FINAL REPORT

Title: Manipulating soil heating patterns to optimize barrens restoration

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List of Abbreviations/Acronyms

ac	acres
AIC	Akaike information criterion
ANOVA	Analysis of Variance
C	Carbon
°C	degrees Celsius
Ca	Calcium
cm	centimeters
CNNF	Chequamegon-Nicolet National Forest
DBH	diameter at breast height
°F	degrees Fahrenheit
FIA	Forest Inventory and Analysis
FIREMON	Fire Effects Monitoring and Inventory Protocol
FOFEM	First-Order Fire Effects Model
ft	feet
g	grams
ha	hectares
HMV	Heat Moisture Vapor model
hr	hour(s)
in	inch
IFT-DSS	Integrated Fuel Treatment Decision Support System
K	Potassium
kg	kilogram
kJ	kilojoule
kW	kilowatt
m	meters
Mg	Magnesium OR megagram (metric ton)
N	Nitrogen
NPS	National Park Service
P	Phosphorus
PRS	Plant Root Simulator (probes)
PyC	Pyrogenic Carbon
SPFI	Soil Postfire Index
TCP	Thermocouple Probe
TDR	Time-Domain Reflectometry
USDA	United States Department of Agriculture
USDI	United States Department of Interior

Keywords

ash, burn severity, char, dormant season burns, duff, FIREMON, FOFEM, fuel consumption, fuel loading, mechanical brush cutting, mesophication, nutrient exchange rates, paint tag, pine barrens, pine woodlands, prescribed burn, pyrogenic carbon, restoration, sandy loam, seed bank, soil fertility, soil heating, soil hydraulic connectivity, thermocouple probe, top kill

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Abstract

Prescribed burning is frequently applied to restore and maintain open-canopy ecosystems such as pine woodlands and barrens. Critical uncertainties on the effectiveness of these burns exist in regard to soil heating and its effects on soil nutrient pools and other biological processes such as hardwood regeneration and seedbank viability. The First Order Fire Effects Model (FOFEM) is widely used for fire planning in such situations, but the embedded soil heating model has not been validated across a wide range of soil and fuel conditions. Understanding how soil heating influences below-ground processes such as soil nutrients, seed germination, and hardwood regeneration is needed to inform effective restoration strategies. We partnered with the Chequamegon-Nicolet National Forest in northern Wisconsin to investigate 1) fire behavior, 2) first-order soil heating, and 3) second-order effects of soil heating within a long-term barrens restoration area, the Moquah Barrens, to evaluate the FOFEM soil heating predictions and the restoration success of initial-entry and repeated prescribed burns.

We established 112 plots and 23 unburned reference plots across a range of current and historic vegetation types with contrasting fuel manipulation treatments. Late dormant season prescribed burns were conducted in 2016 and 2018. We collected prefire, postfire, and 1-year postfire measurements of fuel loads, vegetation, soil nutrients, and seed bank. To capture metrics of fire behavior and soil heating responses, we instrumented a subset of plots to evaluate soil heating model performance against soil properties. We evaluated a suite of burn severity methods, analyzed carbon and nutrient content of the forest floor and mineral soil, evaluated soil nutrient exchange rates, and quantified seedbank and hardwood regeneration responses.

We found FOFEM substantially overestimated the consumption of litter, duff, and woody fuels indicating modeled combustion rates overestimated heat transfer to the soil. Within the Moquah Barrens, even relatively thin duff layers insulated the soil from heat, although biologically relevant belowground heating occurred where duff was especially thin or absent. Fire effects on soil nutrient stocks was most evident in the forest floor horizon that is subject to first-order fire effects. Changes in soil nutrient stocks and exchange rates were ephemeral, reflecting transient nutrient inputs from the ash layer that linked with surface fuel consumption. Brush cut-and-leave and fuel addition treatments lessened brush density 1-year postfire relative to other fuel treatments supporting the hypothesis that increased soil heating with brush fuel load, rather than brush-cutting itself, drive effects on belowground root systems. We found a viable seedbank in duff and mineral soil layers consisting of targeted restoration barren species and related to vegetation condition. Pre- and postfire germinant densities were similar, suggesting fire intensity and severity did not influence seed abundance.

Our study suggests soil heating is not a major factor under late dormant season conditions. While such burns remain effective for maintenance of barrens, our results support expansion of the burn window when soil heating may be more likely. FOFEM evaluation suggested the current model may not be appropriate when applied to sandy soils. Shortfalls identified herein are leading to key improvements, including parameterization of new sandy soil options. A research-management partnership allowed us to accomplish large-scale fire research and answer complex restoration questions through knowledge co-production. This study provides information that will strengthen the success of Lake States burn programs and provides results widely applicable to other eastern US ecosystems under prescribed fire management.

Objectives

Our overarching project goal was to quantify fire behavior, first-order (soil heating), and second-order (hardwood mortality, seed bank, and soil fertility) fire effects within a sandy outwash system of Wisconsin experiencing active restoration of open pine-barren ecosystems. We addressed the following questions fundamental to restoration success of initial-entry and repeated prescribed burns:

- *What level of soil heating is necessary for achieving short-term and sustained reductions in soil nutrient pools, which in turn help maintain a barrens ecosystem?*
- *What level of soil heating creates negative impacts on barrens restoration by killing seeds of desired plant species? How abundant are seeds after long periods of closed-canopy conditions?*
- *What is the ideal balance between fire intensities high enough to eliminate potential hardwood regeneration, but low enough to sustain seed banks for native barren communities?*

To investigate the above questions, we addressed the following specific objectives:

1. Field validation of available soil heating models within sandy soils underlying fire-prone forest and open barren systems of the Lake States region.
2. Investigate second-order relationships between critical ecosystem processes relevant to pine barrens restoration and soil heating, including:
    a. Hardwood stem mortality and resprouting response
    b. Seed abundance and diversity
    c. Soil fertility (total carbon, pyrogenic carbon, nitrogen, cations, and pH)
3. Evaluate field-based severity estimates of post-burn soil impacts to determine relationships between predicted vs actual second-order effects.

Our research represents a strong research-management partnership with the Chequamegon-Nicolet National Forest (CNNF) that enabled empirical measurements across a diversity of restoration stages within the Moquah Barrens restoration area of northwestern Wisconsin. Our research directly addresses the Soil Heating Funding Opportunity Notice by contributing towards the calibration and validation of soil heating models within the widely used First-Order Fire Effects Model (FOFEM) (Reinhardt et al., 1997) that will expand its utility for fire management applications by linking fuel consumption models to soil heating, and simultaneous validation of visual burn severity impact assessments that are the backbone of ecosystem and economic wildfire impact assessments. Research activities focused on sandy soils that are both underrepresented within FOFEM and widely applicable to the most extensive fire prone regions of the glaciated northern and the coastal plain regions of the United States. In addition, our validation efforts enabled investigation of second-order fire effects on soil seed banks, soil nutrients and hardwood tree regeneration, which when combined affect the restoration success of pine barren ecosystems following fire.

The full factorial experimental design originally proposed was modified to accommodate the actual range of factor combinations present within the study area. The modified experimental design balanced adequate replication with evaluation of the effects of the proposed research factors (i.e., historic vegetation, current vegetation, and fuel treatments). Prescribed burns scheduled for the second study year (2017) were postponed due to logistical constraints, including poor weather conditions, lack of qualified personnel after prolonged fire preparedness

nationally, and balancing other management priorities within the CNNF (sharp-tailed grouse re-introduction efforts). As a result, we completed the second set of prescribed burns in 2018 and completed 1-year postfire measurements in 2019. Laboratory processing of summer 2019 soil samples began the following fall. Carbon and nitrogen (C and N, respectively) analysis is nearing completion following COVID-19 laboratory re-openings. For CN results, we have presented data from the 2016 burn plots and the unburned reference plots; we will repeat the analyses presented here once remaining laboratory results are received.

Two proposed elements were dropped from the study plan due to logistical and financial constraints. Individual hardwood survival and resprouting success via tagged individuals was dropped in favor of addressing this variable using prefire, immediate postfire (i.e., top-kill) and 1-yr postfire measurements (i.e., an indicator of resprouting capacity). Laboratory trials examining the thermal tolerance of common plant species required more resources than anticipated. As a substitute, we conducted additional sampling and measurements to provide greater insight into soil processes that may be influenced by fuel consumption, soil heating, ecosystem response. These included collecting ash layer samples after the 2016 burn (Quigley et al., 2019), soil hydraulic conductivity in 2017 (Quigley, unpublished manuscript), and soil microbial community composition and greenhouse gas flux rates in 2017 and 2018.

Background

Fire-dependent communities are declining worldwide due to fire suppression and land development. Such communities represent important reservoirs of biodiversity at risk due to loss of disturbance regimes and resulting habitats in which they have evolved (Pausas and Keeley, 2009). Yet fire also represents a clear risk to human health and safety that has been exacerbated by a surge of rural development (Dombeck et al., 2004). Extensive ecoregions characterized by sandy soils that include glacial sand depositional landforms of the Lake States (Cleland et al., 2004) and the coastal plain of the Northeast U.S. (Boerner et al., 1988) represent hotspots of fire-prone ecosystems in an otherwise fire-resistant biome. Land managers are seeking ways to balance seemingly opposing objectives of maintaining biodiversity and managing risk to human safety and infrastructure in fire-prone hotspots. One strategy is restoring open barren and savannah ecosystems that were historically common on sandy soils throughout the region. These open ecosystems converted to closed canopy either intentionally (via pine plantations) or unintentionally (via fire exclusion and succession). Two successional pathways to forested conditions are possible (Radeloff et al., 2000) – either pine-dominated systems that retain regional conservation value but are also vulnerable to intense crown fires (Sturtevant et al., 2009), or hardwood-dominated systems that are more fire-resistant and have less regional conservation value (Schulte et al., 2007) (Figure 1). Open barrens provide greater ease of wildfire control than closed-canopy pine systems, while simultaneously providing essential habitat for rare species [e.g., sharp-tailed grouse (*Tympanuchus phasianellus*, upland sandpiper (*Bartramia longicauda*), Chryxus artic butterfly (*Oeneis chryxus*)] in an otherwise forested matrix. Pine barrens are considered globally imperiled and regionally imperiled in Wisconsin by the Wisconsin Department of Natural Resources Natural Heritage Inventory.

Long-term persistence of these communities is contingent on their resilience to maintain open, low-stature structure after fire. Composition of pristine barrens is characterized by specific grasses [e.g., little bluestem (*Schizachyrium scoparium*), poverty oat grass (*Danthonia spicata*)], forbs [e.g., goldenrod (*Solidago* spp.), blazing star (*Liatris* spp.)] and low shrubs {e.g., early

low-bush blueberry (*Vaccinium angustifolium*), sweet fern (*Comptonia peregrina*)]. Frequent fire is critical for maintaining these species (Curtis, 1959). Establishment of hardwood tree species [e.g., oak (*Quercus* spp.), aspen (*Populus* spp.) and red maple (*Acer rubrum*)] reduce probability of fire, thereby, reducing probability of retrogression from transitional to savanna or barren stages.

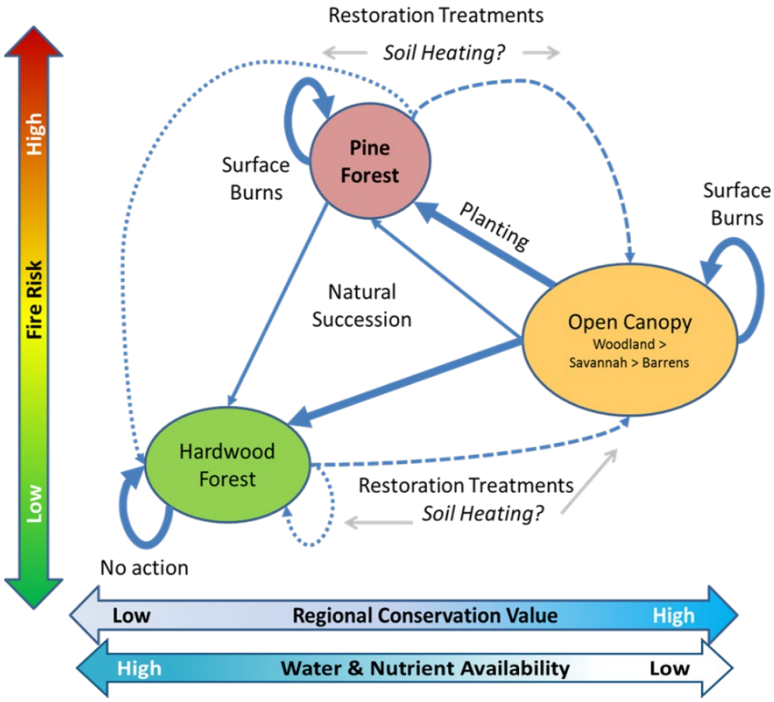


Figure 1. Human and ecosystem drivers affecting vegetation states within historically open canopy pine sandy systems, and the places where soil heating might accelerate restoration goals. Solid lines represent historic drivers with line width indicating theoretical probabilities. Dashed lines represent intentional restoration pathways, dotted lines represent unintentional restoration pathways.

Several factors contrast between the successional stages of closed forest and barrens that tend to retain each system within self-reinforced states. The stability of closed forests is enhanced by greater tree dominance, more overstory canopy cover, lower availability of light in the understory, greater soil organic matter and nutrient content, and higher soil moisture retention – all associated with a regionally consistent phenomenon known as "mesophication" (Nowacki and Abrams, 2008). In contrast, opposite conditions self-reinforce a barrens state: domination by low-stature plants, high availability of light, low soil organic matter, low soil moisture retention, abundance of barren-adapted species capable of reproducing, and (presumably) rich, dense soil seed banks of barren-adapted species. Such non-equilibrium system dynamics suggest that scientifically informed management activities can more rapidly transition intensive and expensive restoration activities to low-cost maintenance of the desired condition. Thus, effective barrens restoration activities must be informed by accurate information on how variation in – and manipulation of – fire intensity affects first-order soil heating patterns and second-order ecosystem processes because these factors affect the probability of successful transition from closed-canopy to barrens structure.

The FOFEM model and its embedded combustion and soil heating models, is poised to become a standard for fuel treatment planning [e.g., Forest Vegetation Simulator, (Reinhardt and Dickinson, 2010), (Reinhardt and Crookston, 2003)] and is expected to be an important component of the new Integrated Fuel Treatment Decision Support System (IFT-DSS). Available soil inputs do not yet include a sandy soil option, which is the dominant soil type underlying fire-

prone ecoregions across the glaciated and coastal plains regions of the Eastern US. Likewise, postfire assessments have recently become standardized to visually estimate above and below-ground impacts based on visual indicators such as vegetation charring, combustion of the forest floor, relative coverage of ash, and coloration of exposed mineral soil e.g., (Jain et al., 2012; Jain and Graham, 2007; USDI National Park Service, 2003). Such visual assessments can be performed rapidly to help scale-up direct fire impacts that are far less easily quantified (R.K. Kolka et al., 2017). Simultaneous evaluation of the soil heating model and postfire burn assessment methodology will enable scaling of soil heating and consequent belowground effects to scales necessary to investigate soil heating effects on hardwood regeneration, soil seed banks, and soil fertility.

The soil seed bank is an important reservoir of vegetation diversity (Taft, 2003). Observations from previous restoration treatments in the Moquah Barrens area suggests that seed banks for barrens vegetation may be greatly diminished in closed-canopy forests even after restoration treatments. Studies in southern systems suggest that some critical functional groups of vegetation may not persist in the seed bank for extended periods (Taft, 2005, 2003). The seed bank has never been directly studied within the Moquah Barrens, so it is not clear whether seed is unavailable or destroyed by the restoration activities (for instance, from intense soil heating, (Maia et al., 2012)). As a rule of thumb, necrosis of unprotected and physiologically-active tissue (including subsurface buds) occurs if their temperatures reach 60 °C (Choczynska and Johnson, 2009). Yet dormant seeds can have much greater tolerance to heat, and germination may be stimulated by fire (Paula and Pausas, 2008). A better understanding of seed bank dynamics and its interactions with forest cover and soil heating associated with fire in barrens would help optimize fuel treatments, burning strategies, and, if necessary, seeding treatments to facilitate more rapid and diverse colonization by native barrens vegetation.

Fire effects on soils in the Lake States suggests that high severity fires may help create low- soil fertility conditions (Miesel et al., 2012) – these conditions would favor the desired barrens vegetation state. Fires must completely combust surface fuel to have major effects on soil pH by releasing base ions (Arocena and Opio, 2003). However, transient effects can occur even under low-intensity prescribed fires, although the magnitude and direction of effect vary among plots (Boerner et al., 2009). Likewise, inorganic (i.e., plant-available) forms of N are known to increase in the short-term following fire before returning to approximately prefire levels (Certini, 2005). The conversion of organic forms of nitrogen (i.e., N contained in live or dead biomass) to inorganic forms that are susceptible to loss by leaching, especially in well-drained, sandy soils underlying barrens ecosystems may support restoration of nutrient-limited systems (Quigley et al., 2020). Thus, fire management practices may not only convert closed-canopy forest to barrens vegetation, sufficient soil heating during fire may also help contribute to self-sustaining barrens vegetation by promoting loss of excess nutrients that have accumulated under closed-canopy forest conditions.

Soil heating should also be directly related to the amount of residual pyrogenic carbon (i.e., charred organic material ranging in size from microscopic to macroscopic particles, formed during the pyrolysis stage of combustion reactions). Pyrogenic carbon (PyC) in soil has been positively correlated with soil pH, total C and N, phosphate availability, and conifer seedling regeneration (MacKenzie et al., 2008; Makoto et al., 2011). The factors that influence the formation of PyC during fires are poorly understood, but laboratory studies have shown that PyC increases with temperature at lower ranges (<500°C) (McBeath et al., 2011), and that higher

formation temperatures may increase the stability of the PyC in the environment (Al-Wabel et al., 2013). Identifying the relationships between fire energy release, soil heating, and PyC and soil nutrients is essential because the impact of soil heating on nutrient status over the short and long terms may be enhanced or mediated by its presence.

Restoration activities occurring on the Moquah Barrens within the Washburn Ranger District of the CNNF (Bushman, 2009) provided a unique opportunity for a research-management partnership to investigate the potential for soil heating to accelerate restoration activities. The forest recently aggregated small burn units into large heterogeneous burn blocks that contain a broad range of vegetation types at different stages in the restoration process. They further employ broad-scaled mechanical brush cut treatments intended as the classical "one-two punch" designed to set back hardwood regeneration by first cutting it and then burning it 1-2 seasons later, resulting in higher intensity late-dormant season burns. Such treatments provided opportunities to integrate fuel loading manipulation experiments to create high contrasts in burn intensity to study the consequences for aboveground tree and vegetation, belowground heat penetration, and belowground processes such as seed bank diversity and vitality, resprouting potential, and soil nutrient availability and fertility. The study further enabled a rigorous field-validation of available soil heating models and the application of broadly applied burn severity metrics in a study area underlain by excessively drained sandy soils that typify extensive fire-prone regions of the eastern United States.

Materials and Methods

Study Area

The Moquah Barrens (latitude 46.6235° N, longitude 91.2123° W) serves as the centerpiece (~5100 ha) of a ~8903 ha area managed under the Northwest Sands Restoration Project (Bushman, 2009), located on Northwest Sands ecological landscape of Wisconsin (Figure 2). This region has flat to gently rolling topography and a relatively short growing season (121 days), with a mean annual temperature of 5.2 °C and mean annual precipitation of 800 mm (Wisconsin DNR, 2015). Soils are well-drained to excessively drained Spodosols (primarily Entic Hapolorthods [Sultz, Vilas, and Rubicon series]) of glacial outwash origin, up to 180 m deep. The Moquah Barrens landscape is managed by the Washburn ranger district of the CNNF, and management activities include timber harvest, invasive species control (i.e. herbicide application), mechanical brush-cutting, and prescribed burning, which together are designed to restore the plant community composition and structure that typify pine barrens. "Pine barrens" of the region are dominated by grasses, low shrubs, and scattered pine trees [primarily red pine, *Pinus resinosa*, jack pine, *P. banksiana,* with some scattered white pine (*P. strobus*)].

Historic vegetation cover data was obtained from a 1951 timber survey (USDA Forest Service, 1954). The timber survey classified regions by plant community types including grassland (i.e. barrens), aspen-paper birch (*Betula papyrifera*), scrub oak, red pine, and jack pine communities. Pine-dominated sites denoted whether communities were natural or the result of planting, which typically involved furrowing of soils during plantation establishment. The presettlement fire return interval for Moquah Barrens was approximately 9–15 years and has since increased to much longer under general fire suppression (Guyette et al., 2016). Prescribed fire management was introduced to Moquah Barrens in the late 1960s, during which time burns were managed in small 50–100 ha units. In the mid-2000s these small units were aggregated to fewer burn blocks

on the order of 200-800 ha in size (Figure 2). At the time this study was initiated, previous burn frequency of specific locations ranged from zero to nine years since 1970 (Quigley et al., 2020). Prescribed burns are generally conducted during the dormant season, typically in mid to late May.

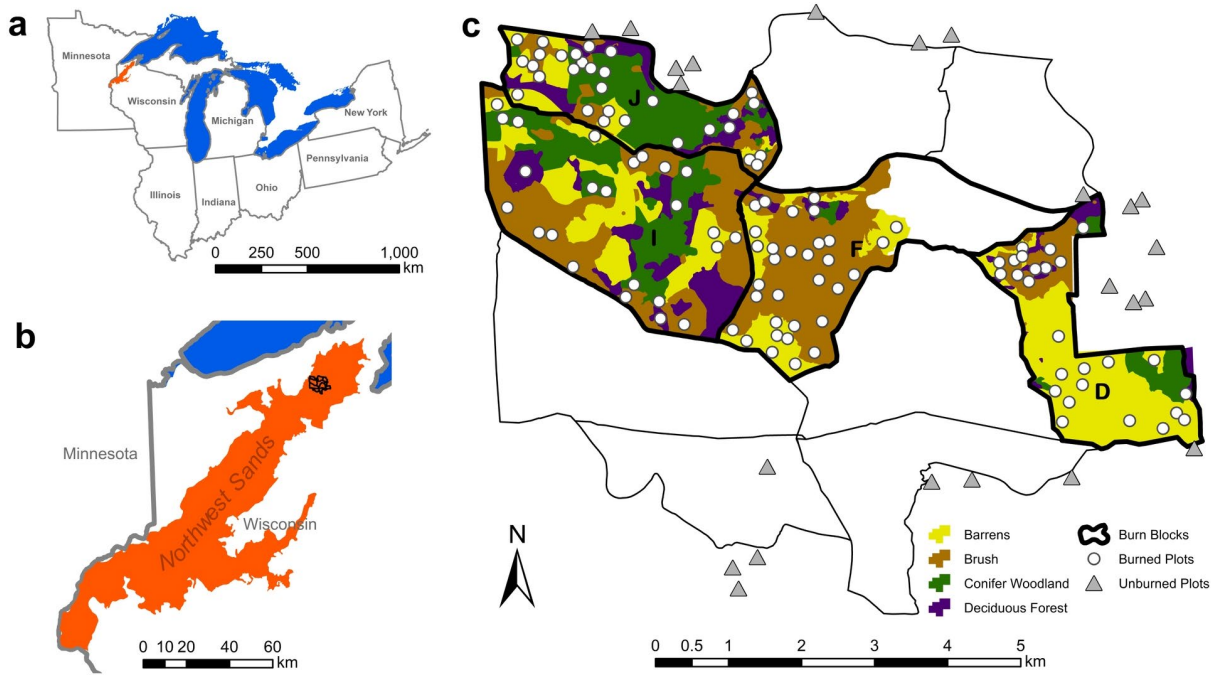


Figure 2. Map of the study area showing (a) the Lake States region of the United States, (b) the Northwest Sands region of Wisconsin, and (c) the Moquah Barrens area. Vegetation cover type is indicated by color in each of the four burn blocks used in this study (J, I, F, D, delineated by dark borders). Locations for burned plots and unburned reference plots are indicated by symbols. Adapted from Quigley et al. (2020).

Study Design

Our study design consisted of three core factors: historic cover type, current cover type, and fuel treatment. Historic cover type was defined as barren, pine, or deciduous, according to the 1951 surveys. In some cases, pine plantation establishment occurred just after the survey, in the early 1960s, based on USDA Forest Service records. Current cover type was defined as pine woodlands, deciduous forest, transitional hardwood brush, or open barrens (Figure 3). Pine woodlands consisted of established pine plantations that had been heavily thinned in the period between 2010 and 2015 to open the canopy, with a target basal area of 7-14 m^2/ha (30-60 ft^2/ac) and minimum tree density of 100 trees per ha (40 trees per ac). Thinning treatments removed crowns and branches to minimize fuel loadings post treatment. Deciduous forest consisted of closed canopy broad-leaved trees at least pole-sized (i.e., greater than 11.5 cm (4.5 in) diameter at breast height (DBH). "Brush" is defined as regenerating deciduous tree species, including tall shrubs such as hazel and willow. To distinguish barrens from brush cover types, we applied the trigger points defined by the CNNF for treatment, where 30% brush cover suggests controlled burn treatments, and 70% brush cover suggests brush cut treatments. Our brush cover types were therefore defined as having greater than 70% brush cover (occasionally as low as 50%), and

barrens was defined as having less than 30% brush cover. However, low-statured woody species that rarely exceed 0.5 m in height, such as sweet fern and lowbush blueberry, are considered typical of the barrens, and therefore were not considered as "brush."

Table 1. Study Design and sample size (plots). Plots were equally distributed across burn years, and (to the extent possible) stratified evenly across two burn blocks in each of the two years.

Current state	Fuel treatment	Historic state (circa 1951)		
		Barrens	*Pine*	*Deciduous*
Burned plots				
Deciduous Forest	None			8
Brush	None		8	8
Brush	Cut & Remove		8	8
Brush	Cut & Leave		8	8
Thinned Pine	None		7	
Thinned Pine	Addition		9	
Barrens	None	8	8	7
Barrens	Addition		8	8
Unburned reference				
Deciduous Forest	None			4
Brush	None			4
Pine Plantation	None		3	
Thinned Pine	None		4	
Natural Pine	None		4	
Barrens	None	4		

Fuel treatments were applied to a subset of the historic and current cover type combinations. Brush cut-and-leave treatment typically deposit on the order of 22 metric tons per ha (10 tons per ac) of light (1 to 100-hr) woody fuel, cut at least one growing season prior to burning to allow the cut brush to cure. One third of the brush plots were placed where brush remained standing, the remaining two thirds were placed within brush cut areas. One half of the brush cut plots were selected for fuel removals (i.e., brush fuel was removed). For the purpose of fuel treatments, plot size was 20m (66ft) radius (0.13ha; 0.33ac). For the pine woodland and restored barrens cover types, brush fuel was added to approximate the loading of a typical brush cut and leave treatment. Brush fuels were moved and placed by hand. Plots with brush removal treatments involved removing all woody fuel types greater than 1hr fuels, and brush additions focused on 10-hr and 100-hr size classes (Figure 3).Our study design was not a complete factorial but consisted of 14 individual factor combinations (Table 1, Figure 3). We selected 8 replicate plots within each factor combination, evenly stratified to the extent possible across two burn blocks, and in each of two burn years (2016 and 2018). Plots were initially selected for even coverage across burn blocks using the 1951 historic timber survey that we digitized and overlaid with current cover types and brush treatment polygons provided by the CNNF, and then confirmed for accurate classification in the field. Plots were placed a minimum of 100m from the burn block edge, a minimum of 100 m (most at least 200m) apart, with plot center at least 25 m from the edge of a contrasting cover type. For factor combinations including fuel treatments, plots were paired for similarity in composition, topography, and brush loading, and one of each pair was subsequently selected for treatment.

We further selected unburned "reference" plots to serve as unburned controls for the seed bank and soil response variables. Reference categories were selected to capture the "bounds" of the cover types present in the broader region, as illustrated in Figure 1, and to represent the major restoration stages sampled as part of our broader study design (Table 1). Four replicate plots were located within each of six vegetation types just outside the perimeter of the Moquah Barrens restoration area that had no record of burning dating back to at least the 1960s. The six types were closed canopy broad-leaved forest, closed canopy pine forest of natural origin, closed

canopy pine plantation, recently thinned pine plantation woodland, brush cover, and open barrens, where all types of plantation origin were established using furrowing (Table 1). The final plot count across all factors included 112 burn plots and 24 reference plots (Figure 2). One of the burn plots was excluded from burning due to its proximity to a sharp-tailed grouse lek area, one of the natural forest plots was burned due to a delay in the burn year for the study, and one closed canopy pine reference plot was compromised, resulting in 112 burned plots, and 23 reference plots.

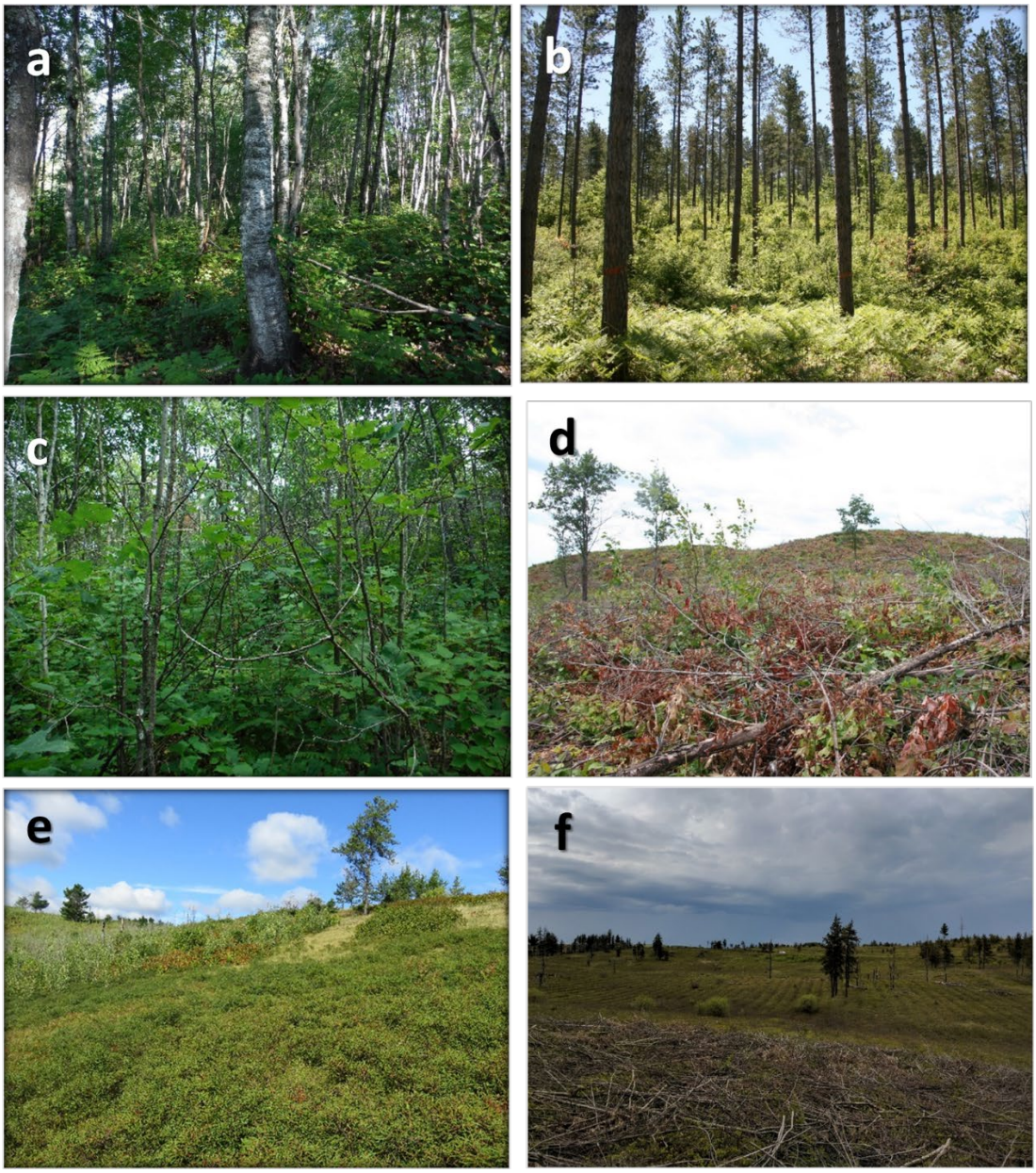


Figure 3. Representative photos showing current cover types and fuel treatments in the Moquah Barrens study area: (a) deciduous forest, (b) pine woodland, (c) brush, (d) brush with cut-and-leave fuel treatment, (e) barrens, (f) barrens with fuel addition treatments (furrows evident in background illustrate the widespread plantation management during the early 1900s).

Prescribed burns for each of the blocks used in this study, as well as the prescribed burns for any of the plot locations that had been previously burned, took place during late dormant season (mid-May) with a target phenological state corresponding to newly emerging aspen leaves between US dime to quarter (1.8-2.4 cm) size. Prescribed burns were conducted with the assistance of aerial ignitions via helicopter fitted with a plastic sphere dispenser that applied ignitions in strips with a 200-400 m spacing within the burn block interior. Resulting fire behavior was a mix of backing, flanking, and small head fire runs throughout the burn blocks based on topography, dominant wind, and ignition patterns (Figure 4). According to the Chequamegon Nicolet National Forest's incident action plan, ecological burn objectives were to:

- Restore the vegetation species and structure that typified the pine barrens ecosystem.
- Use fire as a process in the restoration of the pine barrens.
- Top kill 60-90% of existing brush/shrub cover
- Reduce 1hr and 10hr fuels by 50%
- Minimize mortality to less than 20% mature pine > 12.7 cm (5 in) DBH

Figure 4. Image of the typical mix of backing, flanking, and head fire associated with aerial ignitions applied using a plastic sphere dispenser installed in a helicopter.

Our study was organized according to six broad-level research elements: soil heating, fuel, severity assessment, soil properties, seed bank, and vegetation. Plot layout for sampling six elements is shown in Figure 5 (Lutes et al., 2006).

Soil Heating

Field Measurements. The primary soil heating measurement was conducted using a fine thermocouple probe (TCP) inserted near the interface between the mineral soil surface and duff layer (Figure 6 a). If no duff was present, the TCP was positioned slightly below the surface to ensure that the temperatures of the TCP reflected soil conditions. The TCPs were inserted at soil

severity quadrats in a subset of plots ("instrumented plots" hereafter) chosen to span the range in vegetation and fuels. Thick TCPs (Figure 6 b) where raised at 25cm aboveground to be exposed to the flames. Their responses were used to estimate residence time and, through calibration relationships, fuel consumption (kg m^{-2}) and reaction intensity (kW m^{-2}), following Bova and Dickinson (2008). TCPs were co-located with aluminum tags (Forestry Suppliers, Item #79182) with dots of heat-sensitive paint as an extensive surrogate for maximum heat for locations where TCPs were not installed (see section on **Burn severity**, and detailed methods in (Quigley et al., 2019). The maximum heat experienced by the paint tag was recorded as the respective melting point for the highest level of paint dot that was affected, referred to hereafter as "paint tag temperature."

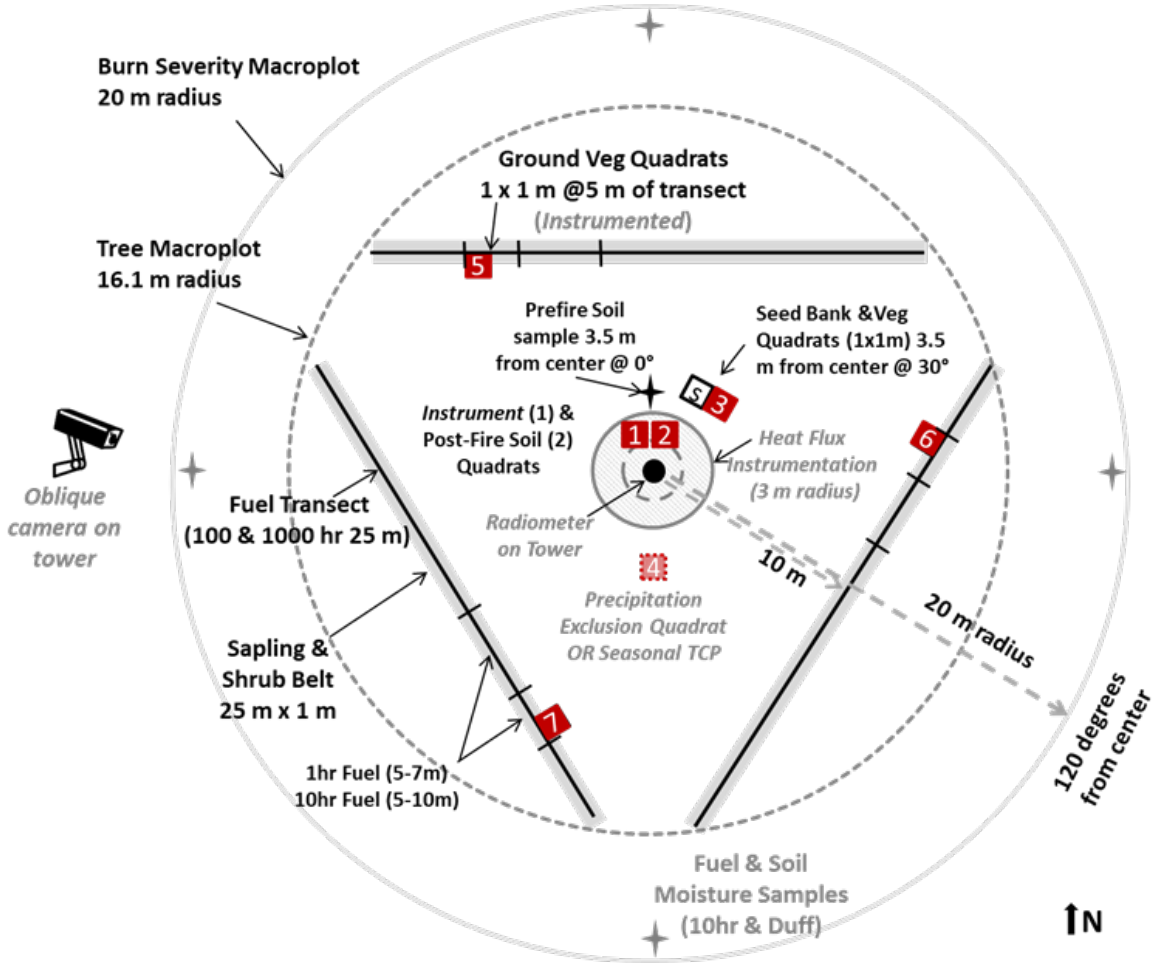


Figure 5. Diagram of plot layout. Italicized grey items relate to instrumentation installed on a subset of plots (N = 10 instrumentation plots per burn block, for a total of 40 instrumented plots used in this study). Items and text diagrammed in black outline and quadrats indicated by red squares (see detail in Figure 7) were included in all study plots.

Instrumented plots were further monitored with video cameras and tower-mounted radiometers. The overhead video cameras and radiometers used to characterize fuel consumption and fire behavior were purpose-built by the Rochester Institute of Technology (Cannon et al., 2014;

Kremens et al., 2012). The soil heat flux sensors were originally developed by Peter Robichaud of the USDA Forest Service's Rocky Mountain Research Station (Robichaud and Brown, 2019) and were installed in a subset of instrumented plots. The soil heat flux and temperature profile measurements will not be discussed further in this report.

Data Analyses. Summary of the soil temperature datasets was automated using MATLAB. Background (prefire) temperature was the 0.5-hr average temperature up to 0.5-hr before peak, a definition adequate for our relatively shallow TCP positioning. Temperature rise was peak temperature minus background. The time at which temperatures started rising was defined as the last time in which the rate of change in temperatures was <zero within a one-minute window in which temperatures were averaged. Time at peak temperature was defined as the first time the temperature reached the maximum measured value.

Figure 6. Thermocouple probe (TCP) positioning for soil temperature measurement and flame characterization. The TCPs are Type K and, as with all thermocouples, temperature is measured at the junction, which is at the tip. Panel (a) shows installation of fine TCP probes (<0.5 mm diameter), inserted ~5 cm horizontally along the interface between the duff and mineral soil (left) from a ~2.5 cm wide trench cut through the duff which was then backfilled after installation. The image shows the miniplug connection with the heat-resistant lead that ran to the logger. The photo shows the trench being elongated after fire to determine the precise positioning of the TCP tip. The thick TCP shown in (b) is exposed to the flames. Panel (b) also shows an insulated datalogger, shielded by ceramic fiber insulation and layers of fire shelter material, in a plot representative of low fuel loads. In plots with high fuel loads, the insulated datalogger package was set 1-2 m from the probes, with fuels removed from around the datalogger package to avoid damage to the electronic logger.

Often, negative biological impacts are assumed only if temperatures reach 60 °C, a convenient threshold based on observations that physiologically-active tissues don't survive exposures to that temperature (e.g., Pingree and Kobziar, 2019). The threshold is actually more of a rule of thumb where the underlying biophysical process is more complicated. In its simplest form, impacts on biological molecules (e.g., proteins) and systems (e.g., membranes) appear to occur as discrete events (lesions) that accumulate until function is sufficiently impaired to result in death of the cell or tissue (Dickinson and Johnson, 2004). The rate at which lesions are caused is dependent on temperature. Typically, rates rise exponentially above about 45 °C for physiologically active cells and tissues. This kind of phenomenon is called a temperature-dependent rate process. Temperature exposures to microbial communities and plant roots in soils during fires rise from prefire levels to a maximum and then fall slowly as the soil cools. In contrast to the heating of "thermally-thin" structures such as foliage and twigs, soil heating is relatively slow. The consequence is that cells and tissues are killed well below the typical 60 °C "threshold" because of the long duration of exposure to elevated temperatures. To demonstrate this, we applied a thermal tolerance model based on protein denaturation. The parameters of the thermal tolerance model are for a collection of proteins (Rosenberg et al., 1971).

Soil Model Parameterization. During the course of this project, the heat moisture-vapor model (HMV) was incorporated into FOFEM (Massman, 2015, 2012). As such, we consider soil heating predictions from HMV and the Campbell model (Campbell et al., 1995, 1994). A range of fuel and weather-related parameters were required to run the soil heating models in FOFEM. We estimated fuel loadings using either the quadrat (litter, herbaceous and shrub fuels) or fuel transect (woody fuels) that corresponding to the soil TCP location (Figure 5). For the centrally located quadrat (i.e., quadrat 1, Figure 5), we applied plot average woody fuel estimates Large diameter (i.e., 1000-hr) fuel was generally sparse and not observed specifically where TCPs were installed, so that fuel class was ignored, as were moss and lichen classes that were present in only sparse amounts. Duff loadings were estimated based on duff depths observed at the point of TCP installation. Percent moisture for duff, 10-hr fuel, and soil were estimated at the plot level. See the section on **Fuel** for more details on the fuel load and moisture sampling. Ambient temperature was defined based on the temperatures recorded by the aboveground thick TCP prior to heating (Figure 6). The FOFEM model currently does not include thermophysical characteristics of loamy sand soil such as is present at the Moquah Barrens. Instead, exploratory simulations presented in this paper were derived using the coarse-loam option.

Soil Model Evaluation. Temperature-time histories of fine TCP probes at or near the interface of duff (where present) and underlying mineral soil were used to evaluate model output for both the HMV and Campbell models. Modeled temperatures were available at the surface and 1 cm increments into the soil. Given the heating we observed, we only considered the top 5 cm of the soil. To match depths between measurements and the models, modeled temperature rise to maximum at a range of depths was fit with a negative exponential model allowing interpolation to the depth at which a measurement was made, if necessary. For this report, only temperature rise is compared between models and measurements. In the future, time-temperature traces will be compared more fully. While we estimated fuel consumption for most fuel elements directly (see **Fuel** for details), for this evaluation we applied FOFEM's internal fuel consumption model as a typical user would apply the model, but also compared the model outputs of percent consumption with our one field estimates.

Fuel

We applied standard Fire Effects Monitoring and Inventory Protocol (FIREMON) methods to measure woody, shrub, herbaceous, and litter fuels (Lutes et al., 2006), supplementing with additional information where needed. Woody fuels were collected using the planar transect method (Brown, 1974) using the three 25m transects shown in Figure 5. One-hour fuels and ten-hour fuels were measured from the 5-7m and 5-10m transect positions, respectively. One-hundred and one-thousand-hour fuels were measured from the full 25m transects. We quantified charring by adapting the methods of Donato et al. (2009) to define char classes (Table 2), assigned based on the visual assessment of char depth at the point of intersection. Though this method is challenging for larger-diameter fuels (Donato et al., 2009), our fuels were dominated by brushy fuel sources that enabled rapid classification of char. A separate study is investigating the proportion of char recorded in each of the size and char class combinations indicated in Table 2. Woody fuels were sampled once for reference plots. For burned plots, woody fuel was measured the year prior to the burn, but following brush-cutting for plots that received that treatment. Woody fuels were resampled following manual fuel treatments (i.e., addition and removal treatments), and all burned plots were resampled for woody fuel within 2-6 weeks following the burn.

Table 2. Char codes and their definitions assigned at the point of intersection for all sampled woody fuel observations. Codes were aggregated increasingly with decreasing size class: 1-hr (U+L; M+D+C), 10-hr (U+L; M+D; C), 100-hr (U+L, M; D; C), 1000-hr (all five char classes).

Char Code Class	Description
U	Unburned – No signs of blackening or charring at point of intersection
L	Light Charring: Signs of burning with less than 10% charcoal of cross-sectional area at point of intersection
M	Moderate Charring: Charcoal is ≥ 10% and < 50% of cross-sectional area at point of intersection
D	Deep Charring: Charcoal is ≥ 50% and < 90% of cross-sectional area at point of intersection
C	Complete Charcoal: Charcoal is ≥ 90% of cross-sectional area at point of intersection

Woody volume was corrected for slope measured for each transect, and the standard piece angle corrections were applied (Brown, 1974). Coefficients necessary to calculate fuel loadings from volume estimates were derived from Forest Inventory and Analysis (FIA) studies. Quadratic mean diameter constants for light fuel classes were derived as the average for the three dominant forest type groups (white/red/jack pine, oak/pine, and aspen/birch) representative of the study area (Woodall and Monleon, 2008) (Table 3). Wood density for light fuels were derived as the average values for tree species in the study area [jack, red, and white pine, red maple, red oak (*Q. rubra*), trembling aspen (*P. tremuloides*), paper birch, and pin cherry (*Prunus pensylvanica*)] (Harmon et al., 2008) (Table 3). Wood density for fuel sizes greater than 7.6 cm diameter were assigned based on tree species and decay class values (Harmon et al. 2008). We assumed char reduced fuel mass by 70% (Donato et al., 2009) and adjusted the mass of each sampled piece using the mid-point proportion of each char class shown in Table 2.

While litter and duff measurements depths were measured at 15 m and 25 m along each of the transect lines according to Lutes et al. (2006), our principle source for these values were derived from litter/duff consumption pins. Three steel tag stakes (Forestry Suppliers, Item #79202) were inserted into the soil such that the bottom of the "eye" of stake was flush with the surface of the litter in three standard locations within each of the 1 m^2 quadrats shown in Figure 5 (Figure 7). Litter depth was measured to the nearest mm at each stake location in the few days before the burn. Depth to surface and depth litter-duff interface were measured within 36 hours of the burn for each pin location, and the depth of the remaining duff for each pin location was measured within 3 weeks of the burn. In 2018, we additionally measured the depth of accumulated light ash. Litter bulk density (g/cm^3) was estimated from the volume and oven-dried mass of litter samples extracted from the same 30cm diameter rings used to collect the forest floor. Duff bulk density was estimated by subtraction from the bulk density derived from the forest floor samples. See methods on **Soil Processes** for additional details.

Table 3. Parameters used to estimate volume and oven-dried mass of woody fuel from planar transects for light woody fuel. QMD = quadratic mean diameter.

Size Class	Diameter (cm)	Angle Correction[a] (dimensionless)	QMD[b] (cm^2)	Density[c] (g/cm^3)
1-hour	0.0 – 0.6	1.13	0.12	0.59
10-hour	0.6 – 2.5	1.13	1.75	0.54
100-hour	2.5 – 7.6	1.13	20.45	0.47

[a]Brown 1974; [b]Woodall and Monleon 2008; [c]Harmon et al. 2008

Herbaceous, graminoid, and shrub fuel loadings were measured at the 1 m^2 quadrat level within 3 weeks prior to scheduled burns (i.e., late April, early May). Herbaceous fuel includes nonwoody herbs and ferns, while shrubs focused on the low shrubs such as sweet fern, blueberry, sand cherry (*Prunus pumila*), and included woody herbs such as wintergreen (*Gaultheria procumbens*). Taller shrubs such as hazel (*Corylus* spp.) and tree seedlings were not included in the coverage. While moss was also estimated, its coverage was typically negligible. Percent coverage of the combined live and dead fuels still anchored in the ground were visually estimated within the percentage classes recommended by Lutes et al. (2006), as was the percentage of that coverage was live (including dormant woody stems) or dead. Three maximum heights of vegetation (nearest mm), sampled where possible from three of four quadrants of the quadrat, were measured within the three fuel types. Maximum height was defined by the top surface of the fuel type *in situ*, rather than measuring the upright height of vegetation that had otherwise settled in place, with the goal of estimating a volume of material within the square meter quadrat.

We established additional quadrats outside of our research plots to develop regression equations relating the volume of each fuel class (i.e., the average height times the percent coverage of the square meter) to biomass (oven-dried weights of the clipped samples). Two plots representative of each of four vegetation types (deciduous forest, thinned pine forest, barrens, and brushland) were selected for a total of eight sampling plots. Within each sampling plot, two quadrats were selected for high concentrations of each of the three fuel types, resulting in six samples per plot, and 48 samples total. A digital photo record of each sample quadrat was archived, and the same measures of fuel type coverage and height outlined above were taken. Plant material for each of the three fuel types were then clipped to the ground surface and bagged separately for later oven-drying. Resulting regression equations were used to estimate prefire fuel loadings at the quadrat and plot scale for study plots.

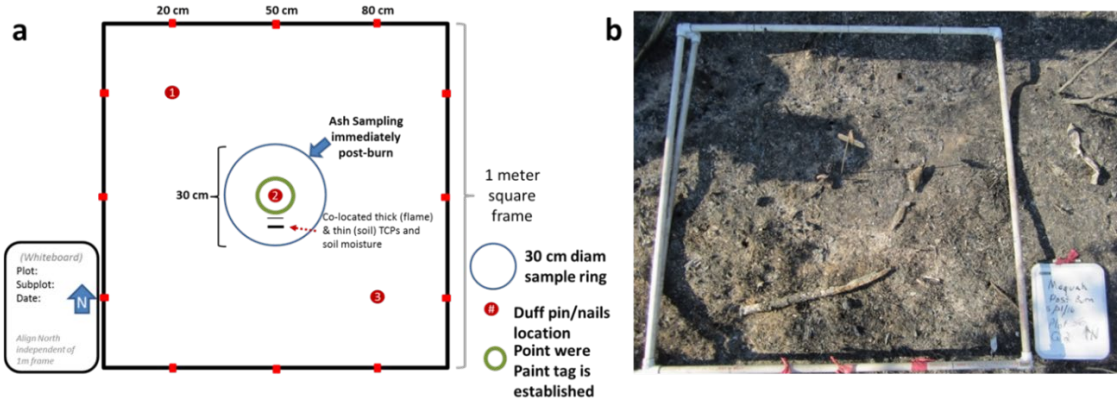


Figure 7. (a) Quadrat diagram showing sampling and measurement locations, with (b) example 24-48 hr postfire image showing paint tags and litter/duff pins.

Fuel loadings used as inputs for FOFEM were calculated at the quadrat and transect scale for each respective TCP location, where woody fuel loadings for the central quadrat was estimated as the plot-level average. Litter and woody fuel consumption were estimated directly via subtraction of postfire mass from prefire mass for each component at the plot scale. Consumption estimates for the herbaceous and shrub fuels were based on simplified FOFEM assumptions (i.e., 100% consumption of grass/herb and 60% of shrub fuels).

Fuel moisture was estimated following the methodological recommendations of Zahn and Henson (2011). Fuel moisture for 10-hr fuel and duff and soil moisture was sampled for all instrumented plots one-two days prior to each burn. Duff was sampled using 6.5 cm-diameter corer, and 10-hr fuels were clipped to approx. 20 cm lengths. We measured volumetric moisture content of the mineral soil using time-domain reflectometry (TDR; Field Scout 300 TDR Meter, Spectrum Technologies, Aurora, IL) at cardinal directions on the 20 m periphery of the study plots (Figure 5).

Four 20 m radius fuel plots were established just outside the burn blocks to be representative of vegetation types within the burn zone (i.e., red pine, jack pine, oak, and barren). During each day of active burning, we acquired hourly samples of fine fuels (herbaceous, graminoid, shrub, and litter) starting at the time of ignition and concluding by the end of burn (generally between hours 0900 and 1900). We also collected 10-hr and duff samples using the same methods used for instrumented plots twice daily (i.e. ~1400 hr and the end of burn).

Data Analyses. Alternative models for the herbaceous, graminoid, and shrub biomass estimates, and relationships between burn severity and fire processes (i.e., paint tag temperature and fuel consumption) were each explored using a combination of general linear models and general mixed linear models, where alternative models were evaluated using AIC, and standard diagnostics of modeling assumptions (i.e., normality, homoscedasticity, outliers, etc.) were evaluated for the best models, all applied within R (packages: car, lme4, MASS, gvlma).

Burn Severity Assessment

Burn severity was assessed in stages and at multiple scales. One to two days prior to each burn, four plot-level digital photos were taken looking out from plot center in each of the cardinal directions. A downward digital photo was then taken from standard orientation over each of the 1

m^2 quadrats (Figure 7). Litter/duff pins were checked for position, and paint tags were installed, hanging from tree stakes at 5 cm and 25 cm above the litter surface (Figure 7), as described in Quigley et al. (2019). Within 36 hours of each respective burn and before rainfall, the digital photo series was repeated, and paint tags were collected. We visually assessed substrate burn severity for each 1 m^2 quadrat using methods of the National Park Service (NPS) (USDI National Park Service, 2003), and the soil postfire index (SPFI; Jain et al., 2012). For internal quadrats (1-3, and 4 where present, Figure 5), we assessed the same metrics for the 30cm diameter area centered on the center duff pin, and for the soil sampling quadrat (2) we applied the metrics to the 30 cm diameter area centered on each of the duff pins. Ash collections described under the soil section were also sampled at this time.

Starting approximately 3 weeks post-burn, we assessed the more persistent indicators of burn severity. Where trees greater than 11.4 cm diameter at breast height (DBH) were present, we estimated the percentage of the plot-scale tree crown that was green, brown (scorched) or black (charred or consumed), using the percentage classes from FIREMON (Lutes et al., 2006). Minimum and maximum char heights were measured to the nearest 0.1 m below 2 m, and to the nearest 0.25 m above 2 m, for a representative tree in the quadrant for each cardinal direction. The aspect for the direction facing out from the side of the maximum char height was also recorded.

Data Analyses. We applied general linear regression to the plot average NPS severity estimate against plot-averaged paint tag temperature and surface fuel consumption estimates. Standard diagnostics of modeling assumptions (i.e., normality, homoscedasticity, outliers, etc.) applied within R (packages: car, MASS, gvlma).

Vegetation

We characterized the successional status (Figure 1) and live fuel structure related to anticipated soil heating effects. Vegetation measurements were adapted from the national FIREMON protocol (Lutes et al. 2006). We measured DBH, species composition of live and dead overstory trees (≥11.4 cm DBH) in a 0.8 ha macroplot; we tallied tree saplings (≥1.4 m tall & <11.4 cm DBH), tree seedings (≥0.5 m and <1.4 m tall), and shrubs (≥0.5 m tall) by species and live/dead status in transects (1 x 22.5 m) (Figure 8); we measured percent cover of ground-layer vascular plant (including trees and shrubs <0.5 m tall) by species in 1 x 1 m quadrats. Prefire measurements occurred after brush-cutting. Immediately after fire, the macroplot and belts were re-measured for alive/dead tallies. One-year after fire, all vegetation was remeasured.

Figure 8. Tree sapling and shrubs were tallied along 1 m × 25 m belt transects. This photo represents the barrens cover type. Photo credit: USDA Forest Service.

Seed Bank

Field Sampling. We characterized the soil seed bank in terms of density and composition of "germinable" seeds (i.e., those seeds that germinate under ideal conditions) across both reference and burned plots (pre and postfire) (Figure 5). We studied both the persistent seed bank buried in the top 5 cm of the mineral soil and the transient seed bank in the duff layer, *sensu* Thompson and Grime (1979). In quadrat S1 (Figure 5), five soil samples were collected in a systematic, equally spaced pattern prefire (May 2017) and, in an adjacent quadrat, postfire (May 2018) in the treatment areas. Reference (untreated) areas were sampled when prefire samples were collected (May 2017). Soil cores were extracted with a 6.5 cm diameter corer. The sample included the duff layer (if it was present) and 5 cm depth of the mineral soil. The soil core was cut at the intersection of the duff and mineral layers and the layers were labelled and put in plastic bags separately. All samples were placed in cold storage until the greenhouse study was initiated a week or two later.

Greenhouse Methods. The germination observations were initiated within 1-2 weeks after the soil samples were extracted from the field. All soil samples were sieved (4 mm mesh) to remove large organic matter (e.g., roots, twigs) and rocks. Then, soil samples were spread evenly over an autoclaved potting mix (1:1:1 Sand:Peat:Vermiculite) in trays in a greenhouse. The greenhouse was set at broadly optimal growing conditions [23 °C (74 °F) day, 20 °C (68 °F) night, 15 hr photoperiod 6:00 AM until 9:00 PM, regular irrigation]. The trays were monitored for germinants for 3 months (similar to the study area growing season length). Germinants were tallied and marked until genus or species identification was determined.

Data analysis. We used Kruskal Wallis rank sum tests to evaluate differences in germinants among burn status and fuel loading within pre- and post-burn plots followed by Wilcoxon signed-rank tests with Bonferroni adjustment for multiple comparisons among treatments where models were significant at $p > 0.05$.

Soil Processes

Soil sampling and laboratory analysis. We assessed changes in forest floor and upper mineral soil total C and total N pools, and pH by collecting forest floor and mineral soil samples at the prefire, postfire, and 1-yr postfire timepoints. We measured forest floor horizon depth and collected volumetric forest floor samples within a 30-cm diameter circular sampling frame before collecting mineral soil samples to 10 cm with a hammer driven bulk density corer (5 cm diameter). The prefire soil sample was collected 3.5 m from plot center along the 0° transect, and the postfire and 1-yr postfire samples were collected from preplanned areas within the soil sampling quadrat, adjacent to the fire behavior instrumentation plot to provide maximum characterization of the soil heating environment while avoiding any influence on fire effects (Q2, Figure 5, Figure 7). During the postfire soil sampling, we also installed plant root simulator probes (PRS, Western Ag, Canada) which remained in situ for approximately ten weeks. Mineral soils were transported on ice to the lab where they were measured for A horizon thickness, split into 0-5 and 5-10 cm increments, weighed, and air-dried to constant mass. We then determined dry mass of the total sample and sieved to 2 mm to record mass of the fine and coarse fractions. We determined pH, P, and cation (Ca, Mg, K) concentrations on the fine fraction mineral soil (Quigley et al., 2020). We then ground the forest floor samples on a Wiley mill to pass a 1 mm screen, and pulverized subsamples of each mineral soil layer to determine total C and N concentrations via combustion (CHN Elemental Analyzer, Costech Analytical Technologies,

Valencia, CA), and pyrogenic C using nitric acid-peroxide digestion (Maestrini et al., 2017; Maestrini and Miesel, 2017; Pingree et al., 2012). We determined bulk density based on oven-dry mass (60 °C for forest floor, 120 °C for mineral soils) for each plot and used these values with nutrient concentrations to calculate nutrient stocks for each plot and measurement event. We removed the PRS probes at the end of the season-long incubation period, processed them following manufacturer instructions, and shipped them on ice to the manufacturer for analysis. All sample mass not used for analysis in this study has been archived for potential future analyses and collaborations.

Data Analysis. We used Analysis of Variance (ANOVA) to evaluate differences in soil nutrient stocks and exchange rates, within soil horizon, followed by Tukey-adjusted pairwise comparisons (e.g. for comparisons among treatments, timepoints, burn severity level, and paint tag temperature) where models were significant at $p>0.05$.

Results & Discussion

Prescribed Burns

Block F was burned on May 19, 2016, and block I was burned the following day. Wet spring weather precluded burning in 2017, so remaining burn plots in blocks D and J were burned in 2018. Block J was burned on May 16, 2018, though a late afternoon shower resulted in a patchy burn on the western half of the block that required a reburn the following day. Block D was burned on May 20, 2018.

Growing season precipitation varied across the period of the study (Table 4) in a way that likely affected fuel and soil moisture during burn periods as well as nutrient fluxes across years. Fuel moisture estimates (Figure 9) confirm that duff was wetter in 2018 relative to 2016, with substantially greater variability, particularly during burn dates. Ten-hour and fine fuel moisture was higher for Block J in 2018, but comparable among the burn dates for the remaining burn blocks (Figure 9).

Table 4. Growing season precipitation, shown by year for each season of nutrient exchange rate measurements.

	Precipitation (cm)			
Month	**2016**	**2017**	**2018**	**2019**
May	9.30	15.06	10.72	11.40
June	7.82	8.05	22.05	6.48
July	11.73	9.40	6.83	4.88
August	5.26	14.12	17.58	7.92
Season total	*34.11*	*46.63*	*57.18*	*30.63*

Fuel Loading and Consumption

Loadings for woody fuels, duff, litter, herbaceous (i.e., herbs, ferns, and grasses), and shrubs are required inputs for the FOFEM soil heating models. We found significant differences between the bulk densities of pine litter (0.025 g cm^{-3}, standard error = 0.0010 g cm^{-3}) versus all other litter types (0.010 g cm^{-3}, standard error = 0.0001 g cm^{-3}) for the study area. These values are lighter than for relevant forest types reported by Woodall et al. (2012), but within the range reported for pine forests in the western US (Sikkink et al., 2009). Our duff bulk density, estimated by subtraction from forest floor bulk densities calculated from the soil samples, was approximately 0.17 g cm^{-3} regardless of vegetation type, similar to the value of 0.15 g cm^{-3} of Woodall et al. (2012). Relationships between dry fuel biomass and fuel volume (each transformed by log base 10) were dependent on fuel type (i.e. grass, herb, shrub), and there was a consistent strong positive relationship between fuel volume and biomass across all fuel types

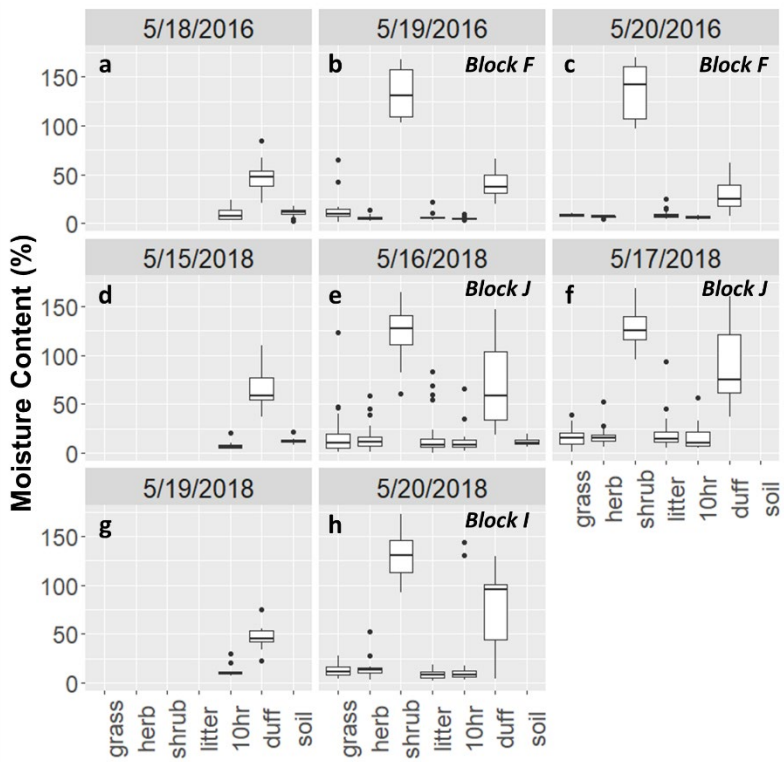


Figure 9. Fuel moisture content immediately prior to and during burn periods, shown by fuel component. Block name is shown to represent the date that block was burned.

($F_{1,133} = 565.87$, $p < 2.0 \times 10^{-16}$, $r^2 = 0.81$; Table 5). Grass fuel biomass was generally lower at a given volume than the other two fuel types. Duff results are summarized in the **Soil Heating** section.

Table 5. Linear regression model parameters (estimate ± standard error) for dry weight of fuel biomass (log_{10} of biomass in g) as a function of the fuel volume (log_{10} m^3) within a 1 m × 1 m sampling quadrat, and fuel type.

	Estimate ± s.e.	t-value	Pr(>|t|)
Intercept	2.584 ± 0.079	32.6	$< 2.00^{-16}$
$Log_{10}(m^3)$	0.741 ± 0.033	22.7	$< 2.00^{-16}$
Herb	0.287 ± 0.065	4.4	2.11^{-5}
Shrub	0.245 ± 0.066	3.7	2.88^{-4}

The distribution of surface fuel types across the study design demonstrate the potential impact of brush-cutting and other fuel treatments on fire behavior and soil heating (Figure 10a).

Fuel additions by either cut-and-leave or manual additions more than doubled total fuel load, primarily in the 10- and 100-hour size categories. Notably, pine litter enhanced the fuel loadings with the "thinned pine" cover type, while remnant barrens carried the lowest fuel loadings overall. The addition of light woody fuels had direct influence on fire behavior (Figure 11) as well as heavier ash deposition (Figure 12) (see ***Soil Processes***).

Fuel consumption patterns were reflective of the prefire fuel loads (Figure 10b). However, our percent consumption rates are substantially less than that assumed by the FOFEM system (Table 6). Some of this discrepancy may be attributed to our methods of including charred wood in the fuel load estimates. Charred wood substantially reduces its mass, but it still represents an important legacy in terms of a persistent carbon pool (Donato et al. 2009), and potential source

of pyrogenic carbon (PyC) to the soil, where it has potential to influence nutrient availability and soil processes (MacKenzie et al., 2008; Maestrini et al., 2017; Makoto et al., 2011). Some postfire deposition of fuel would also decrease the consumption estimates. Nonetheless, our results do show that the CNNF is meeting its targets in terms of fuel reduction.

Table 6. Mean fuel consumption across the entire study.

Fuel Class	Plots (N)	Consumption (%)	95% C.I.
Litter	110	62.2	57.2 - 67.1
1-Hour	110	60.5	53.9 - 67.2
10-Hour	108	59.6	53.2 - 65.9
100-Hour	95	49.9	42.9 - 56.9
1000-Hour	56	55.9	44.8 - 66.9
Total	*110*	*63.7*	*60.2 - 67.3*

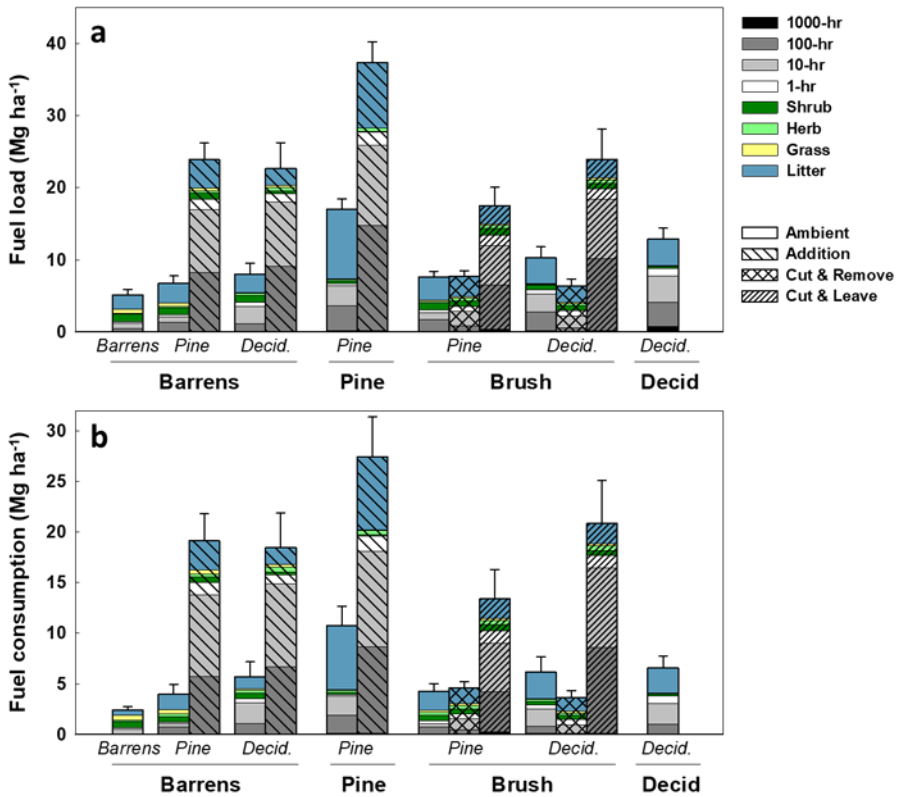


Figure 10. Fuel load (a) and consumption (b) by component across the full study design. Current vegetation shown in bold font, historic vegetation shown in italicized font, and fuel treatment shown by cross-hatching.

Burn Severity

We co-applied two substrate burn severity indices at the square meter quadrat scale immediately (i.e., within 36 hours) after the 2016 burns. We compared their frequency distributions as an indicator of the degree to which the indices could act as a surrogate for the effects of burn intensity on belowground processes within the context of barrens and woodlands restoration via controlled burns. We found that the NPS severity index enabled a broader spread of index values that approximated a normal distribution (Table 7). The SPFI of Jain et al. (2012) was less useful for our purposes, specifically because the thresholds in mineral soil exposure necessary to change levels was too coarse for useful application to a dormant season prescribed burn. SPFI appears far more appropriate for application to more severe wildfires (e.g., Kolka et al., 2017).

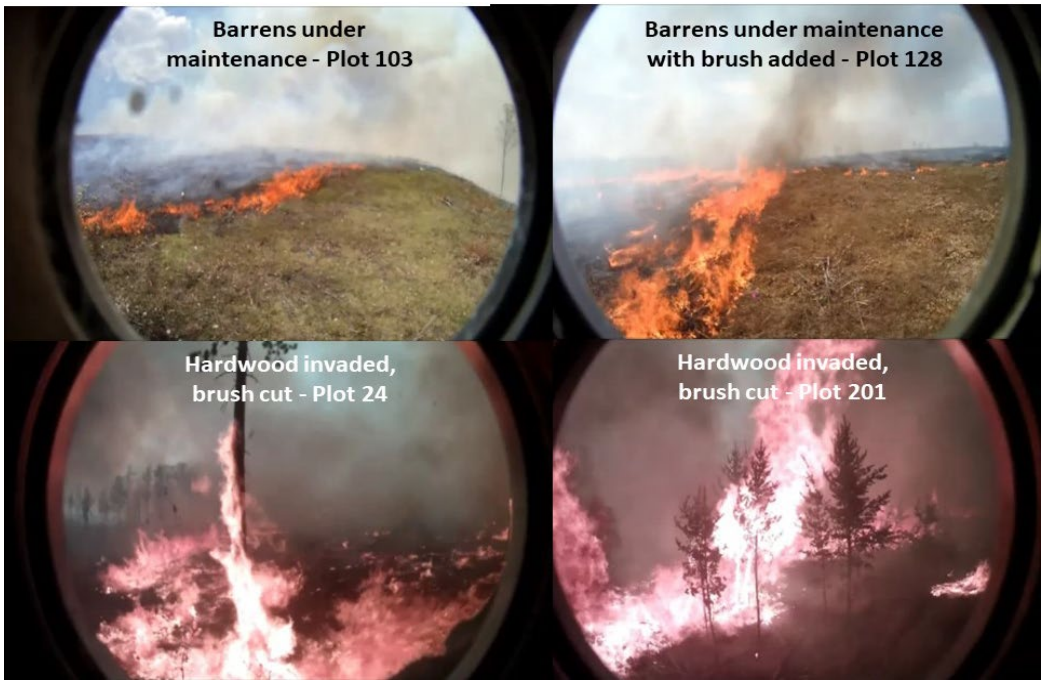


Figure 11. Representative fire behaviors observed across different cover types and fuel treatments, as observed by heat-protected video cameras.

Figure 12. Photo showing the postfire status of an instrumented plot representative of the barrens fuel addition treatment. Photo credit: USDA Forest Service.

Table 7. Matrix comparing frequency distributions of the square meter quadrat-scale estimates of substrate burn severity based on the methods of the National Park Service (NPS 2003) versus the soil post fire index (SPFI; Jain et al. 2012).

	National Park Service					
SPFI	**5**	**4**	**3**	**2**	**1**	**Total**
0	3					***3***
1	4	86	140	48	2	***280***
2.1		4	2	2		***8***
2.2			5	20		***25***
2.3			1	8		***9***
3.2				7		***7***
3.3				1	1	***2***
3.4				1		***1***
4.2				1	1	***2***
Total	***7***	***90***	***148***	***88***	***4***	***337***

Our next question was whether the NPS severity index could serve as a useful surrogate for important processes related to soil heating and belowground impacts, relative to other more intensive measures. Paint tag temperatures suggested heat ranges from nearly ambient to occasionally exceeding the melting point of the aluminum used for the paint tags (661 °C). Regression analyses indicated a second-order polynomial best approximated the curvilinear response between NPS burn severity and the log base 10 of paint tag temperature ($r^2 = 0.68$) (Figure 13a). A linear model best fit the relationship between NPS burn severity and log base 10 of total fuel consumed ($r^2 = 0.40$) (Figure 13b). The year of burn was not significant factor in either model, suggesting a consistent relationship across burn years.

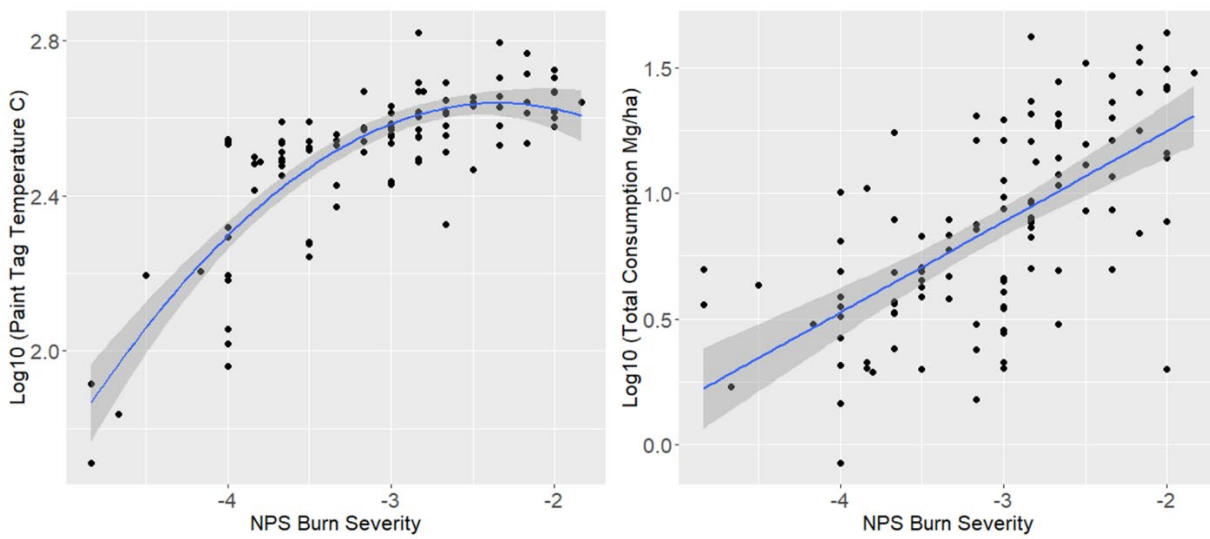


Figure 13. Modeled relationships between the National Park Service burn severity index and (a) paint tag temperature for paint tags installed at 5cm above litter surface, and (b) total fuel consumption. Both dependent variables were transformed using log base 10, and the index was multiplied by -1 to order burn severity from unburned (-5) to heavily burned (-1).

Soil Heating

Soil heating during dormant season burning in the Moquah Barrens was primarily determined by the energy of flaming combustion and depth within the soil (Figure 14), where depth in this case is measured from the top of the duff. Fuel consumption (kg m^{-2}) is a good indicator of total fire energy (kJ m^{-2}). Based on the relationships in Figure 14, where fuel loads were high, and duff was thin or absent, mineral soil heating was greatest.

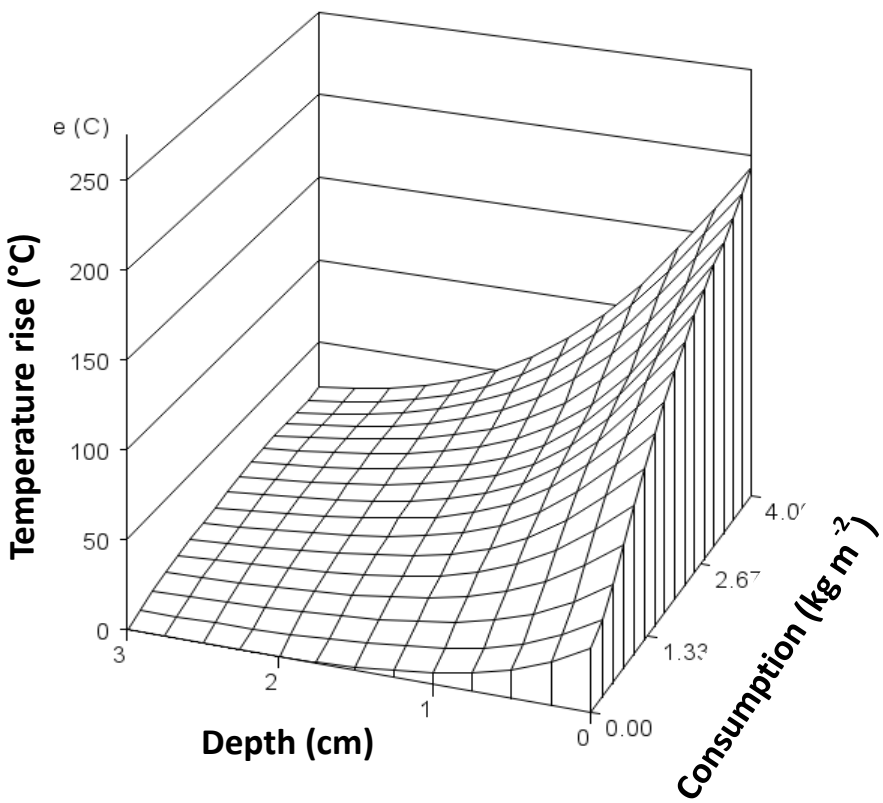


Figure 14. The general relationship between temperature rise in the soil, depth below the duff surface, and fuel consumption. Fuel consumption is predicted from the time-integrated response of thick TCPs exposed to flames. A spline routine was used to create the 3D surface

Table 8. The ends of a continuum of duff effects on soil heating, adapted from Hartford and Frandsen (1992).

Fuels	Behavior	Soil heating
Slash over moist duff	➢ Large flames/high intensity ➢ Duff consumption dependent on external heat subsidy	Minimal
Litter over dry duff	➢ Small flames/low intensity ➢ Independent duff smoldering	Extreme

There were few instances where the duff appeared to be consumed independently (by smoldering combustion) and, as such, duff acted as an insulating layer. Hartford and Frandsen (1992) described the general situation where overlying duff that is not consumed insulates underlying mineral soil regardless of high levels of energy release during flaming combustion (Table 8). Our results suggest that duff (where present) in our study sites insulated the soil during dormant

season burns, despite high levels of fuel consumption on some plots. Duff consumption averaged about 0.5 cm over all measured sub-plots (Figure 15).

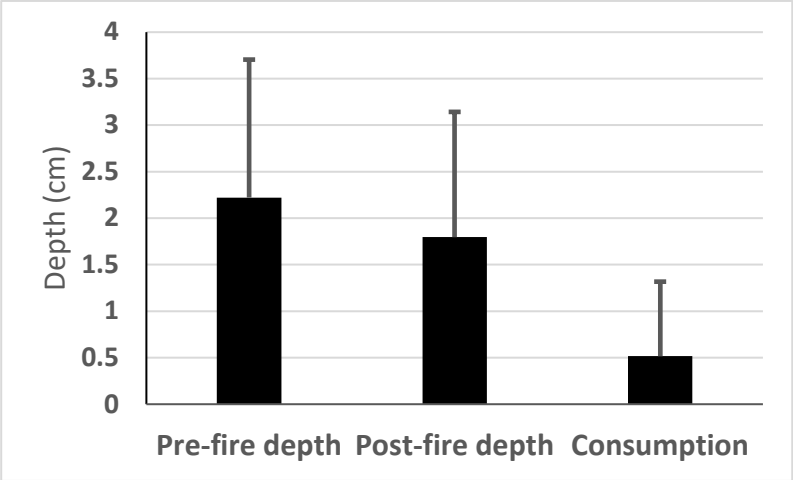


Figure 15. Prefire and postfire duff depth and consumption (the difference between pre and postfire duff depth) during Moquah Barrens dormant season fires.

We derived a relationship between fuel consumption estimated by the thick TCP and the maximum temperatures reached by paint tags (Figure 16). The relationship is useful because paint tags are easier and less expensive to use than TCP and other devices and, thus, can provide an extensively replicated measurement that correlates with fire energy. We used consumption estimated from paint tag temperatures in Figures 17 and 19, below.

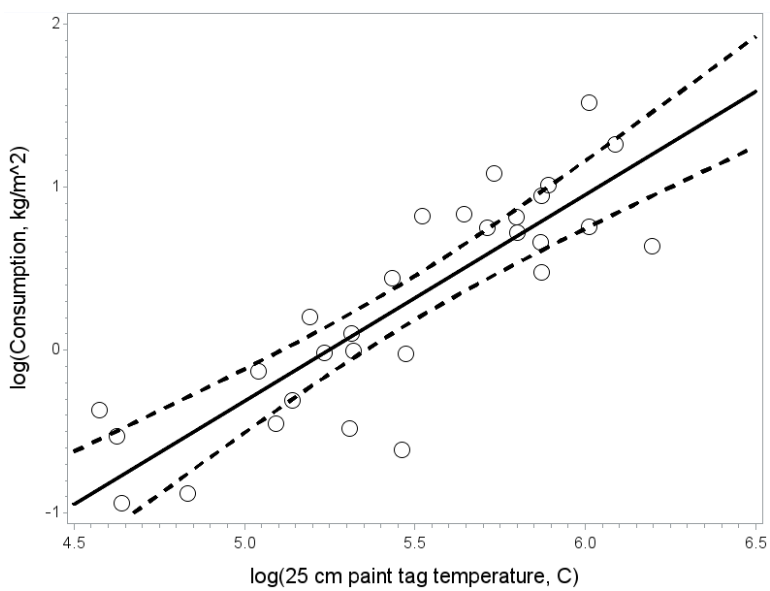
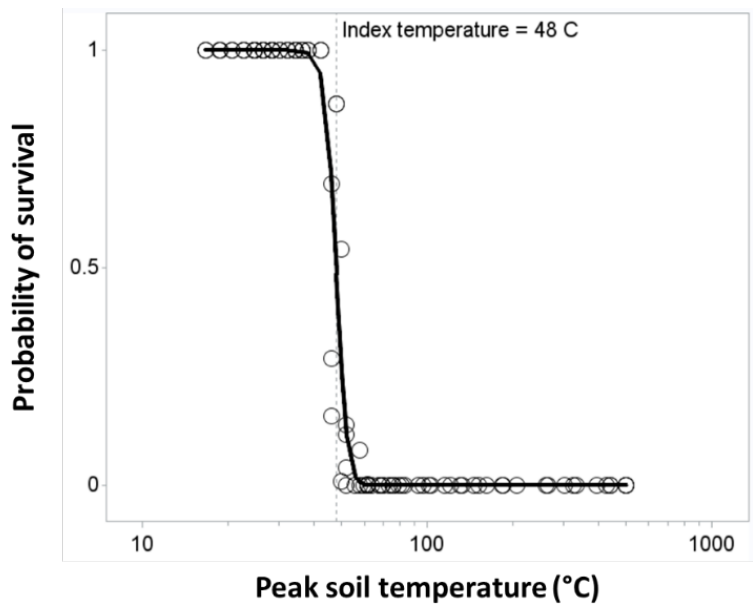


Figure 16. Fuel consumption from paint tag temperature. Paint tag temperature is from the tags set at 25 cm above ground. Fuel consumption is estimated from thick TCPs exposed to flames, also at 25 cm height. Confidence limits (95%) are displayed.

A key objective of this project was to link soil heating regimes with biological impacts and soil nutrient stocks. Based on a thermal tolerance model parameterized with Rosenberg et al.'s (1971) protein denaturation data, we found that, as a rule of thumb, if temperatures in soils reach about 48 °C, the index temperature, the accumulation of denaturation events over the entire temperature regime (including both the periods in which temperatures are rising and falling) is

expected to be sufficient to kill cells and tissues down to the depth at which that temperature was reached (Figure 17). In caveat, there are no studies that we could find that estimate parameters of a thermal tolerance model for physiologically active soil micro-organisms, yet these kinds of data are required to disentangle the joint effects of time and temperature during exposures to elevated temperatures. Dormant structures such as spores and seeds are known to be highly tolerant to elevated temperatures and some relevant data exist in the literature (see Martin et al., 1969). We expected variability based on taxa (see Pingree and Kobziar, 2019) but have no means of examining this problem further without relevant thermal tolerance studies. There are two main implications of our results. First, using 60 °C as a threshold would appear to underestimate the depth of impacts of soil heating on biological systems because it ignores the effects of long exposures to temperature lower than 60 °C during soil heating. Second, impacts on soil nutrient dynamics and other biologically mediated processes carried out by physiologically active micro-organisms would be expected to be severely impacted, at least initially, down to the depth at which 48 °C was reached. The same goes for roots and root collars at the base of stems. Depths to which 48 °C was reached in soils is shown in Figure 18. To date, knowledge of the time course over which soil organisms and soil function recovers after fire remains limited (Dove et al., 2020; Whitman et al., 2019) and is an important topic for further study.

Figure 17. Survival in a putative population of physiologically active cells as a function of maximum temperature reached in the soil resulting from Moquah Barrens fires during the dormant season. The datapoints are modeled survival where soil temperature regimes were measured in quadrats. The inflection point of the curve is 48 °C.



We hypothesize that impacts of soil heating on mobilization and volatilization of soil minerals will be closely tied to thermal degradation of organic material in soils because mineral elements are often bound within organic molecules. In approximation, thermal degradation of organic materials resulting in volatilization and charring begins around 180 °C and its rate peaks around 320 °C, e.g., (Biswas et al., 2007; White and Dietenberger, 2001). As such, we would expect that impacts on mineral stocks and cycling would be correlated with the depth to which those bracketing temperatures were reached. Greater variability was observed in the depth at which 48 °C thresholds was reached (relative to 180 and 320 °C) because it was more affected by differences in prefire soil temperature which, at the surface, sometimes approached 50 °C

(Figure 18). Consumption was predicted by calibration from paint tag temperatures (Figure 16). Our predicted depth results indicate that thermal degradation of soil organic material did not occur often in the mineral soil during dormant-season fires on the Moquah Barrens because 180 and 320 °C temperatures, on average, did not occur below the duff (Figure 18).

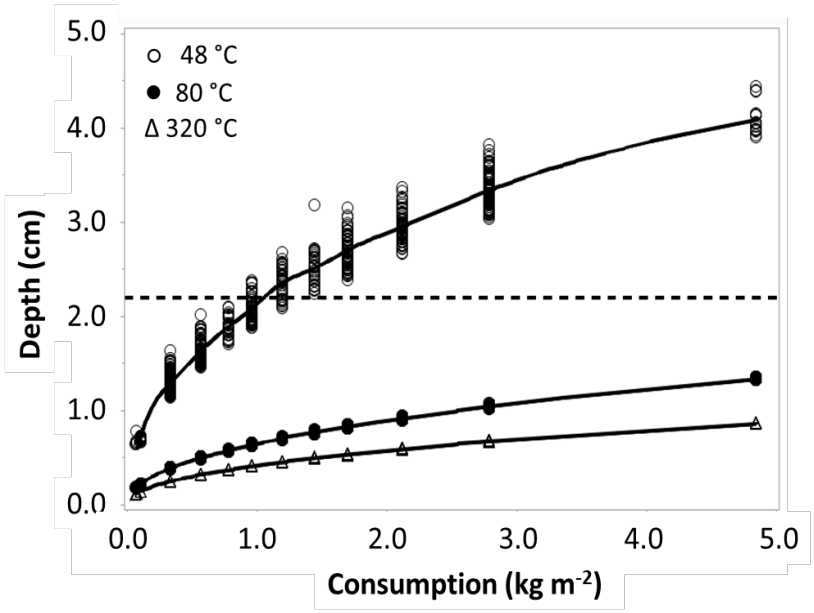


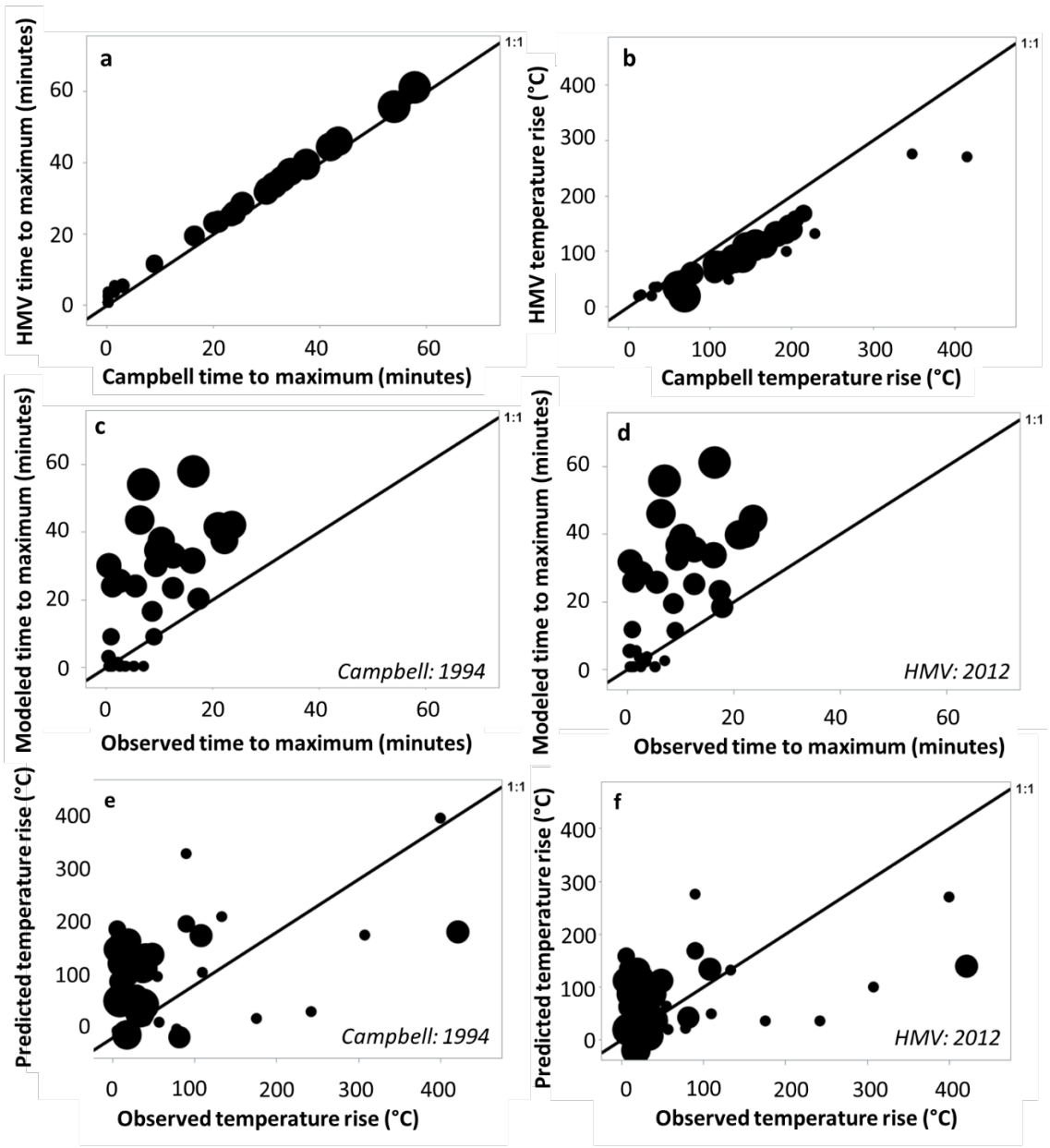
Figure 18. Predicted depths reached by temperatures relevant for organic material thermal degradation (180 and 320 °C) and impact on biological systems (48 °C) relative to fuel consumption. Soil organic material volatilization and charring begins around 180 °C and rates peak around 320 °C. Predicted values obscure variability so this plot represents the average trend. The reference line at the average prefire duff depth (2.2 cm) is provided for context, where depth on the y-axis is defined from the surface of the duff.

Evaluation of FOFEM soil heating routines

The soil heating (temperature) data are from fine thermocouples inserted at or near the duff-mineral soil interface which, in the model, is depth = 0 (Figure 6). There was error in positioning the TCPs, sometimes the tip was in the duff and sometimes the tip ended up deeper in the mineral soil. We only use temperature regimes at the interface or deeper in the mineral soil. Below, we report evaluation data based on temperature rise from background to maximum and the time from when soil temperatures first started rising at a given depth to the time of the maximum. Because FOFEM outputs temperature information at 1 cm increments, we fit nonlinear models to both temperature rise and time of maximum so we could interpolate between depths as needed to compare model with measurements at the same depths within the mineral soil. Temperature rise declines as a negative exponential with depth while time to maximum temperature rose exponentially with depth.

The two FOFEM soil heating models performed similarly in time to maximum temperature and temperature rise (Figure 19a, b). In all figures presented below, the size of the symbol is determined by the duff thickness, with the largest symbols having the deepest duff and the smallest symbols having no duff cover. The HMV model predicted shorter times to reach maximum temperature than the Campbell model while the Campbell model tended to generate slightly higher maximum temperatures than the HMV model. Duff depth did not lead to differences among the models in their performance.

Figure 19. Scatterplots showing results of model comparisons and evaluations. Comparisons of the Heat-Moisture-Vapor (HMV) and Campbell soil heating models run within FOFEM are shown for (a) time from first rise above background temperature to maximum temperature, and (b) temperature rise from prefire soil temperature to maximum temperature. Comparisons between predicted and observed time from first rise above background temperature to maximum temperature are shown for the (c) Campbell and (d) HMV models. Comparisons between predicted and observed mineral soil temperature rise are shown for the (e) Campbell and (f) HMV models. Symbol size is dependent on duff depth with the smallest symbols having no duff cover.



The evaluation against our soil TCP measurements suggest areas for future research and development to improve FOFEM. The time from the first rise above background temperature to maximum temperature was generally long in the models compared with times estimated using

our field data (Figure 19 c, d). Longer times to maximum temperature would suggest that the combustion rates in the BURNUP sub-model in FOFEM that supplies heat to the mineral soil surface are lower than expected. As well, where there was duff, modeled temperature rise was often high relative to observation (Figure 19 e, f). This is consistent with results for duff consumption. In the model, duff consumption averaged 60% whereas field observations suggest duff consumption was more like 20%, with a lot of variability (Figure 17). Likewise, litter, 1-hr, and 10-hr fuels were modeled as fully consumed, and 100hr fuel was nearly so (93%), substantially higher than our field estimates (Table 6). In the future we will test several options in the HMV model that modify the time-course of heat flux at the soil surface, these options include prescribed fire, wildfire, and burn pile. We expect that it will be necessary to evaluate and further develop the BURNUP model. Clearly, improving predictions of soil surface heat flux from combustion behavior should be a focus for future research and development.

In future analyses, we will explore why models and field measurements differed as the basis for recommending future research directions. It appears that the combustion and heat transfer processes determining heat flux at the soil surface under a range of combustion and duff conditions will be a key focus. Specifically, totals and rates of fuel consumption in FOFEM will be examined along with duff consumption predictions and how they related to duff moisture.

We measured about 0.5 cm of duff combustion on average and duff appeared to act much more as an insulator than a heat source. Duff's role as an insulator has relevance for the ability of dormant season burns to heat below the ground surface, because duff depth is related in part to current vegetation type and burn history (Figure 20). Vegetation cover influenced duff depth at reference sites ($F = 3.4$, $p = 0.03$) but not at burn plots according to ANOVA. Post-hoc pairwise comparisons of reference cover types revealed that that pine forest and brush plots had significantly greater duff depth than natural barrens plots. T-tests comparing burned vs. reference site duff depth within each cover type revealed that duff depth was significantly greater at reference brush plots than at burned brush plots ($t = -4.6$, $p < 0.01$), but while that trend generally was consistent across other vegetation types (Figure 20), no other significant differences between pairs were observed.

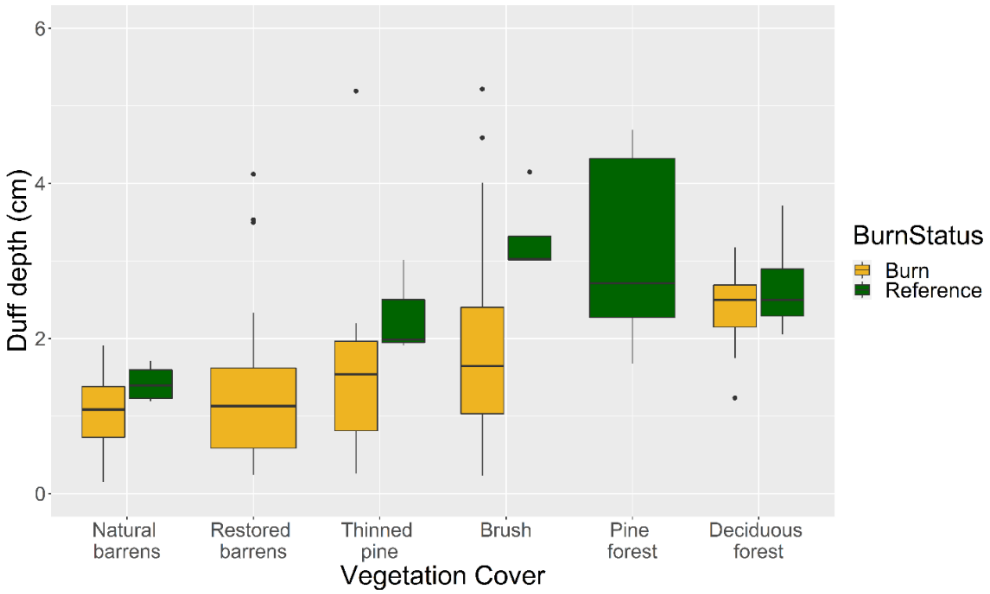


Figure 20. Duff layer thickness (cm) by vegetation type and burn status. Duff thickness within burn plots were measured immediately postfire, and data coms from the litter/duff pin locations within sample quadrats (Figure 7).

Soil Processes

Carbon, nitrogen and pyrogenic C stocks

Total soil C stock was greatest in pine plots, whereas barrens, brush, and deciduous plots supported similar total C stock (Figure 21 a). Differences among cover types was statistically significant only for the 5-10 cm horizon, where the brush and deciduous plots supported lower stocks than barrens plots (Table 9). C stocks decreased from prefire to postfire burn status (Figure 21 b). For the forest floor, C stocks decreased in the postfire and 1-year postfire treatments, relative to the reference and prefire plots (Table 9). This result was expected, as the forest floor horizon experiences the most direct effects of fire, via pyrolysis and combustion, as well as mass loss during wind erosion of ash layers after fire, which can be high in organic C (Bodí et al., 2014; Quigley et al., 2019). The only statistically significant effect of fire on mineral soils occurred in the 0-5 cm layer as a decrease between the prefire to postfire stock (Table 9). The postfire decline in total C was similar for low and high fuel treatments (Figure 21 c), suggesting that the response of total soil C stocks to fire are relatively consistent within a vegetation cover type, regardless of the amount of fuel present.

Pyrogenic C (PyC) comprised approximately 10 – 15 % of total soil C, with deciduous forest having the greatest proportion of PyC among vegetation cover types (Figure 21 d). Upper mineral soils stored the greatest amount of PyC, with greater PyC in deciduous than barrens plots for the 0-5 cm mineral soil (Table 9). Total PyC showed a decline over time after fire, and this was significant for the forest floor, whereas only the 5-10 cm mineral soil PyC stock was lower in the 1-yr postfire measurements relative to prefire measurements (Figure 21 e, Table 9). The decrease in PyC stocks in the forest floor horizon over this time period (Figure 21 e) reflects that although PyC is formed via pyrolysis during fires, pyrolysis represents an early phase of the combustion process. Thus, PyC can also be lost via combustion, postfire erosion of ash layers, or physical translocation over time into underlying mineral soil horizons (Bodí et al., 2014; Miesel et al., 2015; Quigley et al., 2019). This physical translocation of PyC may explain patterns observed in the mineral horizons, where existing PyC stocks would be protected from direct effects of the relatively low-intensity fires in our study system. Future investigations of the influence of topography and precipitation may help clarify these responses. In general, patterns of PyC response to fire and fuel treatments followed those of total C, although greater PyC loss was observed in 0-5 cm mineral soils of pine plots with added brush fuels than those with no fuels added (Figure 21 f).

Total N stocks also followed the general patterns exhibited by total C stocks, across vegetation cover type, burn status, and fuel treatments (Figure 21 g, h, i). Total soil N was greatest at pine plots and lowest at deciduous forest plots, and this was driven by differences in the mineral soil, where 0-5 cm N was greater in pine plots than brush or deciduous plots, and 5-10 cm N was greater in the barrens and pine plots than in deciduous plots (with barrens also differing from brush, whereas pine did not differ from barrens or brush plots) Figure 21 g, Table 9). N stocks showed a small decline postfire, primarily due to loss of forest floor N, and recovery to near-prefire stocks by 1-year postfire, primarily because of an increase in upper mineral soil N (Figure 21 h). The only statistically significant differences within horizon occurred between the prefire and 1-yr postfire measurements for forest floor. There were no differences among prefire, postfire, or 1-yr postfire measurements for either of the mineral soil layers. For the fuel treatments, the only statistically significant differences in N stocks between prefire and postfire

measurements occurred for the mineral soil horizons, with fuel additions decreasing N stocks ($p<0.02$ for all).

Barrens plots showed very minimal decrease in soil N stocks following fire, regardless of fuel treatment (Figure 21 i). Pine plots had decreased soil N stocks postfire under both ambient and fuel addition conditions, but the majority of N loss was observed in the forest floor horizon under ambient fuel conditions as opposed to loss in the mineral soil under brush fuel additions (Figure 21 i). Smaller mineral soil N (and C) stocks in deciduous plots, as well as a relatively high proportion of total N present in the forest floor horizon, may suggest greater nutrient demand by species in the deciduous forest cover type, as well as higher N concentration in deciduous litter. These processes support the idea that mesic species facilitate mesophication of fire-adapted ecosystems, leading to accumulation of relatively nutrient-rich organic horizons that retain moisture and further inhibit fire occurrence and spread (Nowacki and Abrams, 2008; Quigley et al., 2020).

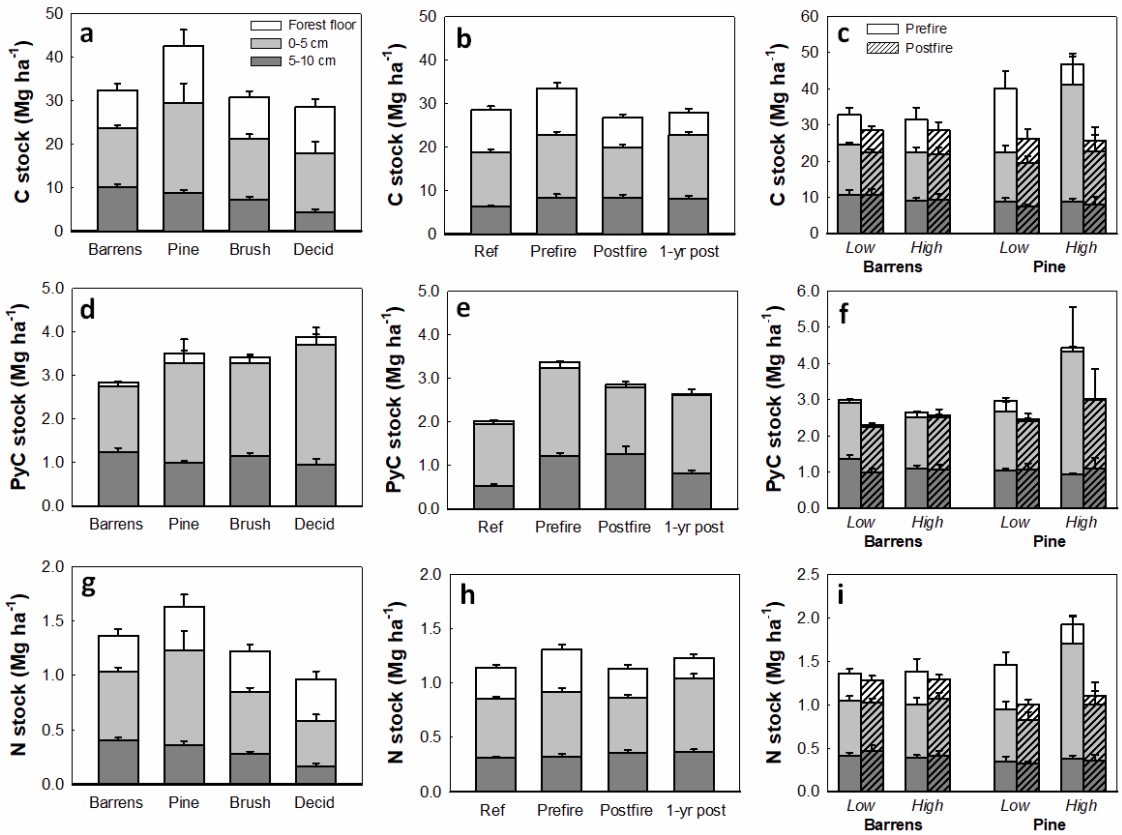


Figure 21. Total carbon, pyrogenic C, and nitrogen stocks in the forest floor, 0-5 cm and 5-10 cm mineral soil horizons. The first column (a,d,g) shows differences among vegetation cover types (fuel additions and removals excluded), the second column (b,e,h) shows differences among burn status groups, and the third column (c,f,i) shows differences between low and high (brush fuel added) fuel treatments. Total bar height represents cumulative stocks across layers, with fill color indicating stock within layer. Error bars represent standard errors within layer. Note differences in y-axis scales among panels.

Table 9. Statistical results for analysis of soil carbon and nutrient stocks by horizon (FF: forest floor, 0-5 cm and 5-10 cm mineral soil layers). Shown are model degrees of freedom (numerator DF, denominator DF), F-statistic and p-values, and Tukey-adjusted pairwise comparisons for models with p ≤ 0.05. Pairwise comparisons for fuel treatments are described in the text.

	Vegetation cover type (prefire)							Burn status (ambient fuels)							Fuel treatment		
Variable	DF	F	P	Barrens	Pine	Brush	Decid	DF	F	P	Ref	Pre	Post	1-yr	DF	F	P
C stock																	
FF	3, 53	0.67	0.557	a	a	a	a	3, 165	6.52	**<0.001**	**a**	**a**	**ab**	**bc**	3, 145	0.905	0.440
0-5	3, 58	2.52	0.067	a	a	a	a	3, 179	3.50	**0.017**	**ab**	**a**	**b**	**a**	3, 155	2.287	0.081
5-10	3, 52	4.78	**0.005**	**a**	**ab**	**b**	**b**	3, 175	4.60	**0.004**	**b**	**a**	**a**	**ab**	3, 156	3.996	**0.009**
PyC stock																	
FF	3, 54	2.24	0.093	a	a	a	a	3, 180	17.49	**<0.0001**	**b**	**a**	**bc**	**c**	3, 155	0.50	0.683
0-5	3, 57	3.24	**0.029**	**b**	**ab**	**ab**	**a**	3, 179	4.73	**0.003**	**b**	**a**	**ab**	**ab**	3, 155	3.05	**0.030**
5-10	3, 52	1.48	0.230	a	a	a	a	3, 175	16.72	**<0.0001**	**b**	**a**	**a**	**b**	3, 153	0.26	0.854
N stock																	
FF	3, 53	0.15	0.926	a	a	a	a	3, 165	5.70	**<0.001**	**ab**	**a**	**ab**	**b**	3, 143	1.00	0.396
0-5	3, 58	3.83	**0.014**	**ab**	**a**	**b**	**b**	3, 179	5.22	**0.002**	**a**	**ab**	**a**	**b**	3, 155	5.49	**0.001**
5-10	3, 52	7.18	**<0.001**	**a**	**ab**	**bc**	**c**	3, 175	1.77	0.155	a	a	a	a	3, 156	7.38	**0.0001**
Ca stock																	
FF	3, 105	3.57	**0.017**	**ab**	**a**	**b**	**ab**	3, 249	12.68	**<0.001**	**b**	**a**	**a**	**b**	3, 292	2.76	**0.042**
0-5	3, 109	2.08	0.108	a	a	a	a	3, 260	4.78	**0.003**	**b**	**ab**	**ab**	**a**	3, 306	1.72	0.162
5-10	3, 106	0.16	0.924	a	a	a	a	3, 262	1.97	0.119	a	a	a	a	3, 309	1.96	0.120
Mg stock																	
FF	3, 105	3.01	**0.033**	**ab**	**b**	**a**	**ab**	3, 249	12.94	**<0.001**	**bc**	**a**	**b**	**c**	3, 292	2.99	**0.031**
0-5	3, 109	0.95	0.421	a	a	a	a	3, 260	16.75	**<0.001**	**c**	**a**	**a**	**b**	3, 306	3.36	**0.019**
5-10	3, 106	2.79	0.044	**ab**	**b**	**a**	**ab**	3, 262	2.31	0.077	a	a	a	a	3,309	2.28	0.080
K stock																	
FF	3, 105	4.95	**0.003**	**a**	**a**	**b**	**ab**	3, 249	15.03	**<0.001**	**b**	**a**	**b**	**b**	3, 292	5.90	**0.001**
0-5	3, 109	0.42	0.738	a	a	a	a	3, 260	5.10	**0.002**	**c**	**ac**	**ab**	**b**	3, 306	8.99	**<0.001**
5-10	3, 106	1.39	0.2515	a	a	a	a	3, 262	8.91	**<0.001**	**b**	**b**	**a**	**b**	3,3,09	6.89	**<0.001**
P stock																	
FF	3, 105	2.83	**0.042**	**b**	**ab**	**a**	**ab**	3, 254	5.44	**0.001**	**ab**	**a**	**ab**	**b**	3, 299	5.25	**0.002**
0-5	3, 110	0.67	0.570	a	a	a	a	3, 261	4.74	**0.003**	**b**	**a**	**a**	**a**	3, 307	8.08	**<0.001**
5-10	3, 106	0.47	0.703	a	a	a	a	3, 262	2.13	0.100	a	a	a	a	3, 309	4.04	**0.008**

There were no significant differences in forest floor or mineral soil C or PyC stocks across the NPS severity classes, or across paint tag temperature categories ($p > 0.10$ for all, Figure 22).

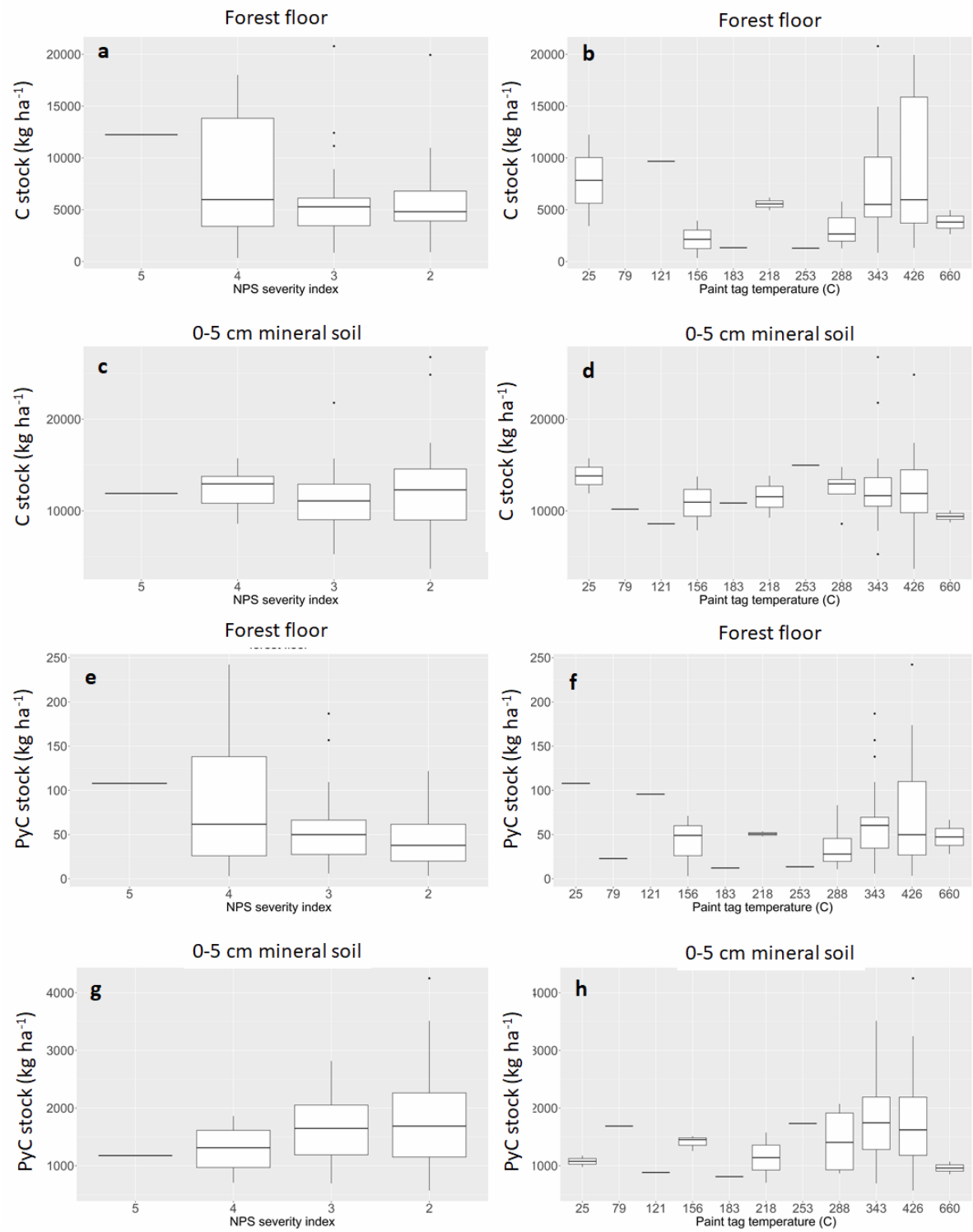


Figure 22. Forest floor and 0-5 cm mineral soil total carbon and pyrogenic carbon stocks, for NPS severity and paint tag temperature.

Cation and phosphorus stocks

Calcium (Ca), magnesium (Mg), potassium (K), and phosphorus (P) are additional essential plant macronutrients. They are present in mineral soil in exchangeable forms, as well as bound in dead plant biomass in organic horizons and soil organic matter. Total Ca stocks, and 0-5 cm mineral soil Ca stocks, were lower in deciduous plots than in all other vegetation cover types, whereas the pine plots supported the lowest forest floor Ca and greatest (and most variable) 0-5 cm mineral soil Ca stocks (Figure 23 a). Across all cover types, overall forest floor Ca stocks were greater in prefire plots than reference plots and increased over time after fire (Figure 23 b). There was high variability in Ca stocks among fuel addition treatments, resulting in no statistically significant differences among treatments (Figure 23 c).

Total Mg stocks were greater in brush and deciduous plots than barrens and pine plots, and this was influenced by forest floor Mg stocks (Figure 23 d). Mg stocks were greater in prefire forest floor and 0-5 cm mineral soil than in reference plots, and forest floor Mg stocks decreased across our postfire measurements (Figure 23 e). There were no effects of fuel addition or fire, in barrens or pine plots on total Mg stocks (Figure 23 f).

Although total K stocks followed the same patterns as Ca across vegetation types, the forest floor K stocks were greater in barrens and brush plots than pine or deciduous plots, and deciduous plots supported lower K stocks than other cover types (Figure 23 g). Relative to reference plots, the prefire restoration plots supported greater forest floor K stocks, with forest floor stocks decreasing across the postfire and 1-year postfire measurements. The K results suggest that fire creates a pulse of K in mineral soils, which diminishes by the 1-year postfire timepoint (Figure 23 h). This pulse is also reflected in the prefire to postfire measurements in barrens plots, with fuel addition treatments trending toward greater K stocks at both timepoints (Figure 23 i).

Phosphorus (P) stocks followed the same pattern as Mg stocks, although the pool sizes were generally 25% of Mg stocks and suggest greater influence of vegetation type in the 5-10 cm mineral soil (Figure 23 j). Total and 0-5 cm mineral soil P stocks were greater in prefire restoration plots than reference plots, and forest floor P stocks decreased from prefire to 1-yr postfire measurements (Figure 23 k). There was high variability in P stocks among fuel treatment plots, and no differences among treatments (Figure 23 i).

A general pattern of decrease in forest floor nutrient stocks from prefire to 1-yr postfire measurements, and parallel increases in mineral soil stocks, suggests that prescribed fires influences nutrient pools, as well as the distribution of nutrients within the soil profile likely a result of leaching of ash and subsequent accumulation in upper mineral soils.

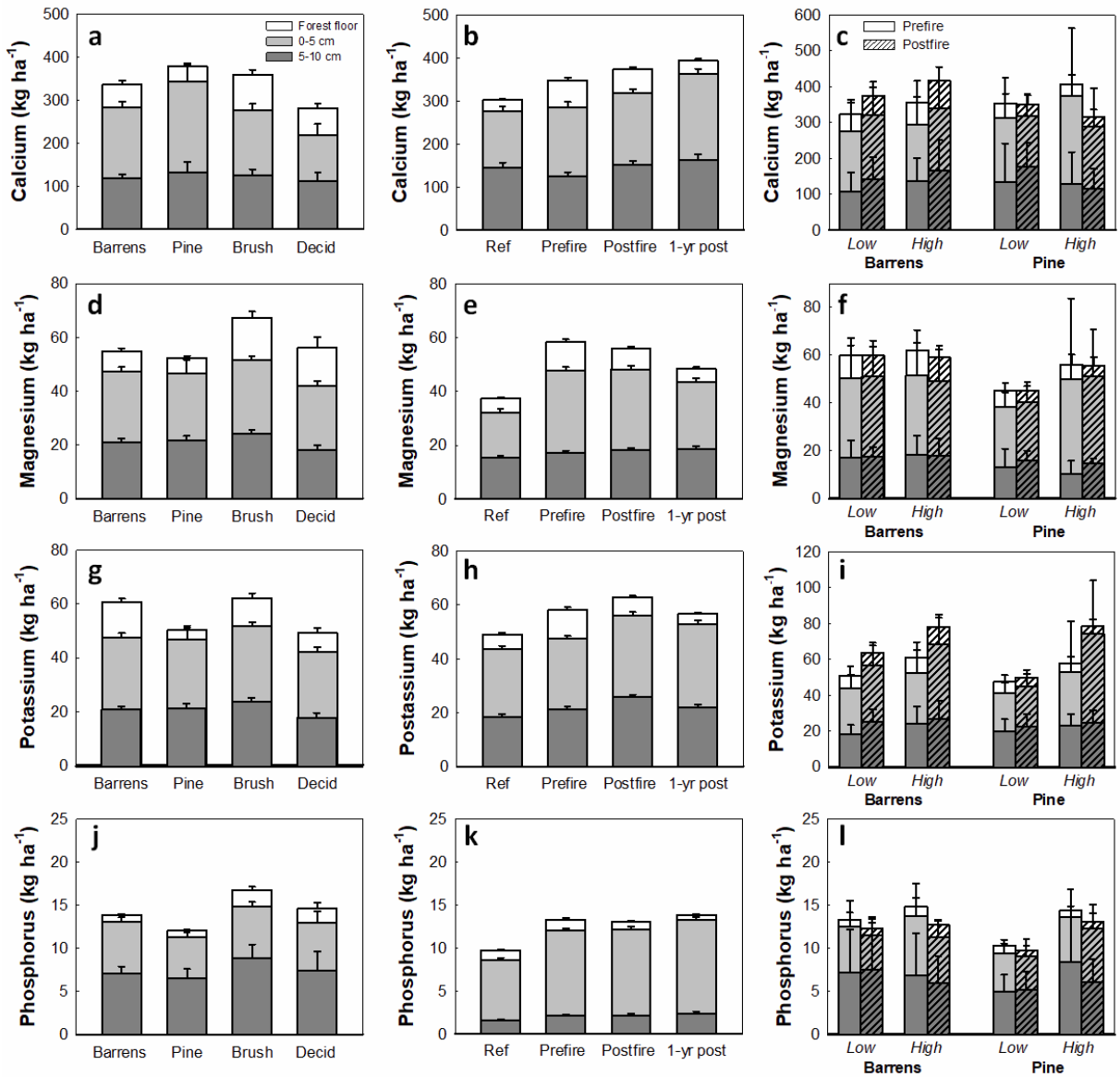


Figure 23. Soil calcium (Ca), magnesium (Mg), potassium (K), and phosphorus (P) stocks in the forest floor and 0-5 cm and 5-10 cm mineral soil horizons. The first column (a,d,g,j) shows differences among vegetation cover types (fuel additions and removals excluded), the second column (b,e,h,k) shows differences among burn status groups, and the third column (c,f,i,l) shows differences between low and high (brush fuel added) fuel treatments. Total bar height represents cumulative stocks across layers, with fill color indicating stock within layer. Error bars represent standard errors within layer. Note differences in y-axis scales among panels.

Soil pH

Mineral soil pH was acidic across all study plots, with mean pH typically between 4.5 and 5. Soil pH was generally lower at unburned reference plots than at burned plots with similar vegetation cover, and there was a general pattern of increased soil pH following fire (Figure 24). Although low severity prescribed burns in this ecosystem produce ash that is highly enriched in cations

(Quigley et al., 2019), the pulse of cations from a single burn event appears to have minimal effects on mineral soil pH, with no statistically significant differences in restoration plots across the study period (Figure 24). The greater pH in burn plots compared to reference plots suggests that repeated prescribed burns may increase soil pH over time, as our study area has been under active restoration treatments for some time (Quigley et al., 2020).

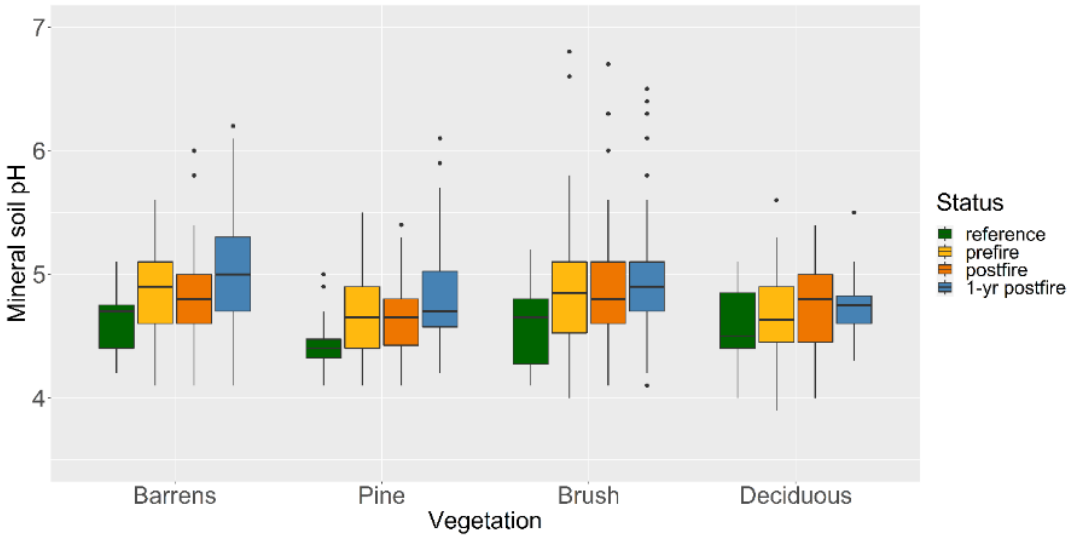


Figure 24. Mineral soil pH (averaged for 0-5 cm and 5-10 cm horizons) by vegetation cover and burn status.

Nutrient exchange rates

Nutrient exchange rates indicate the rate at which nutrients become available for plant uptake. We present the results of ion exchange resins, which imitate the rate of nutrient supply to plant roots (Qian et al., 2008) (Figure 25). Despite large inputs of cation-rich ash following prescribed burns (average: 2.0 Mg ha^{-1}, maximum: 10.2 Mg ha^{-1}) (Quigley et al., 2019), inorganic N (nitrate, NO_3^-; ammonium, NH_4^+) exchange rates were minimally affected by a single burn event.

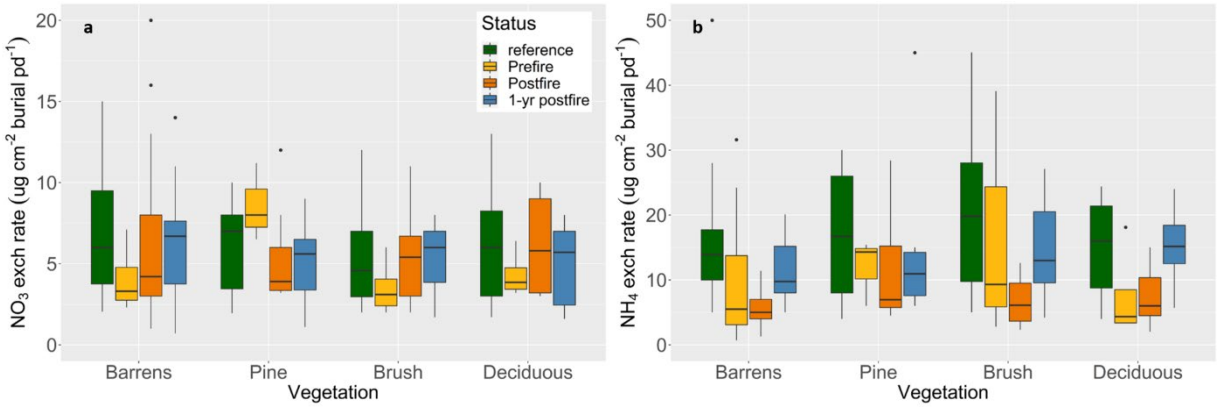


Figure 25. Exchange rates for soil nitrate (a) and ammonium (b), by vegetation cover type and status. Values are reported as exchange rate per unit area of resin strip (µg cm^{-2}) over the probe burial period. Data are shown for plots with ambient fuel loads only (i.e. no fuel treatments).

Burn status had no effect on NO_3^- exchange rates, for any of the four vegetation cover types (all p>0.11, Figure 25 a). Although NH_4^+ exchange rates in barrens experimental plots differed from long-unburned barrens reference plots, there were no differences across burn status (Figure 25 b). For brush plots, the only statistically significant difference occurred between the postfire and reference treatments (p = 0.001, Figure 25 b). There were no differences in pine or deciduous plots.

Phosphorus (P) exchange rates varied by vegetation cover type, with the most notable difference between deciduous and barrens plots (Figure 26 a). Although ANOVA suggested a significant effect of vegetation cover on P exchange rates ($F_{3, 174} = 2.8$, p = 0.04), post-hoc tests revealed that pairwise differences were only marginally significant (p = 0.07 for Deciduous *vs* Barrens). Within vegetation cover types, there were no significant differences in P exchange rates among burn status groups (all p > 0.05).

Ca exchange rates varied significantly by vegetation cover (Figure 26 c; $F_{3, 174} = 4.1$, p < 0.01), with brush plots showing significantly higher Ca exchange rates than barrens plots (p < .01). Within cover types, there was no significant effect of burn status on Ca exchange rates (all p > 0.05). Mg exchange rates did not vary among vegetation cover classes (Figure 26 d; $F_{3, 174} = 2.6$, p > 0.05). Similarly, burn status groups did not vary significantly within any of the vegetation cover types (all p > 0.05). K exchange rates did not vary among vegetation cover classes (Figure 26 b; $F_{3, 174} = 2.5$, p > 0.05). Similarly, burn status groups did not vary significantly in Ca exchange rates within any of the vegetation cover types (all p > 0.05).

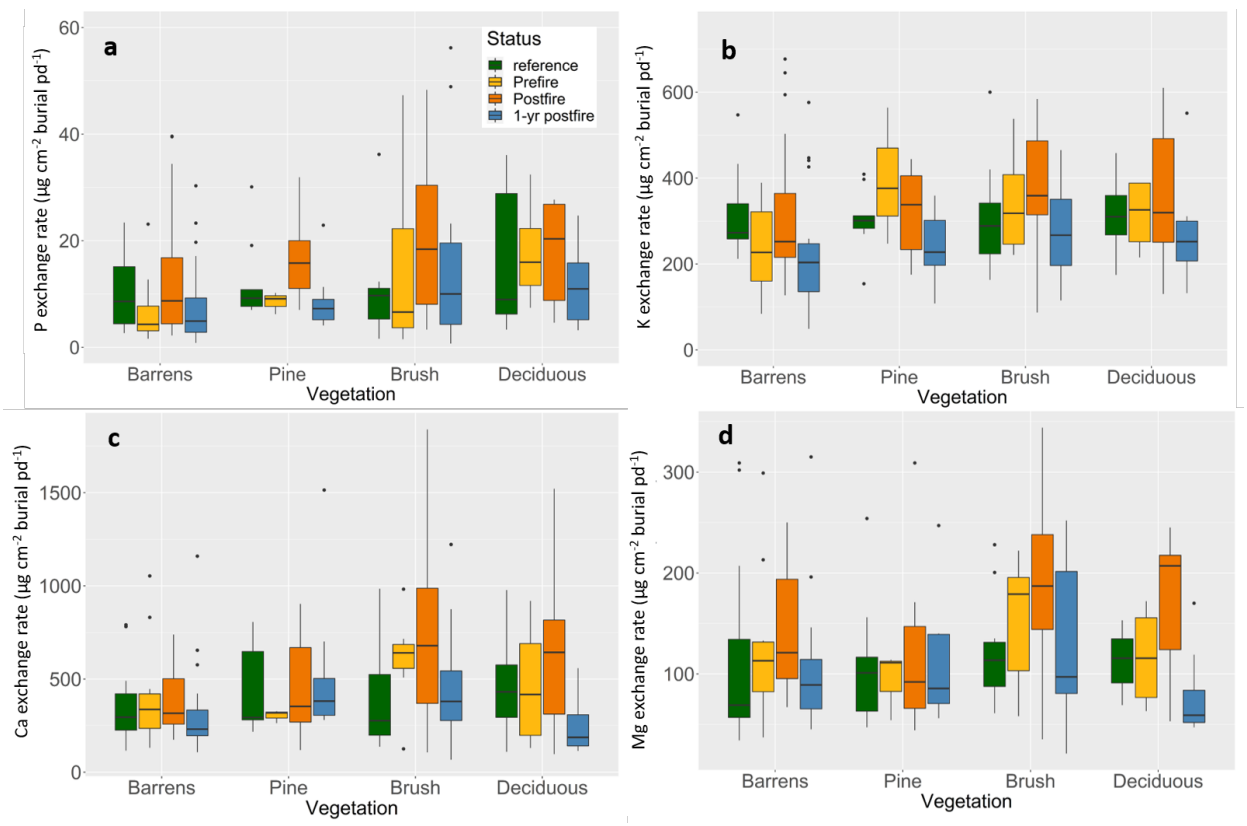


Figure 26. Nutrient exchange rates by vegetation cover type and burn status. Values are reported as exchange rate per unit area of resin strip (µg cm^{-2}) over the probe burial period. Data are shown for plots with ambient fuel loads only (i.e. no fuel treatments).

There were no effects of fuel treatments on inorganic N (NO_3^- or NH_4^+), P or Mg exchange rates in any of the vegetation types, in either the postfire growing season, or one year after fire ($p>0.05$ for all; Figure 27, Figure 28). The only statistically significant effects of fuel treatments on nutrient exchange rates occurred in barrens plots, with increases in K exchange at 1-yr postfire ($p = 0.003$, Figure 28e), and increases in Ca exchange 1-yr postfire ($p = 0.029$, Figure 28h).

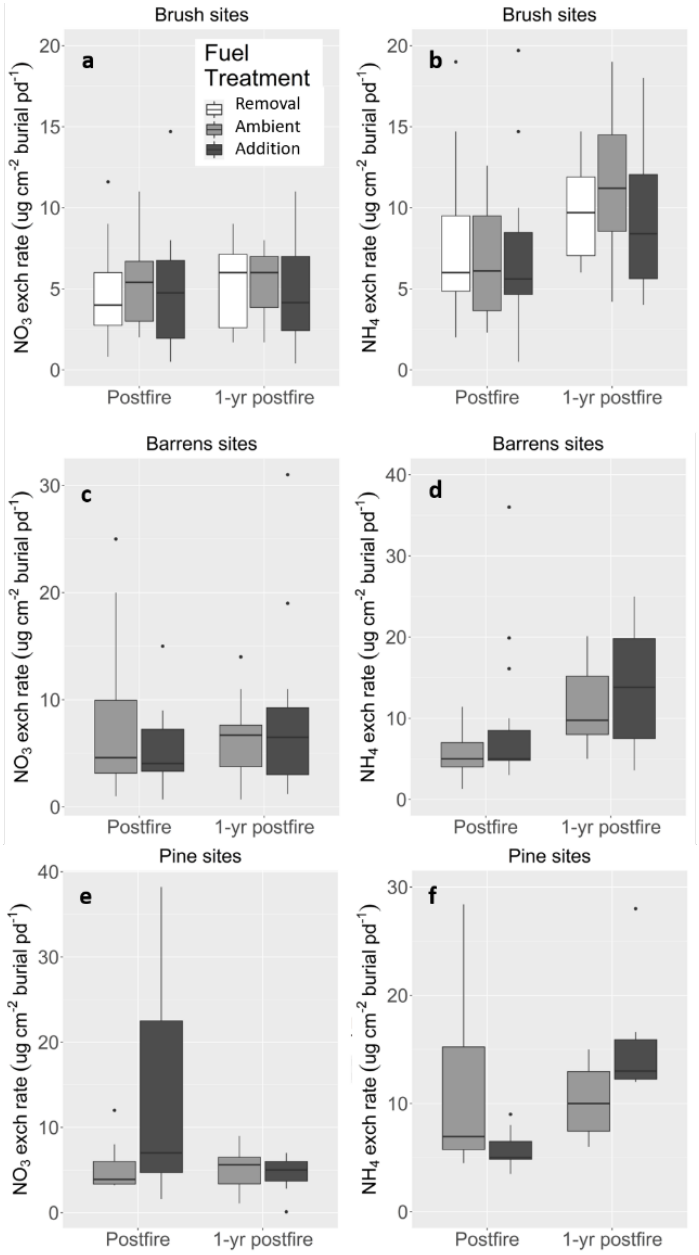


Figure 27. Nutrient exchange rates in relation to brush fuel treatments at brush plots (first row), barrens plots (second row), and pine plots (third row). Addition treatment represents cut brush at brush plots. No fuel manipulation treatments were applied to deciduous forest plots.

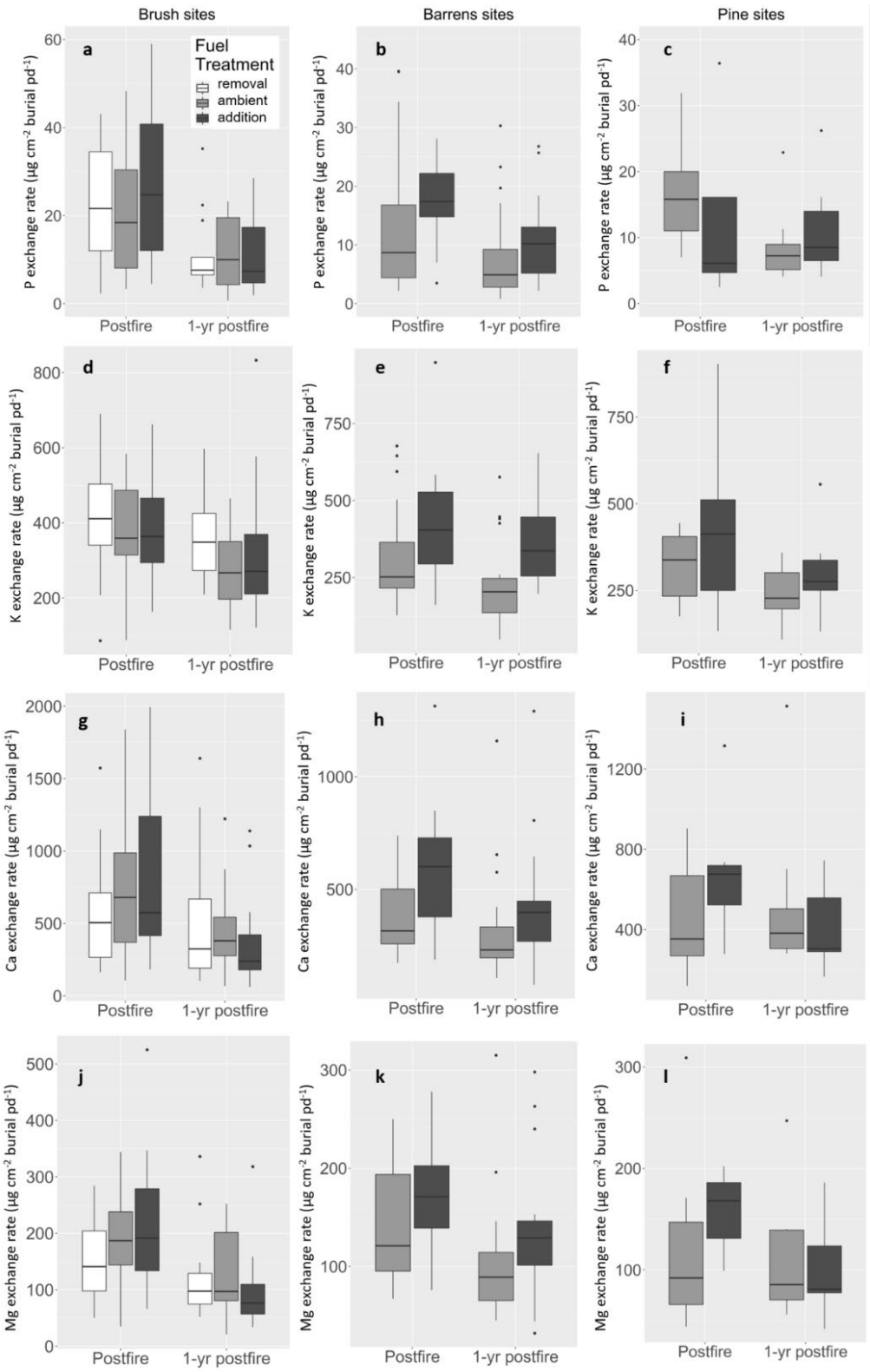


Figure 28. Nutrient exchange rates in relation to brush fuel treatments at brush plots (first column), barrens plots (second column), and pine plots (third column). Addition treatment represents cut brush at brush plots. No fuel manipulation treatments were applied to deciduous forest plots.

Vegetation & Seed Bank

Seed Bank

A total of 1,345 plants germinated (Table 10). Thirty-nine plants were identified to genus or species. Most plants were graminoids (70% (n=943 [germinants])). Shrubs and herbs made up a low amount of the germinants (20% (n=263) and 10% (n=129), respectively). Tree germinants were rare (<1% (n=10)). Most germinants were native (98% (n=1,318). Most species were part of the target restoration composition, although low-priority plants were more common than high-priority plants (58% (n=774) and 40% (n=542), respectively). Non-target plants were rare (2% (n=29)).

Germinant density was similar in reference and in managed areas pre and immediately post-2018 fire (p = 0.09; Figure 29). Pairwise comparisons showed densities prefire were lower than reference areas (p = 0.05), and densities postfire tended to be greater than prefire, but not significantly. However, when breaking down germinant density by vegetation history, the trend of postfire greater than prefire densities was significant for historically deciduous forest conditions (Figure 30 a). Historic barrens had the opposite trend: prefire greater than postfire. Prefire germinant density was greater in historic barrens relative to historic deciduous forest (p = 0.03), but this relationship did not persist immediately postfire. Germinant densities were similar among current vegetation conditions pre and postfire (Figure 30 b).

Figure 29. Distribution of germinant density from the soil seed bank profile (duff+mineral soil layers) in reference and managed area pre- and post-2018 fire in the Moquah Barrens Management Area. The mean is represented by the diamond symbol. Dots represent outliers.

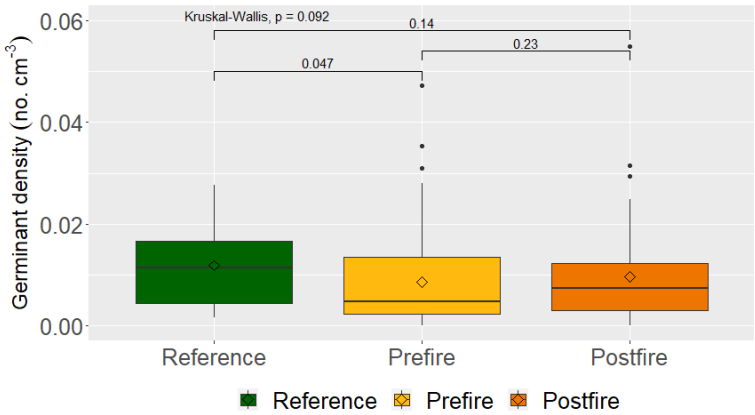


Figure 30. Distribution by a. historical vegetation type and b. current vegetation condition, of germinant density from the soil seed bank profile (duff+mineral soil layers) in managed area pre- and post-2018 fire in the Moquah Barrens Management Area. The mean is represented by the diamond symbol. Dots represent outliers.

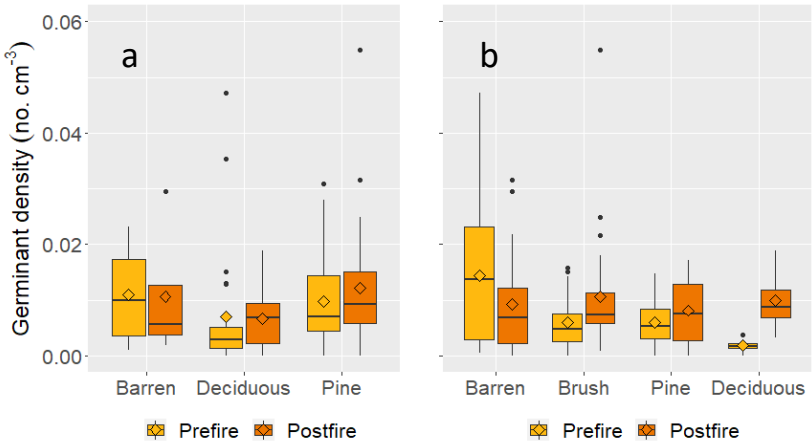


Table 10. Genus-Species list and tally (count) of plants that germinated from soil sampled from the Moquah Barrens Management Area and then processed and placed in a greenhouse for a 3-month germination period (similar to a local growing season length). Species were rated by local managers as high-/low-priority restoration targets for composition or non-target (none).

Species	Total	Ref (2018)	Pre (2017)	Post (2018)	Mineral (2017-18)	Duff (2017-18)	Origin	Restoration Target	Lifeform
Agrostis sp.	54	13	0	41	34	20	Native	Low	graminoid
Anaphalis margaritacea	22	1	6	15	6	16	Native	High	herb
Aster sp.	7	0	7	0	1	6	Native	Low	herb
Calystegia spithamaea	1	0	1	0	1	0	Native	High	herb
Campanula rotundifolia	5	0	1	4	1	4	Native	High	herb
Carex sp.	292	57	97	138	186	106	Native	Low	graminoid
Comptonia peregrina	5	2	0	3	2	3	Native	High	shrub
Conyza canadensis	15	0	13	2	2	13	Native	Low	herb
Corylus americana	69	28	16	25	10	59	Native	Low	shrub
Danthonia spicata	243	45	146	52	83	160	Native	High	graminoid
Diervilla lonicera	126	42	8	76	63	63	Native	High	shrub
Digitaria sp.	4	0	4	0	4	0	Introduced- Naturalized	None	graminoid
Epigaea repens	4	0	0	4	0	4	Native	High	woody
Epilobium angustifolium	1	0	1	0	0	1	Native	Low	herb
Equisetum sp.	1	1	0	0	1	0	Native	None	herb
Geranium sp.	3	0	2	1	1	2	Native	Low	herb
Hieracium sp.	19	1	17	1	2	17	Native	High	herb
Juncus tenuis	1	0	1	0	0	1	Native	Low	graminoid
Lechea sp.	1	0	0	1	0	1	Native	High	herb
Leucanthemum vulgare	10	2	0	8	5	5	Introduced- Naturalized	None	herb
Lonicera canadensis	1	0	1	0	1	0	Native	None	shrub
Oryzopsis pungens	70	20	21	29	32	38	Native	High	graminoid
Panicum sp.	215	7	184	24	129	86	Native	Low	graminoid
Pinus sp.	1	1	0	0	1	0	Native	High	tree
Poa sp.	63	22	0	41	32	31	Native	Low	graminoid
Populus sp.	9	1	1	7	7	2	Native	Low	tree
Rubus sp.	38	8	16	14	15	23	Native	Low	shrub
Rumex acetosella sp.	7	0	5	2	7	0	Introduced- Naturalized	None	herb
Thalictrum dioicum	7	2	0	5	4	3	Native	Low	herb
UNK herb	159	35	76	48	73	86	UNK	UNK	herb
UNK spiderwort	20	7	0	13	4	16	Native	High	shrub
Vaccinium sp.	21	0	6	15	12	9	UNK	UNK	herb
Verbascum thapsus	6	0	5	1	1	5	Introduced	None	herb
Viola sp.	24	3	10	11	17	7	Native	High	herb

In terms of brush/fuel treatments, germinant density was greater in cut-and-leave brush compared to no-brush treatment (i.e., Ambient) in prefire conditions ($p = 0.05$), but this relationship did not persist immediately postfire, suggesting that the dragging and piling of fuel prior to the burn may have influenced the number of seeds on-site (Figure 31). Burn severity and temperature (as indicated by paint tags) were variable without significant relationships to germinant density, but germinant density tended to decline slightly at higher paint tag temperature, measured at 5 cm height above the litter surface (Figure 32).

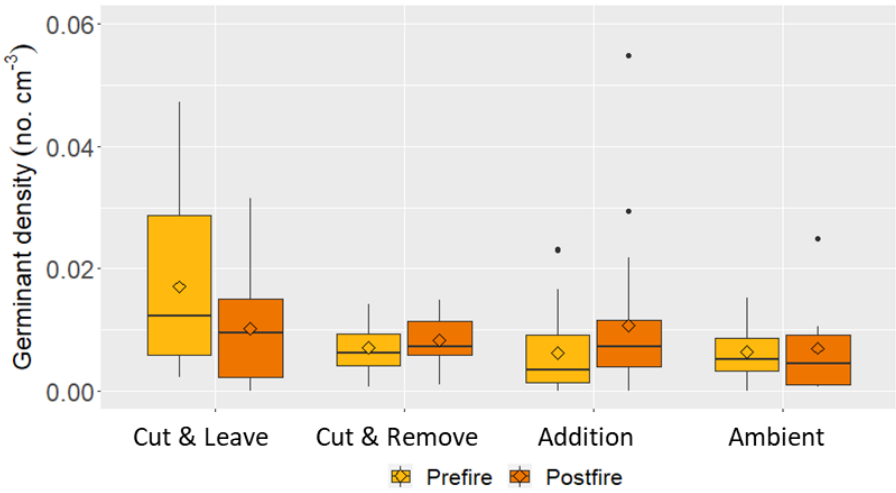


Figure 31. Distribution by fuel treatment of germinant density from the soil seed bank profile (duff+mineral soil layers) in managed area pre- and post-2018 fire in the Moquah Barrens Management Area. The mean is represented by the diamond symbol. Dots represent outliers.

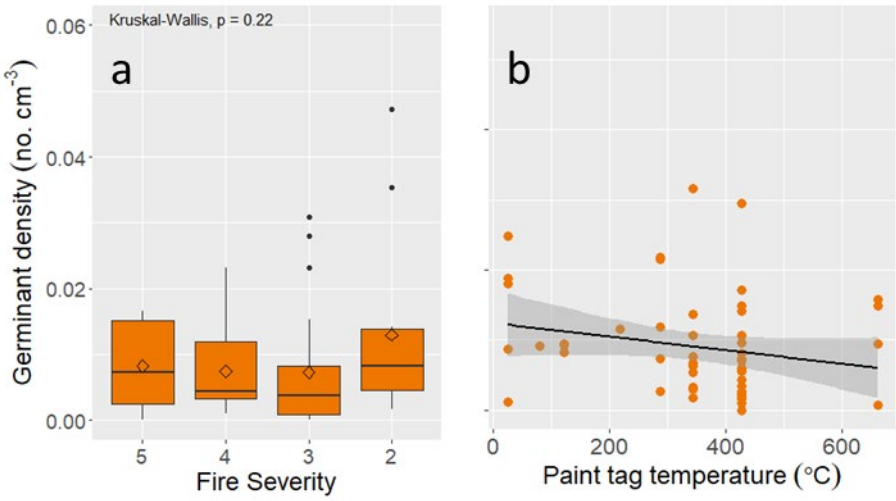


Figure 32. Distribution by a) NPS burn severity and b) by paint tag temperature, of germinant density from the soil seed bank profile (duff+mineral soil layers) in managed area pre- and post-2018 fire in the Moquah Barrens Management Area. The mean is represented by the diamond symbol. Dots represent outliers.

The increase in germinant density postfire suggests prescribed fire may be stimulating successful germination within deciduous plots more so than other vegetation conditions, especially barren conditions. However, the slight decrease in successful germination with increasing paint tag temperature may suggest a threshold at which germination is reduced, but this result may also be due to our small sample sizes in the highest fire temperatures.

Overall, abundance and compositional shifts immediately after prescribed burns were subtle based our study. These results could reflect the minimal effects of soil heating from prescribed burns, the ubiquitous condition of the seed bank in this area, or the limitations of the methods we

used (i.e., 3-month period of nutrient rich, moist, high light, and warm growing conditions of a greenhouse).

Brush and Hardwood Sprouting

Tree sapling and shrub ("brush", alive stems >0.5-tall to 2.54-cm DBH) were more common in areas historically or currently typed as forest (Figure 33). Prescribed fire had similar effects on across vegetation type. Immediately after fire, brush stems were nearly eliminated, but partially rebounded to prefire densities one year later. In total, 23 woody species were identified (Table 11). Species found prior to fire were likely to be present after fire. White pine was the only species unique to references. Ten species were only found in the managed area at low frequencies and, of these, 6 were not found after fire. The results suggest that prescribed burns represent a low to moderate disturbance that partially setback resprouting and slightly shifted composition in the managed area. As such, brush persists broadly in areas of past and present forestlands.

Table 11. Number of plots measured (N) in burned and reference areas before (prefire) and after (postfire and one-year postfire) the 2016 prescribed fire.

Species		Growth habit	Number of plots (N)			
Scientific name	**Common name**		**Reference**	**Prefire**	**Postfire**	**1-yr**
Abies balsamea	Balsam fir	Tree	0	0	1	0
Acer rubrum	Red maple	Tree	17	38	1	39
Acer saccharum	Sugar maple	Tree	0	1	0	2
Betula papyrifera	Paper birch	Tree	5	17	2	22
Pinus banksiana	Jack pine	Tree	6	9	1	0
Pinus resinosa	Red pine	Tree	11	4	2	0
Pinus strobus	White pine	Tree	1	0	0	0
Populus sp.	Aspen	Tree	11	34	5	42
Prunus sp.	Cherry	Tree	6	19	2	18
Quercus sp.	Oak	Tree	17	50	7	52
Acer spicatum	Mountain maple	Shrub	0	1	0	0
Alnus crispa	green or mountain	Shrub	0	4	0	0
Amelanchier sp.	Juneberry	Shrub	15	47	1	44
Corylus americana	American hazelnut	Shrub	13	23	0	29
Corylus cornuta	Beaked hazelnut	Shrub	4	30	1	23
Crataegus sp.	Hawthorn	Shrub	0	3	0	0
Diervilla lonicera	Bush honeysuckle	Shrub	0	0	1	0
Prunus pumila	Sandcherry	Shrub	0	2	0	0
Prunus virginiana	Chokecherry	Shrub	0	5	0	0
Rosa blanda	Wild rose	Shrub	0	0	0	3
Rubus sp.	Raspberry/Blackberry	Shrub	2	0	2	23
Salix sp.	Willow	Shrub	4	36	0	41
Viburnum acerifolium	Maple-leaf arrowhead	Shrub	0	1	0	0

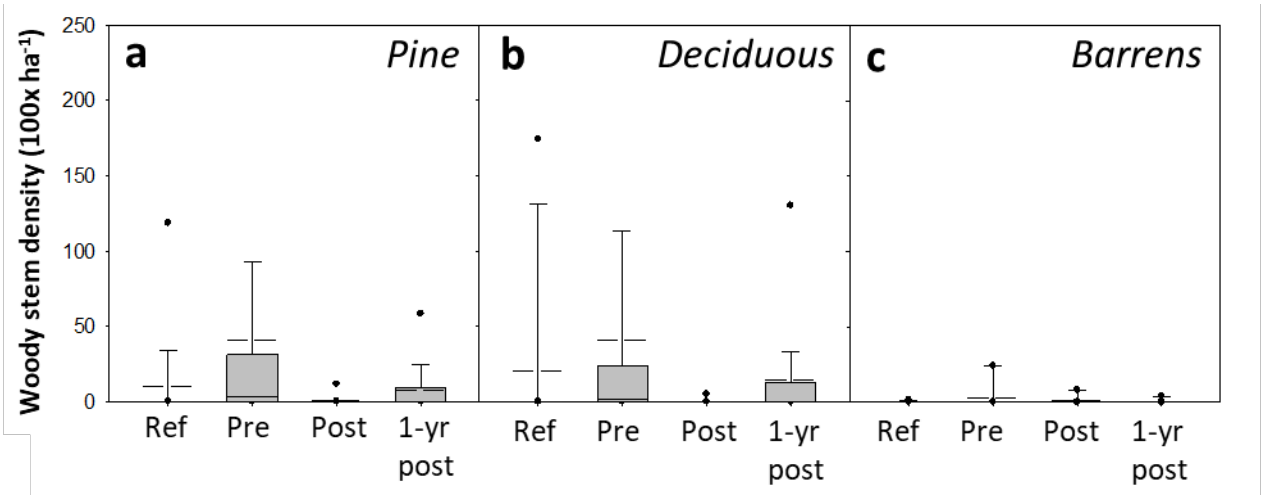


Figure 33. Distribution of tree sapling and shrub density (alive stems >0.5 m tall to ≤ 2.54 cm DBH) in reference areas and in managed areas pre, post 1-month, and post 1-year of a prescribed fire in 2016 by historical vegetation type. The boxplot solid and dashed line represent the median and mean, respectively, and the dots represent the 5th or 95th percentile.

Where eliminating or diminishing brush is a goal, brush treatments are an approach to modify the fuel load and soil heating effects. In our study, cutting brush and leaving it on site tended to decrease brush density 1-year after fire (Figure 34). Adding or piling fuels tended to reduce brush 1-year after fire as well. The opposite was true if the brush was cut then removed. Leaving cut brush or piling it on site increased the fuel load and likely increased soil heating and, consequently, damaged roots and buds, limiting the sprouting ability of woody stems. In terms of the prescription for the Moquah Barrens prescribed burn objectives to top kill 60-90% of the existing brush/shrub cover, our results suggest this was met immediately after fire and partially still met 1-year later. Continued monitoring will be needed to assess the impacts of the prescribed burn on the brush layer persist.

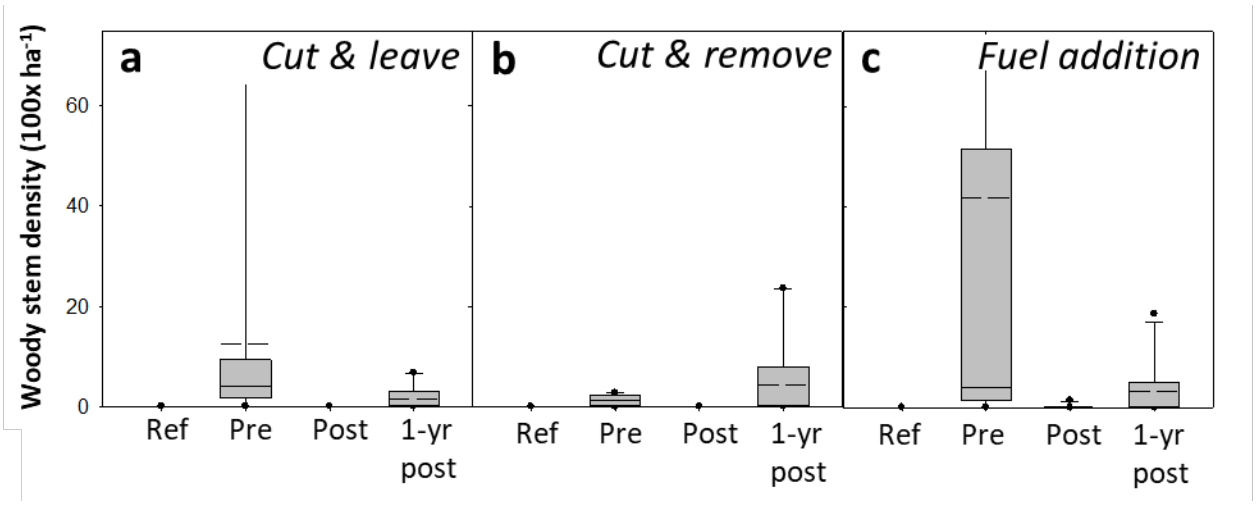


Figure 34. Distribution of tree sapling and shrub density (alive stems >0.5-m tall to <=2.54-cm DBH) in reference areas and in managed areas pre, post 1-month, and post 1-year of a prescribed fire in 2016 by fuel treatment. Stem density measured after cutting for that treatment. The boxplot solid and dashed line represent the median and mean, respectively, and the dots represent the 5th or 95th percentile.

Overstory Trees

Tree density (alive stems >11.4-cm DBH) was high in areas historically or currently typed as forest (Figure 35). Prescribed burns had minimal effects on density across vegetation type. Tree density was highest in the reference plots. In total, 7 tree species were identified (Table 12). Oak and pine were the most common species. Species found prior to fire were likely to be present after fire. White pine was the only species unique to references. Jack pine was the only species found in grasslands. Mortality was moderate for most species. In terms of the prescription goals for the prescribed fire at the Moquah Barrens, the objectives of <20% mortality of mature pine was met for red pine, but not for jack pine. However, mortality was highly variable such that the objective was likely met and unmet depending on location and context to fuel arrangement, forest structure, and fire behavior. Overall, prescribed fire effects on residual stand density and composition appear variable. Future analyses will test effects of fuel treatments on overstory mortality.

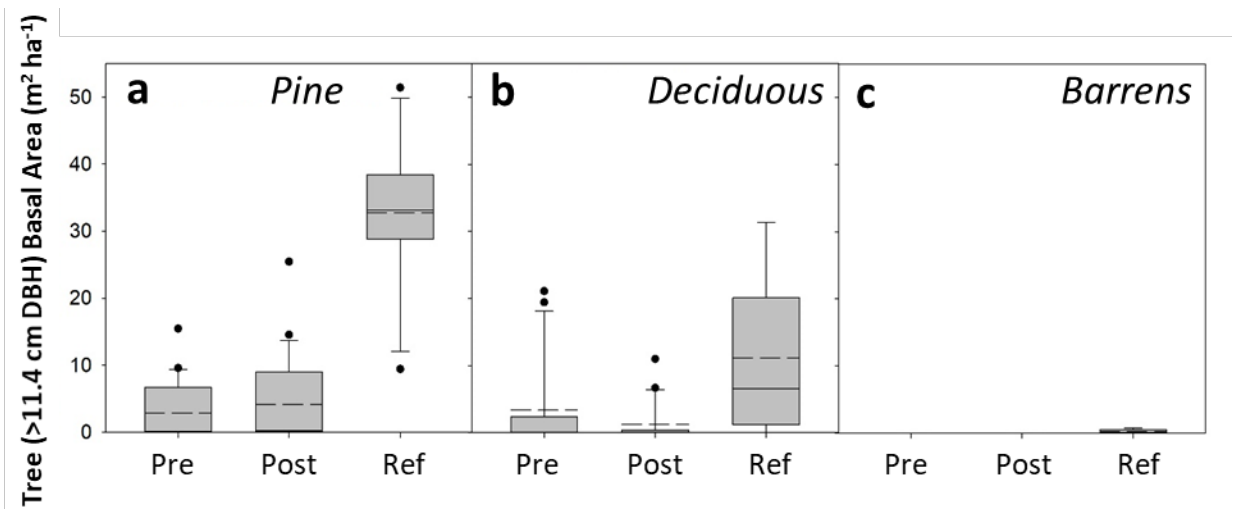


Figure 35. Distribution of tree density (alive stems >11.4 cm DBH) in reference plots and in burned plots pre, post 1-month, and post 1-year of a prescribed burn in 2016 by historical vegetation type. The boxplot solid and dashed line represent the median and mean, respectively, and the dots represent the 5th or 95th percentile.

Table 12. List of species by plot frequency (n) before and after the 2016 prescribed fire and in references areas, and association with historic vegetation type.

Species	Historic Vegetation	N plots			Basal Area (m^2 ha^{-1})			Mortality (% BA)
		Pre	Post	Ref	Pre	Post	Ref	
Acer rubrum	Pine, Decid	7	7	7	1.0 ± 0.5	0.2 ± 0.1	1.4 ± 0.5	49.1 ± 13.8
Betula papyrifera	Pine, Decid	4	4	2	1.6 ± 1.2	0.2 ± 0.2	0.9 ± 0.5	62.0 ± 11.6
Populus sp.	Pine, Decid	6	6	7	1.8 ± 1.0	0.5 ± 0.5	7.8 ± 3.8	64.7 ± 9.7
Quercus sp.	Pine, Decid	9	9	10	4.2 ± 1.6	2.1 ± 0.8	5.5 ± 3.1	57.4 ± 9.2
Picea glauca	Decid	1	1	0	1.1 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	85.8 ± n.d.
Pinus resinosa	Pine, Decid	10	10	11	5.3 ± 1.4	8.6 ± 2.5	27.3 ± 4.3	11 ± 9.9
Pinus banksiana	Pine, Decid, Barrens	9	9	6	2.7 ± 0.9	1.2 ± 0.6	3 ± 1.4	52.1 ± 14.0

Ground-layer vegetation

Over 50 species (or genera) were found at the study site (Table 13). These were largely herbaceous plants. Most plants were native; however, 6 non-native plants were found prefire and two 1-year postfire. Most species were found pre2018 fire (n=100) but the number of species found after fire was reduced to 61. Ferns were the most abundant (in percent foliage cover) growth habit, while non-native herbs, native herbs, and tree seedlings were low in mean abundance. Across historic vegetation type forestlands (coniferous or deciduous) had greater abundance of ferns and herbs than grasslands, while grasslands had a greater abundance of graminoids than forestlands (Figure 36). Prescribed burns had varying effects on the distribution of plant cover. Ferns tended to increase in cover postfire in deciduous forest and grasslands, but herbs had not clear pattern in any historic vegetation type.

All growth habits were present before and after fire, but the number of species populating each growth was reduced. Likely, species without fire adaptations (e.g., underground buds or root systems) or sensitive to fire were diminished. Future compositional analysis will examine trends of species and their traits.

Table 13. List of growth habits by number of species or genera and mean plant cover (%) before and after the 2018 prescribed burns and in references areas in the Moquah Barrens Management Area. The bottom sums the number of species/genera across growth habits and averages mean and standard errors across growth habits.

Growth Habit	**Species or genera (N)**			**Cover (%)**		
	Prefire	**Postfire**	**Reference**	**Prefire**	**Postfire**	**Reference**
Trees (≤ 0.5 m tall)	13	10	6	2.1 ± 0.5	7.0 ± 2.3	2.0 ± 0.5
Shrubs (≤ 0.5 m tall)	14	9	10	3.4 ± 1.2	7.6 ± 1.1	4.4 ± 1.5
Ferns	3	2	3	8.8 ± 7.0	9.9 ± 9.6	11.7 ± 10.2
Herbs	55	32	31	1.9 ± 0.4	3.6 ± 0.4	1.9 ± 0.5
Graminoids	9	6	6	3.5 ± 0.7	7.5 ± 1.0	6.0 ± 2.3
Non-native Herbs	6	2	0	1.7 ± 0.5	1.1 ± 0.1	0.0 ± 0.0
Overall sum species or genera	*100*	*61*	*56*			
Overall mean cover				*3.6 ± 1.7*	*6.1 ± 2.4*	*4.3 ± 2.5*

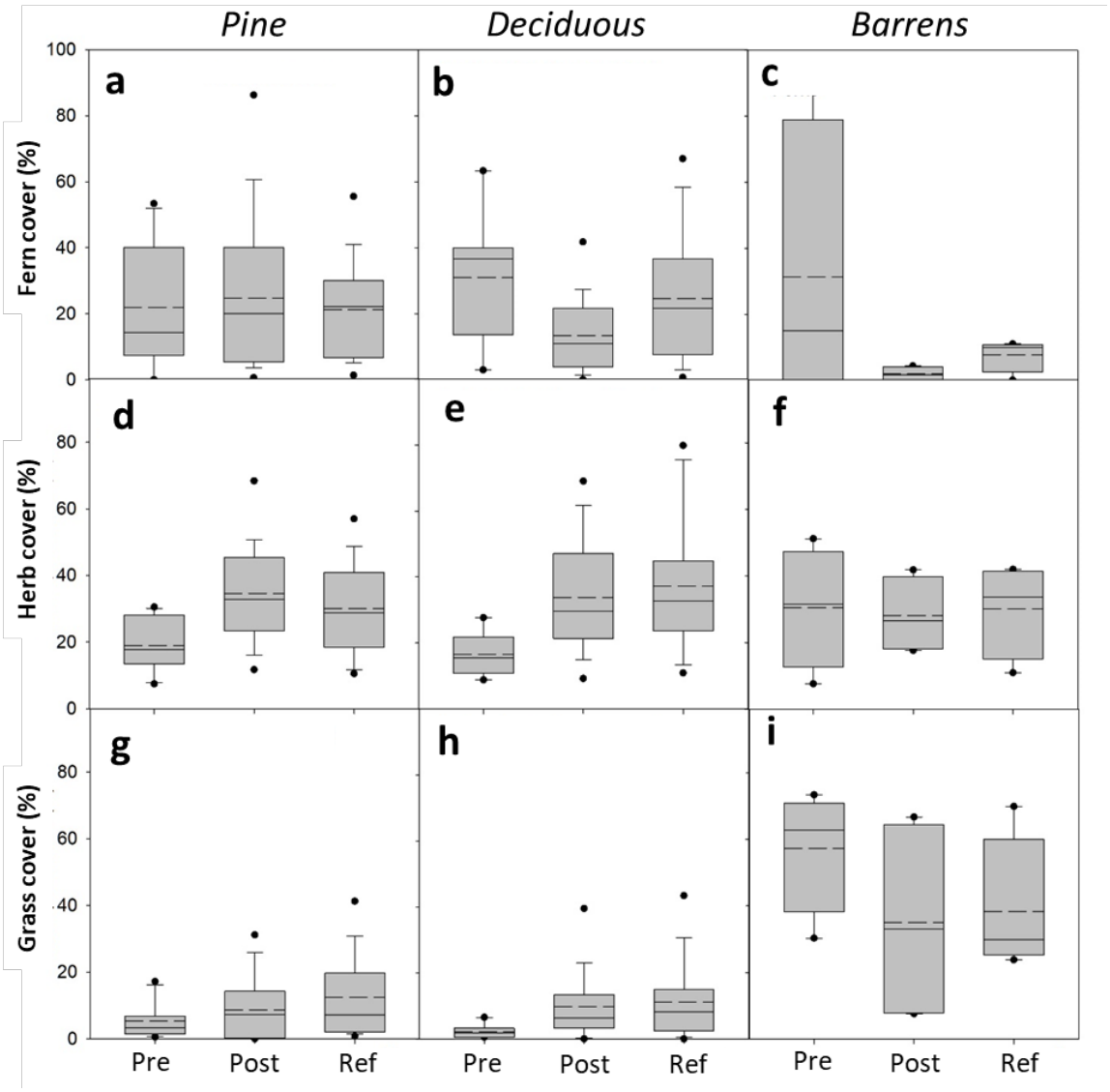


Figure 36. Distribution of ground-layer vegetation (top row: ferns, middle row: herbs, bottom row: grasses) in reference areas and in managed areas pre and post 1-year of a prescribed fire in 2018 by historical vegetation type. The boxplot solid and dashed line represent the median and mean, respectively, and the dots represent the 5th or 95th percentile.

Conclusions and Implications

Resolution of Study Objectives

Soil heating models are critical to understanding the belowground impacts of fire, since direct measurement is so time and cost intensive. A primary objective of our study was to field-validate two soil heating models (Campbell and HMV) available within the FOFEM decision-support system that had not been validated within the upper Midwest. Results indicated such models hold promise for application in the region and on sandy, droughty soils, but more research is needed to accurately predict soil heating effects. Our findings are consistent with field validations occurring within different regions such as the coastal plain of the Southeastern US (Kobziar et

al., 2019). Field measurements of soil heating indicated that soil heating was generally insulated by the duff layer during the late dormant season burns investigated, despite a generally thin duff layer (i.e., 2.2 cm overall average). Independent smoldering in the duff was rare if it occurred at all, such that below-ground heating was entirely contingent on aboveground fuel loads. Despite these system limitations, we observed biologically relevant belowground heating, particularly in locations where the duff was either thin or absent. After applying a thermal tolerance model, based on understanding that protein denaturation underlies cell and tissue injury and subsequent death (Rosenberg et al., 1971), our results suggest that impacts of soil heating regimes on soil microbial communities and other biota would be expected down to a depth at which 48 °C is reached when duration of heating is incorporated. Thus, the threshold for biological activity (60 °C) assumed by FOFEM may underestimate biological thresholds if the temporal duration of heating is not considered.

Our second objective was to evaluate the second-order effects of soil heating on ecosystem processes most relevant to pine woodland and open barrens restoration, organized across a series of interrelated questions:

How abundant are seeds after long periods of closed-canopy conditions?

The seed bank is a potential source for the restoration of species missing from native ecosystem species assemblages. Temporally, seed banks represent current and historical vegetation based on deposition that occurred at time scales ranging from months to decades. Spatially, seed banks accumulate from local seed dissemination by gravity and from long-distance dissemination by wind or animals. We evaluated the expression of the seed bank in an area with a long history of restoration activities across a variety of canopy conditions prior to prescribed burns. We found duff and upper mineral soil layers are indeed seed sources for common plants of the pine barrens, but our prefire results did not support the notion that it was severely impacted by historic or current closed-canopy conditions – either by plantation or native forest. However, seed bank response was both sparse and variable, so additional research may be necessary to fully address this question. Challenges associated with the study of seed banks are well-known (Keyser et al., 2012; Kobziar et al., 2019). Hence any data with respect to the seed bank – even if sparse – holds value for restoration purposes.

What level of soil heating creates negative impacts on barrens restoration by killing seeds of desired plant species?

We evaluated the influence of prescribed burns on the expression of the seed bank immediately postfire. We found no evidence that enhanced woody fuel loads impacted the native seed bank – either positively or negatively. Seed bank response appeared to be unrelated to burn severity or intensities as captured by the NPS severity assessment and paint tags, respectively. It is likely that the insulative properties of the duff served to protect the seed bank against damage from fire. Instead, results suggest vegetation type was the primary driver affecting the existing seed bank.

What is the ideal balance between fire intensities high enough to eliminate potential hardwood regeneration, but low enough to sustain seed banks for native barren communities?

The effects of the fuel treatments on hardwood regeneration were subtle; however, we found some evidence that brush cut-and-leave practices and experimental fuel additions tended to reduce brush density one-year postfire relative to ambient and brush removal treatments. Similar

results across these two treatments indicate soil heating related to fuel subsidies, rather than brush-cutting itself, elicited the response. This result is also consistent with results of the thermal tolerance model that indicated the heat generated by fuel subsidies were sufficient to heat the upper soil to temperatures capable of biological impacts. Yet, our postfire seed bank results suggest that there was no apparent trade-off between the health and vitality of the seed bank versus hardwood regeneration control during dormant season prescribed burns.

> *What level of soil heating is necessary for achieving short-term and sustained reductions in soil nutrient pools, which in turn help maintain a barrens ecosystem?*

Several soil nutrients were significantly affected by fire, but the effects of a single burn were minimal and usually ephemeral. More specifically, increased fuel loads greatly enhanced ash deposition (Quigley et al., 2019), and several nutrients including N and some important cations indicated pulses immediately postfire. For most nutrients these pulses were no longer evident by the 1-year postfire. There was, however, some evidence for greater loss of both PyC and N in the soil under added fuels. This result is striking, as fire is often reported to be a net source of PyC (Santín et al. 2015, Santín et al. 2015), and can contribute positively to soil nutrient status (Makoto et al., 2011; Verheijen et al., 2009). While the additional heat from fuel subsidies have the potential for duff consumption, the consumption of duff overall during our dormant season burns was generally low (i.e., 0.5 cm on average). Our data on soil heating suggests that the direct effects of fire were limited to the uppermost layers of soil only. Although fire can also influence soil nutrient dynamics indirectly via the effects of soil heating on fine roots or soil microorganisms, our measured and modeled soil heating data suggest that any such effects were likely highly localized and minimal, and we detected no significant changes in nutrient exchange rates as a function of temperature or burn severity. These results suggest the principle process affected soil nutrient pools in this study system is not soil heating *per se*, but rather the consumption of surface fuel loads that determines postfire ash loading (Quigley et al., 2019), and the subsequent translocation of ash residues through the upper soil layers. Although nutrient pulses generally would not be retained long within these and excessively well-drained soils, some soil properties (such as pH and hydraulic conductivity; Quigley et al. 2020, Quigley unpublished manuscript) increased with fire frequency (determined from CNNF burn treatment records) tended to increase soil properties such as pH (Quigley et al., 2020) and soil hydraulic conductivity (Quigley unpublished manuscript), whereas others decreased. For example, burned plots often had less duff thickness relative to unburned reference plots), indicating repeated burning can influence soil properties over time.

Our final objective was to evaluate the relative utility of different severity metrics for evaluating soil impacts after prescribed late dormant season burns. Most burn severity applications are focused on impact assessments following wildfire (Jain and Graham, 2007). We found that some indices, in our case the SPFI (Jain et al., 2012), are more suited for wildfire applications that often experience a greater range of fire intensities and resulting soil impacts. Although the fuel additions in the present study resulted in fire intensities occasionally hot enough to melt the aluminum paint tags (i.e., > 660 °C), the relatively high duff moisture content minimized duff consumption – an important component of the SPFI metric, which differentiate severity primarily based on loss of the organic horizon and exposure of the mineral soil. Thus, the SPFI was unable to differentiate within the relatively lower range of severity as occurred from the dormant season burns used in our study system. Under these circumstances, the NPS system (USDI National Park Service, 2003) adopted by FIREMON protocol (Lutes et al., 2006)

provided enough resolution at the lower end of the fire impact spectrum to correlate with processes such as fire intensity and fuel consumption. Such correlations did not translate into significant relationships between the NPS severity class and ecosystem responses, despite some interpretable trends in the seed bank and soil nutrients.

Implications of Results to Management and Policy

The present study represents the most comprehensive investigation of the soil heating process related to fire and its effects in the upper Midwest of the United States, and directly addresses some of the greatest uncertainties impeding our understanding of fire-related soil heating as a tool to accelerate restoration objectives. Our results from the matrix of vegetation cover types and sandy, droughty soils represented in our study at the Moquah Barrens have implications to future restoration activities within similar fire-prone ecosystems. Historically fire-prone regions of the Lake States were often associated with glacial outwash sand deposits such as those underlying the Moquah Barrens and the Northwest sands of Wisconsin (Cleland et al., 2004; Radeloff et al., 1999). Similarly fire prone regions in the Northeastern US are either associated with glacial deposits or the coastal plain (such as the New Jersey Pine Barrens). The physical process of soil heating we quantified is rooted in physics and therefore likely generalizable to many sandy soil ecosystems.

Results from our study may be best interpreted in the context of mesophication often associated with fire exclusion in the eastern United States (Nowacki and Abrams, 2008). Mesophication initiates with woody densification, increasing canopy closure, and the subsequent development of more nutrient-rich forest floor that both retains soil moisture and progressively degrades fuel quality necessary to carry fire. The Moquah Barrens Restoration Area is in initial stages of mesophication, where the primary issue is related to either hardwood encroachment or past pine plantation establishment. As such, the reintroduction of fire through prescribed burns is a logical solution to reverse this trend (Figure 37).

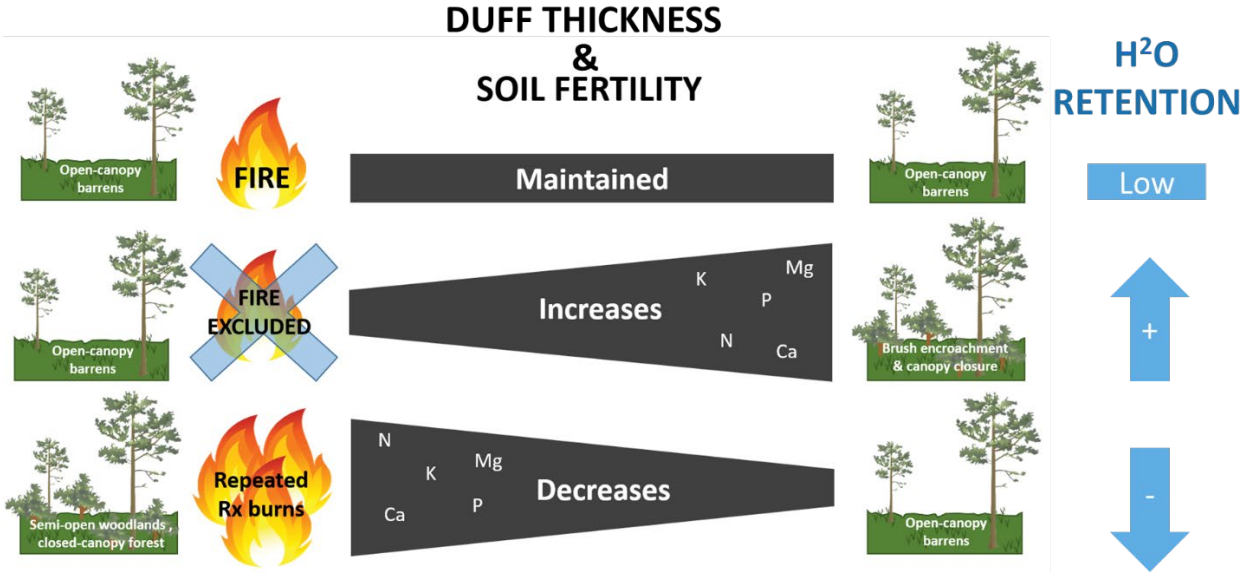


Figure 37. The role of fire in either maintaining or restoring barrens ecosystems by countering the cumulative effects of brush encroachment and consequent enrichment of the forest floor – known more broadly as mesophication. Adapted from Quigley et al. (2020).

Nowacki and Abrams (2008) suggest there is a tipping point within the mesophication process, beyond which reversal may be unrealistic. This tipping point may be more prevalent on richer soils where mesophication proceeds rapidly. Restoration of historically fire-prone systems on excessively-drained sandy soils – like the Moquah Barrens system – is more likely to respond favorably to the reintroduction of fire. However, several regional studies indicate that it is not fire application per se, but "*effective fire*" that enables achievement of restoration goals. Some studies in the Lake States indicate that dormant-season burns may not be effective for restoring native plant communities (Weyenberg and Pavlovic, 2014) for many of the same reasons our study demonstrated. Specifically, soil heating is limited due to the insulative properties of the characteristically moist duff layer during the spring dormant season. Reversing mesophication, therefore, is not as simple as fire versus no fire – effective burns matter.

Our results suggest repeated spring dormant season burns are countering mesophication within the Moquah Barrens Restoration Area to some degree. Although top-kill of hardwood regeneration is near complete, stem densities approach prefire densities within 1-year postfire, which is a trend observed elsewhere in the region (Buckman, 1964). The "one-two punch" of mechanical brush-cutting followed by burning has a detectable impact on stem densities 1-year postfire, but the impact was subtle, and monitoring is needed to determine longer term treatment effectiveness. Duff consumption was likewise minor overall, and changes to soil fertility were generally ephemeral. The primary influence of fuel subsidies on mineral soil was the subsequent ash loading (Quigley et al., 2019). Still, our analyses also suggest an important incremental effect of repeated burns – both soil pH and soil hydraulic conductivity increased with increasing fire frequency (Quigley et al., 2020, Quigley unpublished manuscript). The observed increase in hydraulic conductivity translates into less soil moisture retention and may be related in part to decreasing duff thickness that likewise may result from the cumulative effect of multiple burns. Given the insulative properties of the duff, the sum total of these different effects of burning work synergistically to reverse the effects of mesophication.

Prescribed burning is commonly applied during the spring dormant season within the Lake States region (Melvin, 2015), hence our dormant-season results will have broad relevance regionally. Fire application in the spring dormant season is simplified due to more continuous fuel beds, less smoke production, fewer conflicts with recreation, and availability of fire personnel during a relatively quiet period in wildfires at the national level. Such justification for spring burning is dominated by human, social, and logistical considerations rather than ecological criteria. There is growing recognition in the Lake States and elsewhere that growing season burns may have substantially stronger restoration impacts relative to those applied in the dormant season (Weyenberg and Pavlovic, 2014). Indeed, a classic study (Buckman, 1964) from nearby Minnesota demonstrated that growing season burns can be far more effective at reducing hardwood shrub and tree regeneration relative to spring dormant season burns, and recent remeasurements of that study suggest that frequent growing season burns had long-lasting impact of hardwood regeneration capacity (Scherer et al., 2018). Justified in large part by the results of the present study, the CNNF expanded its burn window and burned an entire burn block (block C) in July of 2018 and the lower half of Block F in July of 2019 (Figure 38).

Figure 38. Immediate results of a growing season burn applied to burn block C at the Moquah Barrens Restoration Area in July of 2018.

Opportunities for Direct Implementation by Users

The First Order Fire Effects Model is a valuable model in part because it's the only fire effects model of its kind and an integral part of existing and developing decision support systems (e.g., Forest Vegetation Simulator and IFT-DSS). Investing in model evaluation by research and development is greatly needed for improvement (Reinhardt and Dickinson, 2010). Recently we worked with the Missoula-based team managing FOFEM on short-term changes will allow us to better evaluate model performance for the Moquah Barrens. Specifically, two new soil types will be added to the suite available in FOFEM. These sandy soils are Quincy Sand from Campbell et al.'s original studies (Campbell et al., 1995, 1994), a high sand content soil, and sandy loam from the Moquah Barrens. Such improvements will significantly enhance applications in the Eastern US. Staff at the Missoula Fire Lab will further estimate the parameters needed to run the model in FOFEM. The FOFEM team is also modifying code to allow for more extensive batch processing and for generating more information-rich output files that will support evaluation. We encourage other studies to interact with FOFEM developers to better understand how their research might be integrated into model improvements to ultimately help guide fire management activities.

Much of our findings can be integrated by managers into decision-making within a monitoring and assessment context. For instance, burn severity metrics are often applied to prescribed burns to assess fire effectiveness towards a restoration goal. Burn severity metrics need to have the resolution to distinguish among ecologically meaningful soil impacts. However, many severity metric systems are designed for wildfire assessment that have less relevance to the range of fire intensities observed under prescribed burn conditions. We affirmed that the NPS substrate severity index recommended within the FIREMON system to be a useful indicator of fire intensity. In addition, our methods employing replicated square meter quadrats was both repeatable and rapid within a standard FIREMON plot monitoring system, and therefore amenable to an integrated fire effects monitoring program.

Semi-open forest systems ranging from woodlands to savannas and barrens, once commonplace across the Eastern US, are now exceedingly rare (Hanberry et al., 2020). The restoration management of semi-open systems requires a different mix of silvicultural approaches than traditional stand management (Bragg et al., 2020). Several outcomes from our research will help support such programs, such as the development of custom restoration fuel profiles related to mechanical brush-cutting, heavily thinned plantations, and characteristic barrens vegetation. Further, our results confirmed important plants within this system are retained in the seed bank even after multiple burns and mechanical treatments.

Implications for Future Research

The present study is part of a growing body of research investigating the physical process of soil heating from fire (Massman, 2015, 2012) and the biological and ecological effects of the heating process on belowground processes (Kolka et al. 2017. Yet we have only scratched the surface of these critical issues. Relative to the soil heating, a critical research need is to improve the combustion processes at the core of FOFEM. BURNUP algorithms generate the soil surface heating required by the Campbell and HMV soil heating models, but they clearly need evaluation and further development. Further, FOFEM appears to overestimate duff combustion on barrens systems. Moreover, neither FOFEM nor any other decision support tool contains a physically based duff combustion model, which is a glaring gap in capability for all ecosystems where duff is an important component of fuel beds (Reinhardt and Dickinson 2010). Improving the flaming and smoldering combustion models that support fire effects prediction is a tall order and will require concerted effort by physical scientists and engineers, in collaboration with ecologists, to integrate model development with laboratory and field measurement campaigns. Fortunately, improvements in the combustion foundation of FOFEM will flow broadly to other decision support systems.

The use of growing season burns as a restoration tactic is gaining significant traction in the Lake States region (Jack McGowan-Stinski, Lake States Fire Science Consortium, personal communication) as it is in other regions of the US (Waldrop et al., 1992; Zald et al., 2020). Wildfires burned in varying seasons and frequencies through presettlement forests of the Lake States (Guyette et al., 2016). Emulating processes and patterns of unmanaged forests, such as presettlement forest conditions, is one approach to restoration (Franklin et al., 2007). Yet, the practical application of burning during the growing season is often limited by unfavorable weather, unavailable personnel, and limited resources (Melvin, 2015). Future research should take advantage of the limited opportunities to apply prescribed fire during the growing season and evaluate its role in reaching restoration goals. Likewise, alternative methods of brush

control, such as herbicide applications, should be explored for efficacy relative to more traditional mechanical methods, both alone and in concert with fire application.

We did not detect an apparent trade-off between restoration objectives belowground with respect to surface fuel consumption and consequent soil heating during the dormant season. Yet there may be trade-offs between fuel loads, consumption, and top kill of residual trees. Restoration of semi-open systems such as woodlands, savanna, and barrens still benefit from the presence of old trees. The mortality we observed (data not shown) was highly variable, but substantial, with > 50% mortality of both the deciduous trees and of jack pine, whereas red pine was clearly more resilient (table Tree 1). Clearly, greater fuel loads contributed to large tree mortality, but in other cases it resulted from the context of fire tactics applied, local topography, and juxtaposition with respect to surrounding heat sources and fire behavior. Future research should investigate the resilience of old, open-grown pines relative to recently thinned plantation pines to different burning tactics. Likewise, burning within the residual deciduous forest may be converting them to more extensive young brush that is far more resilient to repeated burning. Such issues reflect some of the unique challenges underlying the restoration of semi-open forest conditions that were far more prevalent across many ecosystems across the nation. These challenges are the subject of a recent virtual special issue due for release in the journal Forest Ecology and Management within the next month (Sturtevant and Hanberry, unpublished manuscript) inspired in part by the present study.

One of the more frequent issues raised by land management agencies charged with ecosystem restoration is that structural restoration (e.g., tree density, size, spatial arrangement) is relatively easy to achieve through traditional mechanical and fire applications, but compositional restoration – particularly of the ground-layer plants– remains a fundamental challenge (Paul Strong, CNNF Forest Supervisor, personal communication). Structural restoration often meets important goals for wildlife habitat and recreation opportunities. Yet many important ecosystem services, such as berry production, pollinator resources, and biodiversity, are tied more directly to the composition of the vegetation responding to restoration treatments. The seed bank investigation of the current project is an important step in the right direction, but there is still so much we do not know about seed sources and their interactions with species germination responses and fire. Future studies could use more detailed soil heating measurements at the seed scale and/or use alternative field and greenhouse methods that could reveal more details and trends with prescribed fire and other management tools on soil seed banks. Future studies could also focus on the effects of prescribed fire soil heating on other management approaches, such as direct seeding, planting, and translocations of plant species, to meeting target compositional goals. While blueberries and related species are of substantial cultural and social values for the Lake States more broadly, and barren systems in particular – interactions between fire and berry production have received relatively little in the Lake States region. Bracken fern (*Pteridium aquilium*) is well known for its aggressive competitive ability, particularly within woodland and barren/heath ecosystems (Marrs et al., 2000). In some parts of the world, fire appears to enhance dominance by closely-related bracken fern species (e.g., Hartig and Beck, 2003), but it is an open question whether either fire-related soil heating or season of burn have any influence on the competitive ability of bracken fern. Finally, while low-soil fertility is a hallmark of barren ecosystems, there are commonly associated species, such as sweet fern, that fix nitrogen. The importance of such traits on the ecosystem function of barrens systems is poorly understood.

Land managers are frequently presented with an array of potential solutions to vexing ecological restoration problems. Choices are often made based on what is fiscally possible, known to be previously effective, or what is familiar to use. The mission of management does not typically allow for experimentation or alternative methods or if it does, the results of this experimentation are often not captured and shared with other land managers thereby creating a knowledge gap. Conversely, research organizations rarely have the resources and expertise necessary to carry out large scale burn operations that are often required for multidisciplinary fire research. A research-management partnership is mutually beneficial because it can help answer complex restoration questions, explore new methods for achieving results, and capture this knowledge gain for dissemination to the greater research-management community. This was this case with the present research-management partnership, in that it resulted in answers to ecological questions that will guide future management decisions at the Moquah Barrens, documented the results of this research for application by other land managers, and generated more research questions to be explored that will further hone future management actions. The value of a research-management partnership cannot be overstated. This partnership, termed co-production, has been highlighted in a recent push to increase prescribed fire acres nationwide for habitat and hazardous fuels management (Hiers et al., 2020).

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Appendix B List of Deliverables

Peer-reviewed Journal Articles

Quigley K.M., B.W. Wildt, B.R. Sturtevant, R.K. Kolka, M.B. Dickinson, C.C. Kern, D.M. Donner, and J.M. Miesel. 2019. Fuels, vegetation, and prescribed fire dynamics influence ash production and characteristics in a diverse landscape under active pine barren restoration. Fire Ecology 15:5. DOI: 10.1186/s42408-018-0015-7

Quigley, K.M., Kolka, R., Sturtevant, B.R., Dickinson, M.B., Kern, C.C., Donner, D.M. and Miesel, J.R., 2020. Prescribed burn frequency, vegetation cover, and management legacies influence soil fertility: Implications for restoration of imperiled pine barrens habitat. Forest Ecology and Management, 470 DOI: 10.1016/j.foreco.2020.118163

Quigley, K.M., R. Kolka, B.R Sturtevant, M. Dickinson, C.C. Kern, and J.R. Miesel. In review. Prescribed fire frequency and fuels influence soil hydraulic conductivity of sandy, glacial outwash soils. Science of the Total Environment

Manuscripts in-prep

Quigley, K.M., R. Kolka, B.R Sturtevant, M. Dickinson, C.C. Kern, and J.R. Miesel. In prep. Immediate and delayed response of soil nutrient dynamics to prescribed fire in an intensively managed barrens-forest mosaic. Planned submission to *Geoderma*.

Dickinson, M.B. et al., In prep. Wildland fire radiation measurements ignore emissions from hot gasses and underestimate total radiation. Planned submission to *Journal of Geophysical Research*.

Technical Reports

Sturtevant, B.R., J.R. Miesel, M.B. Dickinson, R.K. Kolka, C.C. Kern, D.M. Donner, K.M Quigley, and M.M. Bushman. 2019. Manipulating soil heating patterns to optimize barrens restoration. Final Report, Joint Fire Science Program 15-1-05-13 (THIS REPORT).

Conference or Symposium Proceedings

Miesel, J., Kolka, R.K., Sturtevant, B., Adkins, J. and Quigley, K., 2019, January. How Well Do Burn Severity Metrics Translate into Fire Effects on Biogeochemical Processes in Forest Soil? In SSSA International Soils Meeting. ASA-CSSA-SSSA.

Conference or Symposium Abstracts

Miesel, J., J. Adkins, K. Quigley, M. Dickinson, R. Kolka, B. Sturtevant, K. Docherty, and J. Gutknecht. 2019. What does severity mean for the below-ground ecosystem? Investigating patterns in soil nutrient pools and dynamics across gradients of burn severity. Association for Fire Ecology International Fire Ecology and Management Congress. Tucson, AZ.

Quigley, K.M., J.R. Miesel, R.K. Kolka, and B.R. Sturtevant. 2019. Restoring Wisconsin's critically imperiled pine barrens: how does prescribed fire alter soil properties linked to aboveground communities? North American Forest Ecology Workshop. Flagstaff, AZ. 24 June 2019.

Sturtevant, B.R., M.B. Dickinson, C.C. Kern, K.M. Quigley, J.R. Miesel, R.K. Kolka, and DM Donner. 2019. Woody fuel heat subsidies: Can we accelerate pine woodland and barren restoration during dormant season burns. North American Forest Ecology Workshop. Flagstaff, AZ. 24 June 2019.

Sturtevant, B.R., K.M. Quigley, and B.B. Hanberry. Processes underlying restoration of temperate savanna and woodland ecosystems – A Summary/Synthesis. North American Forest Ecology Workshop. Flagstaff, AZ. 24 June 2019.

Miesel, J., J. Adkins, K. Quigley, B. Sturtevant, and R. Kolka. 2019. Post-fire biogeochemical processes in forest soil: how important is burn severity? Soil Science Society of America Annual Conference. San Diego, CA.

Sturtevant, B.R. R.K. Kolka, J.R. Miesel, A. Singh, P.A. Townsend, C.C. Kern, D.M. Donner, and K. Quigley. Quantifying substrate fire severity: Issues of timing, scale, and data aggregation. 7th International Fire Ecology and Management Congress, Association for Fire Ecology, Orlando, FL, November 28-December 2, 2017 (abstract, pubs 56, 81).

M.B. Dickinson, B.R. Sturtevant, R.L. Kremens, J.R Miesel, C.C. Kern, and R.K. Kolka. Fire Behavior and soil heating during dormant-season fires in a barrens ecosystem under restoration: duff, the great insulator. 7th International Fire Ecology and Management Congress, Association for Fire Ecology, Orlando, FL, November 28-December 2, 2017 (abstract).

C.C. Kern, D. Donner, M. Bushman, R. Kolka, J. Miesel, M. Dickinson, and B. Sturtevant. Response of seedbank composition to soil burn severity across a range of pine barren restoration phases. 7th International Fire Ecology and Management Congress, Association for Fire Ecology, Orlando, FL, November 28-December 2, 2017 (abstract).

Quigley, K., B. Sturtevant, J. Miesel, R. Kolka, M. Dickinson, C. Kern, D. Donner. Burn severity estimates in relation to post-fire soil and ash properties. 7th International Fire Ecology and Management Congress, Association for Fire Ecology, Orlando, FL, November 28-December 2, 2017 (abstract).

Posters

Kolka, R., K. Quigley, J. Miesel, B. Sturtevant, and T. DeSutter. Prescribed fire effects on forest floor and upper mineral soil mercury concentrations and stocks during barrens restoration. Soil Science Society of America, San Antonio, TX. November 10-13, 2019.

Ashley-Grose, E. K Quigley, and J. Miesel. How does prescribed fire influence soil carbon storage in critically imperiled pine barrens? Michigan State University Undergraduate Research and Arts Forum, East Lansing, MI. April 05 2019.

Wildt, R., K. Quigley, and J. Miesel. How do fire history and vegetation cover influence soil hydraulic conductivity? Michigan State University Undergraduate Research and Arts Forum, East Lansing, MI. April 18 2018.

Wildt, R., K. Quigley, and J. Miesel. Quantifying ash characteristics after a prescribed burn in the Chequamegon-Nicolet National Forest .7th International Fire Ecology and Management Congress, Orlando, FL. November 28 – December 2, 2017.

Field demonstration/tour summaries

2018 and 2017 – USFS National Advanced Silviculture Program (NASP) field tour

2016 – Lake States Fire Science Consortium Moquah Barrens field tour

Website development

https://www.mieselecologylab.org/gallery.html

Presentations/webinars/other outreach/science delivery materials

Sturtevant, B., K. Quigley, M. Dickinson, C. Kern, R. Kolka, and J. Miesel. Pine woodland and barren restoration: What is possible with late dormant season burns? Fire in Minnesota Ecosystems, Sustainable Forests Education Cooperative, University of Minnesota, Cloquet, MN. Jan 2020.

Sturtevant, B., K. Quigley, M. Dickinson, C. Kern, R. Kolka, and J. Miesel. Pine woodland and barren restoration: What is possible with late dormant season burns? Lake States Fire Science Consortium Webinar Series. Archived Online Webinar. 18 April 2019. http://lakestatesfiresci.net/webinar_04_18_19.html

Quigley, K., and N. Emery. Prescribed fire presentation. MSU SciFest, East Lansing, MI. April 6, 2019.

Miesel, J. Co-organizer and presenter, Careers in Wildland Fire. In partnership with US Fish & Wildlife Service. Michigan State University, East Lansing, MI. February 17, 2019.

Miesel, J., K.M. Quigley, B. Sturtevant, R. Kolka, M. Dickinson, C. Kern, and D. Donner. 2019. Manipulating soil heating patterns to optimize barren restoration. Fifth Annual Burning Issues Workshop. Battle Creek, MI. 05 February 2019.

Quigley, K. Public presentation and discussion on prescribed fire. Science on Tap, Ashland, WI. June 18, 2018.

Quigley, K., and J. Adkins. Hands-on activity demonstrating soil respiration measurements. MSU SciFest, East Lansing, MI. April 7, 2018.

Miesel, J., B. Sturtevant, C. Kern, and M. Dickinson (organizers) Special Symposium: Connecting direct and indirect measures of soil heating to first- and second-order fire effects using wildfire, prescribed fire, and laboratory investigations. 7th International Fire Ecology and Management Congress, Association for Fire Ecology, November 28-December 2, 2017.

Instructor (Sturtevant) – Fire and fuels monitoring workshop. Lake States Fire Science Consortium workshop series. June 18-20, Cloquet, MN.

Sturtevant, B., and K. Quigley (organizers) Special Session: Processes underlying restoration of temperate savanna and woodland ecosystems. North American Forest Ecology Workshop. Flagstaff, AZ. June 24, 2019.

Sturtevant, B.R. Using prescribed fire to restore Wisconsin's imperiled pine barrens. Trees for Tomorrow, "Nature at Night" seminar series, Eagle River, WI. July 9, 2019.

Miesel, J.R. Restoring historically 'open' ecosystems with prescribed fire: What is the role of soil heating? Department of Environmental Sciences, University of Toledo. Toledo, MI. April 3, 2019.

Moquah Soil Heating Study, Chequamegon-Nicolet National Forest Brown-bag Seminar Series, Rhinelander WI, April 4, 2018.

- Sturtevant, B.R., C.C. Kern, R.K. Kolka, M.B. Dickinson, D.M. Donner, J. Miesel, K. Quigley, M. Bushman, J. Rabuck, V. Hazelton, D. Hinson, B. Heeringa, M. Davalos. Manipulating soil heating patterns to optimize barrens restoration.
- Dickinson, M.B., B.R. Sturtevant, C.C. Kern, R.K. Kolka, K. Quigley, J. Miesel. Fire Behavior and soil heating during late dormant-season fires.
- Quigley, K., B. Sturtevant, J. Miesel, R. Kolka, M. Dickinson, C. Kern, D. Donner. Burn severity estimates in relation to post-fire soil and ash properties.
- C.C. Kern, D. Donner, M. Bushman, R. Kolka, J. Miesel, M. Dickinson, and B. Sturtevant. Hardwood sprouting & seed bank

Quigley, K. Equipment demonstration showing how to measure soil greenhouse gas emissions. Northern Great Lakes Visitor Center. Ashland, WI. Aug 23, 2017.

Quigley, K., and R. Wildt. Presentation on fire science to girls in STEM, sponsored by the Graduate Women in Science. March 4, 2017.

Sturtevant, B.R., C. Kern, and D. Donner. Restoration of fire-dependent pine barren ecosystems in northern Wisconsin – Bridging the gap between research and management practices. Lake States Fire Science Consortium webinar series. November 17, 2016. http://lakestatesfiresci.net/webinar_11_17_16.html

Appendix C Metadata

Data and associated metadata for all research elements (plot installation, soil heating, fuel, severity assessment, soil properties, seedbank, and vegetation) were recorded digitally using a series of Microsoft Excel databases under each respective research element. Data were recorded on paper field datasheets that were digitally scanned and archived. Data were then digitized into Microsoft Excel format, and examined using standard methods to ensure quality control, where metadata are embedded in the same database as the data, with references to the associated scanned field data files. We have a plan in place to submit our data and associated metadata to the USDA Forest Service Research Data Archive, following the standards set by the Federal Geographic Data Committee: Content Standard for Digital Geospatial Metadata and the Biological Data Profile. (https://www.fs.usda.gov/rds/archive/metadata).