FINAL REPORT

Identifying Indicators of State Change in Alaskan Boreal Ecosystems: Testing Previous Hypotheses and Conclusions with Long-term Data

SERDP Project RC-2754

OCTOBER 2019

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REPORT DOCUMENTATION PAGE

Form Approved OMB No. 0704-0188

1. REPORT DATE *(DD-MM-YYYY)*
10-30-2019

2. REPORT TYPE
SERDP Final Report

3. DATES COVERED *(From - To)*
08-04-2017 – 08-04-2019

4. TITLE AND SUBTITLE
Identifying indicators of state change in Alaskan boreal ecosystems: Testing previous hypotheses and conclusions with long-term data

5a. CONTRACT NUMBER
17-C-0049

5b. GRANT NUMBER

5c. PROGRAM ELEMENT NUMBER

5d. PROJECT NUMBER
RC-2754

5e. TASK NUMBER

5f. WORK UNIT NUMBER

6. AUTHOR(S)
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8. PERFORMING ORGANIZATION REPORT NUMBER
RC-2754

9. SPONSORING / MONITORING AGENCY NAME(S) AND ADDRESS(ES)
Strategic Environmental ResearDP Research and Development Program
4800 Mark Center Drive
Suite 16F16
VA 22350-3605

10. SPONSOR/MONITOR'S ACRONYM(S)
SERDP

11. SPONSOR/MONITOR'S REPORT NUMBER(S)
RC-2754

12. DISTRIBUTION / AVAILABILITY STATEMENT
DISTRIBUTION STATEMENT A. Approved for public release: distribution unlimited.

13. SUPPLEMENTARY NOTES

14. ABSTRACT
We collected, analyzed and synthesized additional long-term data on the resilience and vulnerability of Alaskan boreal forests to ecosystem state change following wildfire or fuel reduction management. This research revisited site networks established during the completed SERDP project RC-02109, which focused on identifying factors controlling ecosystem vulnerability to state change after disturbance in black spruce (*Picea mariana*) forests, the vegetation type that dominates Department of Defense (DoD) lands in Interior Alaska. At the time of initial measurement, sites were relatively recently disturbed (<10 years after disturbance events). Our overarching objective was to determine whether the mechanistic links among fire, soils, permafrost, and vegetation succession predictive of state change immediately following disturbance remain robust predictors over longer timescales. Our data confirm that managers can use early post-fire data to predict vulnerability to ecosystem state change after wildfire because tree seedling regeneration and soil temperature regime are entrained soon after fire. In fuel management treatments, however, the prolonged window of conifer recruitment will require longer-term (e.g., decadal) observations of composition to predict successional trajectory.

15. SUBJECT TERMS
Resilience, vulnerability, boreal forest, wildfire, fuel management, permafrost

16. SECURITY CLASSIFICATION OF: U			17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON
a. REPORT	**b. ABSTRACT**	**c. THIS PAGE**	UU	38	Michelle C. Mack
U	U	U			**19b. TELEPHONE NUMBER** *(include area code)* 928-523-9415

Standard Form 298 (Rev. 8-98)
Prescribed by ANSI Std. Z39.18

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Abstract

Introduction and Objectives: We collected, analyzed and synthesized additional long-term data on the resilience and vulnerability of Alaskan boreal forests to ecosystem state change following wildfire or fuel reduction management. This research revisited site networks established during the completed SERDP project RC-2109, which focused on identifying factors controlling ecosystem vulnerability to state change after disturbance in black spruce (*Picea mariana*) forests, the vegetation type that dominates Department of Defense (DoD) lands in Interior Alaska. At the time of initial measurement, sites were relatively recently disturbed (<10 years after disturbance events). Our overarching objective was to determine whether the mechanistic links among fire, soils, permafrost, and vegetation succession predictive of state change immediately following disturbance remain robust predictors over longer timescales.

Technical Approach: In 2017, we remeasured tree regeneration across a regional wildfire network of 90 sites that burned in the record 2004 wildfire year, which we established in 2005 and re-measured in 2006, 2008 and 2011. In 2018, we remeasured tree regeneration and permafrost degradation across 36 fuel management network sites that we established in 2012. Over the two years of the project, we continuously monitored permafrost soil temperature regimes in three paired burned and unburned sites that were established in 2013.

Results: Resampling tree seedling regeneration in our wildfire network showed that spruce seedling density in 2006 was the strongest predictor of spruce density in 2017. Out of the fire, environment, and biotic drivers that we examined, only pre-fire spruce density had a small, positive, direct effect on spruce seedling density in 2017, likely indicating that higher density stands sustained seed rain over a longer post-fire period than low density stands. For deciduous tree seedlings, density in 2006 similarly explained most of the variation in density in 2017. However, organic layer depth had a modest, negative, direct effect on density in 2017, suggesting that deeper organic layers may have suppressed episodic recruitment of deciduous seedlings. Across our soil temperature network, we found that most of the increase in soil temperature and permafrost degradation occurred immediately following fire, with site attributes affecting overall warming to a greater degree than time after fire. All sites showed a trend towards increasing soil temperature over time, showing that fire-induced changes in soil temperature are proceeding against a background of regional warming. Across the fuel management network, we found persistent conifer recruitment across the first decade after treatment. Although deciduous tree seedlings dominated in severely disturbed (shearbladed) treatments, recruitment of conifers, likely from surrounding undisturbed stands, established mixed composition stands that will likely undergo complex successional trajectories in the future. Active layer depth and permafrost degradation increased over time: this was large in the shearbladed and small in the thinned treatments. New measurements of fire rate of spread (ROS) indicated that the initial benefits of reduced ROS diminished to pre-treatment levels by the first decade after treatment. Additionally, shearblading of black spruce-lichen woodlands increased ROS because vegetation composition shifted towards plant species with a high abundance of fine fuels.

Benefits: Our data confirm that managers can use early post-fire data to predict vulnerability to ecosystem state change after wildfire because tree seedling regeneration and soil temperature regime are entrained soon after fire. In fuel management treatments, however, the prolonged window of conifer recruitment will require longer-term (e.g., decadal) observations of composition to predict successional trajectory. New measurements showed that fuel management effects on ROS returned to prefire levels about a decade after treatment. Because shearblading treatments increase ROS in open canopy black spruce woodlands, these ecosystems are not effective candidates for fuel brakes using this method.

Executive Summary

Introduction: We collected, analyzed and synthesized additional long-term data on the resilience and vulnerability of Alaskan boreal forests to ecosystem state change following wildfire or fuel reduction management. This research revisited site networks established during the completed SERDP project RC-2109, which focused on identifying factors controlling ecosystem vulnerability to state change after disturbance in black spruce (*Picea mariana*) forests, the vegetation type that dominates Department of Defense (DoD) lands in Interior Alaska. At the time of initial measurement, sites were relatively recently disturbed (<10 years after disturbance events). We revisited sites to determine whether previous conclusions are consistent with findings 15-20 years after disturbance.

Objectives: At the time of initial measurement, sites were relatively recently disturbed (<10 years after disturbance events). Our overarching objective was to determine whether the mechanistic links among fire, soils, permafrost, and vegetation succession predictive of state change immediately following disturbance remain robust predictors over longer timescales. Our specific objectives were to: (1) Determine the long-term effects of wildfire severity on successional trajectories of tree dominance and permafrost degradation; (2) Determine the long-term effects of fuel management treatments on successional trajectories of tree dominance and permafrost degradation; and (3) Determine the long-term effects of climate and wildfire on permafrost soil temperature regimes.

Technical Approach: In 2017, we remeasured tree regeneration across a regional wildfire network of 90 sites that burned in the record 2004 wildfire year, which we established in 2005 and re-measured in 2006, 2008 and 2011. We used structural equation modeling (SEM) to test causal hypotheses and determine whether causal pathways differed between 2006 and 2017. Across the fuel management network established in 2012, we remeasured tree regeneration and permafrost degradation and made a new measurement of fuel rate of spread (ROS). We used a information-theoretic mixed modeling approach to examine change over time in these treatments. Over the two year of the project, we used soil temperature dataloggers to monitor permafrost soil temperature regimes in paired burned and unburned sites that were established in 2013. We used non-linear regression techniques in a mixed modeling framework to predict temperature with depth and analyze profile time series.

Results and Discussion: In our previous research, results from our wildfire network showed that severity of soil organic layer disturbance was the most important indicator of state change because of its impacts on tree species composition and permafrost degradation. High severity burning of the soil organic layer exposed mineral seedbeds, which were preferentially colonized by rapidly growing deciduous tree seedlings, shifting initial successional trajectories from spruce self-replacement to deciduous tree dominance, a novel state for black spruce forests. Results from revisiting these sites showed that spruce seedling density in 2006 was the strongest predictor of spruce density in 2017 (Figure IA). Out of the fire, environment, and biotic drivers that we examined, only pre-fire spruce density had a small, positive effect on spruce seedling density in 2017 that was independent from its effect on seedling density in 2006, probably indicating that higher density stands sustained seed rain over a longer post-fire period than low density stands. For deciduous tree seedlings, density in 2006 similarly explained most of the variation in density in 2017. However, organic layer depth had a modest, negative direct effect on density in 2017, suggesting that deeper organic layers may have suppressed episodic recruitment of deciduous seedlings (Figure IB). These findings support our previous conclusions and add nuance for understanding biotic and abiotic controls that emerge in the decade after fire.

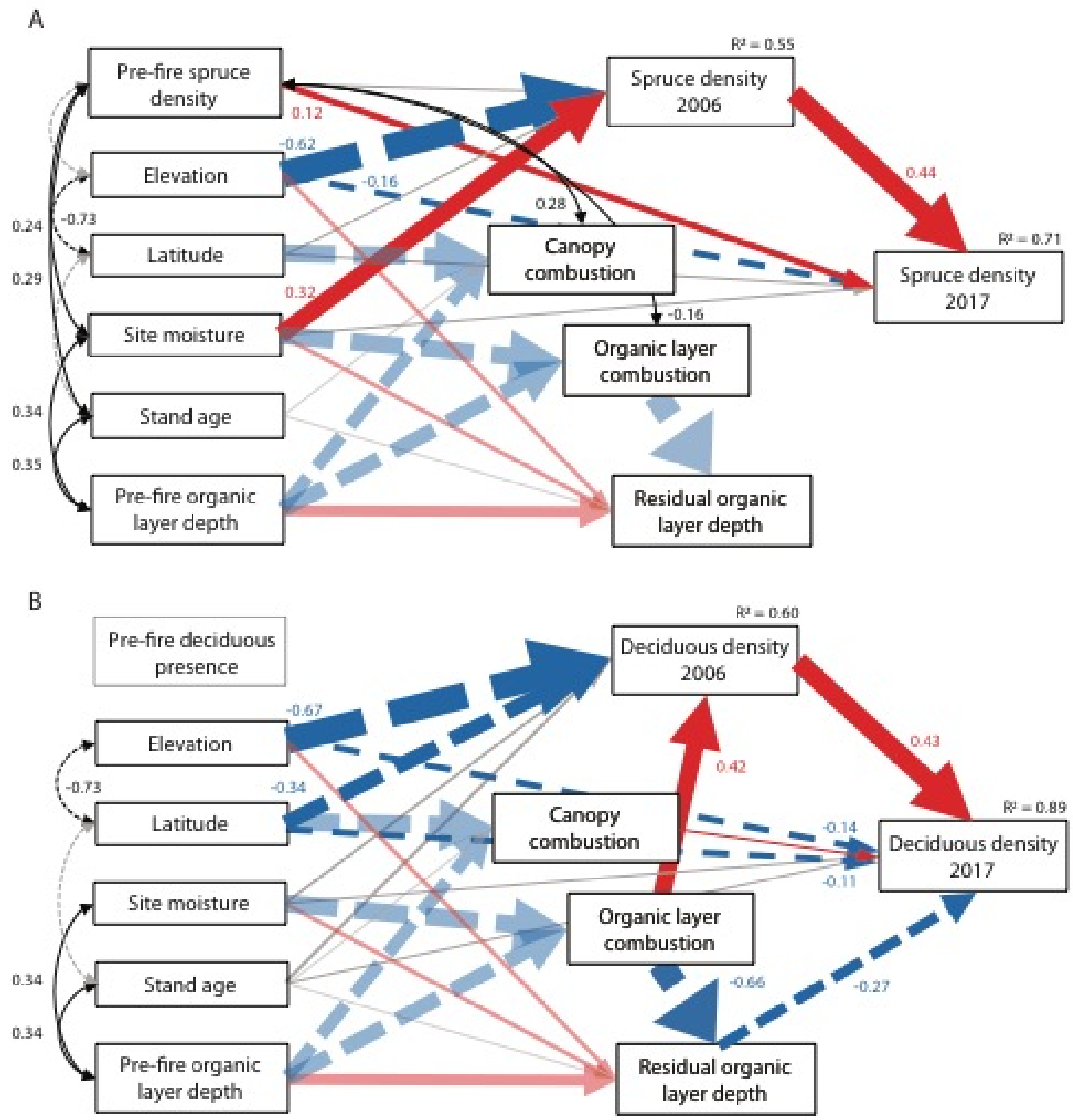

Figure I. *Structural equation model (SEM) results for models of post-fire seedling densities (log-transformed) of a) black spruce and b) deciduous trees. The model for black spruce was simplified based on testing of nested models. In the deciduous model, the presence of pre-fire deciduous trees had no significant effects and was removed from the final model. Paths affecting fire are shown with transparency to emphasize drivers of seedling responses. Directed causal paths are shown as straight lines with single-headed arrows; undirected correlations among variables are shown as curved, double-headed arrows. All paths in the model are illustrated. Line color and thickness indicate the magnitude and direction of effects, with standardized path coefficients shown next to significant paths (red = positive effects, blue + dashed = negative effects, grey = not significant [P>0.05]). Tests of SEM fit indicate no significant differences between the illustrated model structure and actual data (p>0.05).*

Our past research showed that burning reduced ground insulation, leading to deepening of the active layer, permafrost degradation, and in some cases, surface subsidence and changes in site drainage. Across our soil temperature network, we found that most of the increase in soil temperature and degradation of permafrost occurred immediately following fire, with site attributes affecting overall warming to a greater degree than time after fire (Figure II). All unburned sites showed a trend towards increasing soil temperature over time, showing that fire-induced changes in soil temperature are proceeding against a background of regional warming (Figure II).

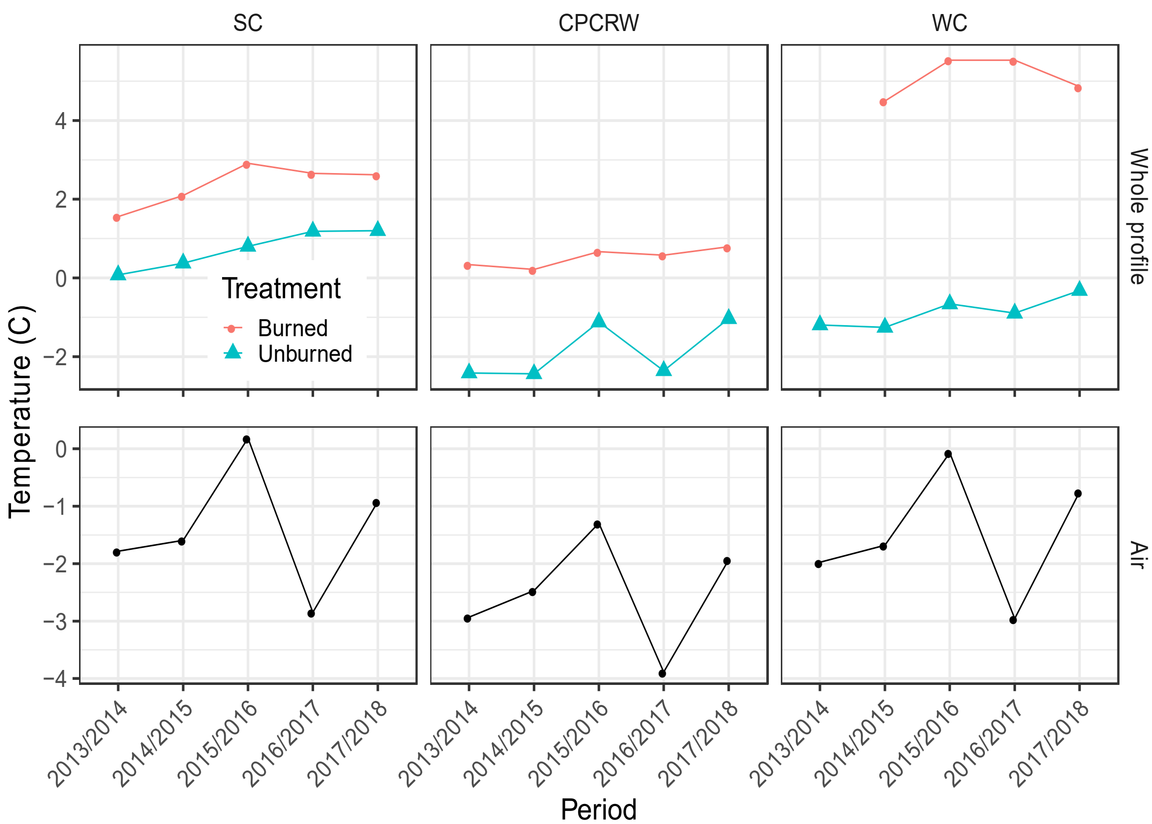

Figure II. *Mean annual (October to September of the following calendar year) temperature for the whole soil profile (5 to 150 cm) at each site and burn treatment. Means were weighted by temperature probe depth to account for disparity in profile depth increments. Air temperatures for CPCRW and WC are from meteorological stations at the sites; SC air temperatures are from Eielson Airforce Base airport, Alaska.*

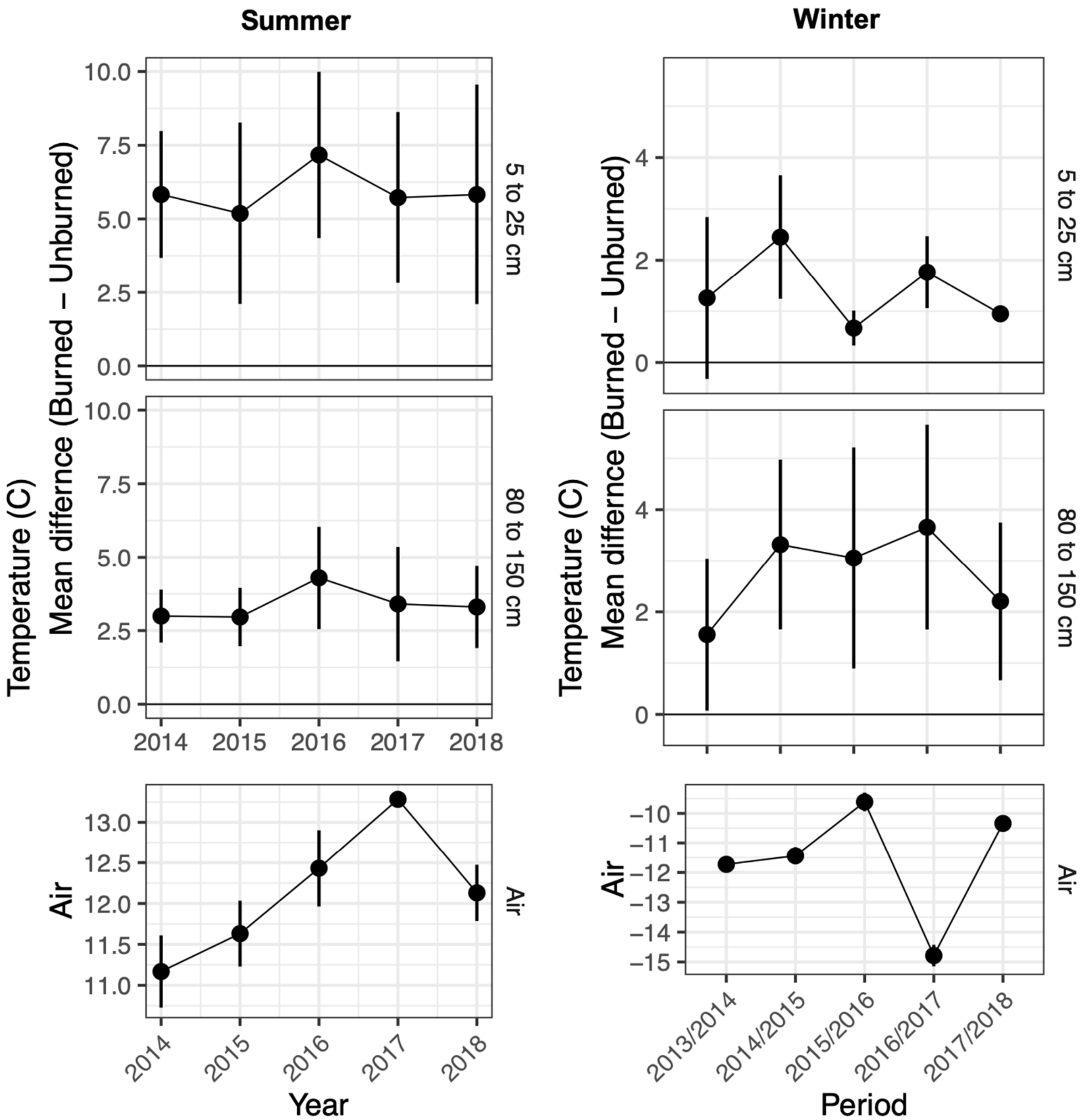

Figure III. *Mean temperature differences (burned-unburned) for shallow (5-35 cm) and deep (80-150 cm) soils. Temperatures for each soil profile are averaged across depths and sites. (left panel) Summer: May-October; (right panel) Winter: October-May of the following calendar year. Air temperatures are from the nearest meteorological station to each site.*

From our past research, we concluded that fuel reduction management treatments that disturbed the soil organic layer (shearblading) similarly catalyzed large changes in permafrost and tree seedling species composition that may lead to state change over longer timescales, while treatment that had limited impact on the soil organic layer (hand thinning) protected the permafrost from thaw and did not alter tree seedling composition. New insights from revisiting this network included observations of consistent conifer recruitment across the first decade after treatment, which is in contrast to our wildfire network (Figure IV).

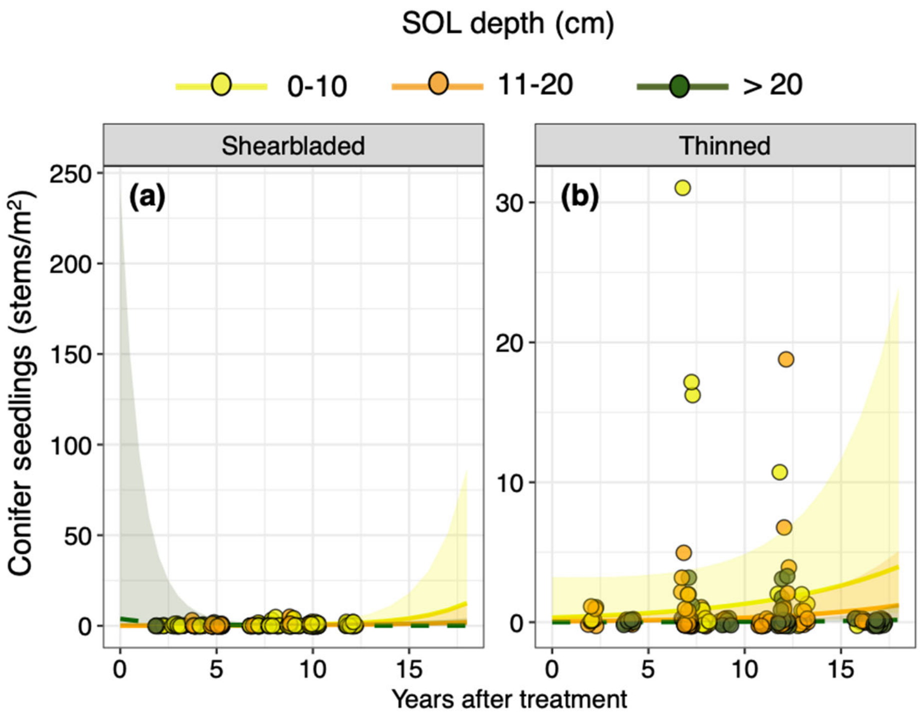

Figure IV. *Generalized linear mixed model (GLMM) results of conifer seedling density as a function of the interaction between treatment type, years after treatment, and SOL depth excluding green moss in shearbladed (a) and thinned (b) sites. Lines represent the relationship between years after treatment and predicted conifer seedling density when the SOL depth is 0-10 cm, 11-20 cm, and > 20 cm (a, b), shading the 95% prediction intervals, and points the raw data values colored by SOL depth. Solid lines indicate significant relationship.*

Although deciduous tree seedlings dominated in severely disturbed (shearbladed) treatments, recruitment of conifers, likely from surrounding undisturbed stands, established mixed composition stands that will likely undergo complex successional trajectories in the future. Active layer depth and permafrost degradation increased over time, but only in the shearbladed treatments; active layer depth also increased in the thinned treatments, but to a lesser degree (Figure V). New measurements of fire rate of spread (ROS) indicated that the initial benefits of reduced ROS diminished to pre-treatment levels by the first decade after treatment (Figure VI). Additionally, shearblading of black spruce-lichen woodlands increased ROS because vegetation composition shifted towards plant species with a higher abundance of fine fuels.

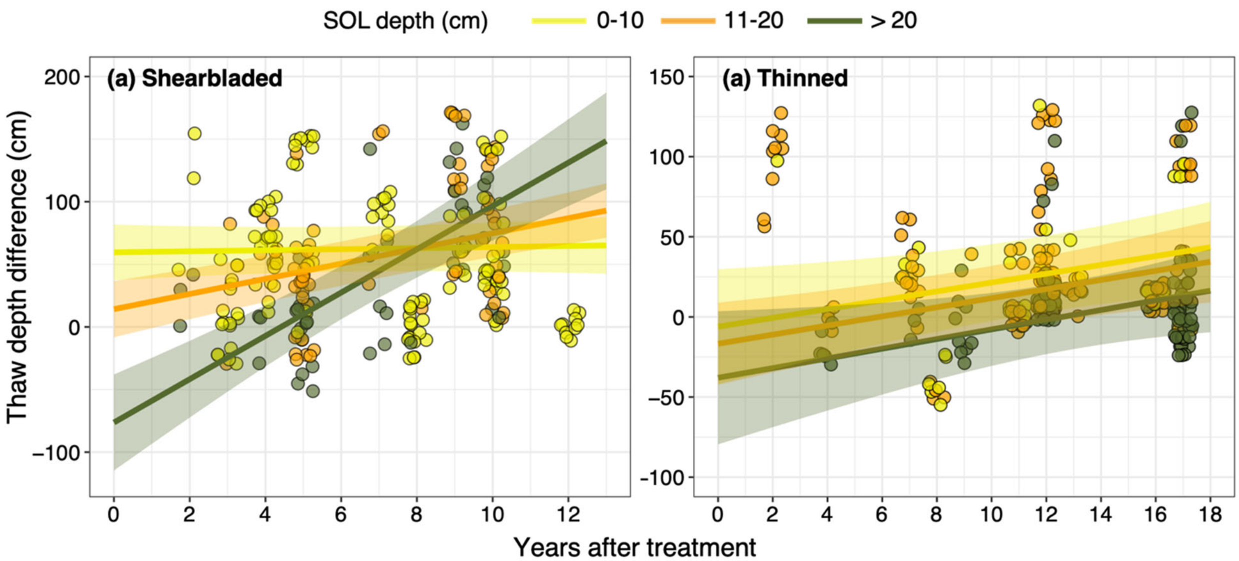

Figure V. *Linear mixed model (LMM) results of thaw depth difference (TDD; treated – unmanaged) as a function of the interaction between years after treatment, treatment type, and SOL depth excluding green moss. Lines represent predicted fits of TDD with years after treatment when SOL depth is between 0 – 10 cm, 11 -20 cm, and > 20 cm in shearbladed (a) and thinned (b) sites, shading the 95% prediction intervals, and points the raw data values. In shearbladed areas, this trend was the strongest when SOL depth was > 20 cm (20.12 cm ± 0.24, p<0.001; Estimate ±* SE*), followed by 11- 20 cm (6.06 cm ± 0.12, p<0.001) and 0 – 10 cm (0.42 cm ± 0.12, p<0.001). In thinned areas this trend was slightly stronger when SOL depth was > 20 cm (3.06 ± 0.06, p<0.001) than 11 – 20 cm (2.75 cm ±* 0.03, p<0.001*) or 0 – 10 cm (2.75 cm ± 0.03, p<0.001).*

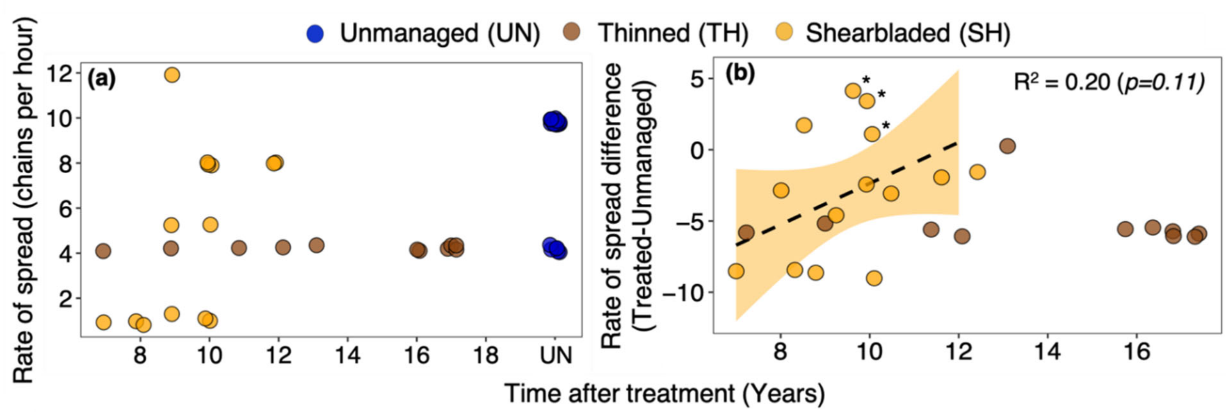

Figure VI. *Rate of spread (ROS) of all subsites in 2018 (a) and the difference in ROS between treated and unmanaged subsites as a function of years after treatment (b). Dashed line represents relationship between ROS difference and years after treatment in shearbladed areas (1.44 ± 0.84, p=0.11), and asterisks on points indicate shearbladed areas that were paired with an open black spruce or open black spruce with paper birch stand where the ROS was greater in the treated than unmanaged stand (b).*

Implications for Future Research and Benefits: Our data confirm that managers can use early post-fire data to predict vulnerability to ecosystem state change after wildfire because tree seedling regeneration and soil temperature regime are entrained soon after fire. In fuel management treatments, however, the prolonged window of conifer recruitment will require longer-term (e.g., decadal) observations of composition to predict successional trajectory. Managers are interested in the idea of a "living fuel break," that treatment-driven changes in species composition could result in establishment of low flammability deciduous stands that could persist for a century. Given the continued recruitment of conifers, we need to follow treatments over more than one decade to determine whether this is a probable outcome. New measurements showed that fuel management effects on fire ROS returned to prefire levels about a decade after treatment. Because shearblading treatments increase ROS in open canopy black spruce woodlands, these ecosystems are not effective candidates for fuel brakes created by this method.

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deciduous trees had no significant effects and was removed from the final model. Paths affecting fire are shown with transparency to emphasize drivers of seedling responses. Directed causal paths are shown as straight lines with single-headed arrows; undirected correlations among variables are shown as curved, double-headed arrows. All paths in the model are illustrated. Line color and thickness indicate the magnitude and direction of effects, with standardized path coefficients shown next to significant paths (red = positive effects, blue + dashed = negative effects, grey = NS). Tests of SEM fit indicate no significant differences between the illustrated model structure and actual data ($p>0.05$).

Figure 9. Structural equation model (SEM) results for models of post-fire seedling biomass (on a log scale) of a) black spruce and b) deciduous trees. Model syntax follows that shown in Fig. 6. Tests of SEM fit indicate no significant differences between the illustrated model structure and actual data ($p>0.05$). Sites with no seedlings of a species were removed from the model ($n=84$ for black spruce and $n=73$ for deciduous trees).

Figure 10. Generalized linear mixed model (GLMM) results of conifer seedling density as a function of the interaction between treatment type, years after treatment, and SOL depth excluding green moss in shearbladed (a) and thinned (b) sites. Lines represent the relationship between years after treatment and predicted conifer seedling density when the SOL depth is 0-10 cm, 11-20 cm, and > 20 cm (a, b), shading the 95% prediction intervals, and points the raw data values colored by SOL depth. Solid lines indicate significant relationship.

Figure 11. Generalized linear mixed model (GLMM) results of deciduous seedling density as a function of the interaction between treatment type and years after treatment in shearbladed (a) and thinned (b) sites, and the interaction between treatment type and SOL depth excluding green moss in shearbladed (c) and thinned (d) sites. Lines represent the relationship between each covariate and predicted seedlings density, shading the 95% prediction intervals, and points the raw data. Solid lines indicate significant relationship.

Figure 12. Generalized linear mixed model (GLMM) results of seedling density as a function of the interaction between treatment type and seedling type in 2018. Black points represent the estimated marginal mean (EMM) of seedling density of deciduous (left) and conifer (right) seedlings based on the model fit, error bars the 95% prediction intervals, and points the raw data values in shearbladed (SH), thinned (TH), and unmanaged (UN) areas. For this figure raw values of seedling density were log-transformed after adding one to all points. Different letters denote significant difference in seedling density between treatments and seedling type.

Figure 13. Linear mixed model (LMM) results of thaw depth difference (TDD; treated – unmanaged) as a function of the interaction between years after treatment, treatment type, and SOL depth excluding green moss. Lines represent predicted fits of TDD with years after treatment when SOL depth is between 0 – 10 cm, 11 -20 cm, and > 20 cm in shearbladed (a) and thinned (b) sites, shading the 95% prediction intervals, and points the raw data values. In shearbladed areas, this trend was the strongest when SOL depth was > 20 cm (20.12 cm ± 0.24, $p<0.001$; Estimate ± SE), followed by 11- 20 cm (6.06 cm ± 0.12, $p<0.001$) and 0 – 10 cm (0.42 cm ± 0.12, $p<0.001$). In thinned areas this trend was slightly stronger when SOL depth was > 20

cm (3.06 ± 0.06, $p<0.001$) than 11 – 20 cm (2.75 cm ± 0.03, $p<0.001$) or 0 – 10 cm (2.75 cm ± 0.03, $p<0.001$).

Figure 14. Non-metric multidimensional scaling (NMDS) across treatments and sampling periods of vascular plant type (a) and ground cover (b). Location of ground cover or plant type name represents mean and points represent individual unmanaged and treated areas sampled in 2012/2013 or 2018. Circles show 95% confidence intervals for each treatment/sampling period centroid type.

Figure 15. Rate of spread (ROS) of all subsites in 2018 (a) and the difference in ROS between treated and unmanaged subsites as a function of years after treatment (b). Dashed line represents relationship between ROS difference and years after treatment in shearbladed areas (1.44 ± 0.84, $p=0.11$), and asterisks on points indicate shearbladed areas that were paired with an open black spruce or open black spruce with paper birch stand where the ROS was greater in the treated than unmanaged stand (b).

Figure 16. Stuart Creek mean daily soil temperatures at depths 5, 20, 50, 80, and 125 cm for unburned and burned sites. Depths are standardized to unburned moss/soil surface = 0 cm.

Figure 17. Mean annual (October to September of the following calendar year) temperature for the whole soil profile (5 to 150 cm) at each site and burn treatment. Means were weighted by temperature probe depth to account for disparity in profile depth increments. Air temperatures for CPCRW and WC are from meteorological stations at the sites; SC air temperatures are from Eielson Airforce Base airport, Alaska.

Figure 18. Mean temperature differences (burned-unburned) for shallow (5-35 cm) and deep (80-150 cm) soils. Temperatures for each soil profile are averaged across depths and sites. (left panel) Summer: May-October; (right panel) Winter: October-May of the following calendar year. Air temperatures are from the nearest meteorological station to each site.

List of Acronyms

DoD – Department of Defense
SON – Statement of Need
NAU – Northern Arizona University
SEM – Structural Equation Model
NMDS – Nonmetric Multidimensional scaling
ROS – Rate of Spread
SOL – Soil Organic Layer
GLMM – Generalized Linear Mixed Model
EMM – Estimated Marginal Mean
LMM – Linear Mixed Model
TDD – Thaw Depth Difference
BLM – Bureau of Land Management
BNZ LTER – Bonanza Creek Long Term Ecological Research
QA/AC - Quality Assurance/Quality Checks
WC – Willow Creek
CPCRW – Caribou-Poker Creek Research Watershed
SC – Stuart Creek

Keywords

Wildfire, forest regeneration, ecosystem state change, permafrost, fuel management, soil temperature, boreal forest

Acknowledgments

We would like to thank Daniel Rees for sponsoring access to sites on DoD lands and assisting with locating sampling sites. We also thank Ray Atwood from the Toghotthele Corporation for granting access to sites on native land in interior Alaska, and Eric Miller from the Bureau of Land management (BLM) for assistance with access. We would like to thank the BNZ LTER for assisting with logistics.

Objective

The objective of this research is to refine our mechanistic understanding of the response of Alaskan boreal forests to wildfires and fuel reduction management by collecting, analyzing, and synthesizing additional long-term data. The research focuses on re-measurement of site networks established during the completed SERDP project RC-2109. In this final report, we describe our methods of data collection and analyses, and summarize our results and conclusions.

The original project focused on identifying factors controlling ecosystem vulnerability to state change after disturbance in black spruce (*Picea mariana*) forests, the vegetation type that dominates Department of Defense (DoD) lands in Interior Alaska. Three regional site networks were developed to test the conceptual and mechanistic basis for threshold change in Alaskan boreal forests following disturbance. For wildfire, the original project results showed that severity of soil organic layer (SOL) disturbance was the most important indicator of state change because of its impacts on permafrost stability and tree species composition (Alexander et al. 2012; Alexander & Mack 2016; Chapin et al. 2008; Genet et al. 2013; Jafarov et al. 2013; Johnstone et al. 2010a; Johnstone et al. 2010b; Melvin et al. 2015; Schuur et al. 2015). High severity burning reduced ground insulation, leading to deepening of the active layer, permafrost degradation, and in some cases, surface subsidence and changes in site drainage. High severity burning also exposed mineral seedbeds, which were preferentially colonized by rapidly growing deciduous tree seedlings, shifting initial successional trajectories from spruce self-replacement to deciduous tree dominance, a novel state for black spruce forests. Our original results also revealed that fuel reduction management treatments that disturbed the SOL (i.e., shearblading) similarly catalyzed large changes in permafrost and tree seedling species composition that may lead to state change over longer timescales, while the treatment that had limited impact on the SOL (i.e., hand thinning) protected the permafrost from thaw and did not alter tree seedling composition (Melvin et al. 2018).

The conclusions from the original project were based on observations across sites that were, at the time of initial measurement, relatively recently disturbed (<10 years after disturbance events). In the current project, we have returned to the original site networks to update our observations and test a revised set of working hypotheses organized around three specific objectives. Overall, we hypothesized that the severity of disturbance of the SOL will determine whether black spruce ecosystems undergo state change after fire or fuel management disturbance. New data collected across the site networks will test whether this overarching hypothesis is consistent with findings 5-20 years after disturbance, *facilitating the confirmation, modification, or abandonment* of initial conclusions. Our proposed research addressed the SERDP statement of need (SON) by *creating the opportunity* to revisit the hypotheses and conclusions from our previous SERDP project. Our overarching objective was to determine whether the mechanistic links among fire, soils, permafrost, and vegetation succession predictive of state change immediately following disturbance remain robust predictors over longer timescales.

Severity of soil organic layer burning and post-fire organic layer depth may still be the best predictors of ecosystem state changes in vegetation and permafrost after fire. However, there are several reasons that re-measurement results may not be linear extrapolations of previous measurements. As vegetation regenerates, ecological processes such as inter-specific competition (Chapin et al. 1994) or herbivory (Brown et al. 2015; Lord & Kielland 2015) may exert increasing controls over tree species composition and successional trajectory. Permafrost that receded rapidly after fire (Yoshikawa et al. 2002) may begin to recover as a new canopy forms (Van Cleve et al. 1991). Finally, regional climate warming has driven warming and degradation of permafrost over the past two decades (Hinzman et al. 2005; Romanovsky et al. 2010), creating a "moving target" or dynamic reference state (Hiers et al. 2012) for understanding the impacts of fire or management disturbance on permafrost stability. Our specific objectives were as follows:

Objective 1: Determine the long-term effects of wildfire severity on successional trajectories of tree dominance and permafrost degradation.

Objective 2: Determine the long-term effects of fuel management treatments on successional trajectories of tree dominance and permafrost degradation.

Objective 3: Determine the long-term effects of climate and wildfire on permafrost soil temperature regimes.

Background

The boreal fire cycle: The boreal region of Interior Alaska comprises a mosaic of evergreen, deciduous, and mixed forest ecosystems interspersed with herbaceous or shrubby wetlands. These ecosystems are often underlain by perennially frozen—permafrost—soils, and range across gradients of soil moisture that vary with topography, from very poorly drained to well-drained soils. Evergreen stands dominated by black spruce are the most abundant forest type in Interior Alaska and are frequently underlain by permafrost (Van Cleve et al. 1991). Black spruce forests are highly flammable and typically burn during stand replacing fires every 70-130 years (Johnstone et al. 2010a). Fire offers an opportunity for plant community reorganization that can be strongly influenced by fire characteristics (Chapin et al. 2006b). After high severity fires, mono-dominant spruce stands can give way to fully deciduous stands where permafrost degrades or is lost completely. After less-severe fires, spruce replaces itself and permafrost can stabilize and recover over succession. Stable cycles of fire disturbance and spruce self-replacement have persisted for over 6kya, since black spruce came to dominate the evergreen forests of Interior Alaska (Chapin et al. 2006a). Forecasted changes in future climate, however, could affect the stability of boreal ecosystems directly, by warming permafrost in undisturbed ecosystems, and indirectly, through an increase in fire size and severity (Figure 1). These direct and indirect effects of climate warming could, together, drive a regional fire regime shift and alter the distribution of boreal ecosystems, changing the landscape of Interior Alaska and the goods and services it provides to humans (Chapin et al. 2008).

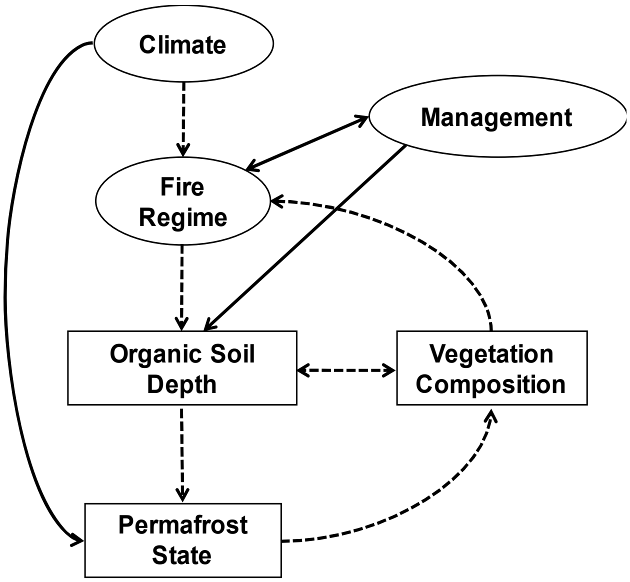


Figure 1. *Conceptual framework for identifying climate-driven state changes in boreal forests. Climate change can affect boreal ecosystems (vegetation/soils/permafrost) directly (solid lines), and perhaps more importantly, indirectly (dashed lines) via the fire cycle. Fire management can influence ecosystems through both its impacts on the fire regime and the effects of fuel management treatments on the soil organic layer.*

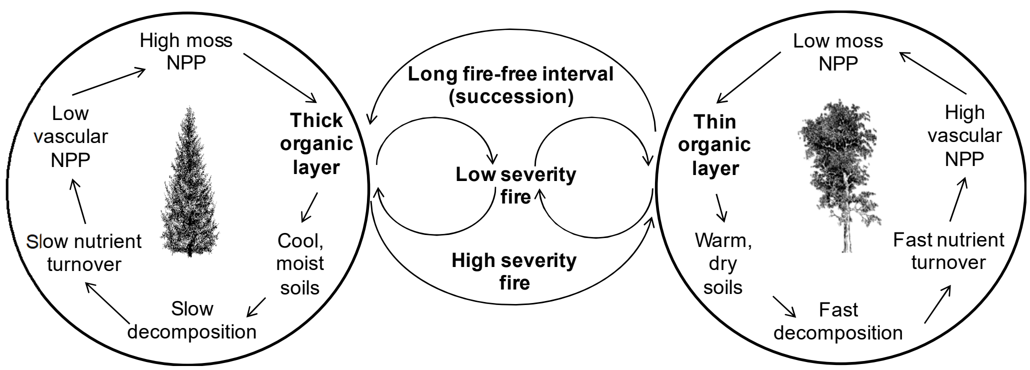
Fire management has the potential to influence the natural fire regime by determining the spatial patterning and timing of fire occurrence, and thus the successional state of the ecosystems within the management area (Figure 1). Recent management trials in Interior Alaska have focused on fire breaks that will buffer structures or training facilities from wildfire movement through highly flammable black spruce forests. Common treatments reduce canopy and ground fuels through hand thinning and stem removal, or shearblading and burning of slash piles. These treatments may result in disturbance of the soil organic layer that could promote the establishment of low-flammability deciduous tree species, which would provide long-term (100 year) maintenance of the buffer zone at minimal management cost. At the same time, disturbance of the soil organic layer could catalyze thaw and degradation of permafrost soils, triggering major changes in surface topography—known as thermokarst—as ice melts and the ground surface subsides (Shur and Jorgenson 2007). Formation of thermokarst wetlands similarly provide long-term fire buffer zones, but also present other management complications caused by ground subsidence and changes in water table depth and drainage.

Permafrost control over ecosystem dynamics: Permafrost is central for understanding potential threshold change faced by boreal ecosystems (Schuur et al. 2008). As the freezing point threshold is passed, permafrost thaw and ground subsidence can have profound biological and physical effects on natural ecosystems. Vegetation is dependent on the surface water table created by permafrost, so much so that ground subsidence and redistribution of soil moisture can have much larger effects on ecological functioning than changes in ground temperature alone (Schuur et al. 2009). In addition, changes in the ground structure due to loss of ground ice can have catastrophic effects on facilities, infrastructure, and military testing and training.

Permafrost temperature, thickness, and geographic continuity are controlled to a large extent by the surface energy balance and thus vary strongly with latitude (Brown et al. 2001). Interior Alaska lies within the discontinuous permafrost zone, where regional temperature is not low enough to sustain permafrost everywhere and existing permafrost is especially susceptible to observed and anticipated climate warming (Hinzman et al. 2006). Here, permafrost stability and distribution is a function of local factors including the vegetation structure and, in particular, by the characteristics of the thick organic soil layer present in many boreal ecosystems. Organic soil layers buffer the response of permafrost to climate variability and delay heat propagation into the soil (Osterkamp et al. 1994). Thus, ecosystem structure, particularly the thickness of the soil organic layer as well as rapid changes in that layer via fire or human activities, can have profound effects on permafrost stability and its degradation in a changing climate.

Vegetation composition and ecosystem dynamics: Plant-soil-microbial (PSM) feedbacks between vascular plants, mosses, and microbial decomposition maintain deep organic soils in the black spruce forests and wetlands of Interior Alaska (Figure 2). This internal feedback has been a key source of ecosystem resilience under the historical fire regime; moist, cold soils, poorly drained due to permafrost, burn at low severity and create a seedbed that favors the re-establishment of black spruce (Johnstone 2006) and the recovery of the organic soil layer. In extreme fires, however, these soils can burn deeply (Boby et al. 2010). When less than ~5 cm of organic soil remains after fire, deciduous tree species such as aspen and birch establish at high densities (Johnstone et al. 2010a) and catalyze a switch to alternate plant successional trajectories that are dominated or co-dominated by deciduous trees. Here, a new PSM feedback domain emerges where shallow organic soils are maintained by rapidly decomposing litter from highly productive deciduous species. Degradation and loss of permafrost is likely once this threshold is reached (Yoshikawa et al. 2002), leading to a state change that permanently alters ecosystem structure and function. Shifts between domains of spruce vs. deciduous dominance and the resulting effects on permafrost have large implications for ecosystem productivity and carbon storage (Mack et al. 2008), feedbacks to regional climate (Randerson et al. 2006), the goods and services that boreal ecosystems provide to humans (Chapin et al. 2008), including ecosystem resilience to anthropogenic activities such as those related to military operations. Because the organic soil layer plays this dual role, mediating climate effects on the ground thermal regime and controlling vegetation successional dynamics, it is a key indicator for understanding resilience and state change of boreal ecosystems on permafrost soils.

***Figure 2:** Conifer and deciduous forest stands maintain distinct ecosystem dynamics through internal cycles mediated by the soil organic layer.*



Managing interactions among fire, vegetation, and permafrost: Fire management has the potential to influence the natural fire regime by determining the spatial patterning and timing of fire occurrence, and thus the successional state of the ecosystems within the management area (Figure 1). Controlled burning of selected evergreen stands can increase deciduous stand frequency on the landscape with the potential for altering spread of natural wildfires because of the differential flammability of boreal ecosystems (Chapin et al. 2006b). White spruce is generally less flammable than black spruce, as illustrated by a long history during the Holocene of white spruce dominance (8-10 kya) with low fire frequency (Higuera et al. 2009). However, the juxtaposition of black and white spruce forest stands on the landscape means that white spruce often burns in tandem within the fire regime of black spruce, as do shrubby or herbaceous wetlands where surface organic soils can serve as a ground fuel to carry fire during dry months. In contrast, deciduous or early successional stands have little ground fuel and while they can burn, they often reduce the spread of fire relative to other ecosystem types (Cumming 2001). Human manipulation of the fire cycle via lower intensity controlled burns has the potential to reduce the risk of high intensity large fires–the type most likely to burn the organic soil deeply and cause threshold changes in permafrost and vegetation– by creating a patchwork of early successional vegetation that could reduce fire spread under some conditions. In sum, fire management has the potential to reduce the risk of the severe fires that are most likely to cause future state changes in boreal ecosystems but has associated localized disturbances that are potentially novel to these ecosystems.

Fire management on military lands: Over 95% of the 7,200 km^2 of Alaskan military land is located in the boreal forest of Interior Alaska, associated with Fort Wainwright and Eielson Air Force Base near the city of Fairbanks, and with Fort Greeley near the city of Delta Junction. These lands cross two ecologically, economically, and culturally important boreal eco-regions: the Tanana-Kuskokwim Lowlands, which covers about 52,000 km^2 of Interior Alaska, and the Yukon-Tanana Uplands, which covers about 102,000 km^2. Fire is the most widespread natural disturbance in these regions. The Yukon-Tanana Uplands have the highest incidence of lightning strikes in Alaska and the Yukon Territory (Dissing and Verbyla 2003). In addition, military lands also experience high human ignition pressures due to the frequency of military testing and training activities. Thus, these military lands are designated in a distinct fire management zone by the Alaska Interagency Coordination Center so that local fire management officers (FMO) can address the unique needs of military land management. Because of this fire management designation, DoD lands offer the potential to understand both the threshold changes associated with severe fires, and the potential for fire management to either contribute to or mitigate vulnerability of DoD lands to threshold change in ecosystem state.

Technical Approach

Objective 1: Determine the long-term effects of wildfire severity on successional trajectories of tree dominance and permafrost degradation.

After the 2004 extreme wildfire season, we established a network of permanent monitoring sites in Interior Alaska to examine the impacts of fire severity on ecosystem recovery. In spring 2005, we identified 90 sites distributed across three large wildfire complexes that span the road system from north of the Yukon River to the Alaska-Canada border in the south (Figure 3; Boby et al. 2010; Johnstone et al. 2009). All sites were dominated by black spruce prior to fire, and sites

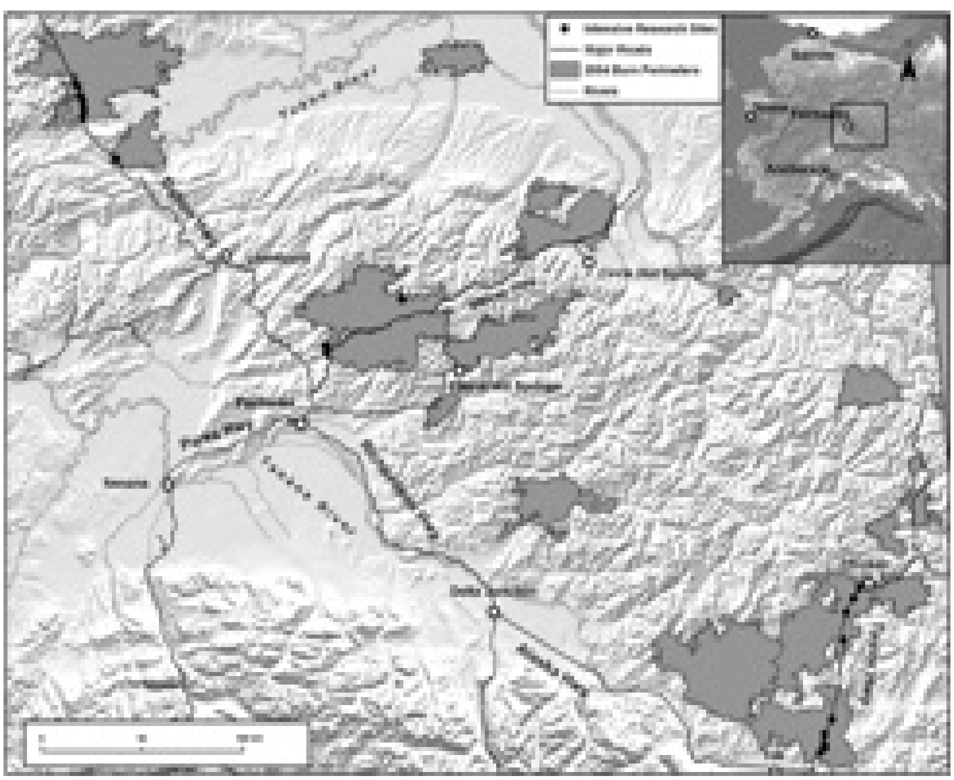


__Figure 3.__ Map of the 2004 wildfire study sites, with the location of the study area in Interior Alaska shown as a square in the inset. Areas burned in 2004 are shown as grey-filled polygons. Sample sites were located in burn complexes along the Dalton, Steese, and Taylor Highways, and are shown as black circles (note that dots overlap for adjacent sites; Johnstone et al. 2009).

were chosen to encompass the range of burn severity and soil drainage that we observed across the region. Each site consists of a 30 m x 30 m area with sides aligned to cardinal directions and marked with center and corner posts for repeated sampling. Within each site, tree seedling species density was surveyed in 20 0.25 m^2 sampling plots in 2005, 2006, 2008, and 2011. In the last survey year, we also measured the diameter and height of a subset of seedlings in each plot and harvested seedlings outside of the plots to construct allometric equations relating diameter and height to seedling biomass. Residual SOL depth and thaw depth (our index of active layer depth) were measured along sampling transects at each site at the end of the growing season in 2005 and 2011. Burn severity of the organic layer at the site level was estimated using a method that estimates depth of combustion from the position of adventitious roots on the stems of burned spruce trees (Boby et al. 2010).

We revisited this site network in July-August 2017 with a 4-5 person sampling team to re-survey tree seedling and sapling density, basal diameter, and height in the permanent plots to evaluate seedling survival and estimate trajectories of forest recovery. We increased the area of individual sampling quadrats to 1 m^2 to accommodate the increased stature of vegetation and sampled 14 replicate quadrats per site, laid out systematically along two parallel 30 m transects. Seedlings and saplings of all tree species present in a quadrat were tallied, which typically included *Picea mariana* (black spruce), *Populus tremuloides* (aspen), and *Betula neoalaskana* (birch), along with rare observations of *Picea glauca* (white spruce) and *Populus balsamifera* (balsam poplar). We selected one seedling of each species present that was located closest to the southwest corner of the quadrat and recorded its basal diameter, stem height, and any signs of current or past herbivory by hare and moose. These seedlings were scaled to biomass using allometric equations. Stem heights ranged from a few centimeters to several meters tall. We also recorded the presence and maximum standing height of any tall shrubs in the quadrat (*Alnus*, *Salix*, or *Betula*) as an index of overstory competition.

We assembled all years of seedling records and compared counts among years and cross-checked inconsistencies against our original field notes to check for errors that may have arisen during data recording or transcription (BNZ LTER data archive record 398). We used site as the sampling unit for all observations, as the sub-sampling design differed among variables, and averaged or summed within-site measurements to obtain site-level values for each measured variable. Seedling counts among quadrats at a site were summed and divided by the sample area to estimate seedling density at each site. Using 2017 seedling densities, we categorized each site into classes of successional trajectories, where deciduous trajectories had $\geq$ 70% of total stem density composed of aspen and birch, spruce trajectories had $\geq$ 70% spruce stems, and mixed stands were intermediate with > 30% and < 70% relative dominance of spruce or deciduous stems. Some stands had very low total stem densities (< 1 stem m^{-2}), and these were put into a separate class of “open” stands.

We used multivariate structural equation models (SEM; Grace 2006) to test our hypotheses about the effects of site-level environmental and fire conditions and biotic factors on 2017 seedling densities and biomass. All data were analyzed in R 3.5.0 statistical software (R Development Core Team 2018) and seedling density and biomass were log(x+c) transformed, where c is the lowest observed value in the dataset. SEM models specifically tested hypotheses that patterns of seedling density and biomass would be influenced by effects of variation in a) environmental factors related to climate and soils, b) fire return interval represented by stand age, c) fire severity represented by proportional combustion losses and effects on residual organic layer depth, d) pre-fire conditions related to stand structure and organic layer depth, and e) biotic interactions such as competition and herbivory (Figure 4). Our 2017 surveys allowed us to compare results from early succession to the initial responses observed just three years after fire (Johnstone et al. 2010) and to assess the impacts of early recruitment on later composition of the

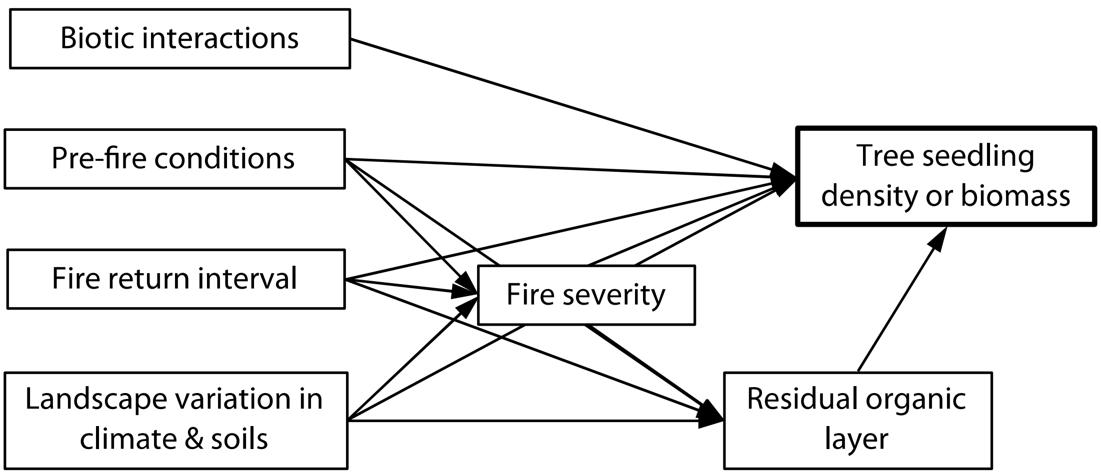


Figure 4. *Conceptual figure of hypothesized factors affecting post-fire tree seedling density or biomass. We tested specific models using structural equation modeling of seedling data from the network of black spruce sites burned in the 2004 wildfires in Interior Alaska.*

recovering tree community. To develop a reasonably parsimonious model for SEM testing, we first screened variables representing our individual hypotheses for evidence of bivariate relationships with seedling density or biomass that showed effects consistent with our hypotheses. Bivariate relationships that were non-significant or opposite in direction to that hypothesized were interpreted as a rejection of the hypothesis and were not included in the formal SEM model. Aspen and birch were combined together into a single group of deciduous broadleaf trees, as these species share similar traits and caused issues due to frequent zeros when analyzed separately. We fit SEMs for stem density and biomass for black spruce and deciduous trees separately, using robust Maximum Likelihood for standard error estimation in the 'lavaan' package (Rosseel et al. 2018). We used likelihood tests to compare simplified alternate models that were nested versions of the full model to develop the most parsimonious model that captured significant relationships apparent in the data. Final models were represented graphically with arrows illustrating all the variables and paths included in a model and estimated path coefficients for significant paths.

Objective 2: Determine the long-term effects of fuel management treatments on successional trajectories of tree dominance and permafrost degradation.

In the late summer of 2018, we re-sampled 19 fuel treatment sites in Interior Alaska that that were previously measured by us in 2012 and 2013 (Figure 5). Results from our first field campaign in 2012/2013 were published during the reporting period (Melvin et al. 2018), as was a fact sheet for Alaska land managers (Appendix 2). At our study sites fuel treatments were carried out by military and state of Alaska land mangers 7 to 17 years prior to our field campaign in 2018. Fuels were reduced at seven sites (10 subsites) by hand-thinning to varying degrees of tree spacing, and at 12 sites (16 subsites) by shearblading, which mechanically removes vegetation and the SOL. Each treated subsite was paired with an adjacent, undisturbed black spruce stand for comparison (henceforth ‘unmanaged’). In 2018 we followed the same field methodology employed during the 2012/2013 field campaign. At the majority of subsites, we sampled along two 20 m transects, approximately 20 m apart, that were established in the 2012/2013 sampling period in both the treated and paired unmanaged stand. Along each transect we documented tree seedling density by species, vegetation and ground cover composition, SOL depth, and thaw depth. Each subsite was also categorized into one of the 57 fuel types described in the updated Fuel Model Guide to Alaska Vegetation (produced by the Alaska Wildland Fire Coordinating Group, March 2018; Appendix 3).

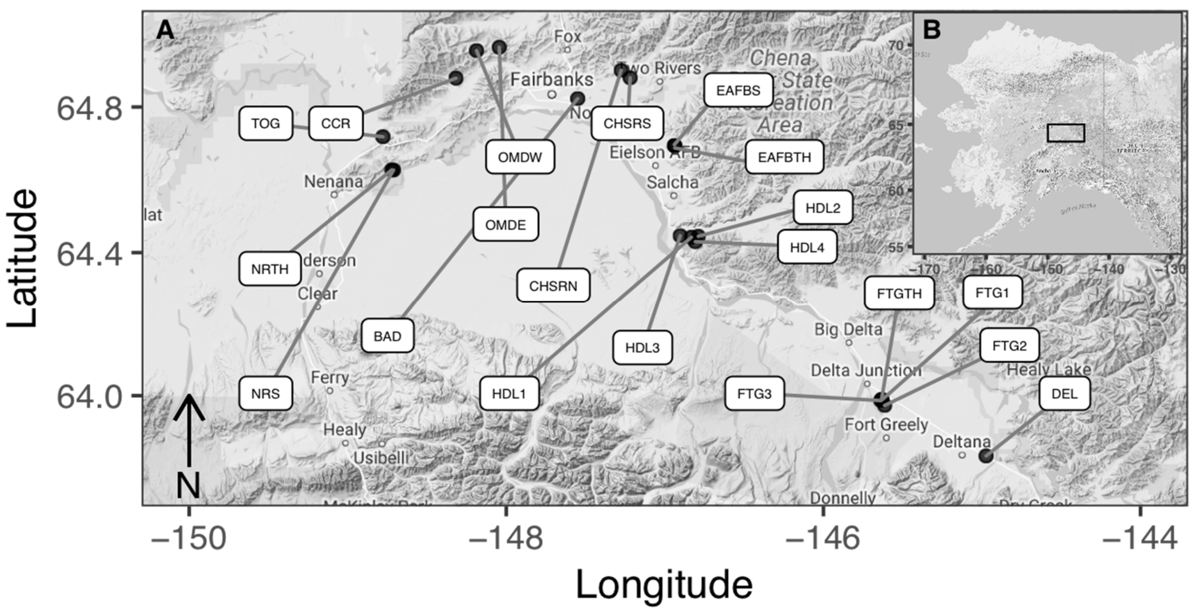


Figure 5. *Fuel management sites in Interior Alaska that were sampled in 2012/2013 and 2018, including military, state, federal, and tribal lands. Each site consisted of a treated area paired with adjacent black spruce dominated stands.*

To estimate the density of tree seedlings and trees (<=1.4 m in height), we placed five 1 x 1 m quadrats at random locations along each 20 m transect. In each quadrat all tree seedlings and small trees were counted, which in included *P. mariana*, *P. glauca*, *P. tremuloides*, and *B. neoalaskana*, and a few observations of *Larix Laricina* (larch). In each of these quadrats, we also characterized and measured the SOL by cutting a 10 x 10 cm block of organic soil using a knife and measuring the depth of each organic horizon. We added these measurements to obtain the total SOL depth from the surface to the mineral-organic interface. Adjacent to each soil sampling location thaw depth was measured using a two m steel probe, which was pushed into the ground until hitting ice.

Using data collected in 2018 and in 2012/2013, we tested the following hypotheses with the specified statistical methods in R 3.5.2 statistical software (R Development Core Team 2018): (1) *Deciduous seedlings will continue to dominate tree regeneration in shearbladed sites because of the severe disturbance of the SOL.* We fit generalized linear mixed models (GLMMs) with a zero-inflated negative binomial distribution in the package 'glmmTMB' (Magnusson et al. 2019) to assess the impacts of treatment type (shearbladed or thinned), years after treatment, and SOL depth on seedling density (stems m^{-2}). Models were built with either deciduous (aspen, birch, and larch combined) or conifer seedling density (white spruce and black spruce combined) as the response variable and the fixed effects of treatment type, years after treatment, SOL, their three-way interaction, and all possible two-way interactions. The random effects of site and subsite nested in site were included in both models to fit varying intercepts by site and subsite nested in site. We built every possible reduced version of the full model for each seedling type and ranked models based on small sample corrected Akaike information criterion (AICc) using the 'model.sel' function in the 'MuMIn' package (Bartoń 2019). We also fit a GLMM to determine if there were differences in seedling density across treatments (including unmanaged sites) and seedling type (deciduous or conifer) in the most recent sampling period (2018). This model included seedling density as the response variable, the fixed effects of seedling type, treatment type, and their interaction, and the random effects of site and subsite nested in site. Model selection was performed as indicated above. Post-hoc tests were performed by computing contrasts of estimated marginal means (EMMs) to determine differences between treatments and seedling types using the 'emmeans' package (Lenth et al. 2019). (2) *Active layer will increase in the shearbladed sites more rapidly than in thinned or undisturbed sites because of severe disturbance of the SOL.* We fit a linear mixed model (LMM) using the package 'nlme' (Pinheiro et al. 2010) to address the impacts of treatment type, years after treatment, and SOL depth on thaw depth. For this analysis we used thaw depth difference (treated – unmanaged; TDD) as the response variable because thaw depth was measured at different times of year during the 2012/2013 and 2018 field campaigns. The model included the fixed effects of treatment type, years after treatment, SOL depth, their three-way interaction, all possible two-way interaction, and the random effects of site and subsite nested in site. Model selection was performed as indicated above. For 2018, we also evaluated differences in thaw depth between unmanaged, thinned, and shearbladed sites using a LMM. This model included thaw depth as the response variable, the fixed effect of treatment type, and the random effects of site and subsite nested in site. Post-hoc tests were performed as indicated for GLMMs. *(3) Vegetation composition in shearbladed sites will not recover to that of unmanaged sites due to SOL disturbance.* We performed non-metric multidimensional scaling (NMDS) using the 'metaNDMS' function in the 'vegan' package (Oksanen et al. 2018) to assess and compare plant composition and ground cover in 2018 and 2012/2013 in thinned, shearbladed, and unmanaged sites. This approach is a rank-based ordination that calculates dissimilarities from an initial matrix using the Bray-Curtis index. The influence of years after treatment and on plant composition and ground cover, and SOL depth on plant composition, was then evaluated by fitting a smooth term for each variable on the response surface. 4) *Fuel treatments vary in their long-term efficacy of transforming vegetation to lower flammability fuel types according to the severity of soil disturbance*. Based on the assigned fuel type we calculated the difference in rate of spread (ROS) between each treated and the paired unmanaged subsite at average moisture conditions. We then performed a simple linear regression to determine the influence of years after treatment on ROS difference at thinned and shearbladed areas.

Objective 3: Determine the long-term effects of climate and wildfire on permafrost soil temperature regimes.

We established three pairs of burned and unburned sites where we have monitored air temperature, active layer temperature, and permafrost temperature dynamics with dataloggers initially installed in 2013 and last measured during the growing season of 2018. During this project period we were able to retrieve previously logged data and new data that was logged between 2017-2018. Direct temperature measurements logged over 30-minute intervals provide finer-resolution assessment of permafrost dynamics compared to single point end-of-growing-season measurements of active layer thickness. The time intervals measured in these reference sites by dataloggers matches the input time scale used by models to represent permafrost dynamics, and thus can be used as a validation dataset for permafrost representation in models. The site pairs span three landscape positions where permafrost susceptibility to fire is expected to vary based on site moisture and soil conditions. The three sites are: 1) Willow Creek (WC), located on a large flat river *floodplain* in the Tanana Flats (burned in 2011), 2) Caribou-Poker Creek Research Watershed (CPCRW), an *upland valley bottom* of a small watershed in the White Mountains managed by the University of Alaska Fairbanks (burned in 2004), and 3) an upland *hillslope* at Stuart Creek (SC), also in the White Mountains (burned in 2013). Sites 1) and 3) first are located directly on DoD-managed land. All site pairs are (or were) mono-dominant black spruce forests with a thick ground layer of moss and soil organic matter, located near a fire perimeter where an unburned site with similar tree density to the burned stand pre-fire was established. All burned sites experienced moderate to high severity fire where a significant proportion of the SOL was consumed. Fire severity, as measured by proportion of canopy and ground layer consumed by fire, was quantified during the initial burned site establishment by measuring remaining adventitious roots and reconstructing the depth and composition of the SOL before fire (Boby et al. 2010).

The temperature probes monitor soil temperature with thermistors every 5 cm for the first 40 cm and every 10 cm for the next 40 cm, with three additional depths to a maximum of 150 cm. Typical unburned forest depth-to-permafrost ranges from 50-100 cm with increased depth-to-permafrost in burned sites. Data was averaged and stored at 30-minute intervals using a Campbell CR1000 data logger powered by a small solar array and a marine cycle battery for continuous year-round measurement. We revisited these installations at the beginning of the summer season of 2017 with data retrieved between the initial maintenance check, at the end of the 2017 growing season, and the beginning and end of the 2018 growing season. Fortunately, we were able to obtain data from the loggers that had been stored over the period when the project was not active, and this was added to the full dataset reported below.

Results and Discussion

Objective 1: Determine the long-term effects of wildfire severity on successional trajectories of tree dominance and permafrost degradation.

Based on our early post-fire surveys, we concluded that successional trajectories of tree species dominance were strongly related to initial severity of organic layer combustion (Brown et al. 2015; Johnstone et al. 2010a). Thus, our *null hypothesis (H_0) was that in the second decade after fire, successional trajectories of tree species dominance would remain strongly linked to the severity of organic layer combustion*. In intermediately burned sites, however, this hypothesis may not be supported. In our 2017 data, 21% of the stands were classified as mixed stands in which spruce and deciduous species were co-dominant (Figure 6). Deciduous tree species have higher relative growth rates than black spruce (Gutsell and Johnson 2002), so for the intermediately burned sites, *we tested an alternative to our null hypothesis (H_1): Mixed stands in the intermediate burn severity range will transition to deciduous dominance.* We have also observed that in areas with high herbivory pressure from snowshoe hare and moose, deciduous tree saplings suffer greater herbivory damage than spruce saplings. *Thus, our second alternative and opposing hypothesis (H_2) was that mixed stands would transition to spruce dominance because of negative effects of herbivory on deciduous tree seedlings.*

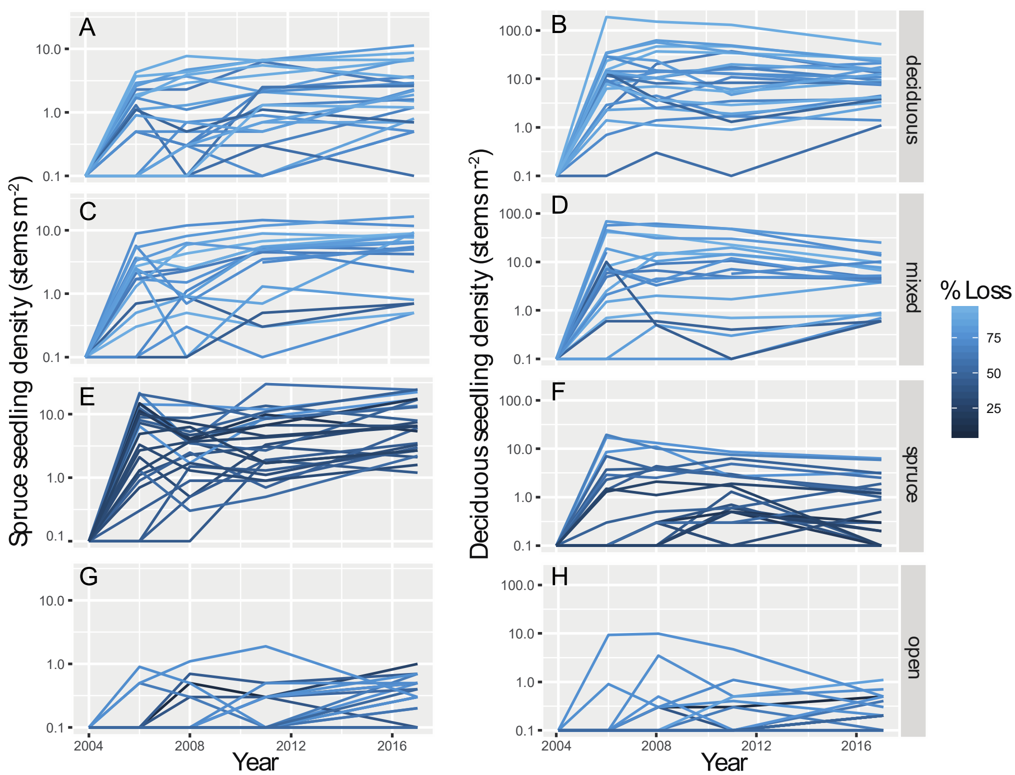


Figure 6. *Changes in seedling densities through time for black spruce (left) and deciduous trees (right). Time series are grouped according to successional trajectories (A,B = deciduous dominated, C,D = mixed; E,F = spruce dominated, and G,H = open stands). Lines connect observations at each site, starting with fire in 2004 and ending with the 2017 survey data. Line shading indicates variation among sites in fire severity, specifically proportional combustion losses of the soil organic layer (SOL). Note that y-axis scales differ between species and are presented on a log scale.*

Seedling densities of black spruce and deciduous seedlings reached almost the full range of observed densities within two years after fire (2006; Figure 6). Maximum observed densities ranged from 0-12 seedlings m^{-2} for spruce and reached an order of magnitude higher for total deciduous seedlings (birch and aspen combined). Following the initial establishment period between 2004-2006, spruce seedling densities were largely stable with a similar range of minimum-maximum densities observed in 2017 compared to 2006. Maximum densities of deciduous seedlings decreased from 2006 to 2017, reflecting moderate declines in deciduous seedling densities across many sites (Figure 6). The strong influence of initial recruitment patterns on subsequent stem densities were reflected in the relative composition of tree seedling

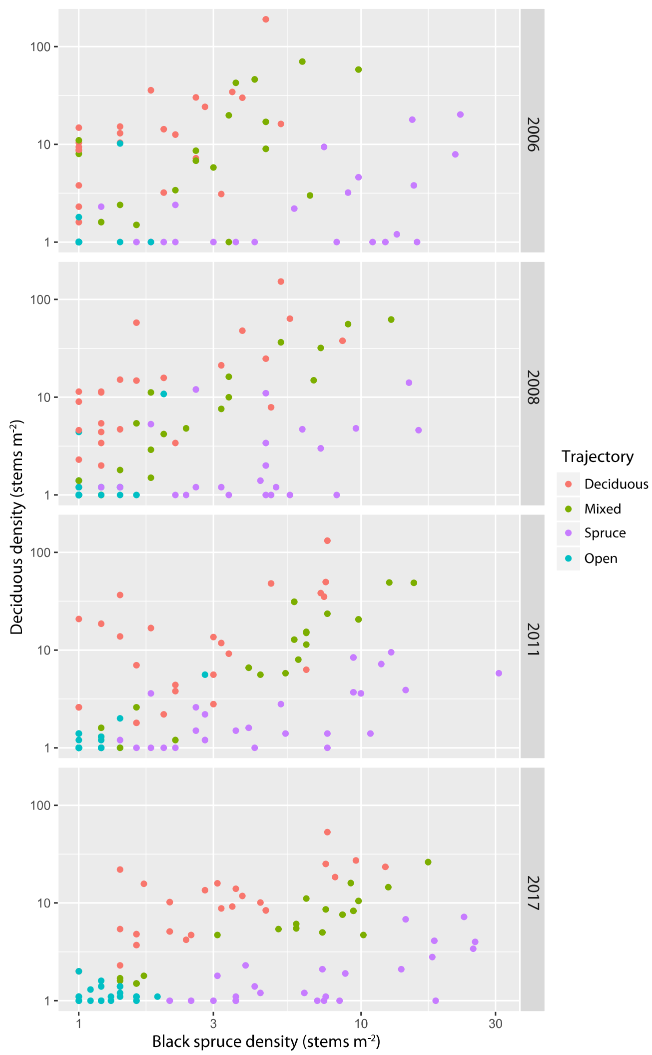


Figure 7. *Time series of tree composition following fire across the network of 90 plots located in black spruce forests of Interior Alaska that were burned in 2004. Each panel presents the density of deciduous broadleaf vs. black spruce tree seedlings alive in different survey years (2006, 2008, 2011, 2017), plotted on log axis scales. Points represent the values for a site and are color coded based on successional trajectories defined by relative densities observed in 2017 (deciduous dominated stands had >70% deciduous seedlings, spruce dominated stands had >70% spruce, mixed stands were intermediate, and open stands had a total tree seedling density <1 stem m^{-2}, indicating transitions to open woodland or non-forest). The time series illustrates that successional trajectories were largely established within 2 growing seasons after the fire, although modest shifts in relative dominance are apparent over the 13-year time series.*

communities. Successional trajectories defined by tree seedling densities in 2017 were largely established by the second summer after the 2004 fire season and have persisted with minor modifications through early succession (Figure 7). Declining densities of deciduous seedlings through time caused some sites that were originally dominated by deciduous seedlings to shift to mixed dominance or open stands by 2017, or from originally mixed stands to spruce dominance, supporting H_1. However, the strong consistency of site composition in 2017 with previous survey years going back to 2006 indicates an overriding influence of early composition, as proposed by our null hypothesis H_0. Sites with the lowest proportional combustions losses to the

SOL by fire had lower rates of deciduous seedling recruitment and were preferentially associated with spruce dominated successional trajectories (Figure 6).

More formal tests of our hypotheses were obtained by fitting SEMs. Models of black spruce and deciduous seedling density were significantly better when they included the effects of initial recruitment density (χ^2 tests of nested models, $p<0.001$). These models indicated strong effects of densities observed in the second growing season after fire on densities observed in 2017 (Figure 8). Stem densities of black spruce were more sensitive to environmental and pre-fire variables than to fire-related variables such as combustion, post-fire organic layer depth, and stand age; removal of fire-related effects on densities did not strongly affect model fit (χ^2 tests, $p>0.05$). In contrast, SEM results showed strong effects of both fire-related and environmental variables on deciduous seedling densities. Together, these results suggest strong effects of organic layer combustion on initial and subsequent seedling composition, largely due to the sensitivity of deciduous seedling recruitment to fire severity. SEM models of seedling biomass also showed significant effects of SOL combustion on seedling sizes of both spruce and deciduous species (Figure 9), indicating a persistent legacy effect of fire severity that interacts with environmental variables to affect patterns of seedling growth apparent in the second decade after fire. Environmental gradients associated with elevation and moisture also influenced seedling densities and biomass, with moist sites supporting higher spruce recruitment and deciduous seedlings showing a greater sensitivity of biomass growth at high elevations than black spruce. In all of our analyses, we found no support for negative effects of woody shrub competition or herbivory on seedling densities or biomass. Instead, weak positive associations between our index of herbivory intensity and deciduous seedling density and biomass suggests that herbivores were attracted to areas of high seedling productivity but did not detectably affect relative dominance or growth, similar to patterns observed in an earlier wildfire in Interior Alaska (Conway & Johnstone 2017). Overall, our results most strongly support H_0 and suggest that variations in fire severity shape patterns of tree seedling recruitment and growth with persistent effects on successional trajectories of post-fire forest recovery.

Our analyses indicate the effects of fire severity on successional trajectories principally arise due to effects of soil combustion and residual organic layer depth on deciduous seedling recruitment. These results are consistent with previous analyses of early post-fire recruitment (Johnstone et al. 2010a), but expand on those early results by demonstrating the strong legacy of initial recruitment for tree composition through early succession. These results are consistent with general patterns observed across the western boreal forest, namely that initial patterns of seedling establishment exert a strong influence over subsequent canopy development and stand composition (Gutsell & Johnson 2002; Johnson et al. 1994; Johnstone et al. 2004; Peters et al. 2002; Shenoy et al. 2011). Other studies have hypothesized that biotic interactions may dramatically alter the compositional outcome of early seedling recruitment, as processes such as competition and self-thinning (Chen et al. 2009; Greene & Johnson 1999) or herbivory (Chapin et al. 2016; Niemela et al. 2001; Olnes & Kielland 2016) may reduce species-specific seedling survival to shift trajectories of canopy dominance. We found little support for these hypotheses in our monitoring of black spruce recovery from fire over a 13-year period.

Variations in relative composition among tree species during early succession were largely driven by initial effects of fire severity and environmental gradients on tree recruitment that were only weakly modified by subsequent effects.

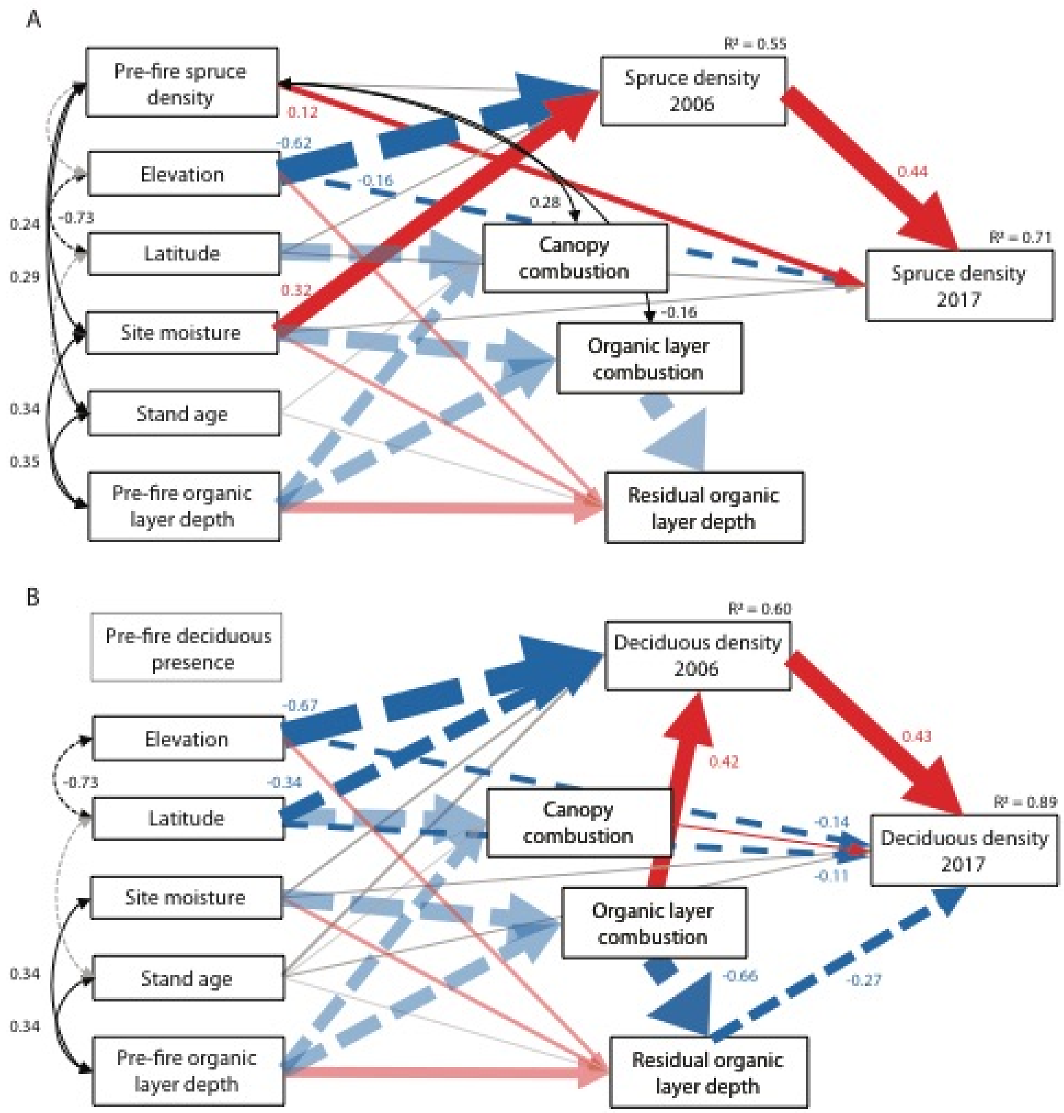


Figure 8. *Structural equation model (SEM) results for models of post-fire seedling densities (log-transformed) of a) black spruce and b) deciduous trees. The model for black spruce was simplified based on testing of nested models. In the deciduous model, the presence of pre-fire deciduous trees had no significant effects and was removed from the final model. Paths affecting fire are shown with transparency to emphasize drivers of seedling responses. Directed causal paths are shown as straight lines with single-headed arrows; undirected correlations among variables are shown as curved, double-headed arrows. All paths in the model are illustrated. Line color and thickness indicate the magnitude and direction of effects, with standardized path coefficients shown next to significant paths (red = positive effects, blue + dashed = negative effects, grey = NS). Tests of SEM fit indicate no significant differences between the illustrated model structure and actual data (p>0.05).*

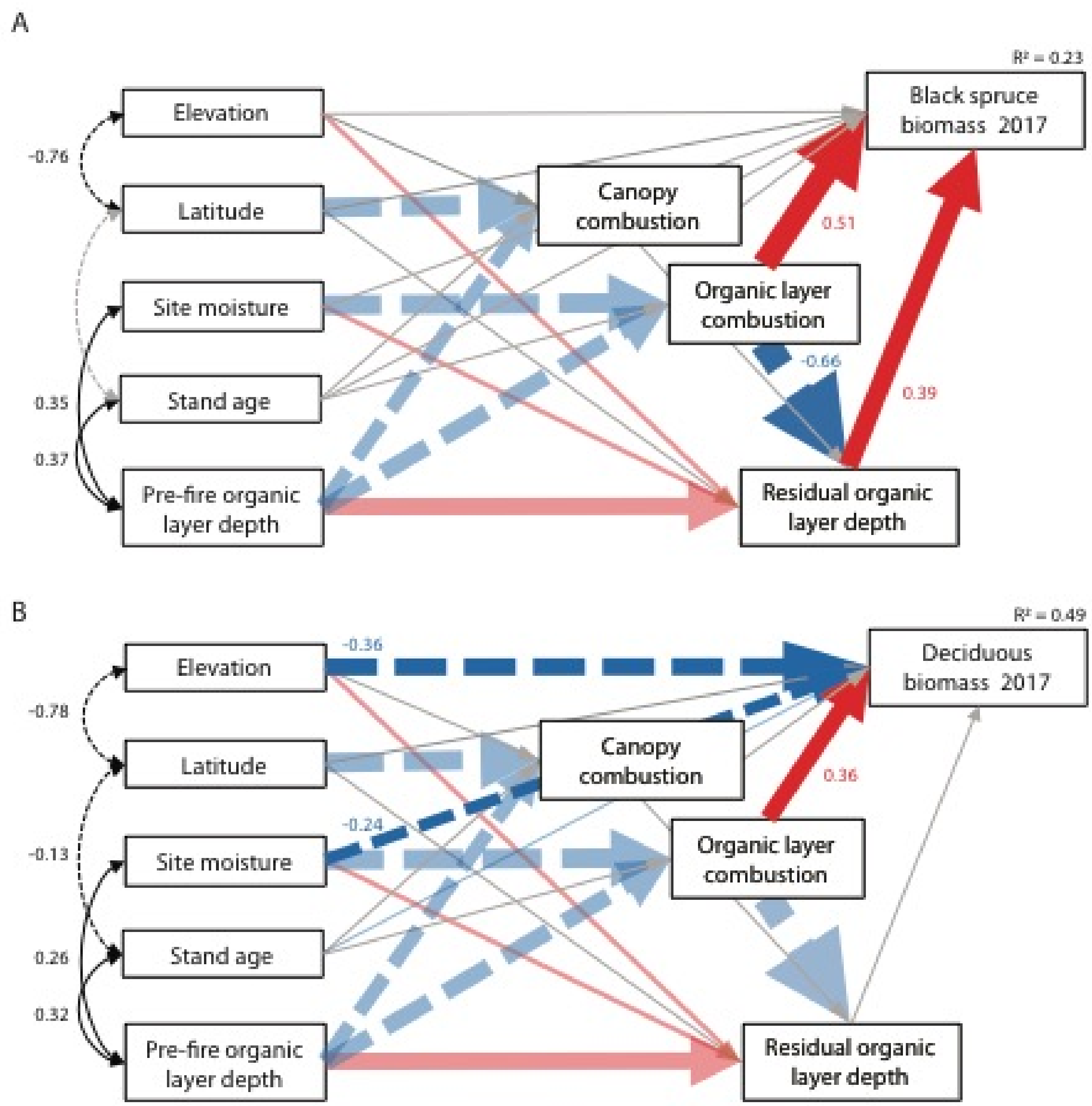


Figure 9. *Structural equation model (SEM) results for models of post-fire seedling biomass (on a log scale) of a) black spruce and b) deciduous trees. Model syntax follows that shown in Fig. 6. Tests of SEM fit indicate no significant differences between the illustrated model structure and actual data (p>0.05). Sites with no seedlings of a species were removed from the model (n=84 for black spruce and n=73 for deciduous trees).*

Objective 2: Determine the long-term effects of fuel management treatments on successional trajectories of tree dominance and permafrost degradation.

Conifer seedling density was positively associated with year after treatment in shearbladed and thinned sites, and this relationship depended on SOL depth. Specifically, the relationship between conifer seedling density and years after treatment was strongest when the SOL was shallow or completely removed in both shearbladed (Figure 8a) and thinned areas (Figure 10b), and there was no change in seedling density when the SOL was > 20 cm (Figure 10a, b). There was no effect of years after treatment on deciduous seedling density in shearbladed sites (Figure 11a), but in thinned areas seedling density slightly decreased within 2-6 years after treatment (Figure 11b). The effect of years after treatment on deciduous seedling density did not depend on SOL depth. However, deciduous seedling density was negatively associated with SOL depth in both shearbladed (Figure 11c) and thinned sites (Figure 11d). Despite these trends deciduous seedlings dominated shearbladed sites in 2018, indicating that deciduous tree recruitment was high initially after treatment and stayed at high densities over time (Figure 12). In addition, there was no difference in deciduous and conifer seedling density in thinned sites in 2018, which also indicates that deciduous tree recruitment was high initially after thinning (Figure 12). These results indicate that severe disturbance of the SOL via shearblading has a larger effect on tree seedling recruitment than thinning, particularly initially after treatment. Although we saw minimal recruitment of conifer seedling in shearbladed areas with greater years after treatment, it is likely that these areas will shift to deciduous dominated landscapes due to the high density and persistence of deciduous seedlings. Decreased deciduous stem density and increased conifer stem density in thinned areas suggests that these stands will not transition to deciduous dominated forests.

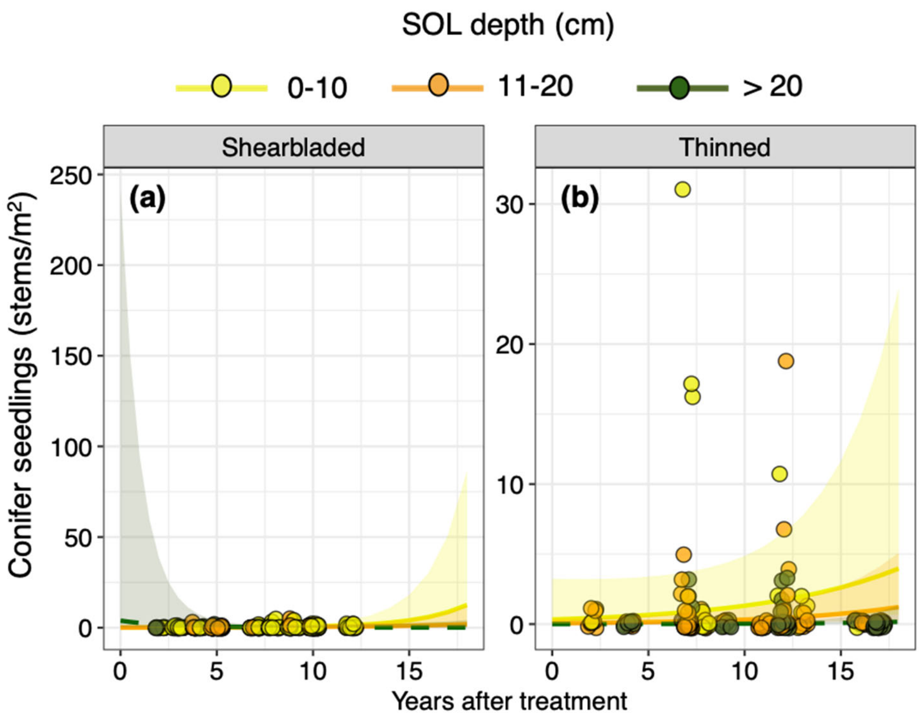


Figure 10. *Generalized linear mixed model (GLMM) results of conifer seedling density as a function of the interaction between treatment type, years after treatment, and SOL depth excluding green moss in shearbladed (a) and thinned (b) sites. Lines represent the relationship between years after treatment and predicted conifer seedling density when the SOL depth is 0-10 cm, 11-20 cm, and > 20 cm (a, b), shading the 95% prediction intervals, and points the raw data values colored by SOL depth. Solid lines indicate significant relationship.*

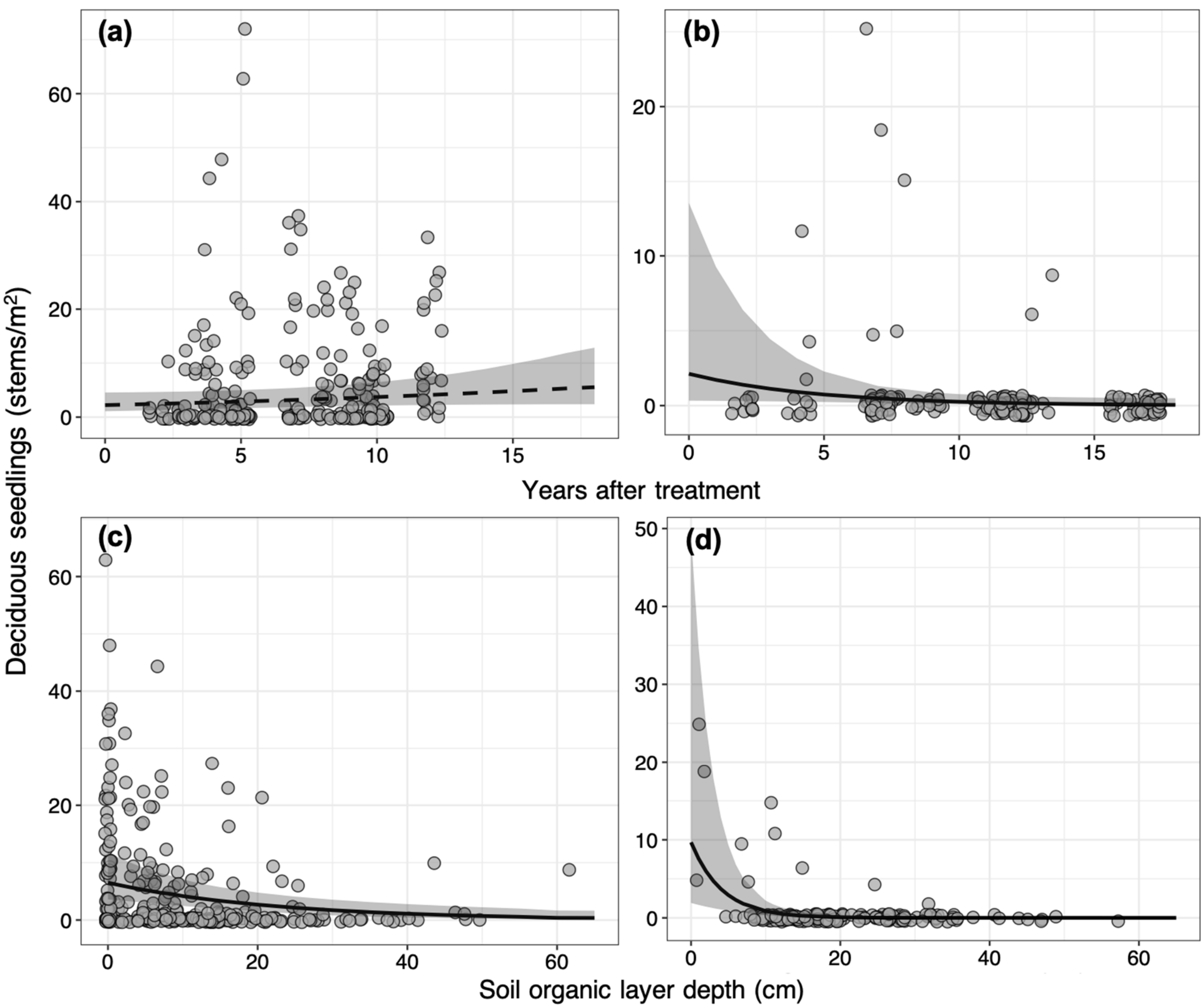


Figure 11. *Generalized linear mixed model (GLMM) results of deciduous seedling density as a function of the interaction between treatment type and years after treatment in shearbladed (a) and thinned (b) sites, and the interaction between treatment type and SOL depth excluding green moss in shearbladed (c) and thinned (d) sites. Lines represent the relationship between each covariate and predicted seedlings density, shading the 95% prediction intervals, and points the raw data. Solid lines indicate significant relationship.*

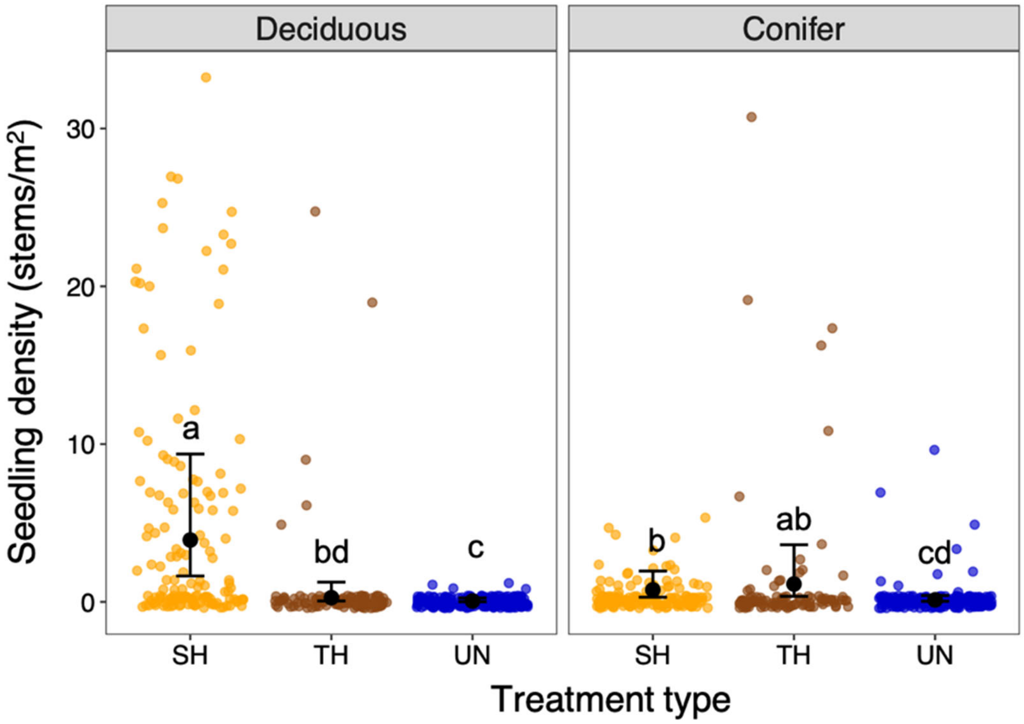


Figure 12. *Generalized linear mixed model (GLMM) results of seedling density as a function of the interaction between treatment type and seedling type in 2018. Black points represent the estimated marginal mean (EMM) of seedling density of deciduous (left) and conifer (right) seedlings based on the model fit, error bars the 95% prediction intervals, and points the raw data values in shearbladed (SH), thinned (TH), and unmanaged (UN) areas. For this figure raw values of seedling density were log-transformed after adding one to all points. Different letters denote significant difference in seedling density between treatments and seedling type.*

When the SOL was deep (> 20 cm), there was a strong positive association between TDD and years after treatment in shearbladed areas (Figure 13). This trend at high SOL depths could be due to differences in topography (Osterkamp et al. 2000), standing surface water (Jorgenson et al. 2010) - which was observed in numerous shearbladed sites - and/or understory vegetation composition (Loranty et al. 2018) to name a few. Thaw depth difference also slightly increased over time in shearbladed areas at shallow SOL depths (< 20 cm). This weaker trend at shallow SOL depths is likely because permafrost thawed rapidly in shearbladed areas initially after treatment when most or all of the SOL was removed (Figure 13). At thinned sites TDD was also highest initially after treatment when the SOL was shallow or non-existent (< 10 cm) but increased over time at a similar rate across all SOL depths (Figure 13). In 2018, thaw depth was greatest at shearbladed sites (104 cm ± 7.00; EMM ± SE), followed by thinned (68.5 cm ± 7.02) and then unmanaged sites (50.0 cm ± 6.28). Overall, these results indicate that shearblading results in greater permafrost thaw than thinning. However, thinning also disrupts permafrost stability, and to the greatest extent when the SOL is shallow or removed.

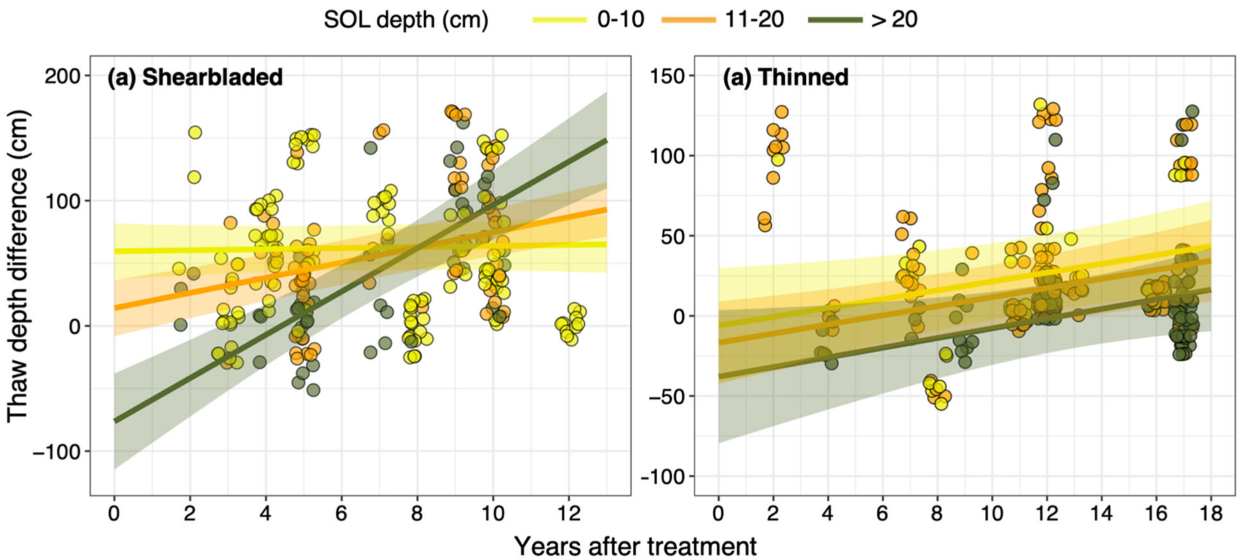


Figure 13. *Linear mixed model (LMM) results of thaw depth difference (TDD; treated – unmanaged) as a function of the interaction between years after treatment, treatment type, and SOL depth excluding green moss. Lines represent predicted fits of TDD with years after treatment when SOL depth is between 0 – 10 cm, 11 -20 cm, and > 20 cm in shearbladed (a) and thinned (b) sites, shading the 95% prediction intervals, and points the raw data values. In shearbladed areas, this trend was the strongest when SOL depth was > 20 cm (20.12 cm ± 0.24, p<0.001; Estimate ±* SE*), followed by 11- 20 cm (6.06 cm ± 0.12, p<0.001) and 0 – 10 cm (0.42 cm ± 0.12, p<0.001). In thinned areas this trend was slightly stronger when SOL depth was > 20 cm (3.06 ± 0.06, p<0.001) than 11 – 20 cm (2.75 cm ±* 0.03, p<0.001*) or 0 – 10 cm (2.75 cm ± 0.03, p<0.001).*

In 2012/2013 and 2018 there were large differences in vegetation composition in shearbladed compared to thinned and unmanaged areas (Figure 14a), which were largely driven by SOL depth ($F_{(9)}$=2.81, p<0.05). Ground cover in shearbladed sites differed from that of thinned and unmanaged sites as well in both 2012/2013 and 2018 (Figure 14b), and was influenced by years after treatment ($F_{(9)}$=3.091, p<0.05). Lastly, the highest ROS was consistently observed in unmanaged areas (Figure 15a). At thinned subsites ROS was always less than at unmanaged subsites (Figure 15b). In shearbladed areas there was a slight, though not significant, positive association between ROS difference and years after treatment, and in some cases ROS was higher in a shearbladed subsite than at the adjacent unmanaged subsite (Figure 15b). This tended to occur when a shearbladed area was paired with an open black spruce or open black spruce with paper birch stand (Figure 15b), and highlights the importance considering stand type before proceeding with fuel reduction treatment.

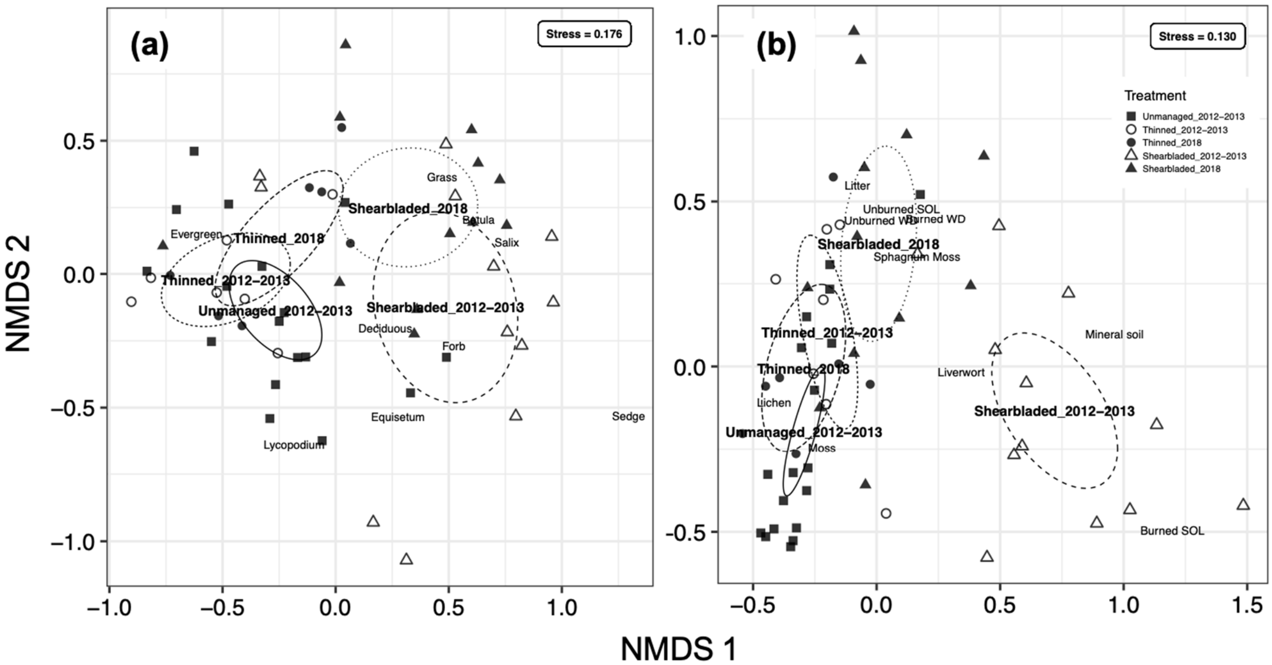


Figure 14. *Non-metric multidimensional scaling (NMDS) across treatments and sampling periods of vascular plant type (a) and ground cover (b). Location of ground cover or plant type name represents mean and points represent individual unmanaged and treated areas sampled in 2012/2013 or 2018. Circles show 95% confidence intervals for each treatment/sampling period centroid type.*

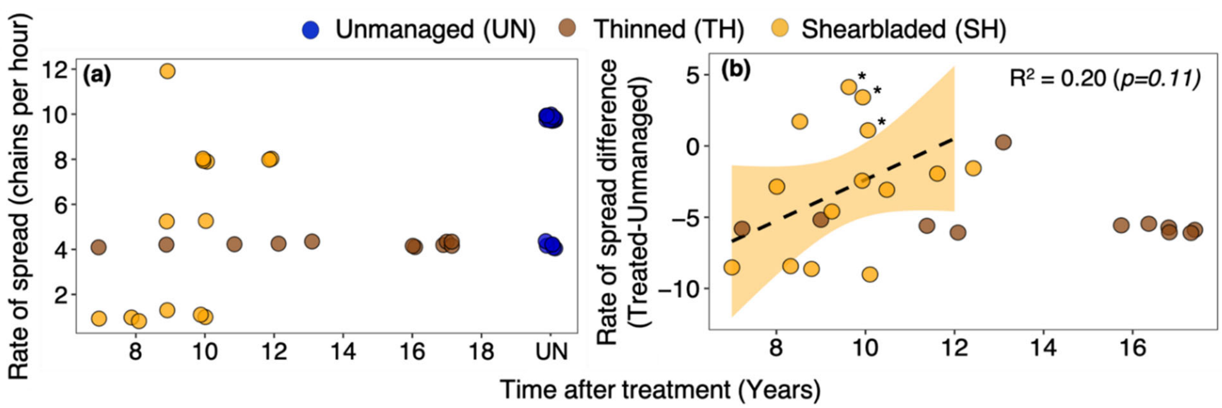


Figure 15. *Rate of spread (ROS) of all subsites in 2018 (a) and the difference in ROS between treated and unmanaged subsites as a function of years after treatment (b). Dashed line represents relationship between ROS difference and years after treatment in shearbladed areas (1.44 ± 0.84, p=0.11), and asterisks on points indicate shearbladed areas that were paired with an open black spruce or open black spruce with paper birch stand where the ROS was greater in the treated than unmanaged stand (b).*

Objective 3: Determine the long-term effects of climate and wildfire on permafrost soil temperature regimes.

A critical feature of our paired site design was the interpretation of soil temperatures at equivalent depths across burned and unburned sites. In most studies, soil depth is referenced at the soil/moss interface to the air (= 0 cm). Because combustion removes the SOL, 10 cm below the soil/moss interface in a burned site actually represents soil that was deeper in the unburned state. The paired burned-unburned site design allowed us to correct all depths to the equivalent depth in the unburned state, where the soil mineral surface was = 0 cm. This allows the soil temperature for any individual reference depth to be directly compared pre- and post-fire in order to assess microclimate changes as a result of fire.

In order to account for differences in the organic layer, soil temperature probe depths were standardized relative to the surface of mineral soil by subtracting the average organic layer thickness for each site from the actual probe depths. Standard depth temperatures were then interpolated using spline functions of the whole soil temperature profile for each 30-minute interval after QA/QC of the data. Reported depth also needed to take into account any soil subsidence that may have changed sensor depth relative to the surface of the soil. To measure changes in subsidence, the distance from top of the probe to the soil surface was measured annually when the data was collected. In cases where there was displacement, the change in depth was assumed to be gradual between measurements; a linear regression was used to correct sensor depths between the two annual measurements. Standardization of depth across burned and unburned sites revealed that combustion of the SOL increased temperatures across the entire soil profile, including at depths that were permafrost in the unburned stand (Figure 16).

Based on our analysis of equivalent soil temperature, burning increased the temperature environment integrated across the entire profile (5 to 150 cm) across all three landscape positions (Figure 17). This effect is a result of combustion of the surface organic layer, which acts as an insulator in the unburned forest. The difference in temperature was greatest in the floodplain (WC) where soils are generally moist and cold when covered with the organic layer. The difference in temperature was smallest at SC where the fire was most recent and the unburned soil temperatures were slightly higher than the other two unburned sites. In two of three sites, the integrated soil temperature moved from below to above zero temperature, indicating significant degradation of permafrost. Across all the sites, there was a statistically significant relationship between air temperature and soil temperature across the measurement period, although this effect was slightly weaker in the burned sites. (Spearman's rank correlation; burned sites, $p=0.097$; unburned sites, $p=0.048$). There was a seasonality to the difference in temperature: summer differences between burned and unburned sites integrated for the soil surface (5 to 25 cm) were approximately 5.0-7.5 degrees C, whereas winter temperature differences were 0.5-2.5 degrees C (Figure 16). Differences at deeper depths (80 to 150 cm) were closer: summer differences were 2.5-5.0 degrees C whereas winter differences were 1.5-4.0 degrees C (Figure 18).

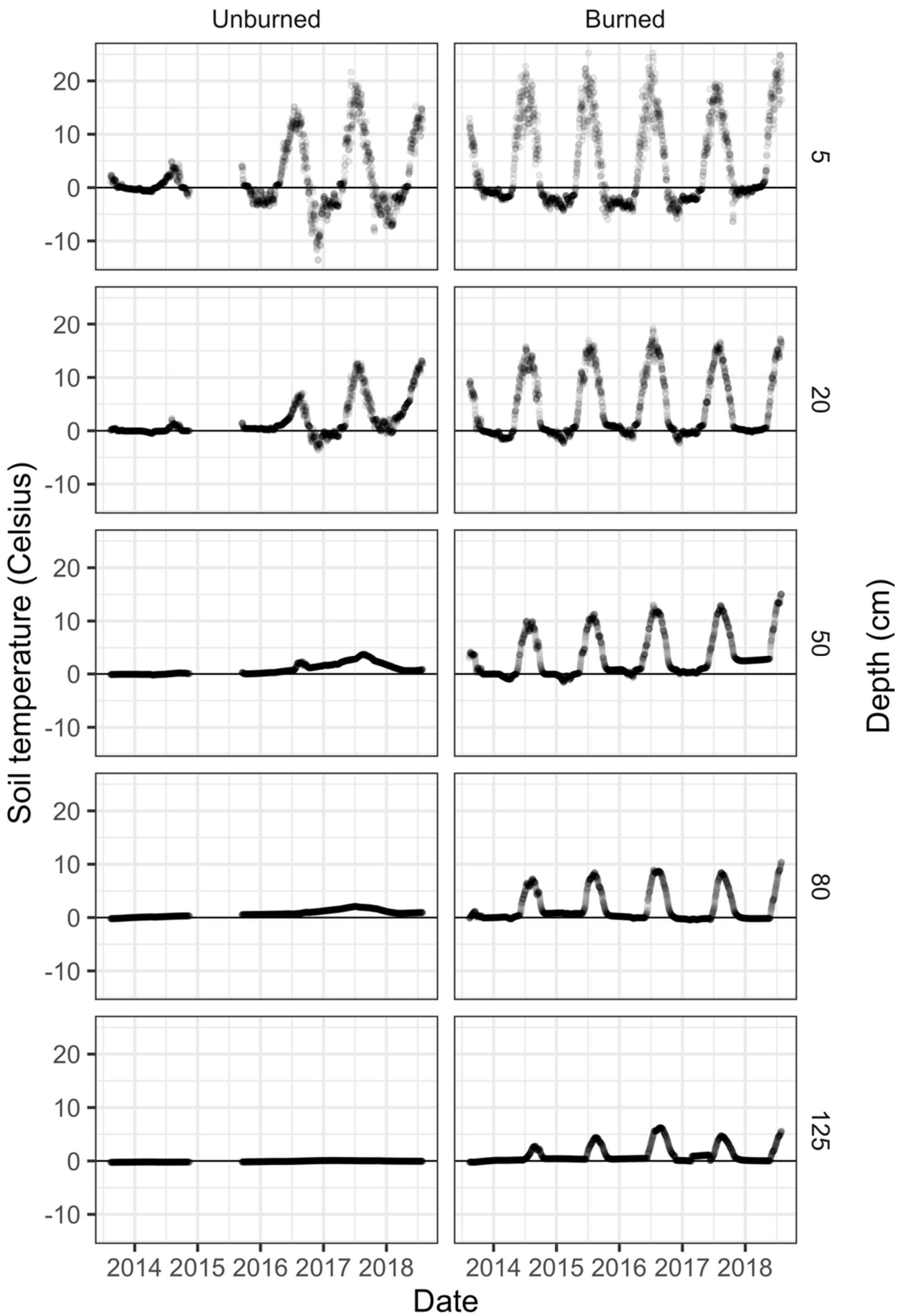


Figure 16. *Stuart Creek mean daily soil temperatures at depths 5, 20, 50, 80, and 125 cm for unburned and burned sites. Depths are standardized to unburned moss/soil surface = 0 cm.*

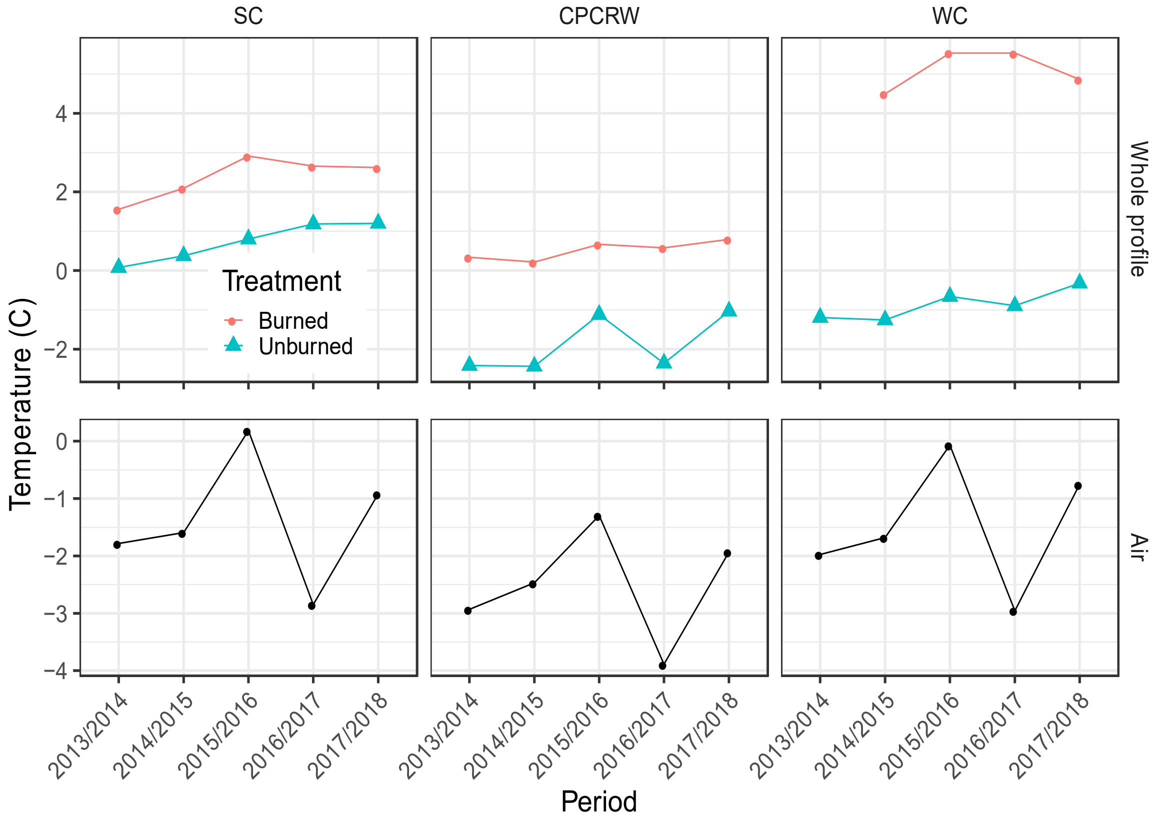


Figure 17. *Mean annual (October to September of the following calendar year) temperature for the whole soil profile (5 to 150 cm) at each site and burn treatment. Means were weighted by temperature probe depth to account for disparity in profile depth increments. Air temperatures for CPCRW and WC are from meteorological stations at the sites; SC air temperatures are from Eielson Airforce Base airport, Alaska.*

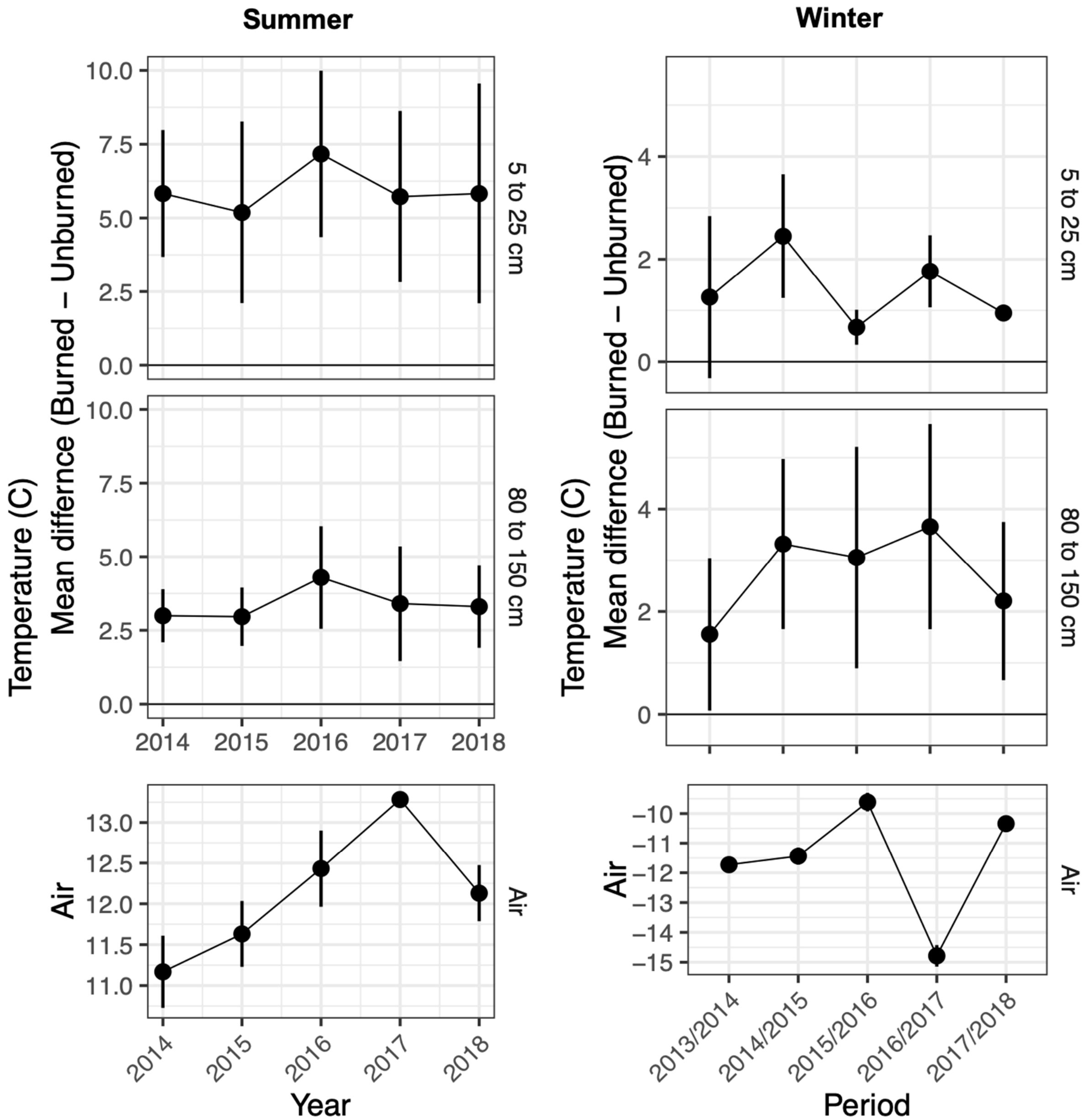


Figure 18. *Mean temperature differences (burned-unburned) for shallow (5-35 cm) and deep (80-150 cm) soils. Temperatures for each soil profile are averaged across depths and sites. (left panel) Summer: May-October; (right panel) Winter: October-May of the following calendar year. Air temperatures are from the nearest meteorological station to each site.*

Conclusions

All of our analyses are finished, and we are in the process of summarizing our findings into manuscript format for publication. Based on the results presented, our conclusions are as follows:

- **Objective 1:** Successional trajectories, as represented by tree seedling densities, were largely established by the second summer after the 2004 fire season and have persisted with minor modifications through early succession to 2017. These trajectories show that proportional combustion losses of the SOL (a measure of fire severity) and residual organic layer depth are important predictors of post-fire trajectories, but these predictors are modified by environmental conditions such as site moisture. Based on these results, black spruce forests that are in drier landscape positions and that are severely burned are the most likely forests to undergo state change after fire to succession dominated by deciduous broadleaved tree species.

- **Objective 2:** Our second survey on the fuel management network indicates that conifer seedling recruitment continued over time in both thinned and shearbladed sites. However, deciduous seedlings continued to dominate shearbladed areas, suggesting a trajectory of deciduous tree dominance in these areas. We also observed large differences in vegetation composition and ground cover in shearbladed versus thinned and unmanaged sites in 2018. However, both thinning and shearblading increased permafrost thaw. We generally observed lower ROS in thinned than shearbladed sites and the highest ROS in unmanaged areas. In some cases, when open black spruce stands were shearbladed the ROS increased relative to the adjacent unmanaged stand. Overall, these results indicate that over long timescales fuel management by shearblading, which severely disturbs the SOL, has the most significant impact on ecosystem dynamics and may result in increased ROS in open black spruce stands. While the impacts of thinning are less substantial, this treatment has a significant impact on permafrost stability, especially if the SOL is completely or partially removed during fuel management.

- **Objective 3:** There was a consistent increase in soil temperature and degradation of permafrost in burned sites relative to unburned sites. Normalizing the temperature depth measurements to account for the combustion removal of the SOL was critical for understanding the temperature changes for specific soil layers. There were correlations between interannual soil temperature and air temperature with unburned sites having slightly higher correlations. Despite interannual fluctuations in temperature, there was an overall increase in soil temperatures over time. Differences in soil temperatures increased more for the surface soil as compared to the deeper soil, especially during the summer. During the winter these differences were reversed, with higher differences at deeper soil depths than at the surface. Most of the increase in soil temperature and degradation of permafrost occurred immediately following fire, with site attributes affecting overall warming to a greater degree than time after fire.

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Appendices

Appendix 1: Scientific/Technical Publications

Jean, M., Melvin, A. M., Mack, M. C., & Johnstone, J. F. (2019). Broadleaf Litter Controls Feather Moss Growth in Black Spruce and Birch Forests of Interior Alaska. *Ecosystems*.

Jean, M., Mack, M. C., & Johnstone, J. F. (2018). Spatial and temporal variation in moss-associated dinitrogen fixation in coniferous- and deciduous-dominated Alaskan boreal forests. *Plant Ecology*. **219**: 837–851.

Melvin, A. M., Celis, G., Johnstone, J. F., McGuire, A. D., Genet, H., Schuur, E. A. G., … Mack, M. C. (2018). Fuel-reduction management alters plant composition, carbon and nitrogen pools, and soil thaw in Alaskan boreal forest. *Ecological Applications*. **28**: 149–161.

Alexander, H. D., & Mack, M. C. (2017). Gap regeneration within mature deciduous forests of Interior Alaska: Implications for future forest change. *Forest Ecology and Management*. **396**: 35–43.

Finger, R. A., Turetsky, M. R., Kielland, K., Ruess, R. W., Mack, M. C., & Euskirchen, E. S. (2016). Effects of permafrost thaw on nitrogen availability and plant–soil interactions in a boreal Alaskan lowland. *Journal of Ecology*. **104**: 1542–1554.

Jean, M., Alexander, H. D., Mack, M. C., & Johnstone, J. F. (2017). Patterns of bryophyte succession in a 160-year chronosequence in deciduous and coniferous forests of boreal Alaska. *Canadian Journal of Forest Research*. **47**: 1021–1032.

Mack, M.C. (2017) Changing disturbances regimes in the warming Arctic. In *Snow Water Ice and Permafrost in the Arctic (SWIPA): Climate Change and the Cryosphere*. pp: 124-138.

Turetsky, M. R., Baltzer, J. L., Johnstone, J. F., Mack, M. C., McCann, K., & Schuur, E. A. G. (2017). Losing Legacies, Ecological Release, and Transient Responses: Key Challenges for the Future of Northern Ecosystem Science. *Ecosystems*. **20**: 23–30.

Walker, X. J., Mack, M. C., & Johnstone, J. F. (2017). Predicting Ecosystem Resilience to Fire from Tree Ring Analysis in Black Spruce Forests. *Ecosystems*. **20**: 1137–1150.

Oral and poster presentations

Mack, M.C. Identifying Indicators of State Change and Forecasting Future Vulnerability in Alaskan Boreal Forests. 5/29/2017. Oral presentation for SERDP and ESTCP Webinar Series.

Mack, M.C. Identifying Indicators of State Change and Forecasting Future Vulnerability in Alaskan Boreal Forests. 11/28/17, Washington DC, Oral presentation for SERDP and ESTCP Annual Symposium.

Mack, M.C. Identifying Indicators of State Change and Forecasting Future Vulnerability in Alaskan Boreal Forests. 11/27/18, Washington DC, poster presentation for SERDP and ESTCP Annual Symposium.

Archived data

Johnstone J., M. C. Mack, T. N. Hollingsworth, F. S. Chapin. 2018. Alaska 2004 Burns: Densities of tree seedlings after fire; measured in 2006, 2008, 2011, and 2017 by Joint Fire Science Program. Environmental Data Initiative.https://doi.org/10.6073/pasta/076b1f5ec550876ff2829e5ca21a64a1. Dataset accessed 6/17/2019.

Melvin A. M., M. C. Mack. 2017. Interior Alaska managed sites: soil organic layer and mineral soil nutrient concentrations and additional characteristics measured one time in either summer 2012 or 2013. Environmental Data Initiative. https://doi.org/10.6073/pasta/77836c63a314214755662e4c517c040d. Dataset accessed 6/17/2019.

Melvin A. M., M. C. Mack. 2017. Interior Alaska managed sites: seedling data measured one time in either summer 2011, 2012, or 2013. Environmental Data Initiative.https://doi.org/10.6073/pasta/001c6a367666a831e259826c0ca1d87c. Dataset accessed 6/17/2019.

Melvin A. M., M. C. Mack. 2017. Interior Alaska managed sites: seedling data measured one time in either summer 2011, 2012, or 2013. Environmental Data Initiative.https://doi.org/10.6073/pasta/001c6a367666a831e259826c0ca1d87c. Dataset accessed 6/17/2019.

Melvin A. M., M. C. Mack. 2017. Interior Alaska managed sites: downed woody debris measured one time in either summer 2012 or 2013. Environmental Data Initiative.https://doi.org/10.6073/pasta/ff93dca5559162842d4aa77d41f55502. Dataset accessed 6/17/2019.

Melvin A. M., M. C. Mack. 2017. Interior Alaska managed sites: tree inventory measured one time in either summer 2012 or 2013. Environmental Data Initiative. https://doi.org/10.6073/pasta/0ea09774a8e7d4c78fcc49f8505182d3. Dataset accessed 6/17/2019.

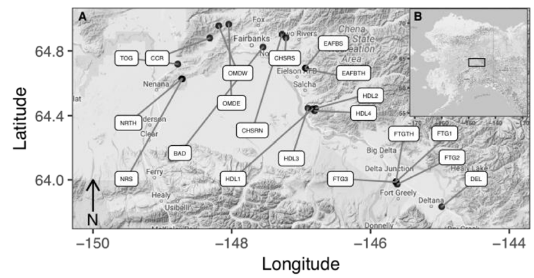
Appendix 2: Fact Sheet for Alaskan Land Managers

Nov. 2017
A. Melvin, M. Mack & R. Jandt

Ecological Impacts of Alaskan Forest Fuel Treatments

Clearing and forest thinning are increasingly seen as strategies to protect private property and infrastructure from boreal wildfires. Property sited in natural spruce-dominated forests are often considered high risk due to the intensity of fires in this fuel type when it burns. Although vegetation treatments can reduce fire potential, they may have unintended ecological effects, but there has been little published on possible impacts—especially for Alaska. So the recent publication (Melvin, *et al.* 2017) of a study on sites managed as fuel treatments by an interdisciplinary team of researchers is an important addition to regional management resources. In fact, it probably represents the FIRST paper specifically on how fuel-reduction affects carbon and nutrient pools, permafrost thaw, and successional trajectories. However, the authors also summarize some published impacts from related management actions like boreal logging and bulldozed firelines.

Thinning treatment near Delta, Alaska, 4 years post-treatment (J. Hrobak, 2006).



Interior Alaska Study Location

The 19 sites in the study (above: Figure 1 from Melvin, *et al.* 2017) are managed by numerous Alaska agencies covering a large swath from Nenana to Deltana, and were sampled at various ages from 2-12 years post-treatment. The most common operational fuel reduction treatments use hand thinning or winter bulldozer clearing—called "shearblading" (intended to clear trees and brush but leave the organic soil and moss mat relatively intact). Mechanical thinning tends to be faster and less expensive than hand thinning, but comes with other trade-offs. A concurrent research project (Little, et al. JFSP 14-05-01-27) is studying the duration of effectiveness and economic impacts of various forest treatments in Alaska.

Shear-bladed experimental fuel treatment at Nenana Ridge site (D. Haggstrom, 2006).

Summary of Results

It should come as no surprise that shearblading eliminated virtually all of the above-ground storage of carbon (C) during the first decade post-treament, as these treatments aim to reduce biomass. C storage in the organic layers of moss were reduced by about half in shearbladed treatments. Thinned treatments were intermediate in carbon storage between shearbladed and untreated stands, but notably most of the organic layer carbon storage—the largest C reservoir-was mostly preserved.

Deciduous seedlings (mostly birch) were most numerous in shearbladed stands (averaging 6/m^2) whereas in thinned stands conifer/deciduous seedlings were about equal in numbers. Unmanaged stands had by far the most conifer seedlings (or “layering” trees)—just over 1/m^2. An important finding was the documentation of impacts of seasonal thaw depth (active layer): thinned stands were thawed about 13 cm deeper than adjacent undisturbed forest. In contrast, shearbladed treatments were thawed an average of 46 cm deeper by mid-July or August. The management implication is that shearblading may not be the treatment of choice where adjacent infrastructure or watersheds rely on keeping permafrost intact. It is also important to remember that climate warming itself is warming permafrost and some areas may be very close to thaw temperature (Lara, et al. 2016)—so fuel reduction treatments are not the only threat to destabilization, but it would be negligent not to consider potential impacts while planning fuel management activities. Differences in understory plant composition were also documented, which will be important for determining potential positive or negative impacts on wildlife habitat and subsistence activities.

Management Implications

While fuel reduction treatments are important for lessening wildland fire risks, they do represent a novel disturbance to the forest inducing changes in forest structure, carbon storage and nutrient dynamics, and permafrost stability. These impacts should be considered in planning treatments.

Full Citation: Melvin, A. M., Celis, G., Johnstone, J. F., McGuire, A. D., Genet, H., Schuur, E. A. G., Rupp, T. S. and Mack, M. C. (2017), *Fuel-reduction management alters plant composition, carbon and nitrogen pools, and soil thaw in Alaskan boreal forest.* Ecol Appl. Accepted Author Manuscript. doi:10.1002/eap.1636

Firefighters burn debris piles in a thinned fuelbreak strip to protect a neighborhood outside Ft. Wainwright’s Badger Road gate in the fall (R. Jandt, 2000).

Citation

Lara, M.J.; Genet, H., McGuire, A.D.; Euskirchen, E.S.; Zhang, Y; Brown, D.R.N.; Jorgenson, M.T.; Romanovsky, V; Breen, A; Bolton, W.R. 2016. *Thermokarst rates intensify due to climate change and forest fragmentation in an Alaskan boreal forest lowland.* Global Change Biology 22(2):816-829.

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Appendix 3: Fuel Model Guide to Alaska Vegetation

Table 1. Crosswalk from Alaska Fuel Types to US fuel models and CANFBP fuel types.[1]

Alaska Fuel Type Number	Alaska Fuel Type Name	FBFM 40	FBFM 13	CANFBP	Alternate Models (See text for usage)
1	Sitka Spruce – Hemlock Forest	TL1	8	C- 5 (closed), C-7 (open)	
2	Closed White Spruce Forest	TU2 upland, TU1 riparian	9	C-3	TU3, FM10
3	Closed Black Spruce Forest and Closed Mixed Black Spruce – White Spruce Forest	TU3	9 adj[2]	C-2	TU1, SH5
4	Open White Spruce Forest	TU5	10	C-3	TU4, TU1
5	Coastal Boreal Transition Open White Spruce – Lutz Spruce Forest	TU1	8	D-1/D-2 or M-1/M-2 w/ low conifer	TU1, TU3, GR4 , FM9
6	Open Black Spruce Forest and Open Mixed Black Spruce – White Spruce Forest	TU4	9 adj[2]	C-2	TU5, TU3, SH5
7	Black Spruce – Tamarack Forest	TU2	10	C-1	
8	Coastal Woodland Rainforest	TL1	8	M-2 w/low conifer % or D-2	
9	White Spruce Woodland and Mixed Black Spruce – White Spruce Woodland with Shrubs	SH2	10	M-2/25% conifer	GR
10	Black Spruce Woodland with Tussocks	GS2	5	C-1	GS3, O-1a/b
11	Black Spruce and/or White Spruce Woodland with Lichen	TU4	9 adj[2]	C-1	GR2
12	Black Spruce Woodland with *Sphagnum* Moss	TU2	10	C-1	TU1, SH2, TU4
13	Closed Black Cottonwood or Balsam Poplar Forest and Closed Red Alder Forest	TL2	8	D-1/D-2	
14	Closed Paper Birch Forest and Closed Quaking Aspen Forest	TU1	8	D-1/D-2	
15	Open Paper Birch Forest	TU1	8	D-1/D-2	M-1, M-2, TU3
16	Open Quaking Aspen Forest	TU1	8	D-1/D-2	
17	Open Balsam Poplar or Black Cottonwood Forest	TL2	8	D-1/D-2	TU1
18	Woodland Paper Birch and Woodland Balsam Poplar	SH1	8	O-1a/b	GR1, SH2
19	White or Black Spruce with Paper Birch and/or Aspen	TU5	10	M-2/50% conifer	TU5, M-1/2
20	White Spruce with Balsam Poplar and Paper Birch	TU1	8	M-2/25% conifer	M-1

Table 1. Crosswalk from Alaska Fuel Types to US fuel models and CANFBP fuel types.[1]

Alaska Fuel Type Number	Alaska Fuel Type Name	FBFM 40	FBFM 13	CANFBP	Alternate Models (See text for usage)
21	Dwarf Tree Mountain Hemlock Scrub and Dwarf Tree Alpine Spruce Shrub	SH1	8	O-1a	
22	Dwarf Tree Black Spruce Scrub	GS1	9	C-1	C-2, GS2, SH5
23	Closed Tall Alder and Closed Tall Willow	TL2	8	D-1/D-2	TU1,SH2, M-1/2
24	Closed Tall Shrub Birch	SH3	9	M-1/M-2	SH2
25	Tall Shrub Swamp	Removed: Low, patchy occurrence, few examples			
26	Open Tall Alder and/or Willow	TU1	8	D-1/D-2	M-1, GS1
27	Open Tall Shrub Birch and Open Tall Shrub Birch – Willow	SH3	9	M-1/M-2	TU4, GS1
28	Closed Low Shrub Birch and Closed Low Shrub Birch – Willow and Closed Low Ericaceous Shrub	SH2	9	D-1/D-2	TU4
29	Closed Low Willow and Closed Low Alder – Willow	SH2	9	D-1/D-2	TU1, M-1
30	Open Low Mixed Shrub – Sedge Tussock Tundra and Open Low Mixed Shrub – Sedge Tussock Bog	GR4	1	O-1a/b	GR5, GS3, SH2
31	Open Low Mesic Shrub Birch – Ericaceous Shrub	GR2	5	O-1a/b	SH7
32	Open Low Shrub Birch – Ericaceous Shrub Bog and Open Low Shrub Birch – Willow	GS2	5	O-1a/b	
33	Open Low Willow and Open Low Sweetgale	SH1	8	O-1a	GR1
34	Open Low Alder and Open Low Alder – Willow	GS1	5	O-1a/b	SH2, FM1
35	Sagebrush – Grass and Grass – Juniper	GR1	8	O-1a/b	FM10
36	Dwarf Shrub Tundra	GS1	5	O-1a/b	FM1
37	*Elymus*	GS2	5	O-1a/b	
38	Grass – Shrub	GS2	5	O-1a/b	GS1
39	Bluejoint (*Calamagrostis*)	GR4	2	O-1a/b	GR7, FM3
40	Bluejoint – Shrub and/Bluejoint – Herb	GR2	6	O-1a/b	GR1
41	Tussock Tundra	GR4	1	O-1a/b	GS3, SH2
42	Mesic Sedge – Grass Meadow or Tundra and Mesic Sedge – Herb Meadow or Tundra	GS1	5	O-1a/b	

Table 1. Crosswalk from Alaska Fuel Types to US fuel models and CANFBP fuel types.[1]

Alaska Fuel Type Number	Alaska Fuel Type Name	FBFM 40	FBFM 13	CANFBP	Alternate Models (See text for usage)
43	Sedge – Willow Tundra and Sedge – *Dryas* Tundra	GR1	5	O-1a/b	FM1
44	Sedge – Birch Tundra	GR2	6	O-1a/b	GR4, GS3
45	Wet Meadow Tundra	GR1	10	O-1a/b	NB6[3]
46	Wet Sedge – Grass Meadow or Marsh	GR1	5	O-1a/b	NB6,[3] FM1
47	Wet Sedge Meadow or Bog and Wet Sedge – Shrub Meadow or Bog	GR1	9	O-1a/b	NB6,[3] FM2
48	Dry Species – Non Burnable	NB7[3]	99		
49	Wet Species – Non Burnable	NB7[3]	99		
50	Mesic Forb Herbaceous	GR1	5	O-1a/b	FM1
51	Foliose and Fruticose Lichen	GR1	2	O-1a/b	GR2, GR3
52	Crustose Lichen	NB9	99		
53	Aquatic Herbaceous	NB8	99		
54	Standing Dead Beetle–Kill Spruce Forest	SB2/SB3	12	M-3	
55	Heavy Stem Breakage/Downed and Jack–Straw Spruce and Aged Post–Mortality Beetle–Kill Forest	SB3	13	C-3	
56	Closed Spruce Forest with Moderate Downed Beetle Kill and Mixed Spruce and Hardwood Forest with Moderate Beetle Kill	TU5	10	M-3	
57	Post–Timber Harvest Areas with Bluejoint Grass and Logging Slash Fuel Beds	GR7	3	O-1a/b	

[1]See Appendix 1 for comparison of fuel model assignments between the 2008 and 2018 guide versions.

[2]The FBFM 13 fuel model "9 ADJ" refers to Norum's (1982) calibration for Alaska Black Spruce. Rate of spread is 1.2 times that predicted for fuel model 9 (Albini 1976, Anderson 1982), and flame length is that predicted for fuel model 5.

[3]NB6 is a custom fuel model referring to areas covered by hydric vegetation types that do not carry fire; NB7 refers to upland (dry species) vegetation types that do not carry fire.

Table 2. Alaska fuels guide classification key for US fuel models and CANFBP fuel types.[1]

Form	Composition	Canopy/Structure	Alaska Fuel Type	40	13	CANFBP
Forest	**Conifer** (over 75% of tree cover contributed by needle leaf species)	Closed (60%+)	(1) Sitka Spruce – Hemlock Forest (2) Closed White Spruce Forest (3) Closed Black Spruce Forest and Closed Mixed Black Spruce – White Spruce Forest	TL1 TU2/TU1 TU3	8 9 9 adj	C-5/C-7 C-3 C-2
		Open (25 – 59%)	(1) Sitka Spruce – Hemlock Forest (see above) (4) Open White Spruce Forest (5) Coastal Boreal Transition/Open White – Lutz Spruce (6) Open Black Spruce and Open Mixed Black Spruce – White Spruce Forest (7) Black Spruce – Tamarack Forest	TL1 TU5 TU1 TU4 TU2	8 10 8 9 adj 10	C-5/C-7 C-3 D-1/D-2 C-2 C-1
		Woodland (<25%)	(7) Black Spruce – Tamarack Forest (see above) (8) Coastal Woodland Rainforest (9) White Spruce and Mixed Black Spruce – White Spruce Woodland w/ Shrubs (10) Black Spruce Woodland w/ Tussocks (11) Black Spruce and/or White Spruce Woodland w/ Lichen (12) Black Spruce Woodland w/ *Sphagnum* Moss	TU2 TL1 SH2 GS2 TU4 TU2	10 8 10 5 9 adj 10	C-1 D-1/D-2 M-1/M-2 C-1 C-1 C-1
	Deciduous (over 75% tree cover contributed by broadleaf species)	Closed (60%+)	(13) Black Cottonwood – Balsam Poplar and Red Alder (14) Paper Birch Forest and Quaking Aspen Forest	TL2 TU1	8	D-1/D-2
		Open (25 – 59%)	(15) Open Paper Birch Forest (16) Open Quaking Aspen Forest (17) Open Balsam Poplar or Black Cottonwood Forest	TU1 TU1 TL2	8	D-1/D-2
		Woodland (<25%)	(18) Woodland Paper Birch and Woodland Balsam Poplar	SH1	8	O-1a/b
	Mixed (25 – 75% admixtures of conifer and broadleaf)	Closed (60%+)	(19) Spruce – Paper Birch – Aspen (20) White Spruce – Balsam Poplar – Paper Birch	TU5 TU1	10 8	M-1/M-2
		Open (25 – 59%)	(19) White or Black Spruce w/ Paper Birch or Aspen (20) White Spruce w/ Balsam Poplar and Paper Birch	TU5 TU1	10 8	M-1/M-2
		Woodland (<25%)	None really described			
Disturbed Lands	Beetle-Kill Spruce		(54) Standing Dead Beetle-kill Spruce Forest (55) Heavy Stem Breakage Downed/Jack-Straw Spruce (56) Closed Spruce/Mixed w/ Mod Downed Beetle-kill	SB2/SB3 SB3 TU5	12 13 10	M-3/M-4 C-3 M-3/M-4
	Post-Harvest		(57) Post-Harvest Bluejoint Grass and Logging Slash	GR7	3	O-1a/b
	Wildfire	1 – 2 yrs since burn	Little fine fuel, spread rate negligible to very slow	TL1	8	D-2
		3 – 5 yrs since burn	Live fuels increasing, still inconsistent creeping spread	TU1	8	D-1/D-2
		6+ yrs since burn	Fine fuels increasing, low-moderate spread with wind	SH2	8	O-1a

Table 2, continued. Alaska fuels guide classification key for US fuel models and CANFBP fuel types.[1]

Form	Composition	Canopy/Structure	Alaska Fuel Type	40	13	CANFBP
Scrub	**Dwarf Tree Scrub** (10%+ dwarf trees cover)	Closed (60%+)	(21) Dwarf Tree Mountain Hemlock and Dwarf Tree Alpine Spruce Scrub	SH1	8	O-1a
		Open (25 – 59%)	(21) Dwarf Tree Mountain Hemlock and Dwarf Tree Alpine Spruce Scrub	SH1	8	O-1a
			(22) Dwarf Tree Black Spruce Scrub	GS2	9	C-2
		Woodland (<25%)	(22) Dwarf Tree Black Spruce Scrub	GS1	9	C-1
	Tall Scrub (shrubs at least 1.5 m or 5 ft tall)	Closed (75%+)	(23) Closed Tall Alder – Willow	TL2	8	D-1/D-2
			(24) Closed Tall Birch Shrub	SH3	9	M-1/M-2
		Open (25 – 74%)	(26) Open Tall Alder and/or Willow	TU1	8	D-1/D-2
			(27) Open Tall Birch/Birch – Willow	SH3	9	M-1/M-2
	Low Scrub (shrubs 20 cm to 1.5m)	Closed (75%+)	(28) Closed Low Birch/Birch – Willow/Ericaceous Shrub	SH2	9	D-1/D-2
			(29) Closed Low Willow/Alder – Willow	SH2	9	D-1/D-2
		Open (25 – 74%)	(30) Open Low Mixed Shrub – Sedge Tussock Tundra/Bog	GR4	1	O-1a/b
			(31) Open Low Mesic Shrub Birch – Ericaceous Shrub	GR2	5	O-1a/b
			(32) Open Low Birch Ericaceous Shrub Bog and Open Low Shrub Birch – Willow Shrub	GS2	5	O-1a/b
			(33) Open Low Willow/Sweetgale	SH1	8	O-1a/b
			(34) Open Low Alder/Alder – Willow	GS1	5	O-1a/b
			(35) Sagebrush – Grass and Grass – Juniper	GR1	8	O-1a/b
	Dwarf Scrub	*Dryas*	(36) Dwarf Shrub Tundra	GS1	5	O-1a/b
Herbaceous	**Graminoid**	Dry	(37) *Elymus*	GS2	5	O-1a/b
			(38) Grass – Shrub	GS2	5	O-1a/b
		Mesic	(39) Bluejoint (*Calamagrostis*)	GR4	2	O-1a/b
			(40) Bluejoint – Shrub/Herb	GR2	6	O-1a/b
			(41) Tussock Tundra	GR4	1	O-1a/b
			(42) Mesic Sedge – Grass – Herb Meadow Tundra	GS1	5	O-1a/b
			(43) Sedge – Willow and Sedge – *Dryas* Tundra	GR1	5	O-1a/b
			(44) Sedge – Birch Tundra	GR2	6	O-1a/b
		Wet	(45) Wet Meadow Tundra	GR1	10	O-1a/b
			(46) Wet Sedge – Grass Meadow Marsh	GR1	5	O-1a/b
			(47) Wet Sedge Meadow – Bog – Shrub	GR1	9	O-1a/b
	Forbs	Mesic	(50) Mesic Forb Herbaceous	GR1	5	O-1a/b
	Bryoid	Lichens	(51) Foliose and Fruticose Lichen	GR1	2	O-1a
			(52) Crustose Lichen	NB9	99	N/A
	Aquatic	Wet	(53) Aquatic Herbaceous	NB8	98	N/A

[1]See Appendix 1 for table displaying fuel model assignments from the 2008 version of the guide.