whereas P increased (Figure 2.2.a–c). We observed persistent decreases in 2015 (>45 years post-fire) for N in the summer biennial treatment, and for P and K in summer annual and periodic treatments (Figure 2.2.a–c). During active burning years, total forest floor Ca decreased, whereas pH increased with increased fire frequency regardless of season (Figure 2.2.d, 2.2.f). However, effects on cations and pH that persisted in 2015 were limited only to the dormant annual burn (increased Ca) and summer periodic burn (decreased pH) (Figure 2.2.d, 2.2.f). Across all years (1959–2015), the summer annual burn increased P, however, summer biennial and periodic burns decreased P and K (Figure 2.2.h, 2.2.i). Increased summer season fire frequency decreased Ca and Mg, however, increased dormant season fire frequency increased Ca and Mg (Figure 2.2.j, 2.2.k). In contrast, pH increased with increased fire frequency regardless of burn season (Figure 2.2.l).

We observed no effects of treatments on upper mineral soil N and K during active burning (1960–1970) years, and effects on P were variable (Figure 2.3.a–b). However, the dormant annual burn resulted in a persistent increase in N and P, measured in 2015 (>45 years post-fire) (Figure 2.3.a, 2.3.b). Summer and dormant season burns increased upper mineral soil Ca and Mg during active burn years (Figure 2.3.d, 2.3.e). All treatments showed a slight increase in pH during active burn years, and a decrease measured in 2015 (Figure 2.3.f). The cumulative effects in upper mineral soil across all years (1959–2015) indicated that N decreased with

increased fire frequency, regardless of burn season (Figure 2.3.g), whereas the effect of summer burns increased P with increased fire frequency and the effects of dormant season burns varied across frequencies (Figure 2.3.h). Upper mineral soil K increased in the dormant biennial treatment and showed a trend towards a decrease with increased summer season fire frequency (Figure 2.3.i). Upper mineral soil Ca showed a significant increase in the periodic burn frequency for both summer and dormant season fires, and in the dormant biennial treatment (2.3.j). Although the cumulative effect size of the dormant annual treatment was not statistically significant, all dormant season burn frequencies suggest a trend toward increases in Ca relative to the control, with the magnitude of increase inverse to burn frequency (Figure 2.3.j). Upper mineral soil pH increased in the dormant biennial and summer annual burns, but decreased in the dormant annual burn (Figure 2.3.l).

Lower mineral soil N decreased during active burning (1960-1970) across all treatments, excluding the summer annual burn, however, the summer annual burn resulted in a persistent decrease in N measured in 2015 (>45 years post-fire) (Figure 2.4.a). In contrast, P increased across active burning years for biennial and periodic burns, regardless of season, and the summer annual burn increased P measured in 2015 (Figure 2.4.b). Ca and Mg increased during active burning across all treatments, excluding the dormant annual burn (Figure 2.4.d, 2.4.e). pH increased across all treatments measured in year 1962 and a decrease in pH persisted for the dormant annual burn in 2015 (Figure 2.4.f). There were few significant overall treatment effects across all years (1959–2015) for the lower mineral soil, except for summer annual (increased P and pH) (Figure 2.4.h, 2.4.l) and dormant biennial (increased K and Ca) (Figure 2.4.i, 2.4.j) treatments. The effects of increased fire frequency within season were evident via trends toward decreased size of effect on N and increased size of effect on pH, for summer burns (Figure 2.4.g,

2.4.1, respectively). In contrast, there were no trends across fire frequencies for dormant season burns for either of these variables. In addition to the results described above, a summary table of statistically significant treatment effects measured in 2015 (>45 years post-fire) is available as supplementary data (Table 2.5.).

2.4.2. Soil Responses and Changes Over Time to Prescribed Fire

Non-metric multidimensional scaling for total forest floor soil properties resulted in a two dimensional solution with a final stress of 2.76. Axis 1 explained 50.6% of the variance in the effect size matrix for the years 1969 and 2015 and was positively correlated with P, K, Mg, pH, mass, and ash soil response variables (Figure 2.5.a). Dormant season burns were situated on the lower end of axis 1 with 1969 treatments in the upper left and 2015 remeasurements in the lower left of axis 1. In contrast, all summer season burns were located along the upper end of axis 1 with 1969 treatments occurring as a loose group in the upper right, whereas 2015 remeasurements occurred as a loose group in the lower right of axis 1. The summer annual burn was arrayed at the extremes of axis 1 and was consistent across years. Axis 2 accounted for 47.4% of variation in the same years and was negatively correlated with N, Ca, and organic matter (Figure 2.5.a). Treatments in 1969 were situated along the upper end of axis 2 and loosely grouped by season, although 2015 remeasurements were located along the lower end of axis 2 and loosely grouped by season.

NMDS ordination for the upper mineral soil converged on a two dimensional solution with a final stress of 5.61. Axis 1 explained 56.4% of the variance in the effect size matrix for the years 1969 and 2015 and was positively correlated with P (Figure 2.5.b). Summer season burns were located on the lower end of axis 1 as were all 2015 remeasurements with the exception of

the 2015 dormant annual burn. In contrast, dormant season burns were located on the upper end of axis 1, with the exception of the 2015 dormant biennial burn, as were all 1969 treatments. NMDS axis 2 accounted for approximately 27.4% of the variability in the same years and was positively correlated with Ca, Mg, and pH (Figure 2.5.b). The 1969 burning treatments occurred as a loose aggregation in the center of the matrix, whereas no pattern in 2015 remeasurements was evident. Across years, the dormant annual burn was arrayed along the right end of axis 2.

NMDS ordination for lower mineral soil properties resulted in a one dimensional solution with a final stress of 6.44. Axis 1 explained 93.1% of the variability and was negatively correlated with K, Ca, and Mg (Figure 2.5.c). Across years, summer and dormant annual burns were positioned along the upper end of axis 1 and the summer annual burn displayed the greatest dissimilarity across time >45 years following the last prescribed fire (Figure 2.5.c).

2.5. Discussion

2.5.1. Short-term Effects of Prescribed Fire on Soil Properties

Our study leveraged a historical study site and dataset to investigate short-, intermediate-, and long-term effects of contrasting prescribed fire treatments in a naturally-regenerated red pine forest. Although Alban (1977) reported short-term findings from only a single year of measurements collected in 1969, we report trends in soil responses to prescribed fire treatments, using existing data collected over >10 years (1959–1969) as well as a complete remeasurement in 2015. While we present a more comprehensive understanding of short-term and intermediate effects of fire treatments, our findings often coincided with Alban (1977). Our results from the active burn period support general short-term findings of prescribed fire effects on soil properties, including decreased organic horizon depth, volatilization of N, increases in pH and base cations,

and inconsistencies in P responses (Certini, 2005; Neary et al., 2005). Short- and intermediate-term soil responses to prescribed burns in our study differed by season of burning and the magnitude of effect size increased with increased fire frequency within season. Repeated burning, whether conducted in summer or in the dormant season, likely magnified the effects of fire by incrementally decreasing organic horizon mass and increasing combustion and subsequent loss of nutrients (Alban, 1977; Busse et al., 2014).

Alban (1977) reported short-term responses in the summer annual treatment resulted in the highest burn severity and greatest mass loss in forest floor horizons; these results corroborate our observations, and support similar findings following 20 years of prescribed fire treatments in loblolly pine (*Pinus taeda* L.) in South Carolina (Wells, 1971). Soil organic matter source material and quantity have direct effects on the amount and retention of nutrients by influencing CEC and pH (Neary et al., 2005). Fire causes changes in soil pH with volatilization of organic acids and an increase in base cations in post-fire ash (Johnson et al., 1991). Our results are comparable to values reported by several studies documenting only short-term increases in pH that are restricted to organic and upper mineral soil (Lunt, 1951; Metz et al., 1961; Smith, 1970; Wells, 1971; McKee, 1982). Nitrogen is a limiting plant nutrient in red pine ecosystems (Elliott and White, 1994) and is often volatilized in large quantities during fire, proportional to fire intensity and soil organic matter loss (Grier, 1975; Neary et al., 2005). This pattern is consistent with the trends we observed for decreased N for summer annual and biennial burns. The loss of N can have significant effects on post-fire plant recovery and long-term site productivity; however, burning may provide conditions that encourages N recovery via fixation of atmospheric nitrogen by leguminous symbiotic bacteria and recolonizing vegetation (McKee,

1982; Wells, 1971). This process may explain the absence of a cumulative treatment effect size we observed for N in the total forest floor horizon.

Soil elements including P, K, Ca, and Mg are resistant to volatilization and often occur as post-fire ash deposits (Bodí et al., 2014; Neary et al., 1999). Retention of these elements in soil is influenced by soil organic matter, CEC, pH, and clay content of post-fire soil (Alban, 1977). Soil elements are retained in the following order: $\text{Ca}^{2+} > \text{Mg}^{2+} > \text{K}^{+}$, whereas P is a negatively charged ion often held as iron and aluminum precipitates and is more susceptible to nutrient losses (Alban, 1977; Lewis, 1974). The presence of base cations and P in post-fire ash is ephemeral, as they are often adsorbed to soil exchange sites, immobilized by soil microorganisms and colonizing vegetation, or translocated off-site via surface runoff or into the mineral soil (Neary et al., 1999; Wittenberg, 2012). The responses we observed in mineral soil properties agree with these patterns.

The trends in short- and intermediate-term responses to prescribed fire across years in mineral soil were similar to trends observed for total forest floor horizon soil responses. However, the magnitude of effect across years was less evident for upper mineral soil and further decreased in lower mineral soil, and supports other soil studies (Metz et al., 1961; Smith, 1970; Alban, 1977; McKee, 1982). Overall, our observations indicated that mineral soil property responses to prescribed fire were relatively minor and often ephemeral, and either remained at—or returned to—pre-burn levels shortly following fire; these results corroborate similar findings from other ecosystem types (Ahlgren, 1970; Smith, 1970; Wells, 1971; Binkley et al., 1992; Franklin et al., 2003). Nutrient retention and CEC of mineral soil is closely related to soil texture and pH as well as soil organic matter (Helling et al., 1964). However, prescribed fire treatments had no influence on mineral soil bulk density or texture at any increment, consistent with studies in other regions

(Lunt, 1951; Metz et al., 1961; Moehring et al., 1966), and trends in pH do not closely reflect nutrient stocks. Thus, short- and intermediate-term trends in upper and lower mineral soil N, K, Ca, and Mg may reflect the effects of increased CEC as post-fire organic matter and nutrients are translocated into the mineral soil (Metz, 1961; Smith, 1970; Alban, 1977; McKee, 1982).

2.5.2. Long-term Effects of Prescribed Fire on Soil Properties

Our study is the first to provide evidence that prescribed fire treatments had legacy effects on organic horizon and mineral soil properties in red pine ecosystems of the Lakes States region, and that effects persisted >45 years since the last prescribed fire treatments. The overall trends we observed in persistent effects reflect similar short- and intermediate-term responses of our meta-analysis, and together suggest that soil responses to prescribed fire differed by season of burning and were further magnified by increased fire frequency within season. The annual fire frequency treatments, regardless of season, accounted for the majority of persistent effects among treatments in organic soil horizons and in upper and lower mineral soil increments.

The results of the NMDS for the total forest floor horizon support the findings that over time (1969–2015), season of burn was the primary contributor to observed trends in soil responses, and annual frequencies within season had a greater effect relative to other frequencies with time since fire. Comparatively, a study implementing a single summer prescribed fire conducted in a jack pine stand in Minnesota concluded that pH and nutrient content (N, P, K, Ca, Mg) increased relative to pre-burn conditions one year post-fire in organic soil, whereas following six years post-fire, only P content was decreased to below that of the pre-burn level (Ahlgren, 1970). Although we detected long-term effects for nutrients in the total forest floor horizon, Johnson et al. (2012) reported that soil variables (C, N, K, Ca, Mg) measured >46 years among a post-wildland fire site

and unburned forest site in California resulted in no persistent differences, with the exception of decreased P in the fire site.

Our results of the upper mineral soil NMDS over time also suggest that overall soil response differed primarily between seasons, whereas frequency of burns and time since fire were both relatively less important. Similar to our observations, a study in Michigan documented no significant differences in physical (bulk density) and chemical (total C, P, K, Ca, Mg, pH) soil properties in the 0–10 cm soil profile, with the exception of decreased total N, between 3–6 year post-wildland fire and undisturbed mature jack pine stands (LeDuc and Rothstein, 2007). However, the long-term increases in N and P in the upper mineral soil for the dormant annual treatment we documented are inconsistent with measurements in a pine plantation (*Pinus halepensis* Miller) recorded nine years following prescribed fire, which indicated decreases in N, P, pH, and C relative to pre-fire values in 0–5 cm mineral soil, although fire season was not reported (Alcañiz et al., 2016). In contrast to our observations, a study reporting the effects of a single spring prescribed burn in ponderosa pine stands in Oregon, documented no differences measured 12 years post-fire between burned and control plots in 0–5 cm mineral soil (Monleon et al., 1997). The persistent responses of soil properties in lower mineral soil we described (decreased N and pH; increased P) are similar to the study by Johnson et al. (2012) mentioned above, who reported long-term decreases in total N, P, and pH in fire sites measured in mineral soil increments at 30–45 cm, 30–90 cm, and 60–75 cm, respectively. Mineral soil is a poor conductor of heat and the effects of fire on mineral soil are often limited to the top few centimeters with the exception of high severity fires (Busse et al., 2014). The few persistent effects in upper and lower mineral soil properties we observed may be attributed to the highly permeable sandy soil and therefore relatively deep translocation of organic matter and soil nutrients at these depths.

2.5.3. Indirect Effects of Prescribed Fire on Soil Properties

The resilience of fire-adapted communities and fire effects on soil properties are often a function of vegetation responses to fire disturbances (Keeley et al., 2011). Rapid recovery of resprouting understory shrubs, including hazel in red pine ecosystems, may mitigate nutrient losses through erosion and leaching and accelerate soil organic matter and nutrient recovery (Nyamai et al., 2014; Tappeiner and Alm, 1975). The original investigators at our study site reported that summer annual and biennial prescribed burns were most effective in reducing hazel densities, whereas dormant season burning resulted in prolific hazel sprouting (Buckman, 1964; Alban, 1977). The effects on hazel have persisted >54 years since initiation of prescribed fire treatments (Scherer et al., 2016) and likely helps explain the trends in soil responses we observed. For example, previous studies in red pine forests have shown that high nutrient content in hazel foliage can increase soil organic matter as well as influence soil chemical composition and rates of nutrient cycling (Tappeiner and John, 1973; Tappeiner and Alm, 1975; Alban, 1977). Weyenberg and Pavlovic (2014) demonstrated that plant community composition in red and white pine stands is similar between pre- and post-burn sites treated with dormant season prescribed fires, whereas summer season burns resulted in statistically significant changes in vegetation including increases in species richness and diversity and a clear successional trajectory of pioneer species being replaced by shade tolerant species. A review of forest soils in Eastern North America concluded that long-term changes in soil were primarily driven by plant nutrient content and variations in soil organic matter quality and quantity, which differ significantly across vegetation types (Johnson et al., 1991). Therefore, quantifying local short- and long-term post-fire vegetation responses, including litterfall contributions and foliar nutrient content, will be critical in ongoing efforts to understand soil and ecosystem responses to fire.

2.6. Conclusions and Management Implications

Our study supports previous short-term findings of prescribed fire effects on soil properties reported in red pine and other ecosystem types and provides evidence that prescribed fire treatments had legacy effects on organic horizon and mineral soil properties >45 years since the last prescribed fire. In general, the legacy effects of summer season burns decreased, whereas dormant season burns increased nutrient stocks in organic and mineral soil horizons, and the effects of fire intensified with increased fire frequency within season. Short- and long-term responses of soil properties to prescribed fire treatments are likely influenced not only by the direct effects of fire intensity, combustion of forest floor horizons, and redistribution of nutrients during fire. In addition, they are also influenced by the indirect effects of post-fire vegetation and litterfall via interactions between the aboveground and belowground components of a post-fire ecosystem, particularly given the permeable sandy soils at our study site. Our results suggest that summer burns may be a valuable approach to increase the variability in burn schedules representative of historical regional fire regimes and facilitate development of fire-dependent species, such as red pine, by reducing organic horizon depths and overall nutrient stocks. Implementing forest management activities that emulate natural disturbance regimes, such as the historical range of wildfire season and frequency, within a given ecological or geographic region, has been recommended for obtaining the best results in restoring and maintaining forest ecosystem structure, species composition, and soil nutrient dynamics (Knapp et al., 2009). To help achieve these ecosystem management objectives, managers could aim to include summer burns where possible, in contrast to the more common application of prescribed fires in the dormant season. Although high frequencies of prescribed fires may be useful for initiating ecosystem restoration (Agee and Skinner, 2005; Knapp et al., 2009), sustained annual and biennial frequencies of burn

schedules are usually not logistically practical, regardless of season, because of weather, budgetary, and personnel constraints (Quinn-Davidson and Varner, 2012; Melvin, 2015). Annual and biennial fires are also more frequent than the historical fire regime in this region and ecosystem type (Bergeron and Brisson, 1990; Guyette et al., 2016). However, the absence of major persistent differences among treatments, and instances of similar direction of effects across treatments for the majority of soil properties we examined, suggest that summer season prescribed fires used to accomplish aboveground management objectives are not likely to result in strongly undesirable impacts to the mineral soil, such as increased nutrient stocks or changes in CEC, soil texture, and bulk density.

Although our results provide a unique comparison of contrasting fire seasons and frequencies, much more detailed information on weather conditions, fuel characteristics, phenology of vegetation, and firing techniques, as well as direct measures of fire intensity remain needed for these and other ecosystem types. These detailed data will be critical for improving our understanding of the relationships between fire behavior and fire effects over the short- and long-term after fire and for increasing the effectiveness of fire management activities to achieve specific management goals.

APPENDIX

Table 2.1. Description of prescribed fire treatments implemented in the original *Red Pine Prescribed Burning Experiment* (1959-1970) in the Cutfoot Experimental Forest, Minnesota, USA. Prescribed fire treatments are shown by season, frequency, interaction of season and frequency, burn dates, and number of times burned, for n=4 replicates per treatment and control. Discrepancies in scheduled burn dates and implementation of treatments were a result of unfavorable burning conditions.

Season	Frequency	Treatment	Burn dates (month/year)	Number of burns
Control	Control	CC	-	0
Dormant	Annual	DA	5/1960, 5/1961, 5/1962, 4/1963, 5/1964, 10/1964, 5/1966, 5/1967, 5/1969, 5/1970	10
	Biennial	DB	5/1960, 5/1962, 5/1964, 5/1966, 5/1969	5
	Periodic	DP	5/1960, 5/1969	2
Summer	Annual	SA	8/1960, 6/1961, 8/1962, 6/1963, 6/1964, 7/1965, 8/1966, 7/1967, 7/1968, 8/1969, 7/1970	11
	Biennial	SB	7/1960, 8/1962, 6/1964, 8/1966, 7/1968	5
	Periodic	SP	7/1960, 7/1967	2

Table 2.2. Means ($\pm$ standard errors) for soil properties remeasured in 2015 (>45 years following the last prescribed fire) at the *Red Pine Prescribed Burning Experiment*, for n=4 replicates per treatment and control. Different lowercase letters within each row indicate statistically significant differences among treatments at $\alpha = 0.10$, determined using analysis of variance (ANOVA) followed Tukey's adjustment for multiple pairwise comparisons. In contrast, the main body of the paper reports standardized effect sizes for multiple years using a meta-analysis approach. Organic horizons investigated included litter (L), fermentation (F), humus (H), and total forest floor (TFF) horizons. Mineral soil depth increments measured in 2015 included 0–10.16 cm, 10.16–50.80 cm, and 50.80–99.06 cm.

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
N total (kg ha ⁻¹)	L	100.67 (15.07)	106.14 (8.15)	100.13 (14.03)	87.77 (9.97)	58.44 (12.94)	106.48 (4.18)	107.89 (13.35)
	F	107.33 (17.16) ab	168.88 (38.76) a	78.25 (9.85) b	107.03 (18.61) ab	106.57 (12.06) ab	128.97 (12.18) ab	133.80 (30.33) ab
	H	749.76 (108.00)	642.03 (115.85)	635.42 (47.55)	783.76 (149.73)	486.16 (130.85)	477.11 (29.16)	524.01 (80.13)
	TFF	957.76 (106.18)	917.04 (153.84)	813.80 (57.05)	978.56 (172.27)	651.17 (128.05)	712.56 (29.96)	765.70 (108.12)
	0-10 cm	739.51 (171.09)	861.47 (239.63)	797.46 (80.56)	956.45 (143.55)	888.84 (96.97)	887.40 (51.40)	1081.81 (87.49)
	10-51 cm	1397.16 (507.89)	2253.38 (295.17)	1207.20 (522.57)	1228.61 (433.39)	540.96 (82.22)	913.40 (300.68)	1444.43 (726.89)
	51-99 cm	259.37 (18.78)	477.99 (92.97)	300.23 (0.51)	353.28 (164.95)	341.95 (39.55)	502.15 (99.32)	453.48 (115.74)
N Kjeldahl (kg ha ⁻¹)	L	104.04 (17.16)	95.08 (11.34)	93.02 (11.62)	86.29 (11.17)	57.44 (11.62)	102.01 (5.78)	97.90 (11.67)
	F	104.24 (19.95) ab	164.54 (39.26) a	78.43 (13.81) b	102.41 (16.92) ab	114.70 (9.47) ab	131.10 (14.01) ab	135.89 (32.20) ab
	H	620.90 (111.19)	641.88 (131.11)	571.81 (14.11)	841.69 (254.52)	449.80 (115.75)	400.40 (22.12)	466.32 (57.57)
	TFF	829.17 (116.69)	901.50 (165.10)	743.26 (23.93)	1030.39 (268.42)	621.94 (114.55)	633.51 (18.64)	700.11 (88.53)
	0-10 cm	725.76 (158.01)	1116.76 (159.06)	702.90 (153.44)	807.78 (88.50)	760.16 (103.46)	883.51 (105.10)	1059.64 (190.24)
	10-51 cm	1098.49 (270.58)	1123.63 (151.18)	917.77 (207.76)	947.21 (74.28)	643.58 (109.11)	1046.05 (105.46)	1386.65 (570.16)
	51-99 cm	388.10 (42.04)	473.96 (171.01)	392.46 (73.32)	430.20 (106.17)	357.88 (59.90)	866.27 (217.30)	415.62 (118.48)

Table 2.2. (cont'd)

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
P (kg ha ⁻¹)	L	8.20 (1.09) ab	7.32 (0.59) a	7.75 (1.83) ab	6.83 (0.66) ab	4.34 (0.70) b	7.63 (0.74) ab	6.80 (0.33) ab
	F	9.57 (1.82) ab	12.67 (1.43) a	4.41 (0.41) b	8.37 (1.97) ab	6.81 (0.73) ab	9.33 (1.16) ab	8.52 (2.45) ab
	H	82.78 (20.71)	67.24 (14.89)	63.10 (8.69)	78.48 (11.34)	37.32 (9.10)	54.15 (15.19)	32.28 (6.48)
	TFF	100.54 (20.73)	87.22 (15.40)	75.25 (9.92)	93.68 (12.54)	48.47 (8.70)	71.11 (14.69)	47.59 (8.67)
	0-10 cm	13.33 (2.80)	25.40 (4.56)	15.45 (3.41)	21.70 (5.56)	21.72 (6.65)	19.32 (4.41)	11.02 (3.94)
	10-51 cm	153.68 (32.46)	158.83 (38.57)	118.62 (21.76)	120.02 (34.58)	217.35 (18.42)	142.87 (39.04)	146.24 (52.66)
	51-99 cm	225.07 (30.03)	268.65 (15.19)	219.36 (43.74)	175.04 (83.01)	280.40 (16.04)	223.66 (23.61)	238.34 (6.98)
K (kg ha ⁻¹)	L	10.30 (1.35) abd	11.73 (0.86) a	10.05 (1.81) abd	9.42 (0.73) abd	6.51 (1.44) cd	12.88 (0.69) ab	10.67 (1.52) abd
	F	6.15 (0.60) ab	9.20 (2.34) a	3.98 (0.63) b	6.03 (0.92) ab	5.90 (0.71) ab	6.90 (0.58) ab	7.47 (1.74) ab
	H	64.63 (15.21)	47.12 (12.56)	52.26 (5.82)	53.15 (9.03)	31.42 (7.91)	42.19 (9.61)	31.80 (5.21)
	TFF	81.07 (15.24)	68.05 (13.64)	66.28 (5.64)	68.61 (9.87)	43.83 (7.21)	61.97 (10.22)	49.94 (6.84)
	0-10 cm	101.39 (26.69)	111.97 (10.69)	84.76 (21.59)	86.56 (14.76)	108.94 (41.38)	104.91 (19.60)	100.72 (8.17)
	10-51 cm	292.89 (113.46)	291.12 (77.53)	298.52 (66.01)	246.22 (42.77)	174.00 (26.11)	239.50 (55.31)	214.93 (40.77)
	51-99 cm	148.00 (0.36)	207.37 (54.80)	228.89 (25.28)	204.59 (29.26)	217.15 (43.21)	222.97 (42.51)	200.41 (32.63)
Ca (kg ha ⁻¹)	L	101.68 (13.09) ab	97.30 (11.97) ab	102.07 (15.23) a	85.53 (7.49) ab	55.89 (13.54) b	104.11 (7.69) a	102.94 (6.30) a
	F	59.43 (12.81) abd	106.71 (27.28) a	38.77 (8.69) cd	67.29 (9.57) abd	52.23 (2.41) abd	73.45 (8.57) ab	64.80 (10.08) abd
	H	125.01 (26.84)	268.72 (99.52)	130.64 (27.57)	212.08 (36.42)	109.02 (30.79)	124.00 (47.78)	97.67 (18.51)
	TFF	286.12 (39.34)	472.73 (126.14)	271.48 (26.75)	364.90 (37.88)	217.13 (37.07)	301.56 (58.07)	265.41 (27.40)
	0-10 cm	622.77 (166.35)	774.11 (101.29)	557.96 (102.78)	671.48 (146.12)	626.42 (41.76)	501.96 (51.05)	684.20 (59.88)
	10-51 cm	2927.16 (1491.15)	1907.03 (717.53)	3458.46 (1213.22)	2862.39 (1289.02)	1240.14 (170.25)	1579.52 (347.36)	2204.49 (996.27)
	51-99 cm	2346.70 (310.35)	1916.25 (511.30)	2935.91 (476.09)	3739.40 (1588.50)	2567.65 (397.16)	2606.43 (573.77)	2176.22 (313.27)

Table 2.2. (cont'd)

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
Mg (kg ha ⁻¹)	L	4.60 (0.69) ab	4.81 (0.27) ab	4.18 (0.63) ab	3.98 (0.38) ab	2.95 (0.57) a	5.09 (0.27) b	4.90 (0.40) ab
	F	2.87 (0.45) ab	4.31 (0.82) a	2.23 (0.35) b	2.67 (0.21) ab	2.80 (0.37) ab	2.97 (0.21) ab	2.96 (0.59) ab
	H	29.80 (11.27)	20.21 (3.27)	24.18 (5.19)	22.51 (2.82)	14.01 (3.31)	13.15 (1.65)	13.55 (0.99)
	TFF	37.26 (11.78)	29.32 (3.30)	30.59 (5.99)	29.17 (3.15)	19.76 (3.16)	21.22 (1.56)	21.41 (1.36)
	0-10 cm	86.27 (24.32)	109.19 (18.36)	75.26 (18.97)	89.05 (19.03)	78.66 (10.25)	72.96 (7.06)	85.83 (7.65)
	10-51 cm	387.68 (233.11)	309.52 (137.31)	520.81 (189.37)	391.68 (166.76)	162.97 (20.79)	236.66 (65.60)	305.58 (132.36)
	51-99 cm	290.23 (68.99)	300.54 (85.39)	429.97 (50.81)	345.03 (38.38)	347.15 (38.78)	387.72 (75.75)	329.32 (87.16)
pH	L	4.93 (0.07)	4.79 (0.09)	4.96 (0.08)	4.82 (0.04)	4.74 (0.15)	4.94 (0.02)	4.77 (0.03)
	F	5.38 (0.09)	5.30 (0.07)	5.40 (0.03)	5.33 (0.07)	5.16 (0.07)	5.48 (0.09)	5.29 (0.01)
	H	5.38 (0.08)	5.18 (0.30)	5.40 (0.12)	5.54 (0.11)	5.27 (0.21)	5.20 (0.06)	4.98 (0.16)
	TFF	5.23 (0.06)	5.09 (0.14)	5.26 (0.05)	5.23 (0.03)	5.06 (0.13)	5.21 (0.06)	5.01 (0.04)
	0-10 cm	5.45 (0.25)	5.20 (0.11)	5.13 (0.09)	5.40 (0.23)	5.35 (0.19)	5.08 (0.09)	5.08 (0.13)
	10-51 cm	5.58 (0.15)	5.15 (0.12)	5.38 (0.05)	5.33 (0.06)	5.45 (0.05)	5.33 (0.14)	5.38 (0.15)
	51-99 cm	6.00 (0.15)	5.67 (0.09)	5.73 (0.12)	6.27 (0.62)	5.93 (0.03)	5.70 (0.21)	5.70 (0.06)
Depth (cm)	L	2.02 (0.29)	1.84 (0.14)	1.78 (0.30)	1.74 (0.33)	1.16 (0.18)	1.63 (0.18)	1.86 (0.13)
	F	1.82 (0.41)	2.25 (0.50)	1.43 (0.08)	1.82 (0.29)	1.26 (0.23)	1.86 (0.18)	1.75 (0.22)
	H	5.57 (1.31)	4.57 (0.60)	5.33 (1.31)	5.31 (0.82)	2.98 (0.54)	4.70 (0.51)	3.84 (0.60)
	TFF	9.41 (1.52)	8.66 (1.13)	8.54 (1.64)	8.86 (1.33)	5.40 (0.64)	8.19 (0.82)	7.46 (0.71)
Mass (Mg ha ⁻¹)	L	11.15 (1.87)	11.69 (0.72)	10.26 (1.95)	9.64 (0.86)	7.66 (1.79)	12.55 (0.67)	11.86 (1.28)
	F	9.52 (0.92)	11.96 (2.29)	6.99 (0.98)	9.10 (1.18)	9.52 (1.53)	10.18 (0.62)	10.34 (2.51)
	H	195.62 (82.65)	82.08 (19.27)	156.48 (47.79)	129.15 (21.20)	82.37 (23.72)	80.00 (13.00)	66.17 (10.16)
	TFF	216.29 (83.84)	105.73 (20.63)	173.73 (50.00)	147.89 (22.94)	99.54 (23.61)	102.73 (13.33)	88.37 (11.98)

Table 2.2. (cont'd)

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
OM (Mg ha ⁻¹)	L	10.37 (1.73)	11.27 (0.72)	9.34 (1.64)	9.22 (0.80)	7.40 (1.74)	12.00 (0.63)	11.43 (1.24)
	F	5.92 (0.95) ab	9.92 (2.44) a	3.93 (0.70) b	6.70 (0.82) ab	6.91 (0.74) ab	7.53 (0.44) ab	7.58 (1.55) ab
	H	25.08 (4.38)	29.48 (3.40)	22.19 (0.89)	26.85 (3.27)	15.76 (2.90)	16.99 (1.23)	20.56 (1.84)
	TFF	41.37 (5.83) abc	50.67 (4.08) a	35.46 (2.23) bc	42.77 (3.54) ac	30.08 (2.69) b	36.52 (1.37) abc	39.57 (1.25) abc
Ash (%)	L	7.35 (2.11)	4.20 (0.57)	8.60 (1.83)	4.80 (0.34)	4.03 (0.37)	4.93 (0.26)	4.18 (0.21)
	F	38.05 (7.62) ab	20.90 (6.17) a	42.93 (7.64) b	26.23 (3.08) ab	26.03 (4.68) ab	26.10 (4.31) ab	25.40 (4.59) ab
	H	83.88 (2.97)	59.63 (8.36)	83.28 (2.87)	77.68 (4.83)	71.85 (10.60)	78.03 (2.09)	65.73 (8.38)
	TFF	43.09 (4.15)	28.24 (1.74)	44.94 (3.95)	36.23 (2.37)	33.97 (4.85)	36.35 (2.12)	31.77 (3.44)
CEC (meq/100g)	0-10 cm	4.67 (0.35)	7.76 (0.96)	5.18 (0.59)	7.07 (1.05)	6.59 (1.03)	7.50 (0.74)	9.06 (1.34)
	10-51 cm	3.65 (1.66)	2.73 (0.95)	5.21 (1.74)	3.74 (1.44)	1.89 (0.21)	3.89 (1.26)	2.81 (0.98)
	51-99 cm	3.07 (0.44)	2.69 (0.73)	3.82 (0.53)	4.71 (1.67)	3.60 (0.49)	3.38 (0.62)	2.85 (0.42)
Sand (%)	0-10 cm	80.97 (2.87)	79.62 (2.94)	77.72 (2.93)	79.65 (2.20)	83.41 (0.84)	83.11 (2.19)	80.82 (1.06)
	10-51 cm	83.86 (3.99)	83.35 (3.34)	80.61 (4.27)	82.02 (2.76)	87.09 (0.88)	87.02 (2.45)	84.24 (2.89)
	51-99 cm	91.08 (0.23)	90.81 (0.71)	89.54 (1.60)	89.49 (0.78)	90.54 (0.30)	90.32 (1.35)	91.84 (1.67)
Silt (%)	0-10 cm	10.97 (2.35)	12.26 (2.54)	13.73 (2.26)	12.66 (1.92)	9.26 (0.85)	9.46 (1.69)	10.57 (0.22)
	10-51 cm	7.47 (2.18)	8.63 (2.02)	9.84 (2.86)	9.41 (1.31)	6.83 (0.89)	5.78 (1.67)	8.13 (1.45)
	51-99 cm	2.95 (0.55)	3.28 (0.17)	2.83 (0.65)	3.68 (0.51)	2.66 (0.65)	2.74 (1.14)	2.40 (1.25)
Clay (%)	0-10 cm	8.06 (0.57)	8.13 (0.39)	8.55 (0.75)	7.69 (0.28)	7.34 (0.29)	7.43 (0.51)	8.62 (1.24)
	10-51 cm	8.67 (1.84)	8.02 (1.34)	9.56 (1.45)	8.57 (1.56)	6.07 (0.42)	7.20 (0.83)	7.63 (1.62)
	51-99 cm	5.97 (0.35)	5.91 (0.64)	7.62 (1.07)	6.82 (0.27)	6.80 (0.43)	6.94 (1.16)	5.77 (0.50)

Table 2.3. Results of ANOVA using a mixed model approach for organic soil response variables in the litter (L), fermentation (F), humus (H), and total forest floor (TFF) horizons remeasured in 2015 (n=4). In contrast, the main body of the paper reports standardized effect sizes for multiple years using a meta-analysis approach. An [ns] indicates no significance at any level, whereas * = $p < 0.10$, ** = $p < 0.05$, *** = $p < 0.01$, and **** = $p < 0.001$.

		N total			N Kjeldahl		
		F	p-value		F	p-value	
L	Season	0.55	0.4672	ns	0.34	0.5652	ns
	Frequency	1.75	0.2027	ns	1.73	0.2055	ns
	Season $\times$ frequency	4.72	0.0225	**	2.72	0.0928	*
F	Season	0.20	0.6572	ns	0.63	0.4383	ns
	Frequency	1.63	0.2244	ns	1.76	0.2012	ns
	Season $\times$ frequency	5.06	0.0189	**	4.25	0.0308	**
H	Season	5.18	0.1598	ns	5.50	0.0556	*
	Frequency	0.56	0.5839	ns	0.92	0.4566	ns
	Season $\times$ frequency	0.17	0.8485	ns	0.27	0.7715	ns
TFF	Season	3.93	0.0687	*	6.33	0.0253	**
	Frequency	0.49	0.6286	ns	0.72	0.5060	ns
	Season $\times$ frequency	0.38	0.6945	ns	0.34	0.7183	ns

Table 2.3. (cont'd)

		P			K			Ca			Mg		
		F	p-value		F	p-value		F	p-value		F	p-value	
L	Season	1.85	0.2022	ns	0.14	0.7090	ns	0.64	0.4354	ns	0.00	0.9850	ns
	Frequency	2.02	0.1817	ns	1.86	0.1837	ns	2.89	0.0819	*	1.32	0.2927	ns
	Season × frequency	3.07	0.0897	*	6.09	0.0096	***	3.69	0.0454	**	5.54	0.0133	***
F	Season	0.04	0.8403	ns	0.16	0.6915	ns	0.56	0.4740	ns	0.17	0.6815	ns
	Frequency	1.67	0.2161	ns	1.98	0.1667	ns	2.22	0.1742	ns	2.35	0.1237	ns
	Season × frequency	5.89	0.0108	***	4.58	0.0247	**	7.98	0.0137	***	3.34	0.0583	**
H	Season	6.98	0.0166	***	3.77	0.0678	*	5.70	0.0586	**	11.40	0.0044	***
	Frequency	0.12	0.8907	ns	0.33	0.7256	ns	0.72	0.5259	ns	0.12	0.8858	ns
	Season × frequency	1.01	0.3843	ns	0.16	0.8515	ns	1.83	0.2419	ns	0.29	0.7510	ns
TFF	Season	7.16	0.0154	***	3.46	0.0792	*	5.41	0.0749	*	10.03	0.0073	***
	Frequency	0.08	0.9258	ns	0.32	0.7334	ns	0.56	0.6024	ns	0.08	0.9250	ns
	Season × frequency	1.36	0.2813	ns	0.49	0.6193	ns	3.08	0.1321	ns	0.04	0.9595	ns

		pH			Depth			Mass			OM			Ash		
		F	p-value		F	p-value		F	p-value		F	p-value		F	p-value	
L	Season	0.38	0.5535	ns	1.75	0.2026	ns	0.02	0.8863	ns	0.11	0.7476	ns	4.96	0.0498	**
	Frequency	5.18	0.0306	**	0.99	0.3901	ns	0.84	0.4482	ns	0.62	0.5497	ns	6.11	0.0181	***
	Season × frequency	0.03	0.9731	ns	1.75	0.2018	ns	3.64	0.0469	**	4.28	0.0302	**	2.69	0.1153	ns
F	Season	0.39	0.5446	ns	0.84	0.3773	ns	0.25	0.6375	ns	0.29	0.5998	ns	1.12	0.3041	ns
	Frequency	4.40	0.0535	**	0.25	0.7816	ns	0.95	0.4875	ns	2.89	0.0966	*	2.90	0.0809	*
	Season × frequency	1.18	0.3583	ns	3.01	0.0959	*	1.47	0.3689	ns	4.43	0.0379	**	2.76	0.0902	*
H	Season	3.52	0.0770	*	3.74	0.0730	*	5.21	0.0382	**	17.59	0.0004	****	0.09	0.7732	ns
	Frequency	0.14	0.8739	ns	1.30	0.3024	ns	1.05	0.3739	ns	2.46	0.1415	ns	2.35	0.1269	ns
	Season × frequency	2.47	0.1124	ns	0.22	0.8023	ns	1.36	0.2888	ns	1.53	0.2694	ns	1.62	0.2275	ns
TFF	Season	2.58	0.1484	ns	3.06	0.0972	*	3.52	0.0768	*	13.26	0.0026	***	0.77	0.3915	ns
	Frequency	4.09	0.0574	**	0.76	0.4839	ns	0.83	0.4529	ns	2.39	0.1266	ns	4.12	0.0336	**
	Season × frequency	2.45	0.1449	ns	0.79	0.4688	ns	0.33	0.7235	ns	10.12	0.0018	***	2.34	0.1250	ns

Table 2.4. Results of ANOVA using a mixed model approach for mineral soil response variables by increment depth (0–10.16 cm, 10.16–50.80 cm, 50.80–99.06 cm) remeasured 2015 (n=4). In contrast, the main body of the paper reports standardized effect sizes for multiple years. An [ns] indicates no significance at any level, whereas * = $p < 0.10$, ** = $p < 0.05$, *** = $p < 0.01$, and ***** = $p < 0.001$.

		N total			N Kjeldahl		
		F	p-value		F	p-value	
0-10 cm	Season	0.57	0.4640	ns	0.05	0.8257	ns
	Frequency	1.03	0.3864	ns	0.71	0.5056	ns
	Season $\times$ frequency	0.07	0.9312	ns	2.88	0.0823	*
10-51 cm	Season	2.80	0.1283	ns	0.18	0.6773	ns
	Frequency	0.47	0.6437	ns	0.39	0.6841	ns
	Season $\times$ frequency	3.71	0.0856	*	2.03	0.1610	ns
51-99 cm	Season	0.46	0.5180	ns	0.61	0.4499	ns
	Frequency	0.01	0.9927	ns	0.96	0.4125	ns
	Season $\times$ frequency	2.69	0.1356	ns	1.13	0.3580	ns

Table 2.4. (cont'd)

		P			K			Ca			Mg		
		F	p-value		F	p-value		F	p-value		F	p-value	
0-10 cm	Season	0.99	0.3330	ns	0.29	0.5943	ns	0.60	0.4504	ns	0.35	0.5654	ns
	Frequency	1.64	0.2224	ns	0.26	0.7769	ns	1.68	0.2148	ns	1.34	0.3038	ns
	Season × frequency	1.43	0.2659	ns	0.75	0.4850	ns	0.32	0.7322	ns	0.84	0.4593	ns
10-51 cm	Season	1.77	0.2003	ns	2.45	0.1379	ns	2.52	0.1300	ns	2.26	0.1498	ns
	Frequency	1.88	0.1814	ns	0.32	0.7312	ns	0.97	0.3969	ns	0.85	0.4436	ns
	Season × frequency	0.17	0.8489	ns	0.33	0.7258	ns	0.33	0.7227	ns	0.24	0.7864	ns
51-99 cm	Season	0.98	0.3605	ns	0.00	0.9971	ns	0.46	0.5195	ns	0.00	0.9455	ns
	Frequency	1.01	0.4170	ns	0.22	0.8054	ns	0.49	0.6300	ns	0.95	0.4156	ns
	Season × frequency	0.74	0.5146	ns	0.03	0.9711	ns	1.10	0.3836	ns	0.24	0.7938	ns

		pH			CEC			Sand			Silt			Clay		
		F	p-value		F	p-value		F	p-value		F	p-value		F	p-value	
0-10 cm	Season	0.31	0.5833	ns	1.96	0.1784	ns	3.80	0.0669	*	4.64	0.0526	**	0.43	0.5220	ns
	Frequency	0.63	0.5450	ns	1.78	0.1966	ns	0.20	0.8174	ns	0.14	0.8737	ns	0.24	0.7904	ns
	Season × frequency	1.05	0.3699	ns	2.21	0.1387	ns	0.48	0.6245	ns	0.19	0.8322	ns	1.61	0.2280	ns
10-51 cm	Season	1.18	0.2921	ns	1.15	0.2979	ns	2.83	0.1098	ns	2.52	0.1299	ns	2.69	0.1183	ns
	Frequency	0.13	0.8781	ns	2.23	0.1366	ns	0.25	0.7803	ns	0.20	0.8216	ns	0.58	0.5712	ns
	Season × frequency	1.28	0.3035	ns	0.01	0.9935	ns	0.25	0.7817	ns	0.33	0.7262	ns	0.15	0.8585	ns
51-99 cm	Season	0.27	0.6178	ns	0.47	0.5116	ns	0.97	0.3501	ns	1.09	0.3176	ns	0.36	0.5598	ns
	Frequency	0.55	0.6011	ns	0.31	0.7443	ns	0.21	0.8198	ns	0.06	0.9465	ns	1.85	0.1989	ns
	Season × frequency	1.31	0.3291	ns	1.37	0.3070	ns	0.82	0.4859	ns	0.29	0.7504	ns	1.62	0.2377	ns

Table 2.5. Persistent effects of prescribed fire on soil properties measured in 2015 (>45 years post-fire), determined using a meta-analysis approach, shown by horizon, response variable, treatment, and direction of change (+ increase, - decrease) relative to the unburned control, for n=4 replicates per treatment and control. Organic horizons investigated included litter (L), fermentation (F), humus (H), and total forest floor (litter, fermentation, humus) horizons. Mineral soil depths measured in 2015 included upper (0–15.24 cm) and lower (15.24–91.44 cm) increments. Statistically significant effects at $\alpha = 0.10$ are reported; non-significant effects are not shown.

Horizon	Variable	Units	Treatment	Change
Litter	Depth	cm	SA	-
Humus	Depth	cm	SA	-
Total forest floor	Depth	cm	SA	-
	Organic matter	Mg ha ⁻¹	SA	-
	Ash	%	DA	-
	Ash	%	SP	-
	N	Mg ha ⁻¹	SA	-
	P	Mg ha ⁻¹	SA	-
	P	Mg ha ⁻¹	SP	-
	K	Mg ha ⁻¹	SA	-
	Ca	Mg ha ⁻¹	DA	+
	pH	-	SP	-
Upper mineral soil	N	Mg ha ⁻¹	DA	+
	P	Mg ha ⁻¹	DA	+
Lower mineral soil	N	Mg ha ⁻¹	SA	-
	P	Mg ha ⁻¹	SA	+
	pH	-	DA	-

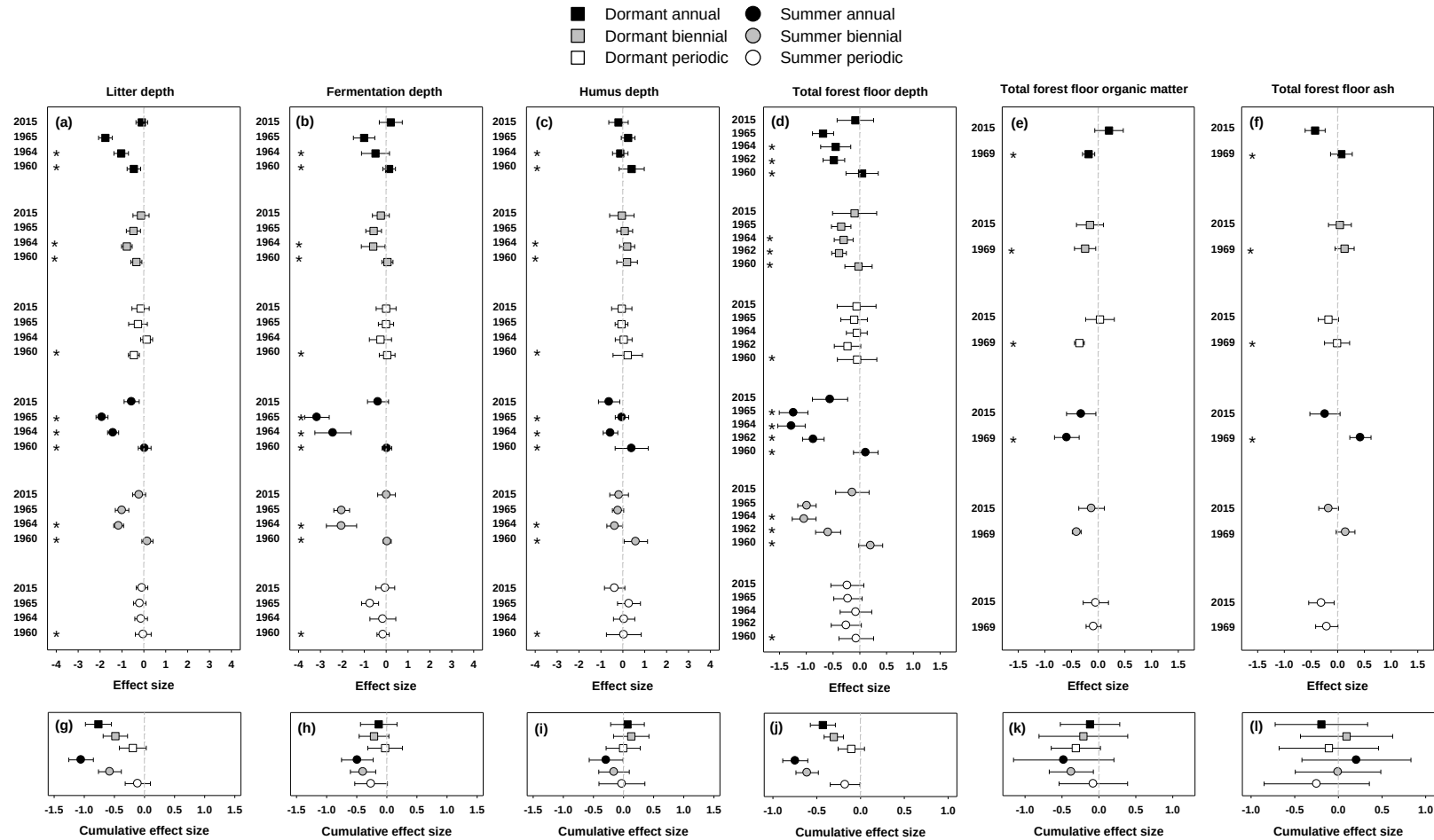


Figure 2.1. Standardized effect sizes ($\pm 90\%$ confidence intervals) for organic horizon litter, fermentation, humus, and total forest floor (litter, fermentation, humus) depth and total forest floor organic matter and ash content. Within-year effect sizes are shown in upper panels, and cumulative effect sizes (across all years) are shown in lower panels. Symbol shape represents prescribed fire season, whereas shading represents frequency, for $n=4$ replicates per treatment. Asterisks [*] in upper panels indicate the years in which prescribed fire treatments were conducted. Error bars that do not overlap the 0 effect size indicate a statistically significant treatment effect relative to the control, and non-overlapping error bars indicate statistically significant differences among treatments ($\alpha = 0.10$). Note changes in x-axis scaling between panels.

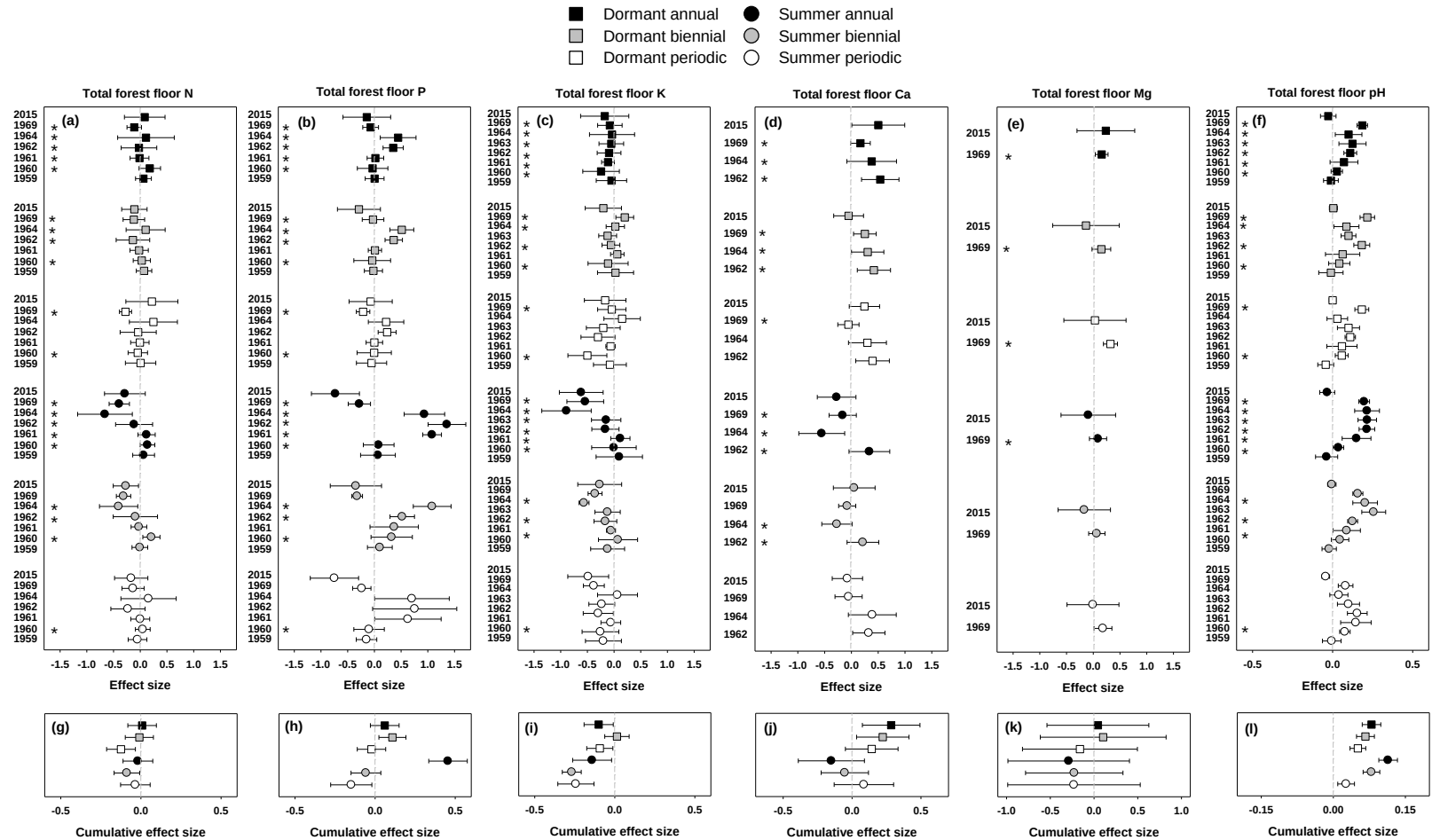


Figure 2.2. Standardized effect sizes ($\pm$ 90% confidence intervals) for total forest floor (litter, fermentation, humus) horizon N, P, K, Ca, Mg, and pH. Within-year effect sizes are shown in upper panels, and cumulative effect sizes (across all years) are shown in lower panels. Symbol shape represents prescribed fire season, whereas shading represents frequency, for $n=4$ replicates per treatment. Asterisks [*] in upper panels indicate the years in which prescribed fire treatments were conducted. Error bars that do not overlap the 0 effect size indicate a statistically significant treatment effect relative to the control, and non-overlapping error bars indicate statistically significant differences among treatments ($\alpha = 0.10$). Note changes in x-axis scaling between panels.

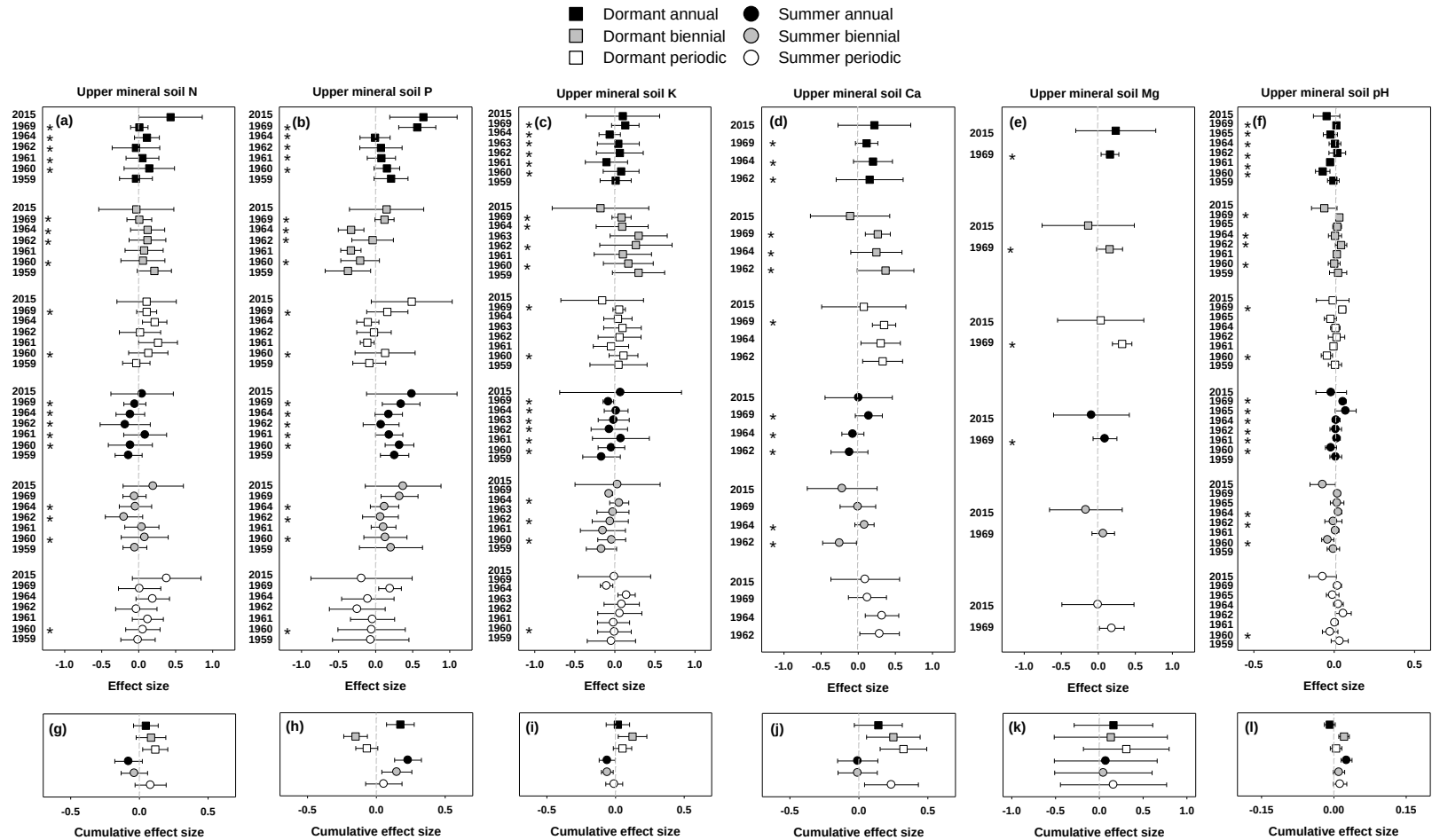


Figure 2.3. Standardized effect sizes (± 90% confidence intervals) for upper (0–15 cm) mineral soil N, P, K, Ca, Mg, and pH. Within-year effect sizes are shown in upper panels, and cumulative effect sizes (across all years) are shown in lower panels. Symbol shape represents prescribed fire season, whereas shading represents frequency, for $n=4$ replicates per treatment. Asterisks [*] in upper panels indicate the years in which prescribed fire treatments were conducted. Error bars that do not overlap the 0 effect size indicate a statistically significant treatment effect relative to the control, and non-overlapping error bars indicate statistically significant differences among treatments ($\alpha = 0.10$). Note changes in x-axis scaling between panels.

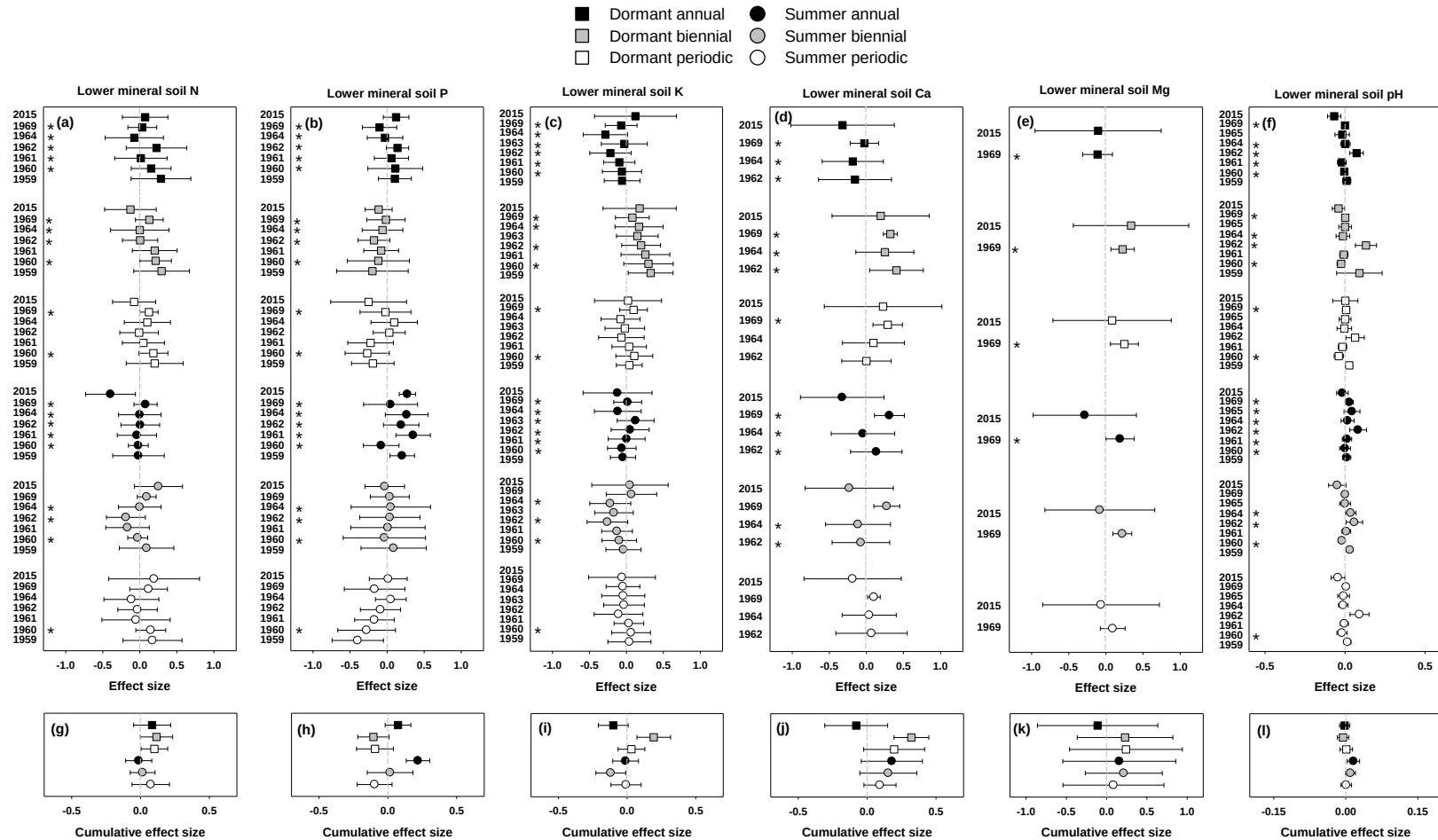


Figure 2.4. Standardized effect sizes ($\pm$ 90% confidence intervals) for lower (15–91cm) mineral soil N, P, K, Ca, Mg, and pH. Within-year effect sizes are shown in upper panels, and cumulative effect sizes (across all years) are shown in lower panels. Symbol shape represents prescribed fire season, whereas shading represents frequency, for $n=4$ replicates per treatment. Asterisks [*] in upper panels indicate the years in which prescribed fire treatments were conducted. Error bars that do not overlap the 0 effect size indicate a statistically significant treatment effect relative to the control, and non-overlapping error bars indicate statistically significant differences among treatments ($\alpha = 0.10$). Note changes in x-axis scaling between panels.

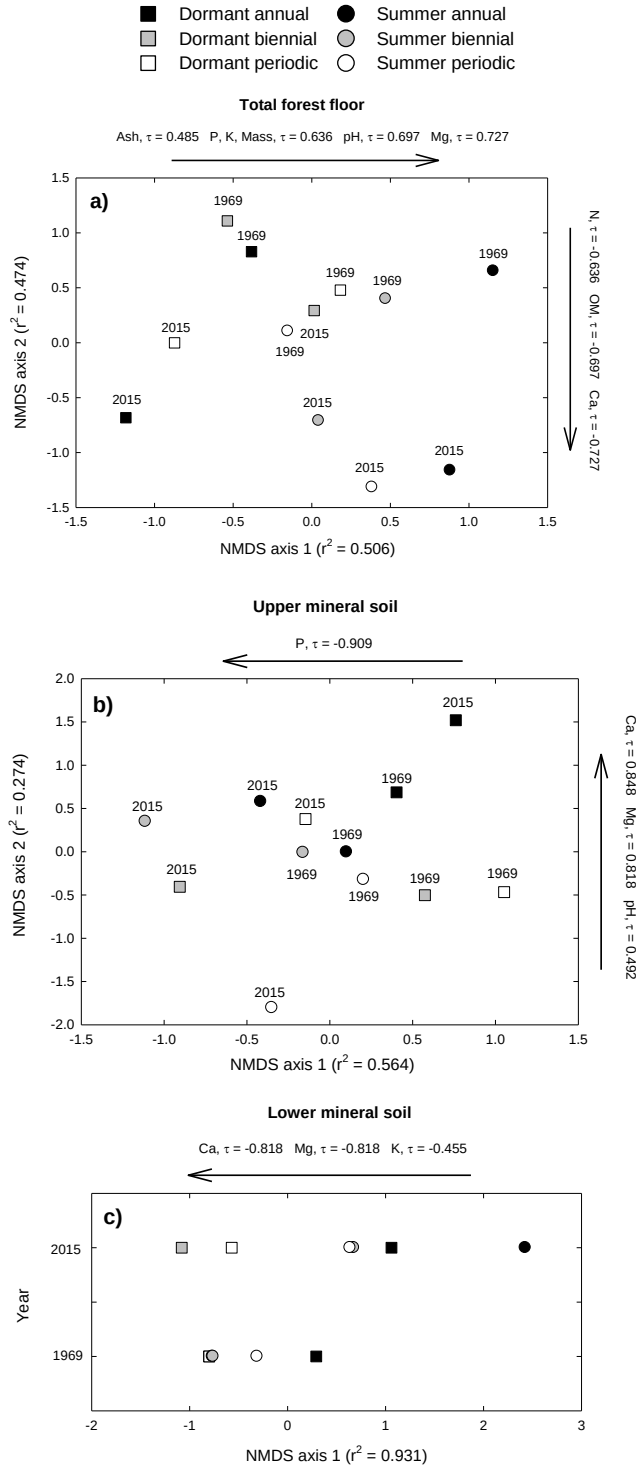


Figure 2.5. Non-metric multidimensional (NMDS) ordination of standardized effect sizes (ES) of soil variable responses measured in 1969 and 2015 (>45 years post-fire) in the total forest floor (litter, fermentation, humus) horizon and mineral soil upper (0–15cm) and lower (15–91 cm) increments. Symbol shape represents prescribed fire season, whereas shading represents frequency, for $n=4$ replicates per treatment. Correlation coefficients (τ) between individual soil responses and NMDS axes at $\alpha = 0.10$ are shown.

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CHAPTER 3

LONG-TERM EFFECTS OF SEASON AND FREQUENCY OF PRESCRIBED FIRE ON SOIL C AND PYC STOCKS AND PYC CONCENTRATIONS IN A *PINUS RESINOSA* FOREST IN NORTHERN MINNESOTA

3.1. Abstract

Soils in fire-dependent ecosystems represent a major pool of fire-affected carbon (C). Soil C stocks and concentrations can be influenced by vegetation and soil properties, as well as by disturbance patterns, such as fire season and frequency. The effects of fire on soil C pools can vary among regions, and adequate knowledge is important for making informed forest management decisions. However, data on soil response to contrasting fire treatments remains limited, especially in the Lake States region. Therefore, we leveraged a historical fire study (conducted 1959–70) with measurements in 2015 to evaluate effects of prescribed fire season (dormant, summer), frequency (annual, biennial, periodic), and time since fire (>45 years post-fire) on soil C and PyC stocks and PyC concentrations in a red pine (*Pinus resinosa* Ait.) forest in northern Minnesota, USA. We used analysis of variance (ANOVA) to evaluate treatment effects on C and PyC stocks and PyC concentrations and simple linear regression to assess the relationship of PyC with soil properties.

Our study indicated few long-term differences among treatments within soil layers excluded to the summer annual burn. We observed a persistent decrease in PyC stocks for the summer annual burn in the humus and total forest floor horizons relative to the control. PyC concentrations decreased for the summer periodic burn in the fermentation horizon, and for the dormant periodic and all summer burns in the humus horizon, relative to the control. Total C stocks in the total soil profile (combined forest floor and mineral soil (0–91 cm)) were lower for

the summer annual burn relative to the dormant annual and summer periodic burns. Overall, measurements indicated 62% of total C and 76% of total PyC stocks in the mineral soil (0–91 cm) relative to the forest floor. Although prescribed fire treatments may result in initial losses of C and PyC stocks in forest floor horizons, these effects may be moderated by the relatively larger pools of C and PyC stocks and increased PyC concentrations in mineral subsoils, as well as by forest floor C recovery since the last prescribed fire. Our results suggest prescribed fire seasons and frequencies used to accomplish aboveground management had minimal impacts on subsoil and total C and PyC stocks, and may be compatible with C sequestration objectives.

3.2. Introduction

Soil is the largest terrestrial pool of carbon (C), storing more C than aboveground vegetation and the atmosphere combined (IPCC et al., 2000; Scharlemann et al., 2014; Schlesinger, 1997). Fire is the principal driver of fire-dependent forests, and forest soils represent a significant pool of fire-affected C. Globally, vegetation fires burn approximately 464 Mha yr⁻¹ (Randerson et al., 2012) and emit on average, 2.0 Pg C year⁻¹ (Van Der Werf et al., 2010). Understanding changes in global and regional soil C storage and cycling in response to fire is important to guide forest management decisions that influence forest productivity and climate change mitigation efforts (DeLuca and Aplet, 2008). Estimates across ecosystem types have suggested that up to 28% of total C affected by fire may be converted into pyrogenic carbon (PyC) (Preston & Schmidt, 2006; Santín et al., 2015). PyC is a thermally resistant, super-passive form of C produced from incomplete combustion of organic material that exists on a continuous spectrum ranging from charred material to soot (Bird et al., 2015; DeLuca and Aplet, 2008), and may represent up to 40% of total organic C in terrestrial soils (Forbes et al., 2006; Reisser et al.,

2016). The various forms of PyC produced and local mean residence time of PyC is dependent on a variety of biotic (e.g., source material, microbial activity) and abiotic factors including climate, site characteristics, soil properties and depth, and fire effects (e.g., fire temperature and frequency) (Bird et al., 2015; Knicker, 2011; Preston and Schmidt, 2006; Santín and Doerr, 2016). Although PyC in soils may be initially lost during fire through volatilization of CO₂ into the atmosphere or carried off-site by erosion, much (>80%) of the PyC remains in soil shortly following a fire event (Forbes et al., 2006). PyC is ubiquitous in fire-dependent ecosystems (Preston and Schmidt, 2006), and has been shown to exert important effects on soil physical, chemical, and biological properties (Liang et al., 2006; Briggs et al., 2012) including plant productivity and soil nutrient cycling (Mingxin, 2016; Pingree and DeLuca, 2017). Because PyC contains a high C content and has a long soil residence time ranging from decades to millennia (Bird, 1999; Singh et al., 2012), PyC has the capacity to sequester large amounts of C within the soil and represents an important pool of C in the global C budget (Santín et al., 2015).

Although wildland fires have contemporarily been the interest regarding C cycling (French et al., 2011), discrepancies exist among the relatively few studies investigating the use and effects of prescribed fire on C cycling and sequestration within and among regions (Lavoie et al., 2010; Wiedinmyer et al., 2010; Campbell et al., 2012). For example, a meta-analysis indicated that forest soil C stocks and concentrations were affected by local geographic variations (i.e., forest, fire, and soil type), and emphasized the need for regionally-specific estimates of soil C and fire management plans (Nave et al., 2011). Prescribed fires are often not representative of wildland fires (Certini, 2005). Prescribed fires are limited by weather conditions, and are commonly conducted out-of-season relative to that of natural fires. As a result, prescribed fires often differ in the direct effects (i.e., fire intensity) and indirect effects

(i.e., vegetation responses) of burning relative to wildland fires (Schmidt et al., 2000). In addition, interactions among forest and soil types, and differences in the time since fire may produce novel changes in soil nutrient cycling and C and PyC stocks at local scales.

Furthermore, the effects of prescribed fire season, frequency, and time since fire on C stocks and PyC production and subsequent stability of residual C and PyC in soils remains unclear. Seasonal effects of increased fire severity (e.g., forest floor combustion) associated with summer burns relative to dormant season burns (Alban, 1977), may increase the amount of PyC produced (Maestrini et al., 2017), whereas repeated fires may consume existing PyC and decrease its stocks, concentrations, and distribution in soils (Foereid et al., 2011). Moreover, studies investigating C and PyC have been limited to surface soils (Santín et al., 2015). Although deep soil horizons are relatively unaffected by the direct effects of fire, nutrients and organic material including C and PyC are often translocated into the soil profile following a fire event and accumulate in deep mineral soil horizons (Bird et al., 2015; Foereid et al., 2011; Knicker, 2007). Therefore, deep soil increments may serve as a depositional reservoir for C and PyC stocks, although deep soil is often unaccounted for when quantifying C and PyC stocks (Dungait et al., 2012; Lorenz and Lal, 2014).

Managing for multiple objectives is challenging, as there are often numerous competing management goals for the use prescribed fire including forest regeneration, ecosystem restoration, wildlife habitat creation, and fuels reduction (Dickmann, 1993; Marschall et al., 2014; Scherer et al., 2016). Of the many objectives, estimating and managing forest C stocks has received recent research and management consideration (Boerner et al., 2009; Meigs et al., 2009; Hurteau & Brooks, 2011). Forest C is important to land managers as it may be used to assess fuel loading and help mitigate the effects of climate change, which is predicted to exacerbate

wildland fire activity (IPCC, 2005; McKenzie et al., 2004). The use of prescribed fire may, therefore, represent an opportunity for land managers to also incorporate C management into prescribed fire objectives (Hurteau et al., 2011).

There remains a limited understanding on the effects of fire type, and particularly prescribed fire, on soil C and PyC stocks, distributions, and cycling within geographic or ecological regions. Nonetheless, prescribed fire is being increasingly implemented in North American forests to restore fire to fire-dependent ecosystems (Ryan et al., 2013). Red pine (*Pinus resinosa* Ait) forests of the Lakes States region are an example of an ecosystem type that has been negatively impacted by alterations in fire regime due to fire suppression that has led to changes in forest species composition (Cleland et al., 2004; Frelich, 1995), fuel loads, and soil nutrient cycling (Alban, 1977; Miesel et al., 2012). Prescribed fire use within this region is often conducted to reduce the risk of high severity wildland fires and meet management objectives for silvicultural applications and ecosystem restoration (Dickmann 1993; Knapp et al., 2009; Ryan et al., 2013). Dormant season (i.e., spring or fall) prescribed fires are commonly implemented in red pine forests due to logistical constraints of summer fires (Melvin, 2015; Quinn-Davidson and Varner, 2012), although these dormant season burns often differ in fire conditions and behavior from those of summer burns and wildland fires. Despite the ecological and economic significance of red pine forests, there are few studies investigating the short- and long-term effects of season and frequency of prescribed fire in red pine ecosystems of the Lake States region on soil properties (Miesel et al., 2012) related to forest health and to achieve specific management objectives including C sequestration.

To address our over-arching research questions of how the season and frequency of prescribed fire and time since fire affect soil properties, we leveraged a historical prescribed fire

study to measure forest floor and mineral soil C and PyC stocks and PyC concentrations. Our study capitalized on the *Red Pine Prescribed Burning Experiment* (Alban, 1977; Buckman, 1964; James et al., 2018; Scherer et al., 2016) that investigated the effects of season and frequency of prescribed fire on site productivity and soil properties in a naturally-regenerated red pine forest in northern Minnesota (USA). Prescribed fire treatments were implemented along with measurements of soil properties from 1959 to 1969 (Alban, 1977). Our study used the original historical study site with new measurements in spring 2015 collected in forest floor horizons and mineral soil increments (0–91 cm) to address the following objectives: (1) evaluate the effects of prescribed fire treatments and time since fire (>45 years post-fire) on total C and PyC stocks and PyC concentrations and distributions; and (2) explore the relationship of PyC with soil physical and chemical properties. We hypothesized that: (1) total C and PyC stocks and PyC concentrations would be greatest for the control treatment and decreased for the summer annual treatment and (2) PyC stocks measured across all treatments would have a strong relationship with organic matter content (total C, depth, mass), bulk density, and overall nutrient stocks.

3.3. Methods

3.3.1. Study Area

We investigated the area established by the U.S. Forest Service in 1959 for the *Red Pine Prescribed Burning Experiment* located in the Cutfoot Experimental Forest (CEF) within the Chippewa National Forest, in northern Minnesota, USA (latitude 47°40'N, longitude 94°5'W). The site characteristics and experimental design have been described in detail by previous studies conducted in the CEF (Alban, 1977; Buckman, 1964; James et al., 2018; Scherer et al.,

2016). The climate in the CEF is continental with summer temperatures exceeding 32°C and minimum winter temperatures below -35°C (U.S. Forest Service, 2009). Annual precipitation ranges between 500–640 mm. Winter snowfall depths range from 1–2 m and prolonged summer droughts are common. The forest stand at our study originated following a natural wildfire in 1870, and fire scars indicate multiple fires in the mid to late 19th century (U.S. Forest Service, 2009). Prior to the initiation of the prescribed fire experiment in 1959, mature red pine (90–years-old) was the dominant overstory species with an average of 30.7 cm dbh (diameter at breast height, 1.37 m) and a site index of 15.2 m at 50 years was estimated. Other species included jack pine (*Pinus banksiana* Lamb.), eastern white pine (*Pinus strobus* L.), paper birch (*Betula papyrifera* Marsh.), and quaking aspen (*Populus tremuloides* Michx.) (U.S. Forest Service, 2009). To create uniform forest conditions, stand density was reduced by thinning to an overstory basal area of approximately 28 m² ha⁻¹ in the winter of 1959 (Alban 1977). The slash from the thinning was removed from the study site to reduce site variability, fuel loading, and overstory tree mortality from the use of prescribed fire.

Soils of the study site are classified as the Eagleview soil series, a mixed, frigid, Lamellic Udipsamment formed in glacial outwash parent material (NRCS, 2017). The soil is deep and well-drained with a fine to medium sand texture. The soil prior to implementation of the burning experiments in 1960 was described as weakly developed with the forest floor approximately 8 cm thick and underlying mineral soil consisted of loamy sand including A (0–1 cm), E (1–11 cm), and B (11–47 cm) horizons (Alban, 1977). Stratified sands and gravels interspersed with thin lenses of very fine sandy loam were measured below the B horizon and calcium carbonate occurred intermittently below 127 cm.

3.3.2. Experimental Design and Treatments

A randomized complete block design was established in 1959 for the *Red Pine Prescribed Burning Experiment*. Seven prescribed fire treatments representing contrasting fire season, frequency, and their interaction were randomly assigned to 0.4 ha compartments within each of four blocks (n=1 replicate per block) and implemented from spring 1960 through the summer of 1970. The 28 compartments were each surrounded by a fire exclusion perimeter and contained a 0.08 ha circular plot with a permanent center marker. Dormant season burns were conducted in the spring or fall, when leaves were absent, whereas summer burns were applied from late June through mid–August. Frequency of treatments were categorized as annual (every calendar year), biennial (every other calendar year), and periodic (every 6–9 years). The seven treatments implemented included: dormant annual (DA), dormant biennial (DB), dormant periodic (DP), summer annual (SA), summer biennial (SB), summer periodic (SP), and an unburned control (CC) for reference conditions.

Prescribed fire treatments were administered 5–15 days following a rain event (Buckman 1964; Alban 1977) and resulted in forest floor horizon moisture content averaging approximately 100% of dry weight in dormant season burns and 40% in summer burns (Alban, 1977). Prior to burning, fire lines were constructed to mineral soil surrounding each compartment and snags were felled and slash was removed from compartments. Burns were initiated using backing fires followed by strip headfires from 3–6 m in width. Red pine needles were the primary fuel source and fires exhibited low-to-moderate fire intensities with < 1 meter flame heights. Measurements in 1969 indicated the litter horizon was consumed for all burn treatments and the fermentation horizon for annual and biennial frequencies for both dormant and summer season burns (Alban, 1977). Organic matter content in the forest floor decreased by approximately 50% for summer

annual burns and exposed mineral soil in < 5 % of the burn compartments following complete consumption of the forest floor horizons (Alban, 1977). The last prescribed fire treatments were implemented in 1970 and resulted in 10–11 burns in annual treatments, five burns in biennial treatments, and two burns in periodic treatments. No additional prescribed fire treatments or changes to the experimental units have been made since the summer of 1970.

3.3.3. Field Methods

We resampled the original *Red Pine Prescribed Burning Experiment* forest plots in June 2015 and collected forest floor and mineral soil samples. Soil samples from 1959–1969 were collected along a NE (45°) to SW (225°) transect bisecting the plot center. However, all soil sampling locations along the transect had been previously sampled. Therefore, we established a new sampling transect along an adjusted NE (22.5°) to SW (202.5°) azimuth following the original author's guidelines. We measured and collected organic horizons and mineral soil increments at 3.05 m from the plot center along each NE and SW azimuth within each of the 28 compartments for a total of 56 subsampling points. We placed a 30 cm diameter circular frame at each subsampling point to record organic horizon (litter (O_i), fermentation (O_e), and humus (O_a)) depth at three locations along the circumference of the circular frame. Four locations were used and averaged if any anomalies (i.e., tree roots, rocks, etc.) were present. Each of the three organic soil horizons were collected from within the circular frame following cutting with a serrated gardening knife around the inside circumference of the circular frame. Tree cones, bark, and all woody debris were included, whereas woody fuels > 0.64 cm (0.25 inches) were omitted from collection within each horizon. Following removal of all organic soil horizons, we collected mineral soil samples within the circular frame. The original study had collected two different sets

of mineral soil depth increments. We adopted the most recent increment depths at 0–10.16 cm, 10.16–50.80 cm, and 50.80–91.44 cm (Alban, 1977). We used a slide hammer with attached cup and sleeve to collect the 0–10.16 cm increment, a t-handle soil probe to collect the 10.16–50.80 cm increment, and a slide hammer with attached soil probe to collect the 50.80–91.44 cm increment.

3.3.4. Laboratory Analysis

Remeasured 2015 soil response variables in the organic soil horizons included total nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), magnesium (Mg), pH, depth, mass, organic matter (OM), ash content, and bulk density along with mineral soil N, P, K, Ca, Mg, pH, cation exchange capacity (CEC), soil texture, and bulk density. Laboratory and chemical analysis methods of previously mentioned soil response variables were first described by Alban (1977) and repeated in 2015 (James et al., 2018). For this study, we also quantified total C and PyC stocks and PyC concentrations in each organic horizon and mineral soil depth increment; these response variables were not measured in the original study and are the focus in our study. Organic soil horizons were oven-dried at 32°C and stored at ambient air temperatures prior to processing. Each organic horizon subsample was weighed after all living material (i.e., plants, roots, lichens, moss, insects, worms, etc.), scat, and stones were discarded. Within-plot subsamples were composited into one soil sample per plot.

Organic horizons were ball milled (Spex SamplePrep 8000D, USA) and then oven-dried at 60°C for 48 hours prior to chemical analysis. Organic soil horizon total C was determined on a dry combustion elemental analyzer (Costech, Italy, combustion temperature 1,000 °C). We derived PyC content by estimating the mass fraction of charred material within the litter and

fermentation horizons by acquiring mid-infrared (MIR) spectra from dry, finely-ground samples using a Bruker Vertex 70 (Bruker Optics, Billerica, MA USA) with a wide-range Si-based beam-splitter and MIR detector with cesium iodide windows. We attained spectra on undiluted (neat) samples using diffuse reflectance (Pike Autodiff accessory, Pike Technologies, Madison, WI) from 6000-180 cm^{-1} , with 4 cm^{-1} resolution. We also obtained a background spectrum (average of 60 scans) for each set of samples, and subtracted the background spectrum from the sample reflectance spectra (also an average of 60 scans). We used a previously validated, partial least squares regression (PLSR) model (developed using The Unscrambler X software, CAMO Inc.) to predict char concentration (i.e., mass fraction of char in total sample mass). The PLSR model was developed using laboratory standards consisting of known mixtures of pine needle litter and char produced from pine needles or pine wood at temperatures of 300 and 550 °C, with char mass fraction varying from 0 to 100% in 5% increments (Miesel et al., in prep). This variance in char was successfully captured via a 2-factor PLSR model with 20-fold cross-validation, resulting in R^2 of 0.97 and root mean square error (RMSE) of 5.0%. We then applied corrections to estimate PyC mass fraction in the litter and fermentation horizons, by multiplying FTIR-predicted char mass by the mean values for total C concentrations (60.41%) and percent of total C in char identified as PyC (33.90%), derived from chemical oxidation of the pine needles pyrolyzed at 300 and 550 °C (Maestrini et al., 2017). The PLSR model used to estimate litter and fermentation char mass fraction was developed using organic soils (>20% organic matter). However, because many of the humus horizon samples had high mineral content, we used a modified approach to estimate char mass in this layer, as follows: first, for each sample location, we assumed that the ratio between the FTIR-predicted char mass fraction and the organic matter (OM) mass fraction in humus horizon was identical to the same ratio in the overlying

fermentation horizon, assumed the ratio between mass fractions of organic matter and of char was similar between these layers. We then used this ratio and the known OM mass in the humus horizon to calculate char mass, and then applied the same calculations described above to estimate PyC mass fraction in the humus horizon. PyC concentrations (%) were determined as the PyC mass fraction of total C within soil horizon and treatment.

We oven-dried mineral soils at 32°C and stored samples at ambient air temperatures prior to analysis. Soil subsamples were processed after removal of all visible organic material. Each mineral subsample was sieved through a 2 mm screen and the fine fraction within each appropriate increment was composited into one soil sample per plot and used for chemical analysis. We ball milled mineral soils and oven-dried samples at 105°C for 48 hours prior to chemical analysis. Mineral soil total C was determined on a dry combustion elemental analyzer (Costech, Italy, combustion temperature 1,000 °C). Mineral soil PyC was measured using a weak nitric acid digestion in the Kurth-MacKenzie-Deluca (KMD) method (Kurth et al., 2006). We digested 0.5 g of pulverized mineral soil at 100° C in 10 mL of 1 M nitric acid and 20 mL of hydrogen peroxide (30%) in a block digester (SEAL Analytical Inc., United Kingdom). We then measured the mass of the digested sample and C concentration using elemental analysis as described above, and corrected for mass loss during digestion (Buma et al., 2014). The soil C quantified after chemical digestion is considered PyC. Mineral soil PyC concentrations (%) were determined as the PyC mass fraction of total C within soil horizon and treatment.

3.3.5. Statistical Analysis

The effects of prescribed fire on forest floor and mineral soil total C and PyC stocks and PyC concentrations across treatments measured spring 2015 were analyzed using analysis of

variance (ANOVA) in a mixed model approach using season, frequency, and their interaction as fixed effects and block as a random effect with PROC MIXED procedures in SAS 9.3. Because the control treatment was not a level of season nor frequency, the control was temporarily removed to enable analysis and reflect the appropriate error degrees of freedom. The assumption of normality of residuals and homogeneity of variance was assessed between all treatments and data transformations were applied when appropriate to meet model assumptions. Where ANOVA indicated significant differences among prescribed fire treatments (n=4), we performed pairwise contrasts using Tukey's adjustment for multiple comparisons. We then conducted a Dunnett's test to contrast prescribed fire treatments to the control. The final model selected was assessed by Akaike Information Criteria (Akaike, 1974). The probability of a Type I error $\alpha = 0.10$ was used in statistical testing due to the high variability of responses in studies of soils. We also used correlation analysis to assess the relationship of PyC with the soil properties measured in 2015 (James et al., 2018), across all treatments (n=28) within soil layers. All variables were evaluated for normality by assessing the variance of residuals and log-transformed if necessary to meet assumptions of normality. Relationships were considered statistically significant at the $\alpha = 0.10$ level.

3.4. Results

3.4.1. Forest Floor C and PyC Stocks and PyC Concentrations

We found no significant differences among treatments within forest floor horizons (litter, fermentation, humus, total forest floor) for total C stocks, although a trend in total C stocks decreased with increasing summer burn frequency; in contrast, total C stocks increased with increasing dormant season burn frequency in forest floor horizons (Table 3.1., Figure 3.1.).

Among treatments, total C stocks was greatest in the humus horizon (105.94 Mg ha⁻¹), whereas the C stocks in the litter (36.52 Mg ha⁻¹) and fermentation (25.98 Mg ha⁻¹) horizons were 66% and 76% lower, respectively (Figure 3.1.). Total PyC stocks among treatments in the litter (1.17 Mg ha⁻¹) and fermentation (3.32 Mg ha⁻¹) horizons were 91% and 74% smaller compared to the humus (12.82 Mg ha⁻¹) horizon, respectively (Figure 3.2.). We found a persistent decrease in total PyC stocks by 71% for the summer annual burn (1.02 Mg ha⁻¹, $p = 0.0912$) relative to the control (3.46 Mg ha⁻¹) in the humus horizon, and by 64% for the summer annual burn (1.55 Mg ha⁻¹, $p = 0.0902$) relative to the control (4.31 Mg ha⁻¹) in the total forest floor (Figure 3.2.). We observed a significant decrease in PyC concentrations for the summer periodic burn (9.99%, $p = 0.0600$) relative to the control (22.51%) in the fermentation horizon, and for the dormant periodic (9.21%, $p = 0.0651$), summer annual (9.41%, $p = 0.0747$), summer biennial (9.80%, $p = 0.0961$), and summer periodic (8.83%, $p = 0.0505$) burns relative to the control (17.41%) in the humus horizon (Figure 3.3.).

3.4.2. Mineral Soil C and PyC Stocks and PyC Concentrations

We found no significant differences among treatments for total C and PyC stocks and PyC concentrations, for any of the mineral soil depth (0–10 cm, 10–51 cm, and 51–91) increments (Table 3.2, Figure 3.1., 3.2., and 3.3.). In general, trends in total C and PyC stocks measured in the mineral soil (0–91 cm) decreased with increasing summer burn frequency but increased with increasing dormant season burn frequency. Among treatments, total C stocks were 23% less in the 0–10 cm (107.53 Mg ha⁻¹) and 70% less in the 51–91 cm (41.37 Mg ha⁻¹) increments relative to the 10–51 cm (140.47 Mg ha⁻¹) increment (Figure 3.1.). Total C stocks in the mineral soil (0–91 cm) increased by 30% for the dormant annual burn (53.83 Mg ha⁻¹) and

decreased by 29% for the summer annual burn (29.34 Mg ha⁻¹) relative to the control (41.40 Mg ha⁻¹). Among treatments by horizon, total PyC stocks were 61% less in the 0–10 cm (11.07 Mg ha⁻¹) increment and 22% less in the 51–91 cm (22.12 Mg ha⁻¹) increment relative to the 10–51 cm (28.43 Mg ha⁻¹) increment (Figure 3.2.). Total PyC stocks in the mineral soil (0–91 cm) increased by 16% for the dormant annual burn (8.55 Mg ha⁻¹) and decreased by 39% for the summer annual burn (6.09 Mg ha⁻¹) relative to the control (8.94 Mg ha⁻¹). PyC concentrations were greatest in the 51–91 cm increment and ranged from 39% for the dormant periodic burn to 65% for the control (Figure 3.3.).

3.4.3. Total Soil Profile C and PyC Stocks

Similar to the forest floor and mineral soil, total C and PyC stocks within the total soil profile (forest floor and mineral soil (0–91 cm) combined) decreased with increasing frequency of summer burns; in contrast, total C and PyC stocks increased with increasing frequency of dormant season burns. Total C stocks measured among treatments in the total soil profile (447.47 Mg ha⁻¹) indicated 62% of total C in the mineral soil (279.03 Mg ha⁻¹) relative to the forest floor (168.44 Mg ha⁻¹), and total C stocks were 5.4 times greater in the 10–51 cm increment (140.47 Mg ha⁻¹) compared to the fermentation horizon (25.98 Mg ha⁻¹). Within the total soil profile, we found a persistent decrease in total C stocks for the summer annual burn (48.03 Mg ha⁻¹) relative to the dormant annual (83.03 Mg ha⁻¹, $p = 0.0505$) and summer periodic (67.73 Mg ha⁻¹, $p = 0.0323$) burns (Table 3.3.). Total PyC stocks measured among treatments in the total soil profile (73.41 Mg ha⁻¹) indicated 76% of total PyC in the mineral soil (56.08 Mg ha⁻¹) compared to the forest floor (17.33 Mg ha⁻¹), and total PyC stocks were 24 times greater in the 10–51 cm increment (28.43 Mg ha⁻¹) relative to the litter horizon (1.17 Mg ha⁻¹). Although not significant,

PyC stocks in the total soil profile decreased by 42% for the summer annual burn (7.64 Mg ha^{-1}) relative to the control (13.25 Mg ha^{-1}).

3.4.4. Relationships Between PyC and Other Soil Properties

In general, soil properties displayed a weak but statistically significant relationship with PyC in the forest floor and mineral soil (Table 3.4.). Out of the 69 correlation analyses performed, 29%, 12%, 14%, and 12% were significant at $p < 0.0001$, $p < 0.001$, $p < 0.010$, $p < 0.05$, and $p < 0.10$ respectively, whereas 33% of the relationships were not statistically significant. Hereafter we provide the results of the strongest relationships with $R > 0.50$ (35%). Overall, the number of relationships with $R > 0.50$ was greater in mineral soil (58%) relative to the forest floor (42%). The number of significant relationships increased in organic horizons moving towards the humus (33%) horizon, whereas the number and strength of significant relationships in the mineral soil was greatest in the 10–51 cm increment (29%, $R = 0.80$), respectively. There were no clear trends in the number or strength of relationships of soil properties with PyC across soil depths (Table 3.4.). However, the relationships between Mg and organic matter (content, mass, total C) displayed stronger R values relative to other soil properties, and Mg within the humus horizon displayed the strongest relationship with PyC ($R = 0.84$).

3.5. Discussion

3.5.1. Total C and PyC Stocks and PyC Concentrations by Soil Horizon

We found that within the total soil profile (combined forest floor and mineral soil (0-91 cm)), there was almost twice as much total C in the mineral soil (62%) relative to the forest floor

(38%). Our findings of large C stocks in the mineral soil agree with those of a meta-analysis which showed that mineral soil C was greater by a factor of two in unburned sites and by a factor of five in burned sites, relative to the forest floor in the same sites (Nave et al., 2011). Our results indicated total C stocks were lowest in the fermentation horizon, whereas the majority of total soil C was measured in the 10–51 cm mineral soil increment and supports a global review of studies reporting soil C stocks in mineral soil layers at these depths (Jobbágy & Jackson, 2000). The increased proportion of C stocks in the 10–51 cm increment may be attributed to a combination of factors. The direct effects of fire are often limited to surface mineral soils (0–5 cm) (Neary et al., 2005; Certini, 2014) and would not be expected to directly impact C stocks at the 10–51 cm increment depth. However, combustion and removal of organic horizons and mortality of aboveground vegetation and belowground root biomass (Schmidt et al., 2011) during fire increases movement of residual C debris into the mineral soil after fire. The highly permeable sandy soils at our study likely increased the translocation of organic matter and nutrients to the 10–51 cm increment, which corresponds to the B horizon and zone of illuviation. Microbial biomass, diversity, and decomposition rates are often greater in topsoil (Sanaullah et al., 2011; Staddon et al., 1997) and generally decreases with mineral soil depth (Blume et al., 2002; Fierer et al., 2003), and therefore are expected to influence the long-term stability of C with increased mineral soil depth (Dungait et al., 2012). In addition, the differences in C stocks we observed in the 0–10 cm, 10–51 cm, and 51–91 cm increment depths likely reflects the influence of microbial activity, as well as the influence of soil depth measured within each increment (i.e., 10cm, 40cm, and 40cm, respectively).

Our findings showed that PyC stocks increased by more than a factor of three in the mineral soil (76%) relative to the forest floor (24%), which is similar to a wildland fire study in

California that reported 81% and 82% of PyC stocks were stored in the mineral soil (0–5 cm) in low-to-moderate and high fire severity classes, respectively, relative to the forest floor two years post-fire (Maestrini et al., 2017). PyC stocks in our study were lowest in the litter horizon as expected, whereas most PyC was measured in the 10–51 cm mineral soil increment. PyC close to the soil surface may act as fuel and be susceptible to consumption by fire (Preston & Schmidt, 2006), and potentially contribute to decreased PyC stocks in forest floor horizons and the 0–10 cm increment. Vertical translocation through the mineral soil profile can be driven by PyC properties and soil characteristics (Schmidt et al., 2000) as well as by bioturbation or other physical processes (Preston and Schmidt, 2006; Schmidt et al., 2000). Haefele et al. (2011) reported that 50% of PyC moved below 30 cm in sandy soils following four years after application, and this vertical movement and accumulation of PyC in deeper soil horizons may contribute to its preservation (Dungait et al., 2012; Lorenz and Lal, 2014).

The 51–91 cm mineral soil increment displayed the greatest PyC concentration among treatments. Most vegetation fires do not exceed the temperatures required to initiate charring (<200°C) a few millimeters below the mineral soil surface (Gonzalez-Perez et al., 2004), therefore, most of the PyC in mineral soils likely originates from the burning of aboveground or forest floor material (Bodí et al., 2014; Boot et al., 2015), and subsequent movement into the mineral soil. Increased PyC concentrations and stabilization of C in the 51–91 cm increment is likely due to the deep translocation through the sandy soils at our study site. The resistant form of PyC and limited accessibility by microbes and exo-enzymes at these depths has been suggested as a mechanism for subsoil C sequestration (Dungait et al., 2012; Golchin et al., 1997). In addition, environmental conditions (e.g., moisture and temperature) are more stable at these subsoil depths relative to surface soils (Sanaullah et al., 2011) and may increase physical and

chemical C stability, and in particular, more resistant forms of C such as PyC (Dungait et al., 2012).

3.5.2. Total C and PyC Stocks by Prescribed Fire Season, Frequency, and Time Since Fire

The patterns in total C and PyC stocks among treatments corresponded with overall nutrient stocks measured in these soil layers (James et al., 2018). These observations suggest, in general, that soil responses differed by season of burning and were further magnified by increased fire frequency within season. For example, we observed a trend in increased total C stocks in the dormant season burns, and in contrast, decreased stocks in the summer burns with increasing fire frequency within season. Comparatively, Kolka et al. (2014) demonstrated no differences in 0–10 cm or 10–20 cm mineral soil C pools measured across soil burn severity levels, immediately post-fire and one year post-fire in a pine dominated (e.g., *Pinus banksiana* Lamb. and *Pinus resinosa* Ait.) site in northeastern Minnesota impacted by wildland fire. The patterns in total C stocks we observed among treatments are likely attributed to the short- and long-term (>45 years post-fire) effects of local vegetation responses to prescribed fire (Buckman, 1964; Scherer et al., 2016), and supports a review by Jobbágy & Jackson (2000) that indicated vegetation type significantly affects the vertical distribution of soil organic C. Studies have shown complete recovery of forest floor C within 40 years of a fire event (Nave et al., 2011), which agrees with our observations and of no differences in total C stocks among the dormant or summer periodic treatments compared to the control in the total soil profile.

We found PyC stocks in the total soil profile were lowest for the summer annual treatment and greatest for the control. The increased PyC stocks in the control may reflect PyC production from historical wildland fires, with the most recent documented wildfire in 1918, and

supports the production and long-term stability of PyC. PyC stocks decreased with increased fire frequency within both seasons, and these observations may support an abiotic loss mechanism and reflect consumption of residual PyC by subsequent fires (Czimczik et al., 2005; Preston & Schmidt, 2006; Kane et al., 2010; Santín et al., 2015). However, increased fire frequency in summer burns resulted in a more pronounced decline in PyC stocks relative to dormant season burns. Seasonal dissimilarities and lower fuel moisture content and higher fire intensities, characteristic of summer burns (Alban, 1977; Govender et al., 2006), may explain the depletion of PyC stocks, although there are few studies quantifying consumption of existing PyC (Santín et al., 2013).

The most pronounced effect of treatment and time since fire on PyC concentrations was observed in the 51–91 cm mineral soil increment, ranging from a PyC concentration of 39% for the dormant periodic treatment to 65% for the control. These observations among treatments may be due to lower post-fire erosion events and rates of PyC (Bodí et al., 2014; Santín et al., 2015) in the control relative to other burn treatments, although slopes were minimal (1-8%) across our study site. In general, PyC concentrations were greater for summer burns compared to dormant season burns, whereas total PyC stocks experienced a greater decrease for summer burns. Relative to dormant season burns, the greater fire temperatures that occur in summer burns and wildland fires, have been shown to increase recalcitrance of PyC (Singh et al., 2012; Whitman et al., 2013), and likely explains the increase in PyC concentration for summer burns at these depths.

3.5.3. Relationship of PyC with Soil Properties

The increased number of relationships between PyC and soil properties detected, particularly in the humus horizon and 10–51 cm mineral soil increment, are similar to increased PyC stocks, organic matter content (i.e., total C, mass, depth), and overall nutrients stocks (James et al., 2018) measured in these horizons relative to other horizons. The relationship of PyC to organic matter content we observed in the humus horizon and 0–10 cm mineral soil increment may be due to the conversion of organic material to PyC during fire events (Bird et al., 2015; Bodí et al., 2014). PyC has also been shown to increase plant nutrient availability and soil fertility (Lehmann et al., 2006; Biederman et al., 2013), and may stimulate plant growth and subsequent organic matter contributions in these soil layers. Greater nutrient stocks (P, K, Ca, Mg) in the humus horizon and 10–51 cm mineral soil increment may reflect the ability of PyC to increase cation exchange capacity (observed in the 10–51 cm increment) and to influence nutrient cycling and biogeochemical processes in the soil (Liang et al., 2006; Biederman et al., 2013) that affect forest recovery following a fire event.

3.6. Conclusions and Management Implications

Persistent differences among prescribed fire treatments and time since fire (>45 years) had minimal effects on C and PyC stocks with exception of the summer annual burn. Deep mineral soils appear to function as a quantitatively relevant reservoir of C and PyC (Lorenz and Lal, 2014; Schmidt et al., 2011) and are important when estimating total soil and forest C stocks. Although prescribed fire treatments may result in initial losses of C and PyC stocks in forest floor horizons, these effects may be moderated by the relatively larger pools of C and PyC stocks in the mineral soil which are primarily unaffected by the direct effects of fire, the deep storage of

C and PyC and increased proportion of resistant forms of PyC in deep mineral subsoils, as well as by forest floor C recovery since the last prescribed fire.

Our results suggest that infrequent summer prescribed fires, including summer periodic burns, may be a valuable approach to increase the variability in burn schedules to be more representative of historical regional fire regimes, and are also compatible with other management objectives such as C sequestration. Sustained annual and biennial frequencies of prescribed fires are often not logistically feasible – and would be much more frequent than historical fire regimes prior to Euro-American settlement in this ecosystem type (Bergeron and Brisson, 1990; Guyette et al., 2016) – but may be valuable for fuels reduction or for the early stages of ecosystem restoration (Agee and Skinner, 2005; Knapp et al., 2009). Summer fires in red pine ecosystems have been shown to have desirable effects on soil properties (Alban, 1977; James et al., 2018) and to increase plant species richness and diversity (Weyenberg and Pavlovic, 2014) while reducing understory competition (Buckman, 1964; Scherer et al., 2016). However, there are several logistical constraints to conducting summer prescribed fires including weather conditions, resource availability, and safety concerns (Melvin, 2015; Quinn-Davidson and Varner, 2012). Our results help address the need for regionally-specific estimates of soil responses to fire. Increased availability of regionally-specific studies such as ours will enable greater understanding of potential beneficial or detrimental consequences of fire and forest management activities within and across regions, thereby helping to increase the effectiveness of prescribed fire in fire-dependent ecosystems.

APPENDIX

Table 3.1. Results of analysis of variance (ANOVA) using a mixed model approach for organic soil layer response variables in the litter (L), fermentation (F), humus (H), and total forest floor (TFF) horizons measured in 2015 (>45 years post-fire) in the Cutfoot Experimental Forest in northern Minnesota, USA. An ns: not significant at any level, whereas *p <0.10, **p <0.05, ***p <0.01, ****p <0.001.

		Total C			PyC			PyC (%)		
		F	p-value		F	p-value		F	p-value	
L	Season	0.10	0.7586	ns	0.15	0.7089	ns	0.55	0.4739	ns
	Frequency	0.23	0.7988	ns	1.79	0.2236	ns	2.50	0.1432	ns
	Season × frequency	3.45	0.0587	*	1.04	0.3948	ns	2.53	0.1412	ns
F	Season	0.00	0.9787	ns	0.00	0.9937	ns	0.54	0.4798	ns
	Frequency	0.64	0.5609	ns	0.09	0.9161	ns	0.86	0.4517	ns
	Season × frequency	1.14	0.3816	ns	0.55	0.5907	ns	0.32	0.7362	ns
H	Season	11.32	0.0056	***	4.40	0.0603	*	1.86	0.1972	ns
	Frequency	0.89	0.4510	ns	0.34	0.7225	ns	1.72	0.2380	ns
	Season × frequency	0.70	0.5298	ns	0.22	0.8084	ns	0.81	0.4760	<u>ns</u>
TFF	Season	0.00	0.9774	ns	3.76	0.0793	*	1.30	0.2821	ns
	Frequency	0.07	0.9302	ns	0.34	0.7179	ns	1.55	0.2613	ns
	Season × frequency	0.45	0.6525	ns	0.19	0.8282	ns	0.91	0.4350	ns

Table 3.2. ANOVA using a mixed model approach for mineral soil response variables by increment depth (0–10.16 cm, 10.16 – 50.80 cm, 50.80 – 91.44 cm) measured in 2015 (>45 years post-fire) in the Cutfoot Experimental Forest in northern Minnesota, USA. An ns: not significant at any level, whereas *p <0.10, **p <0.05, ***p <0.01, ****p<0.001.

		Total C			PyC			PyC (%)		
		F	p-value		F	p-value		F	p-value	
0–10 cm	Season	0.34	0.5690	ns	0.33	0.5755	ns	2.09	0.1659	ns
	Frequency	0.79	0.4755	ns	0.98	0.3956	ns	1.16	0.3355	ns
	Season × frequency	0.49	0.6247	ns	0.77	0.4794	ns	0.78	0.4754	ns
10–51 cm	Season	1.85	0.2115	ns	1.80	0.1960	ns	0.21	0.6513	ns
	Frequency	0.53	0.6168	ns	0.60	0.5574	ns	0.31	0.7390	ns
	Season × frequency	1.64	0.2770	ns	0.71	0.5052	ns	0.16	0.8492	ns
51–91 cm	Season	0.49	0.5078	ns	0.01	0.9081	ns	0.18	0.6758	ns
	Frequency	0.82	0.4842	ns	0.37	0.7024	ns	0.18	0.8401	ns
	Season × frequency	0.44	0.6631	ns	0.21	0.8168	ns	0.96	0.4007	ns

Table 3.3. Mean ($\pm$ standard error) of total carbon (C) and pyrogenic carbon (PyC) stocks, and PyC concentration mass fractions in prescribed fire treatments, shown for litter (L), fermentation (F), humus (H), total forest floor (TFF; litter, fermentation, humus) horizons, mineral soil depth increments (0–10.16 cm, 10.16–50.80 cm, 50.80–91.44 cm), and total soil profile (forest floor and mineral soil (0–91 cm) combined) measured in 2015 (>45 years post-fire) in the Cutfoot Experimental Forest in northern Minnesota, USA. Different letters within each row indicate statistically significant differences among treatments at $\alpha = 0.10$.

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
Total C (Mg ha ⁻¹)	L	5.40 (0.92)	5.87 (0.32)	4.76 (0.91)	4.67 (0.38)	3.99 (1.02)	5.96 (0.37)	5.87 (0.72)
	F	3.24 (0.52)	5.11 (1.18)	2.55 (0.45)	3.47 (0.40)	3.55 (0.35)	3.90 (0.27)	4.16 (0.97)
	H	17.97 (3.48)	18.22 (2.05)	14.75 (2.62)	19.51 (2.68)	11.16 (2.59)	11.88 (0.46)	12.45 (0.55)
	TFF	26.61 (3.67)	29.20 (2.75)	22.06 (3.42)	27.65 (3.13)	18.69 (2.21)	21.74 (0.16)	22.49 (0.89)
	0–10 cm	11.37 (2.52)	17.97 (5.99)	11.11 (2.10)	16.36 (4.30)	15.65 (2.17)	15.75 (1.29)	19.32 (3.00)
	10–51 cm	26.78 (7.98)	31.56 (10.30)	16.45 (4.18)	18.66 (3.92)	10.14 (0.55)	15.44 (0.59)	21.44 (7.49)
	51–91 cm	4.33 (1.50)	5.72 (1.56)	6.01 (1.01)	7.99 (3.33)	4.73 (1.56)	6.60 (1.78)	5.99 (0.38)
	Total profile	68.01 (8.23) ab	83.03 (7.77) a	54.13 (5.68) ab	68.67 (13.72) ab	48.03 (3.04) b	57.87 (1.63) ab	67.73 (5.68) a
PyC (Mg ha ⁻¹)	L	0.19 (0.04)	0.14 (0.04)	0.25 (0.06)	0.13 (0.01)	0.09 (0.03)	0.13 (0.02)	0.24 (0.14)
	F	0.65 (0.13)	0.49 (0.04)	0.39 (0.13)	0.45 (0.09)	0.44 (0.08)	0.50 (0.03)	0.40 (0.12)
	H	3.46 (1.71) a	1.83 (0.53) ab	2.48 (1.09) ab	1.76 (0.33) ab	1.02 (0.25) b	1.15 (0.16) ab	1.12 (0.27) ab
	TFF	4.31 (1.85) a	2.47 (0.51) ab	3.12 (1.23) ab	2.34 (0.40) ab	1.55 (0.32) b	1.79 (0.19) ab	1.76 (0.21) ab
	0–10 cm	1.32 (0.38)	2.06 (0.37)	1.42 (0.32)	1.58 (0.40)	1.56 (0.17)	1.44 (0.08)	1.69 (0.13)
	10–51 cm	5.09 (1.51)	4.48 (1.56)	4.36 (1.43)	4.39 (0.72)	2.25 (0.28)	3.45 (0.35)	4.41 (0.93)
	51–91 cm	3.38 (0.37)	2.68 (1.07)	3.56 (0.24)	3.22 (0.86)	3.05 (0.25)	3.19 (0.52)	3.04 (0.34)
	Total profile	13.25 (1.95)	11.01 (2.73)	11.57 (1.22)	10.74 (1.64)	7.64 (1.24)	9.08 (0.79)	10.14 (0.29)

Table 3.3. (cont'd)

Variable	Horizon	Control	Dormant annual	Dormant biennial	Dormant periodic	Summer annual	Summer biennial	Summer periodic
PyC	L	3.42 (0.25)	2.29 (0.69)	5.70 (1.70)	2.90 (0.33)	2.26 (0.41)	2.26 (0.37)	4.21 (2.62)
(%)	F	22.51 (7.35) a	11.26 (2.41) ab	15.81 (5.62) ab	12.64 (1.67) ab	12.08 (1.15) ab	13.00 (0.98) ab	9.99 (1.89) b
	H	17.14 (5.03) a	9.95 (2.73) ab	14.63 (4.47) ab	9.21 (1.86) b	9.41 (1.15) b	9.80 (1.49) b	8.83 (1.77) b
	0–10 cm	11.31 (1.43)	20.42 (10.22)	12.55 (0.94)	9.79 (1.07)	10.17 (1.01)	9.40 (1.05)	9.09 (0.78)
	10–51 cm	21.01 (4.57)	21.47 (10.02)	26.41 (3.33)	24.77 (3.88)	21.95 (1.70)	22.48 (2.58)	23.80 (4.83)
	51–91 cm	64.94 (5.36)	48.46 (12.60)	62.25 (9.31)	38.96 (22.60)	50.80 (1.02)	55.68 (15.46)	51.13 (6.78)

Table 3.4. Results of simple linear regression between PyC and soil properties regardless of treatments (n=28), for each soil layer measured >45 years post-fire in the Cutfoot Experimental Forest in northern Minnesota, USA. Soil layers shown include the litter (L), fermentation (F), and humus (H) horizons and mineral soil increments (0–10.16 cm, 10.16–50.80 cm, 50.80–91.44 cm.). Soil properties measured included: depth, mass, bulk density (BD), ash, total C, N, P, K, Ca, Mg, pH, and cation exchange capacity (CEC). Soil correlation coefficients (R), direction of relationship (+/-), and p-value. Significance level is indicated by number of asterisks, for p <0.10 (*), p <0.05 (**), p <0.01 (***), and p<0.001 (****), whereas ns indicates not significant.

Variable	Layer	R	Relationship	p-value	Significance
Depth	L	0.36	+	0.0618	*
	F	0.27	+	0.1604	ns
	H	0.71	+	<.0001	****
Mass	L	0.33	+	0.0838	*
	F	0.65	+	0.0002	****
	H	0.81	+	<.0001	****
	0–10 cm	0.61	-	0.0006	****
	10–51 cm	0.49	-	0.0080	****
	51–91 cm	0.10	-	0.6584	ns
BD	L	0.01	+	0.9569	ns
	F	0.43	+	0.0228	**
	H	0.59	+	0.0009	****
OM	L	0.30	+	0.1200	ns
	F	0.33	+	0.0839	*
	H	0.66	+	0.0001	****
	0–10 cm	0.73	+	<.0001	****
	10–51 cm	0.79	+	<.0001	****
	51–91 cm	0.50	+	0.0206	**
Ash	L	0.35	+	0.0700	*
	F	0.41	+	0.0295	**
	H	0.26	+	0.1795	ns
	0–10 cm	0.71	-	<.0001	****
	10–51 cm	0.77	-	<.0001	****
	51–91 cm	0.50	-	0.0206	**
Total C	L	0.29	+	0.1340	ns
	F	0.38	+	0.0467	**
	H	0.73	+	<.0001	****
	0–10 cm	0.78	+	<.0001	****
	10–51 cm	0.45	+	0.0159	**
	51–91 cm	0.35	+	0.1217	ns

Table 3.4. (cont'd)

Variable	Layer	R	Relationship	p-value	Significance
N	L	0.41	+	0.0297	**
	F	0.30	+	0.1163	ns
	H	0.36	+	0.0601	*
	0–10 cm	0.64	+	0.0002	****
	10–51 cm	0.50	+	0.0065	***
	51–91 cm	0.01	+	0.9486	ns
P	L	0.44	+	0.0204	**
	F	0.37	+	0.0519	*
	H	0.53	+	0.0034	***
	0–10 cm	0.25	-	0.2027	ns
	10–51 cm	0.72	-	<.0001	****
	51–91 cm	0.16	+	0.4746	ns
K	L	0.19	+	0.3398	ns
	F	0.37	+	0.0496	**
	H	0.60	+	0.0007	***
	0–10 cm	0.49	+	0.0087	***
	10–51 cm	0.71	+	<.0001	****
	51–91 cm	0.33	+	0.1453	ns
Ca	L	0.50	+	0.0064	***
	F	0.17	+	0.3816	ns
	H	0.07	+	0.7367	ns
	0–10 cm	0.58	+	0.0013	***
	10–51 cm	0.77	+	<.0001	****
	51–91 cm	0.25	+	0.2842	ns
Mg	L	0.34	+	0.0760	*
	F	0.57	+	0.0014	***
	H	0.84	+	<.0001	****
	0–10 cm	0.59	+	0.0011	***
	10–51 cm	0.80	+	<.0001	****
	51–91 cm	0.38	+	0.0933	*
pH	L	0.18	+	0.3537	ns
	F	0.02	-	0.9334	ns
	H	0.11	+	0.5827	ns
	0–10 cm	0.07	+	0.7084	ns
	10–51 cm	0.01	+	0.9565	ns
	51–91 cm	0.15	-	0.5234	ns
CEC	0–10 cm	0.40	+	0.0361	**
	10–51 cm	0.73	+	<.0001	****
	51–91 cm	0.25	+	0.2839	ns

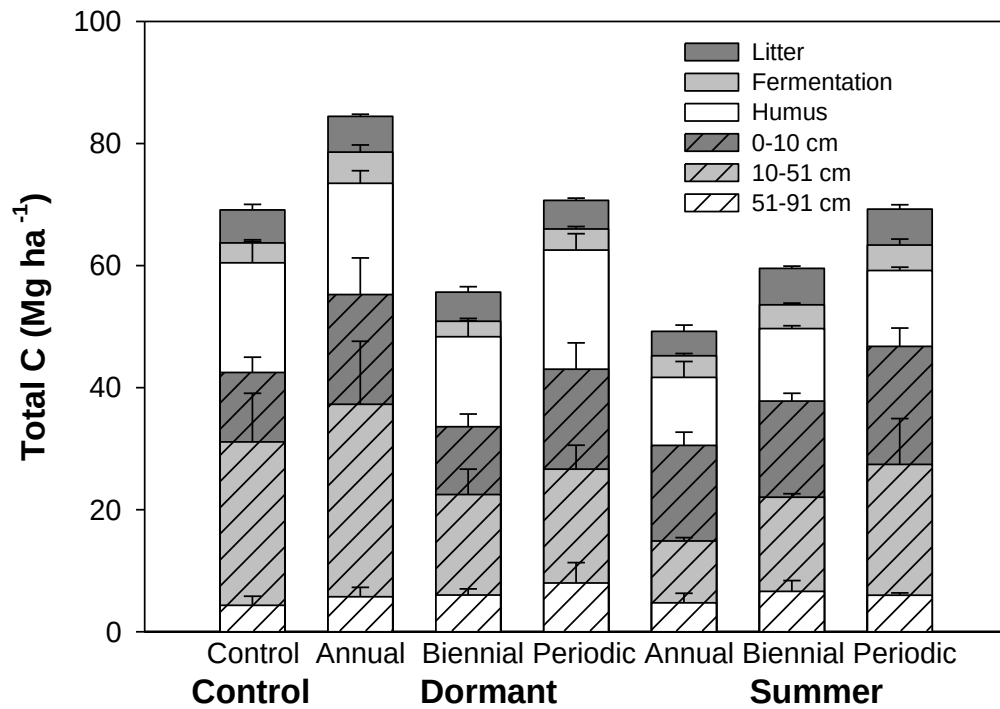


Figure 3.1. Stacked bar charts showing total C stocks in unburned control areas and contrasting prescribed fire treatments measured in 2015 in the Cutfoot Experimental Forest, >45 years post-fire. The total height of the bars represent mean total C stocks within treatment for $n=4$ replicates, whereas shading represents mean ($\pm$ standard error) C stocks in organic horizon and mineral soil depth increments. Lowercase letters indicate statistically significant differences across treatments within soil layer at $\alpha = 0.10$.

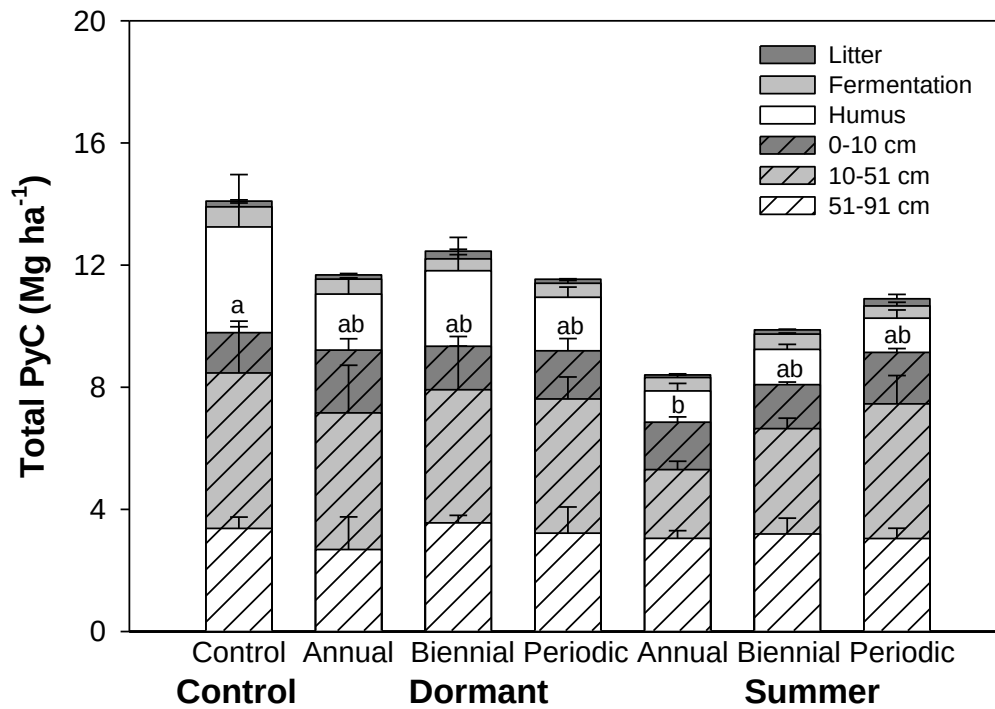


Figure 3.2. Stacked bar charts showing total PyC stocks in unburned control areas and contrasting prescribed fire treatments measured in 2015 in the Cutfoot Experimental Forest, >45 years post-fire. The total height of the bars represent mean total PyC stocks within treatment for $n=4$ replicates, whereas shading represents mean ($\pm$ standard error) PyC stocks in organic horizon and mineral soil depth increments. Lowercase letters indicate statistically significant differences across treatments within soil layer at $\alpha = 0.10$.

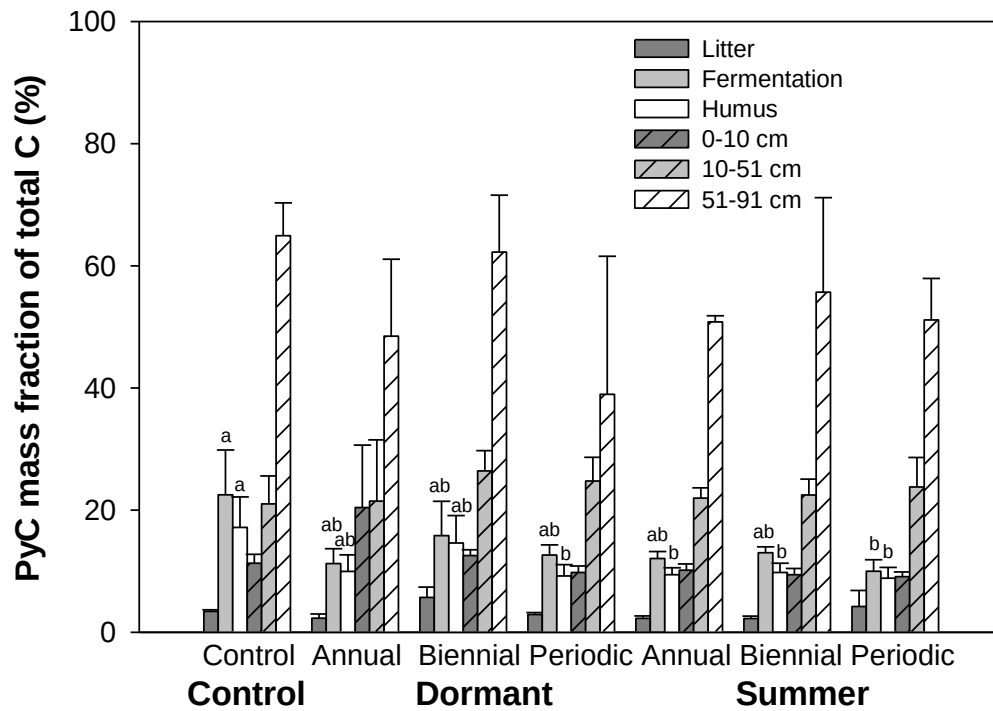


Figure 3.3. Bar charts showing mean PyC concentrations in unburned control areas and contrasting prescribed fire treatments measured in 2015 in the Cutfoot Experimental Forest, >45 years post-fire. The total height of the bars represent mean PyC concentrations within treatment ($\pm$ standard error) and organic horizon and mineral soil depth increments for $n=4$ replicates. Lowercase letters indicate statistically significant differences across treatments within soil layer at $\alpha = 0.10$.

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CHAPTER 4

RESEARCH BRIEF FOR RESOURCE MANAGERS: SHORT- AND LONG-TERM EFFECTS OF PRESCRIBED FIRE SEASON AND FREQUENCY ON SOIL PROPERTIES IN A RED PINE FOREST IN NORTHERN MINNESOTA

4.1. Introduction

Fire-dependent red pine (*Pinus resinosa* Ait.) forests of the Lake States region have important economic and ecological value. Similar to other fire-dependent ecosystems, red pine forests have been affected by prolonged fire suppression that has led to changes in forest structure and soil nutrient cycling (Frelich, 1995). Forest soils respond to changes in fire regime (i.e., fire season, frequency, severity) and influence ecosystem responses to fire including nutrient availability and vegetation recovery (Alban, 1977). Soils in fire-dependent ecosystems also store large amounts of carbon, including a type of carbon (pyrogenic carbon) produced from fires (Scharlemann et al., 2014). Fire can alter soil carbon stocks and long-term storage of carbon that affects soil nutrient cycling and is important for alleviating the effects of climate change due to increased carbon in the atmosphere (IPCC et al., 2000).

Prescribed fire is a management tool used to mitigate the effects of fire suppression and is often conducted to reduce the risk of high severity wildfires and meet management objectives for silvicultural applications and ecosystem restoration (Figure 4.1.) (Ryan et al., 2013). Managing for multiple objectives is challenging, as both ecological effects (i.e., soil responses, carbon storage, stand regeneration) and logistical constraints (i.e., safety, operational, weather, financial) are important considerations when implementing prescribed fire (Figure 4.2.). Dormant season (i.e., spring or fall) burns are often conducted due to weather and safety restrictions of summer burns. Yet, the understanding of short- and long-term effects of season and frequency of prescribed fire

on soils in the Lake States region is limited (Miesel et al., 2012). To address this knowledge gap, we leveraged a historical study (Alban, 1977), conducted from 1959-1970, with remeasurements of soil responses in 2015 to investigate the effects of prescribed fire season and frequency on soil properties (Figure 4.3.).

4.2. Objectives

Our specific objectives were to (1) evaluate individual (1959-1969) and cumulative (1959-2015) effects of fire treatments on soil responses over >55 years and (2) determine soil responses and changes over time since the last fire (>45 years).

4.3. Methods

Our study is located in the Cutfoot Experimental Forest, MN (Figure 4.4.). Prescribed fire treatments were established using a randomized complete block design. Seven treatments, including an unburned control, were randomly assigned to compartments and implemented from spring 1960 through the summer of 1969 to test the effect of season and frequency of prescribed fire (Table 4.1.) on organic and mineral soil horizon properties (Figure 4.5., Table 4.2.). No additional fire treatments or alterations to the experimental units have been performed since the summer of 1970.

4.4. Results and Management Implications

- **Regional studies are required** to accurately measure soil responses to fire that affect the development of local fire management plans.
- **Prescribed fire treatments had long-term effects on soil properties** in organic and mineral soil horizons > 45 years post-fire. Long-term effects of burns are likely attributed

to the interaction of the direct effects of organic matter combustion and loss/redistribution of nutrients, indirect effects of post-fire vegetation, and movement of organic matter and nutrients through the sandy soils (Figure 4.6.).

- **Prescribed fire season appeared to drive short- and long-term soil responses.** Summer burns had immediate and long-lasting desirable effects (i.e., decreased organic horizon depths, and nutrient stocks) without undesirable persistent effects (i.e., increased nutrient stocks or changes in soil texture, bulk density, and cation exchange capacity) in the mineral soil. Additionally, summer fires favor fire-adapted vegetation such as red pine and have been shown to increase species richness and diversity compared to dormant season fires.
- **Prescribed fire frequency magnified short- and long-term seasonal responses.** Annual frequencies of contrasting dormant and summer burns accounted for the majority of persistent effects. However, sustained annual and biennial frequencies of prescribed fires are often not logistically feasible but may be valuable to initiate ecosystem restoration.
- **Deep mineral soils served as a long-term storage of carbon** across treatments. Mineral soils are relatively unaffected by the direct effects of fire and may moderate the influence of fire on combustion and carbon loss in organic soil horizons.
- **Managing for multiple objectives is challenging.** Summer burns may be a valuable approach to increase the variability in burn schedules representative of historical regional fire regimes and be compatible with carbon management objectives. However, competing objectives and logistical constraints of summer burns pose challenges to land managers.

APPENDIX

Table 4.1. Description of prescribed fire treatments in the original *Red Pine Prescribed Burning Experiment* (1959-1970) in the Cutfoot Experimental Forest, Minnesota testing the effects of season and frequency of prescribed fire.

Treatment				
Season	Frequency	Trt	Burn dates (month/year)	Number of burns
Control	Control	CC	-	0
Dormant	Annual	DA	5/1960, 5/1961, 5/1962, 4/1963, 5/1964, 10/1964, 5/1966, 5/1967, 5/1969, 5/1970	10
	Biennial	DB	5/1960, 5/1962, 5/1964, 5/1966, 5/1969	5
	Periodic	DP	5/1960, 5/1969	2
Summer	Annual	SA	8/1960, 6/1961, 8/1962, 6/1963, 6/1964, 7/1965, 8/1966, 7/1967, 7/1968, 8/1969, 7/1970	11
	Biennial	SB	7/1960, 8/1962, 6/1964, 8/1966, 7/1968	5
	Periodic	SP	7/1960, 7/1967	2

Table 4.2. Forest floor and mineral soil horizon variables remeasured in 2015 included: depth, mass, organic matter (OM), nitrogen (N), phosphorous (P), potassium (K), calcium (Ca), magnesium (Mg), ash, pH, cation exchange capacity (CEC), bulk density (BD), carbon (C*), and pyrogenic carbon (PyC*). An asterisk [*] indicates soil variables not measured in the original 1959-1969 study.

	Depth	Mass	OM	N	P	K, Ca, Mg	Ash	pH	CEC	BD	C*	PyC*
Forest Floor	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Mineral soil	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓



Figure 4.1. Prescribed fire use in the Cutfoot Experimental Forest, MN (USFS, 1960).

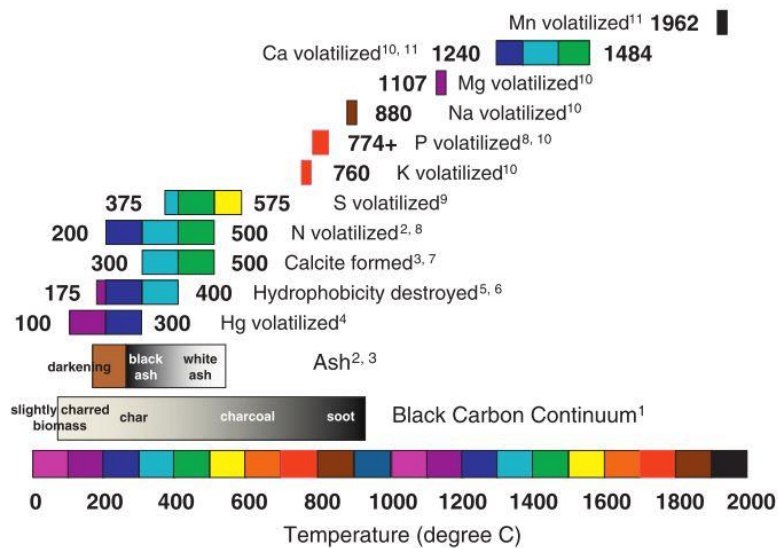


Figure 4.2. Direct effects of fire temperatures on soil chemistry (Bodi et al., 2014).



Figure 4.3. The *Red Pine Prescribed Burning Experiment* study site and experimental units are intact and remain unaltered since the last prescribed fires conducted in 1970 (James, 2015).

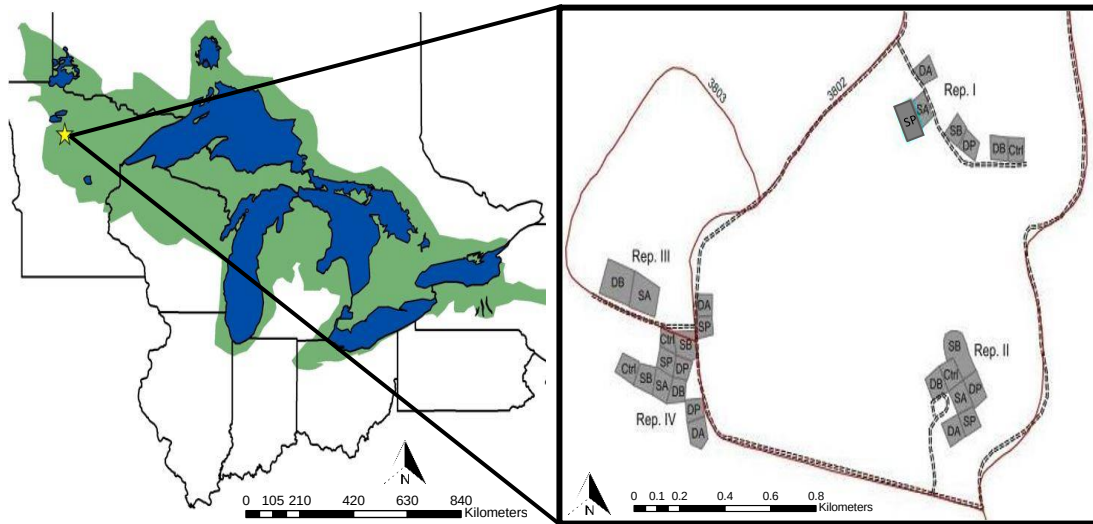


Figure 4.4. Map of the Lake States region and experimental design (www.lakestatesfireshci.net).



Figure 4.5. Study site soil profile representing the forest floor (litter, fermentation, humus) and mineral (0-10 cm, 10-51 cm, 51-91 cm) soil horizons (a) along with field collection methods in 2015 (b).

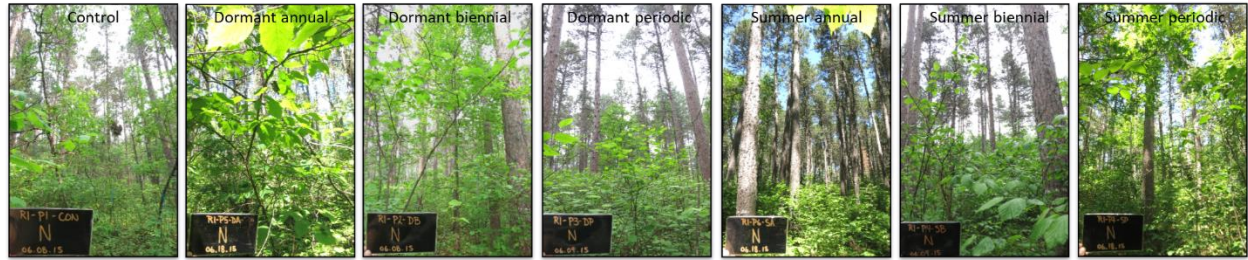


Figure 4.6. Photos taken in 2015 from plot center orientated at a 0° azimuth (north) documenting visual changes in forest structure and composition to prescribed fire treatments > 45 years since the last burn treatments.

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CHAPTER 5

CONCLUSION

Chapter 2 supports previous short-term findings of prescribed fire effects on soil properties reported in red pine and other ecosystem types and provides evidence that prescribed fire treatments had legacy effects on organic horizon and mineral soil properties >45 years since the last prescribed fire. In general, the legacy effects of summer season burns decreased, whereas dormant season burns increased nutrient stocks in organic and mineral soil horizons, and the effects of fire intensified with increased fire frequency within season. Short- and long-term responses of soil properties to prescribed fire treatments are likely influenced not only by the direct effects of fire intensity, combustion of forest floor horizons, and redistribution of nutrients during fire; but also by the indirect effects of post-fire vegetation and litterfall via interactions between the aboveground and belowground components of a post-fire ecosystem, particularly given the permeable sandy soils at this study site.

Chapter 3 indicated that persistent differences among prescribed fire treatments and time since fire (>45 years) had minimal effects on C and PyC stocks and PyC concentrations with exception for the summer annual burn. Deep mineral soils appeared to function as a reservoir of C and PyC, indicating that mineral subsoils are important when estimating total soil and forest C stocks. Although prescribed fire treatments may result in initial losses of C and PyC stocks from forest floor horizons, these effects may be moderated by the relatively larger pools in the mineral soil and the deep storage of C and PyC and increased proportion of resistant forms of PyC in deep mineral subsoils, as well as by forest floor C recovery since the last prescribed fire.

The combined results of Chapter 2 and Chapter 3 indicate the need for regional studies to accurately measure soil responses to prescribed fire that affect the development of local fire management plans. My findings suggest that infrequent summer prescribed fires, including summer periodic burns, may be a valuable approach to increase the variability in burn schedules more representative of historical regional fire regimes and facilitate development of fire-dependent species, such as red pine, by reducing organic horizon depths and overall nutrient stocks. Implementing forest management activities that emulate natural disturbance regimes, such as the historical range of wildfire season and frequency, within a given ecological or geographic region, has been recommended for obtaining the best results in restoring and maintaining forest ecosystem structure, species composition, and soil nutrient dynamics. To help achieve these ecosystem management objectives, managers could aim to include summer burns where possible, in contrast to the more common application of prescribed fires in the dormant season.

Although high frequencies of prescribed fires may be useful for initiating ecosystem restoration or fuels reduction - sustained annual and biennial frequencies of burn schedules are usually not logistically practical, regardless of season, because of weather, budgetary, personnel, and safety constraints - and are also more frequent than the historical fire regime in this region and ecosystem type. However, summer season prescribed fires used to accomplish aboveground management objectives are not likely to result in strongly undesirable impacts to the mineral soil, such as increased nutrient stocks or changes in CEC, soil texture, and bulk density, and are compatible with C sequestration objectives.

My results in Chapter 2 and Chapter 3 provide a unique comparison of contrasting prescribed fire seasons and frequencies as well as time since fire on soil properties in a red pine

forest. Future research will require assessing the interaction of the direct and indirect effects of prescribed fire and ecosystem components to better predict soil and ecosystem responses to fire. Information on fire (i.e., weather, fuels, ignition patterns, temperature), vegetation (i.e., phenology, quantity, quality), and microbial interactions affecting soil responses and cycling remain needed for these and other ecosystem types. Future field collection methods could include a more robust sampling design by increasing the number and distribution of sub-samples within plots, whereas chemical analysis to quantify PyC could include development of additional standard reference materials to better predict PyC in organic horizons that have potential to contain a high mineral content, such as the humus horizon. These detailed data and methods will be critical for improving our understanding of the relationships between fire behavior and fire effects over the short- and long-term after fire and for increasing the effectiveness of fire management activities to achieve specific management goals.