THESIS

THE EFFECTS OF POST-FIRE LOGGING ON MICROCLIMATE AND SURFACE FUELS

Submitted by

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ABSTRACT

THE EFFECTS OF POST-FIRE LOGGING ON MICROCLIMATE AND SURFACE FUELS

Wildfire is increasing in size and severity in forests of the western US, driven by climate change and land management practices during the 20th century. Altered fire regimes have resulted in a greater need for knowledge on best practices for managing burned landscapes, especially in instances where a return to a previous forested ecosystem is desired. Our study location was the Spring Creek Fire in the Rockies of Colorado, where we examined soil moisture, soil temperature, and soil disturbance as well as surface fuel loading and understory vegetation recovery in areas that burned at low and high severity, a subset of which received post-fire logging treatments. Two years post-fire, we found no difference in understory vegetation response; however, logged sites demonstrated lower daily average and minimum soil moisture and higher fuel loading across most fuel size classes, and were more likely to show evidence of compaction, erosion, and rutting. This suggests that post-fire logging may create unfavorable conditions for tree regeneration while increase short term site susceptibility to reburns. Careful consideration should be taken when conducting post-fire logging to prevent detrimental ecological effects.

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CHAPTER 1

BURNING, FELLING, THEN NO TELLING:
EXPLORING UNCERTAIN ECOLOGICAL TRAJECTORIES AFTER POST-FIRE LOGGING

1.1 Introduction

Forests worldwide are facing unprecedented threats from changing disturbance regimes. Throughout the 20th century, western US forests were impacted by logging, grazing, and fire suppression (Kaufmann et al. 2000, Veblen et al. 2000), but the paramount threat affecting these forests today is how anthropogenic climate change has altered disturbance regimes and post-disturbance recovery (Abatzaglou and Williams 2016). Pressures from a changing climate can alter how historical disturbances manifest (Seidl et al. 2017) while simultaneously upending forest recovery (Johnstone et al. 2016). Climate warming has resulted in increased drought conditionsp, which combined with high temperatures can create more favorable opportunities for catastrophic wildfire, which has increased severity and extent in western U.S. forests over the past several decades (Abatzaglou and Williams 2016, Parks et al. 2016, Westerling 2016). Further, climate warming can impact post-fire recovery trajectories, through limiting tree regeneration (Davis et al. 2019a, Coop et al. 2020).

Thus, we propose a testable conceptual model for site specific tree regeneration (Figure 1). The model incorporates environmental and climatic factors influencing tree recruitment following wildfire and post-fire management. We consider seed availability and post-fire seedbed temperature and moisture conditions as essential requirements that need to be met for successful tree recruitment to occur, although other influential factors driving recruitment success include topographical characteristics such as elevation, aspect, and fire severity. Following disturbance, distance to seed source has been shown across much of the literature to be a primary driver of tree regeneration (Chambers et al. 2016, Kemp et al. 2016, Welch et al. 2016, Stevens-Rumann and Morgan 2019), and is primarily affected by fire severity. In recent years, the percentage of burned area that has burned at high severity has increased, due in part to climate change (Parks and Abatzaglou 2020). After fire, if seed trees are close enough to provide seed, then soil moisture and soil temperature are key factors driving successful establishment. Many western conifers are more sensitive to drought and temperature extremes in the germinant stage, with seedling survival largely dependent on soil moisture and/or soil temperature falling within a narrow favorable range (Bell et al. 2014, Kueppers et al. 2017, Petrie et al. 2016, Hankin et al. 2019). Soil moisture and temperature are influenced by many factors, including slope, aspect, elevation, and canopy cover, and microsite conditions can vary even within a stand. Where temperature and precipitation no longer fall within the necessary parameters for tree recruitment of previously dominant tree species, ecosystem conversions are likely (Stevens-Rumann et al. 2018, Davis et al. 2019a, Coop et al. 2020). In places where climate change has pushed temperature and moisture conditions beyond thresholds necessary for successful recruitment, mature trees may persist in conditions where seedlings will not (Johnstone et al. 2016). Climate effects on regeneration can manifest in both positive and negative ways, as reduced snowpack may promote earlier seasonal germination of seedlings, while hotter and drier summers can lead to increased seedling mortality (Werner et al. 2019). The rate of change in suitable climate conditions and wildfire frequency may outpace species adaptation and capacity for range shift.

When forests burn, the resulting landscapes differ from their unburned counterparts on multiple biotic and abiotic levels. Wildfire can remove canopy cover, which increases light availability at the surface, and can contribute to increased soil-warming and drying (Davis 2019b, Marcolin et al. 2019). The conversion of accumulated litter, duff, and understory vegetation to ash alters nutrient inputs, while fire can expose mineral soil and alter edaphic characteristics such as soil structure, water infiltration and available water capacity, and increase hydrophobicity and erodibility of soils (Neary et al. 2005). As adult trees perish, new seedlings are required to maintain a forested state (Johnstone et al. 2016). When soil moisture and temperature conditions are unsuitable for germination and survival, adult trees may persist while new recruitment fails (Davis et al. 2019a). The current habitat occupied by many western tree species is largely a product of temperature and moisture variables, with models indicating that climate change will lead to significant shifts in viable habitat for these species over the coming decades (McKenney et al. 2007, Gray and Hamann 2013, Bell et al. 2014, Rehfeldt et al. 2014, Campbell and Shinneman 2017, Davis et al. 2018). While canopy cover can help buffer climate changes for the understory (Davis et al. 2019b), canopy loss such as that caused by wildfire can reduce recruitment (Dobrowski et al. 2015). Canopy removal and subsequent increased solar exposure may lead to microsite soil temperature and soil moisture conditions outside of the window for successful germination or seedling survival; in extreme cases this may lead to ecosystem type conversion shifts from mixed conifer to deciduous shrubland by advantaging the survival and propagation of re-sprouting species adapted to different climatic conditions (Coop et al. 2016, Coop et al. 2020). Species located at trailing edges of their ideal climate envelope are particularly vulnerable to conversion to non-forested states (Parks et al. 2019). Vegetation conversions may be exacerbated or reinforced with subsequent disturbances whether the disturbance is natural in cause, e.g. subsequent wildfire or flooding, or human caused, e.g. post-fire logging, prescribed fire, or invasive species introduction.

The desire to manage post-fire landscapes has increased alongside the expanding presence of wildfire in the western US. One common form of forest management following wildfire is post-fire logging, often termed "salvage" logging, in which trees burned by the fire are felled and chipped or masticated on site, and/or removed and sold as timber. In general, logging disturbs ecosystems differently than wildfire, and in sequence there is a potential for disturbance interaction. These interactions occur when one disturbance (e.g. wildfire) is followed by another (e.g. logging) in the same location, impacting ecological resilience and recovery differently than the effects of either discrete disturbance (Paine et al. 1998, Buma 2015, Johnstone et al. 2016). Post-fire logging is most frequently implemented to recover lost timber value (Peterson et al. 2009), although other justifications for intervention include its perceived value as a fuels reduction treatment to mitigate risk of future fire, and as a means of aesthetic enhancement (Müller et al. 2018). The impact of post-fire logging on ecosystems varies with many factors, including forest type, seasonal timing and method of tree harvest, and elapsed time since fire (Beschta et al. 2004, Keyser et al. 2009, Leverkus et al. 2020). The effects of post-fire logging on surface fuel loading and continuity vary in the research and differ based on harvest method and intensity. With the logging method of *in situ* hand-felling and limbing, some studies found short term increases in fuels (Donato et al. 2006, McIver and

Ottmar 2007, Keyser et al. 2009, Donato et al. 2013, Campbell et al. 2016, Leverkus et al. 2020), converging with fuel loadings at unlogged sites between 5-13 years after logging (Campbell et al. 2016, McIver and Ottmar 2018, Leverkus et al. 2020). Studies examining sites where whole tree harvesting methods were employed found little to no change in fuels (Keyser et al. 2009, Ritchie et al. 2013). Modeling future fire behavior in post-fire logged sites also shows mixed results (Monsanto and Agee 2008, Fraver et al. 2011). Surface fuel accumulation is of particular interest to forest managers seeking the return of forests to burned landscapes, given an increasing body of research indicating that subsequent burns can instigate or hasten transitions to non-forested states (Coop et al. 2016, Johnstone et al. 2016, Stevens-Rumann and Morgan 2016, Tepley et al. 2018, Whitman et al. 2019, Coop et al. 2020). The effects of post-fire logging extend beyond simple changes to fuel structure. For instance, where high severity wildfire reduces canopy cover, high intensity post-fire logging removes it. The process of removing trees from a burned area can also contribute to soil disturbance (e.g. compaction, erosion, topsoil displacement), which may further alter conditions at the microsite level.

Early successional regeneration following disturbance plays an outsized role in determining later ecosystem structure, particularly in ecosystems affected by changing climate (Gill et al. 2017). Initial post-disturbance soil moisture and temperature may be ameliorated by conditions created by the surface fuel accumulation resulting from tree fall and decay (e.g. nurse logs; Castro et al. 2011). Shrubs and early successional species may also play a facilitating role in later tree recruitment in areas lacking forest canopy by providing cover and altering microsite climate (Gómez-Aparicio et al. 2004, Keyes and Maguire 2005, Kennedy and Sousa 2006, Ettinger and HilleRisLambers 2017). Numerous studies demonstrate that post-fire logging can result in long-term impacts to recovering landscapes (Karr et al. 2004, Donato et al. 2006, Keyser et al. 2009, Castro et al. 2011, Blair et al. 2016). However, little is known about post-fire logging across a range of forest types within the same fire footprint, and how logging affects microsite conditions necessary for forest recovery. Given the interactions between climate change and disturbances, it is of increasing concern to scientists, land managers and stakeholders to further understand how ecosystem recovery functions in a novel climate, and how various pre- and post-disturbance management actions may influence that recovery.

To test this model, and understand the impacts of post-fire logging on fuels, microsite conditions, and regeneration across an elevational gradient, we focused our study on a fire in southern Colorado and designed it to answer the questions: (1) How does soil temperature and soil moisture differ among burn severities and areas that received post-fire logging? (2) How does short-term post-fire vegetation recovery and tree regeneration differ among severities with and without logging? (3) How does post-fire logging affect surface fuel loads, compared to unlogged sites that burned at low and high severity?

1.2 Methods

1.2.1 Study Area

The Spring Creek Fire burned 43,000 ha in 2018 in the Sangre de Cristo range and Spanish Peaks Wilderness of the Southern Rocky Mountains of Colorado, USA (Figure 2) and was the third largest fire in Colorado history at the time, and still ranks among the largest in the state. The fire burned across an elevational gradient from lower elevation shrub and grasslands up to high-elevation subalpine forests, impacting private, state and federal land. Sites ranged in elevation from roughly 2600-3000m. Environmental Site Potential (landfire.gov last accessed November 7, 2020) across our study area included: Southern Rocky Mountain Mesic (and Dry-Mesic) Montane Conifer Forest and Woodland Inter-Mountain, Basin Aspen-Mixed Conifer Forest and Woodland, *Quercus gambelii* Shrubland Alliance, and Southern Rocky Mountain Ponderosa Pine Woodland. Sites spanned many of these forest types with mixed-conifer forest characterized by Ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Psuedotsuga menziesii*), and white fir (*Abies concolor*) dominance at lower elevations. Higher elevation sites fell within two dominant forest types, the first being mixed conifer with predominantly Limber pine (*Pinus flexilis*) mixed with white fir (*Abies concolor*), and to a lesser extent, Douglas-fir, Ponderosa pine, and quaking aspen (*Populus tremuloides*). The second group of higher elevation forest was dominated by lodgepole pine (*Pinus contorta*) and quaking aspen, with occasional white fir. This variability in forest type across a relatively small (approximately 400m) elevational gradient demonstrates the forest type diversity found in this region.

1.2.2 Site Selection

Across the study area, 31 sites were stratified into three disturbance types: 1) low to moderate severity fire (<50% tree mortality) without post-fire treatments (LS, n = 9), 2) high severity burned areas (100% tree mortality) without post-fire treatments (HS, n = 11), and 3) high severity burned sites that were subsequently logged (LO, n = 11; Table 1). Severity was assessed visually according to tree mortality within the boundaries of the site, which consisted of a 0.08 ha circular plot. A total of nine blocks were established, where each contained one plot in each of the three site categories such that all were in close geographic proximity and part of the same contiguous pre-fire forested area, with similar forest type and topographical characteristics for all sites in a given block, including elevation, slope and aspect (Table 2). On average, for all sites in a block, the range of elevation was within 50 meters, slope was 5%, and aspect was 45°, though it should be noted that the greatest variation in aspect occurred on sites with slopes < 5%. For sites in aspen/lodgepole forest, no LS burned areas were present, and so two blocks of sites containing HS and LO were established. A total of 9 sites were established in 2019, including 4 sites that were measured before logging operations began in 2019 and measured again in 2020. In 2020, an additional 22 sites were established across the three disturbance types. Sites were selected to avoid areas with significant pre-fire treatment, e.g. thinning or prescribed fire, that would alter burn severity or introduce other variables into post-disturbance. Logging on private land was commissioned shortly after the fire, and all sites in this study where post-fire treatment was conducted were logged by October 2019 (Figure 3).

Burn severity was assessed as a percentage of live tree density 1-2 years post-fire on a site. Areas that burned and were subsequently logged had an average retention of 9 trees ha^{-1}. In LS and HS plots, no post-fire treatments were conducted. Logging methods varied but were largely focused on the removal of merchantable timber, with varying methods for removal or mitigation of remaining fuels.

1.2.3 Sampling Design

Each site consisted of a 0.08-ha circular plot (adapted from Ott et al. 2018), with four, 16.1m long transects oriented in the cardinal directions from plot center (Figure 4). Each plot center was staked, adult trees were tagged, and site measurements such as slope, aspect, elevation and coordinates were recorded. Distance to live seed source by species was also recorded up to 150m from plot center using a Nikon Forestry Pro range finger. Adult tree species and relative decay classes, when applicable, were noted for all trees within the circular plot. Tree seedling regeneration by species was quantified within the entire 0.08 hectare plot. Conifer seedlings were aged using whorl counts similar to Urza and Sibold (2013).

At each site, fine woody fuels were measured using modified Brown (1974) transects, where 1, 10 and 100 hour fuels (corresponding to size classes 0-0.64, 0.65-2.54, 2.55-7.62 cm) were measured along 7, 64, and 64 m total of the intersecting transects, respectively. Substrate and vegetation cover measurements were collected at 68 locations along transects, noting substrate (litter, bareground, rock) and, if present, plant functional group (graminoid, forb, shrub, tree). Litter and duff depth measurements were taken three times along each of the four transects for a total of 12 measurements and averaged per plot. Canopy cover was determined using a densitometer at 30 points along both transects and averaged per site. Coarse woody fuels (1000hr fuels; >7.62cm) were measured within a 6.9 radius subplot, where any log with a diameter exceeding 7.62cm was recorded as well as the diameter at both ends (Keane and Dickenson 2014). Distance to live seed source was measured with a laser rangefinder, with distances for the three closest living individuals averaged by species. The rangefinder limit was 150 meters, and so this was the maximum distance recorded. At sites lacking a visible seed source tree for relevant species, 150 meters was recorded as the distance.

1.2.4 Soil temperature, moisture, and disturbance

Two to three temperature sensors were installed in each of our 31 plots. The sensors used were a combination of 'HOBO Pendant Temperature Data Loggers' and '8-bit Temperature Smart Sensors,' both from Onset Computer Corporation (ONSET, Product ID: #UA-002-08, #S-TMA-M006). Sensor selection and placement was consistent within blocks, e.g. Pendant Sensors were placed at one set of LS, HS, and LO sites, and 8-bit Temperature Smart Sensors at another set, however both types of temperature sensors were installed at 7 sites in order to verify consistency of collected measurements.

Two 'Soil Moisture Smart Sensors' were installed at depths of 4 centimeters below soil surface at 4 blocks, or a total of 12 sites, with measurements for these sensors recorded using 'HOBO Weather Stations' and 'HOBO Micro Stations' (ONSET, Product ID's #SMA-M003, #H21-001, #H21-002, #H21-USB). All temperature and soil moisture sensors were programmed to record measurements at intervals of 30 minutes, beginning with their installation in the field during the early growing season (May 2020) and continuing until their removal at the end of August and early September. This created a time series of soil temperature and soil moisture measurements across site categories for the duration of the field season, and also allowed for an examination of specific daily measurements (e.g. max daily temperature, minimum daily moisture) for all sites. Data collected by HOBO Weather and Micro Stations experienced periodic gaps in the time series collection, ranging from 5 days to a month, as a result of equipment failure due to weather and wildlife interference. As such, a continuum of soil moisture and soil temperature data is present for all dates throughout the field season, but not all sites are necessarily represented on every given date. As sensor deployment and removal occurred over a span of several days, the dataset was trimmed to remove the initial few days while sensors were adjusting to site conditions, and the final stretch of days spanning sensor removal at all sites, for a total record of 95 days extending from May 24th to August 27th, 2020.

Additional soil moisture readings were taken along transects using a FieldScout TDR 150 Soil Moisture Meter (Spectrum). Measurements were taken along the two 30-meter transects, for a total of 50 measurements per plot. These measurements were collected for all plots on three occasions during the field season, in mid-June, late July, and mid-August, and validated with 17 soil samples collected and weighed in the field and later oven dried to obtain the gravimetric water content (Quinones et al. 2003).

Soil disturbance was assessed at all sites using the USFS Forest Soil Disturbance Monitoring Protocol (FSDMP) Volume I: Rapid Assessment (Page-Dumroese et al. 2009). This protocol examines several soil and forest floor attributes at various points along a random transect, including compaction, rutting, char, forest floor depth above soil surface, topsoil displacement, and others. The FSDMP is designed for applications where forest soil disturbance can be assessed before and after any significant disturbance for monitoring purposes, but due to the unpredictability of wildfire, the speed with which post-fire logging followed the wildfire, and the timeline of this project, we were limited to post-disturbance evaluation. The protocol was modified slightly to focus on the effects of management (e.g. logging activities) on soil disturbance as opposed to the wildfire disturbance, which affected all plots. Transects for the rapid assessment were located randomly within plot vicinity while remaining within the relevant site category with a total of 50 points sampled per plot.

1.2.5 Statistical Analysis

All statistical analysis was performed in R version 4.0.3 (2020-10-10) (RStudio Team 2019) with an $\alpha=0.05$ as the threshold for statistical significance. Surface fuel loadings were stratified by fuel size class (1, 10, 100, and 1000-hour fuels) prior to analysis. Surface fuels, surface substrate, vegetation cover and canopy cover were analyzed using a two-way analysis of variance (ANOVA) using block and site category (LS, HS, LO) as predictor variables, followed by Tukey's HSD test when significance was determined.

Soil temperature and soil moisture data was collected between late May through late August. Temperature sensors produced a time series of maximum daily soil temperature data from all N=31 sites, and soil moisture data from N=12 sites. Within blocks, we subtracted the daily maximum temperature of LO from LS and also of HS from LS, creating two sets of data showing the maximum daily temperature difference between logged and low severity sites, and the difference between high and low severity sites. Maximum daily temperature differences (from LS), average daily soil moisture and minimum daily soil moisture were also analyzed using a two-way analysis of variance (ANOVA) using block and site category (LS, HS, LO) as predictor variables, followed by Tukey's HSD test when significant. Discrete soil moisture measurements were analyzed separately using Spearman's rank order correlation in order to compare measurements from the FieldScout TDR 150 Soil Moisture Meter with gravimetric readings of volumetric water content. Soil disturbance was analyzed as a one-way analysis of variance comparing percentage area of site demonstrating signs of erosion, compaction, topsoil displacement, and rutting, using low severity sites as the baseline for least disturbed.

1.3 Results

1.3.1 Overstory

Mean canopy cover was 61.5% in LS, which was significantly greater than in HS ($p = 0.0047$) where cover averaged 40.5%, which in turn was greater than cover at LO ($p < 0.001$) where canopy cover averaged 0.3% (Table 1). Mean live and dead stand density, LS, HS, and LO contained 565, 500, and 9 trees ha^{-1}, respectively. LO sites had standing trees remaining on only 1 of 11 sites. When considering both live and dead standing trees, there was no significant difference between overall stand density in high and low severity treatments ($p = 0.7293$), though there was a significant difference in live stand density ($p < 0.0001$). Post-fire tree mortality averaged 57.5% in low severity, 99.8% in high severity, and 100% in logged treatments.

1.3.2 Soil temperature, moisture, and disturbance

During the late spring (05/26/20 – 06/18/20), maximum daily soil temperature was significantly hotter in logged sites than other treatments. During this period, peak daily temperature was, on average, 1° C hotter in LO than HS ($p = 0.0299$; Figure 5). Soil temperatures in HS exceeded logging treatments during the later field season (07/23/20 – 8/27/20) and were 0.9° C hotter than logged sites ($p = 0.0234$). During the hottest part of the summer (06/24/20 – 07/17/20) there was no statistical difference ($p = 0.9505$) between max daily temperature at LO and HS nor when evaluating the season as a whole (05/24/20 – 08/27/20; $p = 0.5834$). Throughout the entire field season, daily average and daily minimum soil moisture was greatest in LS ($p = 0.0044$, $p < 0.0000$, respectively). Our data was limited by equipment malfunction, but the trends we saw indicated that daily average and daily minimum soil moisture levels in LO spiked above levels in HS during rainfall events, but with greater time since rain, LO sites exhibited lower soil moisture than HS sites (Figure 7) in LO sites than HS sites. Measurements collected with the Spectrum TDR FieldScout Probe on discrete occasions in mid-June, mid-July, and mid-August demonstrated no significant difference in average soil moisture between treatments ($F = 1.515$, $p = 0.225$; Figure 8).

Using the modified FSDMP, soil disturbance was greater in LO, driven primarily by topsoil displacement, rutting and compaction, which were relatively absent from HS and LS sites. Certain sites demonstrated frequent occurrence of specific soil disturbance characteristics, while at other sites that same characteristic was relatively uncommon, and so median values may provide a more insightful depiction of disturbance than averages. At LO sites, management-induced rutting was observed at 13% (average = 9%), topsoil displacement was observed at 2% (average = 14%), erosion was observed at 12% (average 18%), and near-surface (0-10 cm) compaction was observed at 40% (average = 37%) of site area. For all four soil disturbance characteristics, coverage was significantly greater ($p < 0.0005$) at LO than either HS or LS sites.

1.3.3 Vegetation responses and tree regeneration

Besides graminoids, which had significantly higher cover at LS sites than either LO or HS ($p = 0.0016$, $F = 8.25$), no differences were observed in the cover of functional groups (forbs, shrubs or resprouting trees, or all vegetation cover combined) between site categories (Figure 9). Both Gambel oak and quaking aspen demonstrated comparable regeneration across all site categories, with seedlings or re-sprouts present in 100% of sites that previously contained the species. While we did not distinguish between resprouts and new germinants, many appeared to be resprouting out of top-killed individuals. Mean densities of *Quercus gambelii* at LS, HS and LO sites were 14500, 6250, and 10000 stems ha^{-1}, respectively. Mean densities of *Populus tremuloides* at LS, HS and LO sites were 3500, 7500, and 3250 stems ha^{-1}, respectively. Between LS, HS and LO, there was no significant difference in the density of resprouting Gambel oak ($F = 2.2169$, $p = 0.1714$) or quaking aspen ($F = 1.1839$, $p = 0.3331$) nor when totals for both species were combined ($F = 0.3999$, $p = 0.6743$).

Mature pre-fire lodgepole pine was present at 10 sites, and lodgepole pine seedlings occurred at 90% of these (n=9). Excluding lodgepole pine, conifer seedling data was zero-inflated, with seedlings observed at only 12 of 27 sites. Of these 12 sites, seedlings occurred at

6 sites in LS, 4 sites in HS, and 2 sites in LO. When analyzing sites with a presence/absence metric of conifer seedlings while excluding deciduous resprouting or serotinous conifer species with predictably strong post-fire response, we found no significant difference in the number of sites with conifer presence between site categories ($F = 1.846$, $p = 0.18$). Mature pre-fire Douglas-fir was present at 21 sites and seedlings occurred at 42.9% of sites ($n = 9$). Mature pre-fire white fir was present at 27 sites and seedlings occurred at 14.8% of sites 9 ($n = 4$). Mature pre-fire limber pine was present at 9 sites and seedlings occurred at 11% of sites ($n = 1$). Mature pre-fire ponderosa pine was present at 22 sites and no seedlings were present. Distance to live seed source for non-serotinous, non-sprouting species averaged 49, 113, and 136 meters at LS, HS and LO sites, respectively. There was no significant difference in average distance to seed source between LO and HS ($F = 0.018$, $p = 0.897$; Figure 10), but distances were significantly reduced in LS ($F = 47.7$, $p < 0.0001$). Note that 150 meters was the upper limit of the rangefinder, and in some instances live trees were well outside of this range if visible at all but were recorded as 150 meters, leading to potentially underestimated distances recorded for trees outside of this range.

1.3.4 Surface fuels

Among areas that burned at high severity, logged sites (LO) contained statistically higher surface fuel loading than their unlogged counterparts (both LS and HS) across all fine woody debris classes ($F = 102.4$, $P < 0.0001$; Figure 11, Table 3), with the exception of the one-hour fuel class, where there was no significant difference between fuel loads between LO and LS sites. Similarly, for 1000-hour fuels, logged sites had significantly higher fuel loading than unlogged high severity or low severity sites (F = 8.991, p = 0.0010). Surface substrate was grouped into two categories: 1) bare ground and rock or 2) litter and woody debris. Sites in LS had higher cover of combined litter and woody debris than LO, which in turn had greater cover than HS (F = 32.66, p < 0.0001; Figure 9). As follows, percent coverage of combined bare ground and rock was greatest in HS, followed by LO then LS. LS sites had greater depths of litter (F = 44.155, p < 0.0000) and duff (F = 13.066, p <0.0000) than other sites. While LO sites had a greater area covered by litter than HS, there was no significant difference in depths of litter of duff and litter between site categories (F = 0.413, p = 0.5266).

1.4 Discussion

Soil temperature and moisture varied between LS, HS, and LO sites at different times throughout the growing season which may influence plant growth and survival. Higher surface fuel loads at sites with logging overrode the differences resulting from burn severity, and high severity areas that received logging treatment demonstrated greater cover of litter and debris than those where logging did not occur. In other words, logging resulted in higher surface fuel loadings and greater fuel continuity, which under the right conditions lead to increased short-term fire risk potentially even allow for repeat fire. Across all sites, seedling density was variable, with several conifer species showing little to no e arly post-fire regeneration. The largest plant response observed was from resprouting deciduous species, indicating the potential for transition from conifer to deciduous dominance.

1.4.1 Canopy and surface fuels

Microsites, e.g. the specific climatic, environmental and topographical conditions at a given point on the landscape, including soil moisture, temperature, substrate, etc. can play an outsized role in the germination stage for trees (Rother et al. 2015). We found that canopy cover and standing and downed fuels varied by disturbance type, which may have long-term implications for site suitability for tree regeneration, as others have found (Donato et al. 2006, Keyser et al. 2009, Marcolin et al. 2019). In our study area, canopy cover was greatest in LS sites where live trees were still present, followed by HS sites where standing dead trees contributed to canopy cover. Meanwhile, canopy cover was completely absent at all but one site in LO. Canopy cover has been shown to help moderate temperature and moisture at the surface (Maher et al. 2005, Davis et al. 2019b). As canopy becomes sparser, moderating effects decrease (von Arx et al. 2013). Canopy removal leads to greater solar exposure of the surface and leads to subsequent soil heating and drying (Marcolin et al. 2019). Some studies have shown the sheltering effects of woody debris (7.6 - 10+ cm in diameter) to be more important in promoting establishment than the amount of solar exposure from canopy gaps (Gray and Spies 1997, Hill and Ex 2020), with some studies indicating retained downed timber and slash resulting from logging can offset higher temperatures and moisture deficits in microsites (Castro et al. 2011, Marcolin et al. 2019).

Logging more than doubled surface fuel loading across all size classes compared to burned only sites, except for one-hour fuels where LO and LS sites exhibited similar fuel quantities. Increased fuel loading in LO is similar to findings in other studies (e.g. Donato et al.

2006, Peterson et al. 2015, Leverkus et al. 2020). Though CWD increased in logged sites, mean loading was 19.8 mg ha$^{-1,}$ which falls within the optimum range of CWD for balancing fire risk with ecological benefits of downed materials as identified by Brown et al. (2003). Thus, while loadings were higher, at least with these logging activities, the remaining fuels did not present excessive hazard in the event of a subsequent fire. While not statistically significant, combined litter and woody debris cover was more than doubled at LO sites compared to HS sites, indicating that logging resulted in a greater continuity of fuels, consistent with the findings in Donato et al. (2013).

Surface fuels naturally accumulate as standing dead material begins to fall with longer time since fire, as others have found across the intermountain west (e.g. Roccaforte et al. 2012, Fornwalt et al. 2018, Stevens-Rumann et al. 2020). On HS sites, our CWD loadings were much less than those found by Roccaforte et al. (2012), Donato et al. (2013) and Stevens-Rumann et al. (2020) but similar to those found by Keyser et al. (2009), while FWD loadings were similar to amounts found in all three studies. On LO sites, FWD and CWD loadings were 2 and 4 times greater, respectively, than those found by Keyser et al. (2009), but again much less than those found by Donato et al. (2013). This may be due to the more productive sites measured in Donato et al. (2013) compared to our study area. We observed small increases in fuels on HS sites from 1 to 2 years post-fire but this is very early in the post-fire period and we would not expect high accumulations until 6-14 years post-fire (Roccaforte et al. 2012, Stevens-Rumann et al. 2020, Fornwalt et al. 2018). As such, HS sites may equal or exceed surface fuel loadings of LO sites in years to come, though as McIver and Ottmar (2007) found, it may take up to 20 years. As LS sites continue to grow and shed aerial fuels, we predict that FWD on LS sites will have similar loadings to LO, while CWD at HS sites will eventually exceed loadings at LO. We note that winter weather halted logging operations at one property and it is possible that this accounted for some variability in surface fuel removal.

1.4.2 Soil temperature, moisture, and disturbance

Our research showed that LO sites demonstrated reduced daily minimum soil moisture and daily average soil moisture between May and August. Soil temperature varied between HS and LO sites during these months but did not vary significantly across the season as a whole, unlike previous studies (Maher et al. 2005, Rother et al. 2015). Research shows that seedling emergence occurs during periods of moderate soil temperature and increased soil moisture, while seedling mortality can result from higher soil temperatures (Petrie et al. 2016). Survival of seedlings emerging in late spring may be undermined by the higher early growing season temperatures we observed in LO sites. Low soil moisture during the summer months is also associated with recruitment failure (Davis et al. 2019a), and so reductions observed in LO site soil moisture may cross thresholds to recruitment failure in LO sites before HS sites with greater soil moisture. At lower elevations, the reduced soil moisture in LO sites may exacerbate the harsh post-fire climatic conditions that others have shown may lead to regeneration failure and possible long-term conversion (Coop et al. 2020). Furthermore, the reduced soil moisture in LO sites may undermine active reforestation efforts if moisture deficits and drought-stress exceed the tolerance of planted seedlings.

Across LO sites, we observed increased compaction, erosion, rutting and, to a lesser degree, topsoil displacement. From an ecological standpoint, soil disturbance can both hinder and facilitate post-disturbance plant colonization. Soil depressions, displacement and depositions resulting from post-disturbance logging may contribute to microsite variability and create a more diverse array of potential habitats, although that does not always lead to changes in vegetation (Peterson and Leach 2008). Greater microsite variability may allow for the establishment of a greater number of tree species (Zald et al. 2008). Compaction can lead to suppressed seedling growth and increased seedling moisture stress, though effects vary with soil type (Gomez et al. 2002). Logging can result in increased soil erosion and runoff potential during rainfall events, though effects are reduced with greater vegetation recovery (Malvar et al. 2017). For many tree species, the majority of viable seeds reside within 5cm of the soil surface (Kramer and Johnson 1987). Erosion displaces soil and can lead to seed removal (Cerdà and García-Fayos 2002). While this study did not examine exotic invasive species, research indicates that increased soil disturbance from wildfire and logging may increase vulnerability of logged sites to invasion (Neary et al. 2001, Korb et al. 2005, Lee and Thompson 2012, Burrows et al. 2013).

1.4.3 Vegetation responses and tree regeneration

For most vegetation functional groups, there was no significant difference in cover between site categories, either individually or as a whole. Graminoid cover was significantly greater in LS sites, which may be evidence of a patchy burn mosaic where some plants were not impacted, while those that did burn may be experiencing resprouting in the case of perennial grasses or seedbank reestablishment in the case of many annual species. Areas that burned at high severity (both HS and LO) may have experienced greater plant mortality due to increased surface temperatures and/or residency time during the actual fire event. Overall, burn severity had no significant impact on total understory plant cover, which is similar to second year post-fire findings from the 2002 Hayman Fire by Fornwalt and Kaufmann (2014), which found that time since fire generally had a greater impact that burn severity on understory vegetation response.

Tree regeneration was highly variable, in part because we captured a wide range of forest types in this study. Lodgepole seedlings, Gambel oak, and quaking aspen resprouts were prevalent at most sites where they were present pre-fire, which aligns with research examining other recent fires in the region (Rodman et al. 2020). Non-lodgepole conifer seedlings were observed in less than half of sites, spread across the three site categories. The absence of other conifer seedlings is not necessarily surprising, given that this study occurred two years post-fire, and others have found it may take 5-10 years for 50% or more of seedlings to establish (Stevens-Rumann et al. 2018). Gambel oak and quaking aspen recruits or re-sprouts occurred both 1 and 2 years post-fire at 100% of sites where they had been present previously, with Gambel oak averaging 10,000 stems ha^{-1}, and aspen averaging over 5000 stems ha^{-1}. Post-fire dominance of deciduous resprouting species has been observed in multiple studies (Roccaforte et al. 2012, Coop et al. 2016, Kaufmann et al. 2016, Stoddard et al. 2020). Research on the Leroux Fire in Arizona, which burned through similar forest types as the Spring Creek Fire, found vigorous aspen response in the first year after fire, but observed that densities declined to below pre-fire levels over the following decade in all burn severity classes (Stoddard et al 2018).

Distance to seed source for previously dominant species exceeded 113 and 136 m on average for HS and LO sites, respectively. Distance to live seed source presents a significant obstacle to recruitment for obligate seeding conifers in burned areas (Kemp et al. 2016), particularly species with heavier seeds (e.g. ponderosa pine, limber pine) that are less likely to benefit from long distance wind-dispersal (Chambers et al. 2016). A review of 26 studies between 2005-2018 examining post-fire seedling density noted that the vast majority of studies identified a negative correlation between seedling density and distance to seed source (Stevens-Rumann and Morgan 2019). Other research also indicates that conifer seedling densities decrease as burn severity increases, not only as a result of increased distance to seed source but also increased competition between seedlings and post-fire shrub response (Welch et al. 2016). Other research shows shrub cover positively associated with increased recruitment (Davis et al. 2019a). Of course, tree regeneration depends not only on seed availability and competition, but also climate conditions conducive to recruitment (Davis et al. 2019a). The effects of increased warming and reduced soil moisture are already being felt in some forest types at the lower elevational range, where burned forests reveal climate conditions no longer conducive to tree regeneration (Parks et al. 2019, Coop et al. 2020). As such continued monitoring on these sites is critical to understand where along this elevational and forest type gradient forest conversion to either non-forest or deciduous dominated forest may occur versus those areas where conifer regeneration is simply delayed.

1.4.4 Management Implications

Our study found that post-fire logging resulted in increased surface fuel loading and fuel continuity compared with unlogged sites that burned at high severity, and as such this management action should not be used for short-term surface fuels reduction. Logging treatments may still be useful for site preparation for restoration actions and may be desired to recoup some timber value. If land managers seek to restore burned areas to a forested state and intend to implement a logging treatment, minimizing heavy equipment usage and increasing snag retention may be beneficial for restoration efforts (DeLong and Kessler 2000, Lindemayer and Noss 2006). Greater snag retention may not only reduce long term fuel loads, but the retained canopy cover may help reduce soil moisture loss from solar exposure, increase favorable microsites for tree regeneration, and provide wildlife habitat. Some studies indicate reduced ecological impact when logging occurs during the winter (Wolf et al. 2008) but snow cover and road access can limit the practicality of conducting winter logging in many areas including our study area, where the onset of winter precipitation prematurely terminated some logging activities. Other methods of logging, including cable or helicopter-based tree harvesting methods, have been shown to reduce fuel deposition and may also lessen soil disturbance, but are cost-prohibitive in areas with low timber value (Lindenmayer and Noss 2006, Leverkus et al. 2018). While areas receiving post-fire treatment in our study area logged one year after fire, the ecological impact of logging may also be lessened by increasing elapsed time between fire and logging by 2-4 years (Leverkus et al. 2020) however, delays in harvesting may reduce initial stand regeneration (Blair et al. 2016) and may come at the expense of timber merchantability.

Though this study focused on early post-fire periods, we found high resprouting of deciduous species with variable conifer regeneration. Where conifer dominance is desired post-fire, planting may be warranted to assist recovery to conifer dominant forest. This may be especially helpful in areas subject to frequent drought or located at the lower elevational or latitudinal ranges of desired tree species, where increased soil temperatures and moisture deficits may not be favorable for unassisted regeneration.

1.4.5 Limitations and future research

Our sites spanned 400 m in elevation, and varied in slope, aspect and forest type. As such, these sites can be stratified in multiple ways. Wildlife interference with soil moisture and temperature data logging equipment led to occasional equipment failure, reducing sample size during certain periods of data collection throughout the field season. Soil moisture measurements collected by the Spectrum FieldScout probe required meticulous consistency in instrument rod insertion, potentially reducing accuracy in soils with significant rock content, as was common on a number of our sites (personal observation). Additionally, distance between sites prevented the collection of all Spectrum Fieldscout measurements within the same hour and day for all sites, possibly resulting in temporal fluctuations in soil moisture. The time elapsed and interaction between disturbances (fire and logging) created unique challenges in conducting the FDSMP, leading us to modify the protocol to focus on the effect of post-fire management.

Future studies conducting the FSDMP in logged sites 1) prior to post-fire logging operations, 2) immediately after post-fire logging, and again after one or two years have passed may provide valuable insights on the persistence of changes to soil disturbance following management. Additionally, future research is needed to explore if the canopy created by early seral plant species (particularly by resprouting quaking aspen and Gambel oak) may offset soil moisture deficits and stabilize temperature differences in logged sites, potentially creating more favorable conditions for conifer regeneration. Monitoring changes in seedling density at 5 and 10 years post-fire may offer valuable insights into recovering forest trajectories across site categories, potentially indicating if conifer forest recovery is likely or if strong initial response of resprouting species may foreshadow transition to deciduous-dominant states. While we did not measure species richness, research by Blair et al. (2016) found that post-fire logging can reduce species richness more than the original fire event. If additional logging operations occur within the burn, comparisons between plant community richness, cover and composition in those areas with sites logged one year after fire may be useful in determining if increased time elapsed since fire influences recovery trajectories.

1.4.6 Conclusions

This study examined site microclimate in areas that burned at low severity, high severity, and areas burned at high severity that were subsequently logged. Our results support a correlation between canopy cover and soil moisture throughout the growing season, with seasonally variable effects on soil temperature at key periods in the growing season. As the climate in western forests continues following trends of warming and drying, previously dominant tree species at the lower elevational edges of their range will encounter fewer favorable episodes for regeneration. Post-fire logging may further affect site microclimate and therefore regeneration success. As increases in wildfire frequency and severity result in ever-expanding areas of conifer mortality, landowners will continue to seek strategies to improve conditions for forest regeneration while reducing risk of repeat fire. Changes to site microclimate, soil disturbance and surface fuel structure resulting from post-fire logging should continue to be examined when developing post-fire management and restoration plans. As forests are pushed beyond conditions suitable for natural tree regeneration, post-fire logging may require increased planning and considerations to offset potential negative ecological impacts.

FIGURES AND TABLES

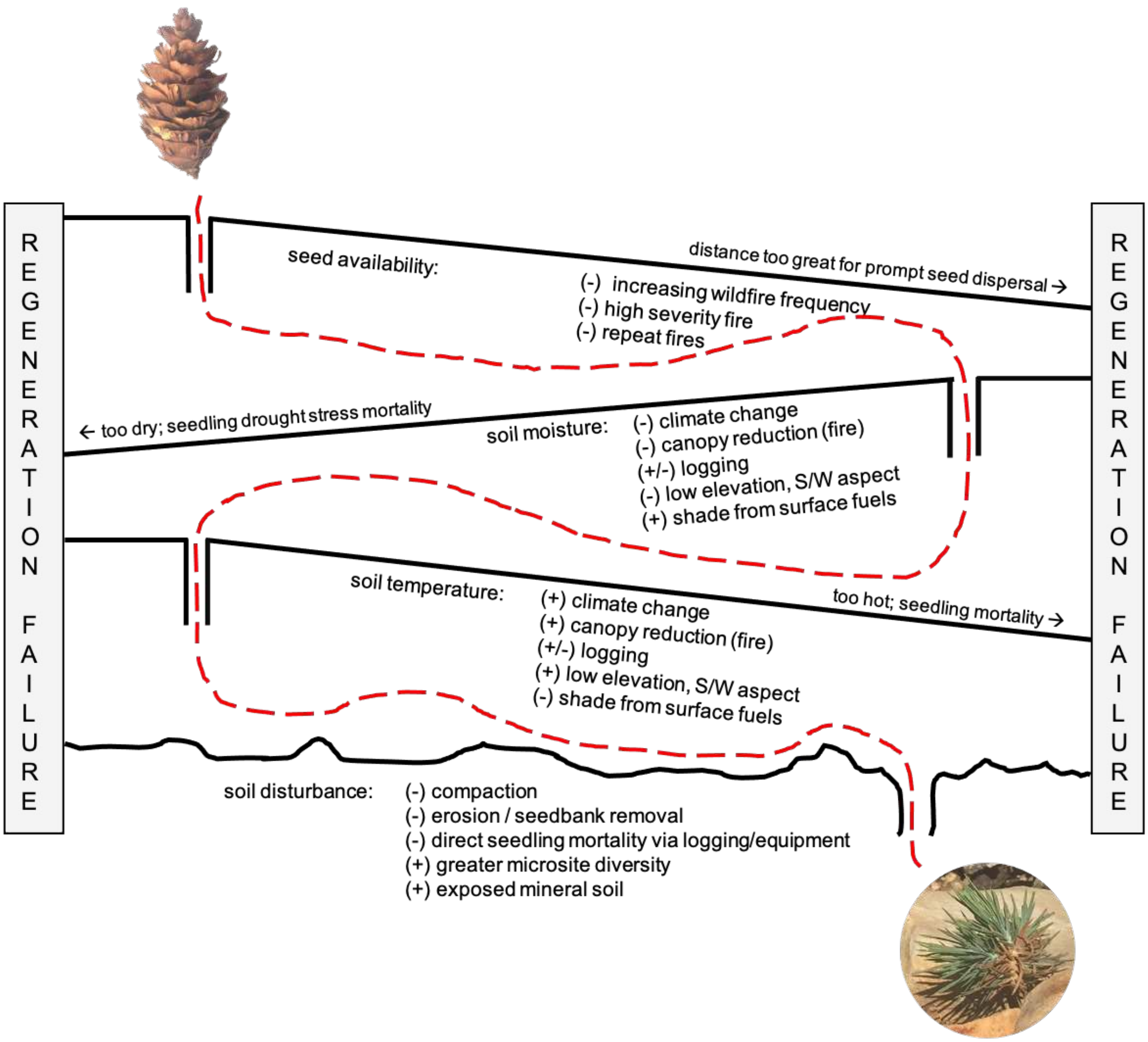

Figure 1: Proposed model illustrating environmental factors and bottlenecks that must be navigated through for successful conifer regeneration to occur. Successful regeneration requires seed availability and soil temperature and moisture conditions suitable for regeneration to occur. Soil disturbance can help or hinder regeneration depending on species preferred soil characteristics. Wildfire, climate change, post-fire logging and other factors may have positive, negative or neutral effects on regeneration.

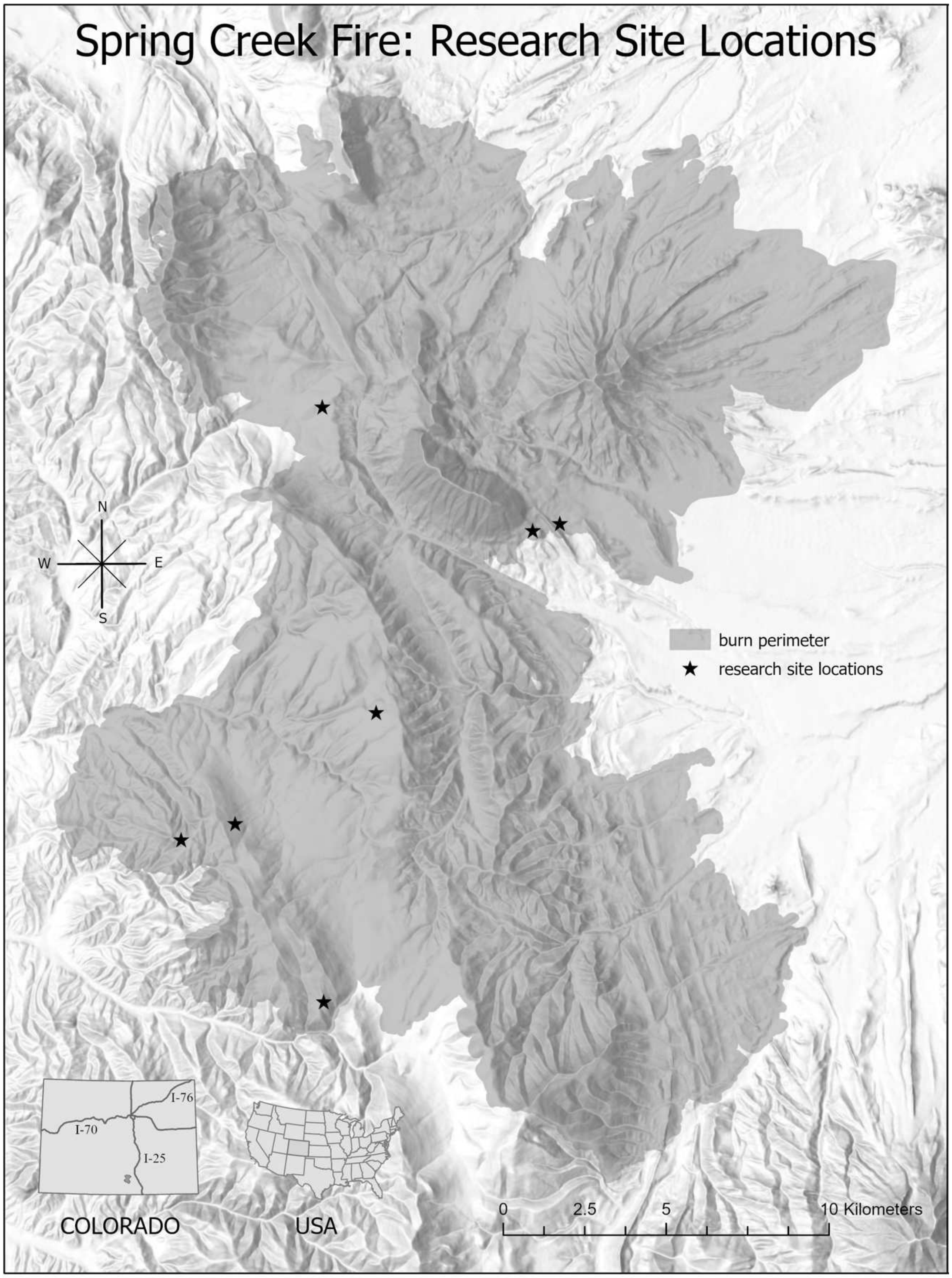
Spring Creek Fire: Research Site Locations
N
W
E
S
burn perimeter
research site locations
I-76
I-70
I-25
COLORADO
USA
0
2.5
5
10 Kilometers

Figure 2: Map of the Spring Creek Fire of 2018. Stars represent approximate locations of research sites. Burn boundary derived from BAER soil burn severity map.

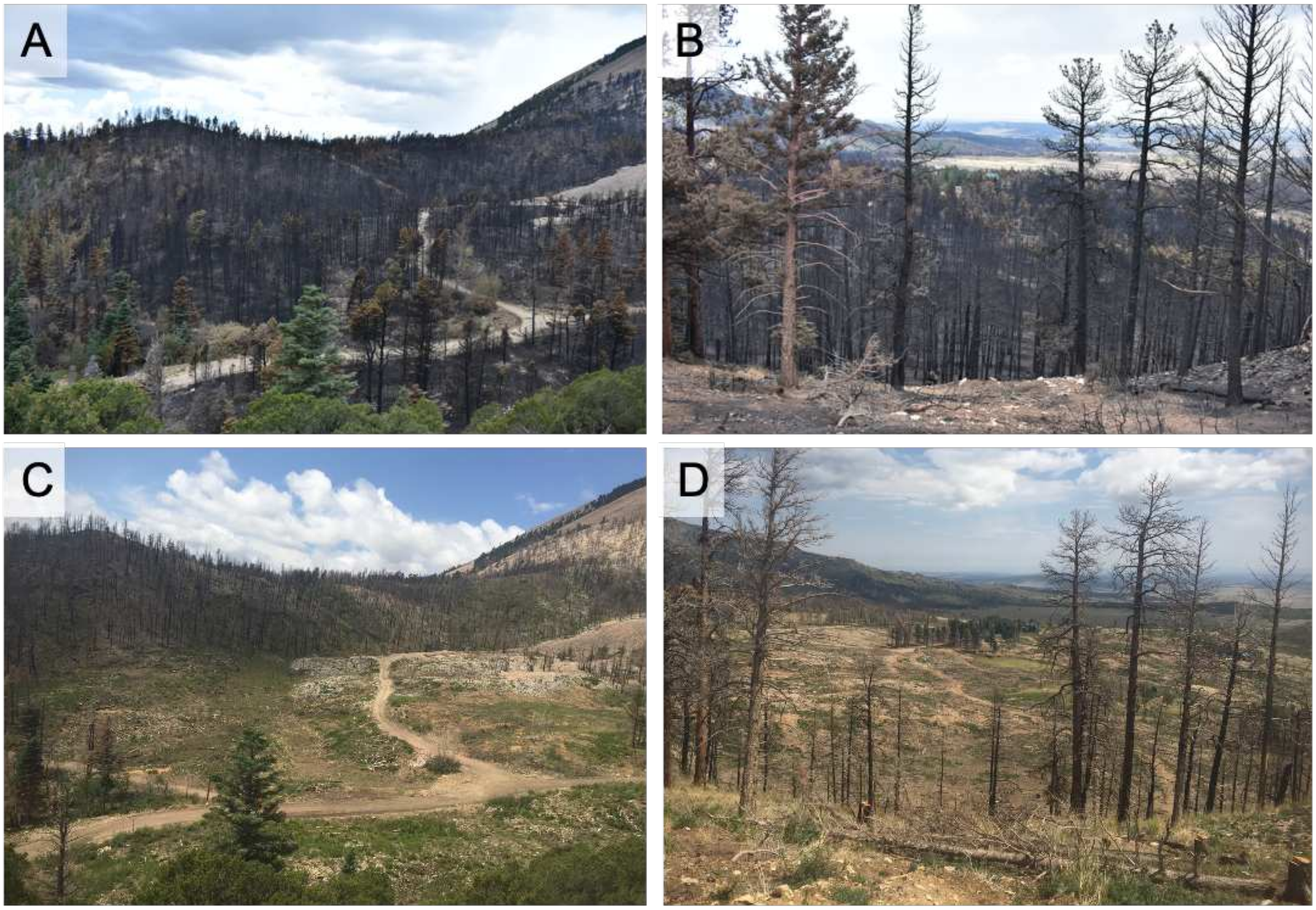

Figure 3: Photos before (A, B) and after (C, D) logging operations in the Spring Creek Fire. Pre-logging photos taken by Steve Keppers in July 2018. Post logging photos taken by the author in August 2020.

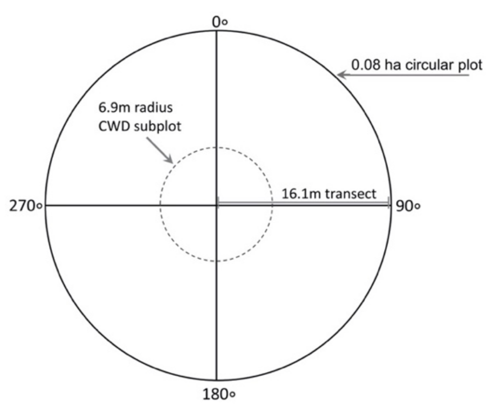

Figure 4: Plot schematic, adapted from Ott et al. 2018. Fuels: 10 and 100 hour tallied across full length of all four transects, one hour fuels tallied on outside 6 meters of each transect. CWD fuels tallied within subplot. Understory vegetation, substrate and canopy cover measured using a line point intercept along all transects. Seedlings and adult trees inventoried within the entire circler plot. Soil moisture and temperature sensors placed one meter from plot center.

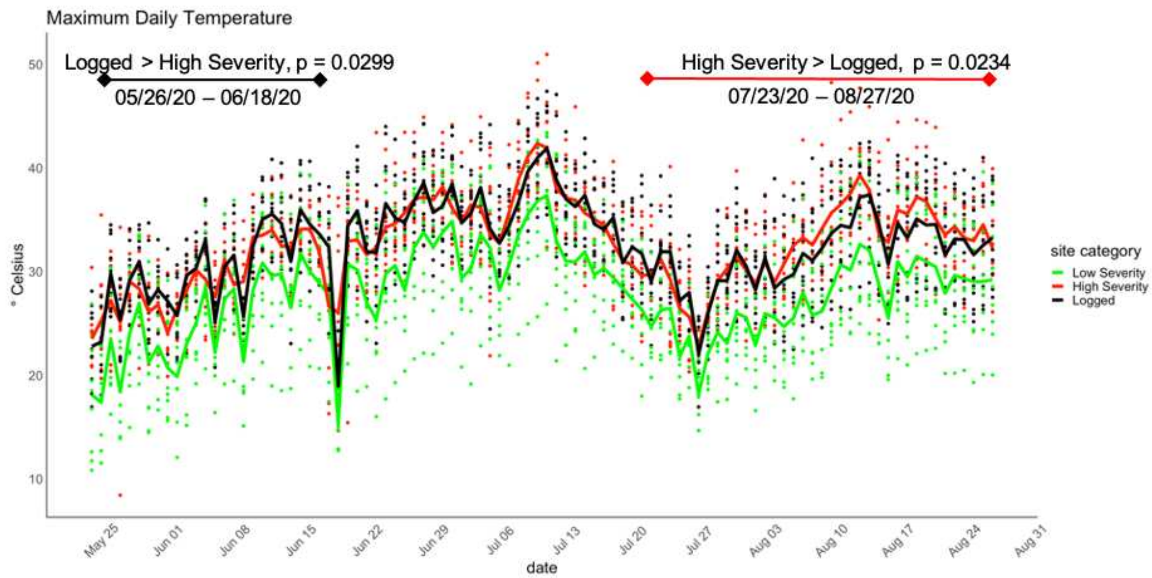

Figure 5: Dots represent the maximum daily temperature recorded at a site, averaged between sensors present at the site. Lines above temperature readings indicate periods of statistical significance. LO sites were significantly hotter than HS sites during early season (May 26th – June 18th) and HS sites were significantly hotter than LO sites during later season (July 23rd – August 27th).

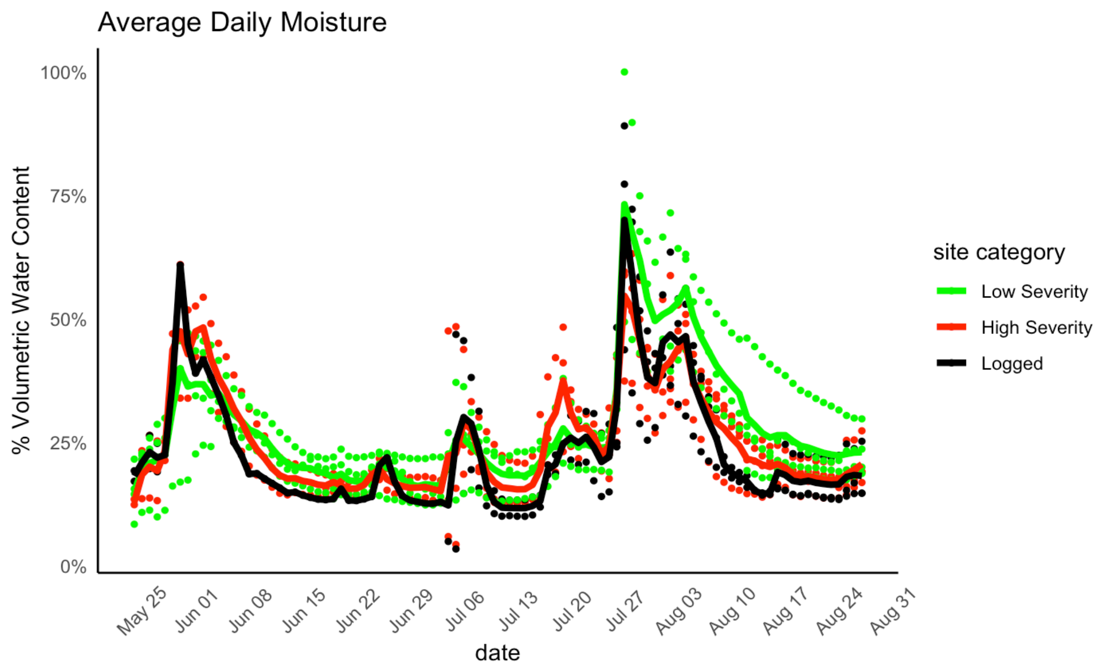

Figure 6: Dots represent the average daily moisture recorded, averaged between sensors at each site. Line above moisture readings indicate periods of statistical significance. Average daily soil moisture was trending lower in LO than HS and LS sites throughout the duration of the data collection period (May 24th – August 27th).

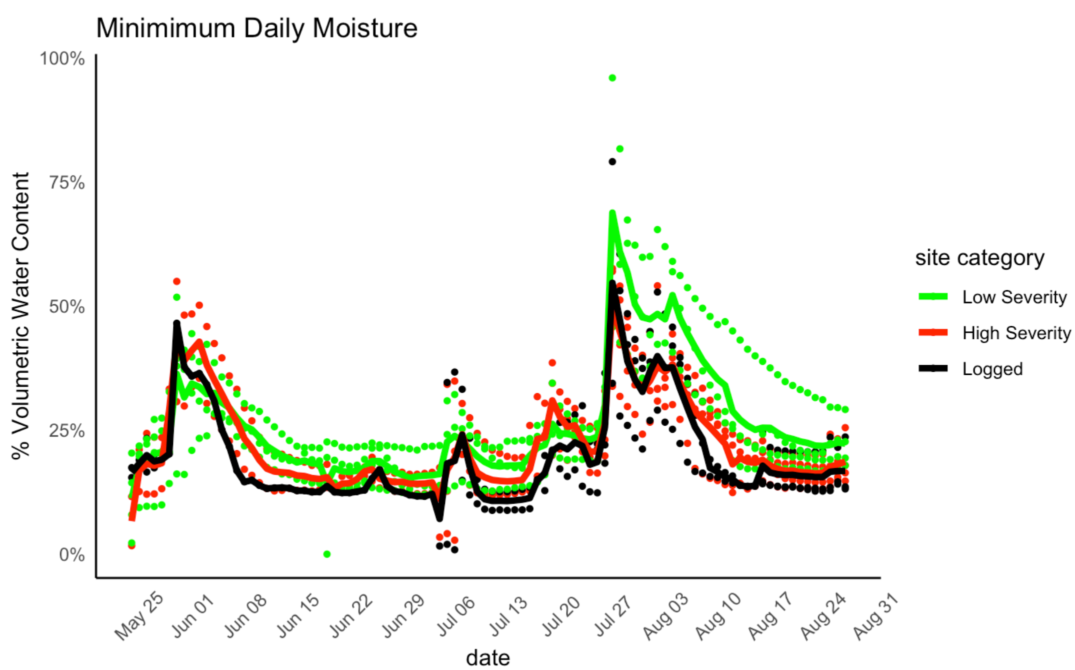

Figure 7: Dots represent the minimum daily moisture recorded, averaged between sensors at each site. Line above moisture readings indicate periods of statistical significance. Minimum daily soil moisture was trending lower in LO than HS and LS sites throughout the duration of the data collection period (May 24th – August 27th), though this trend only approach statistical significance in periods after some time had elapsed since rainfall.

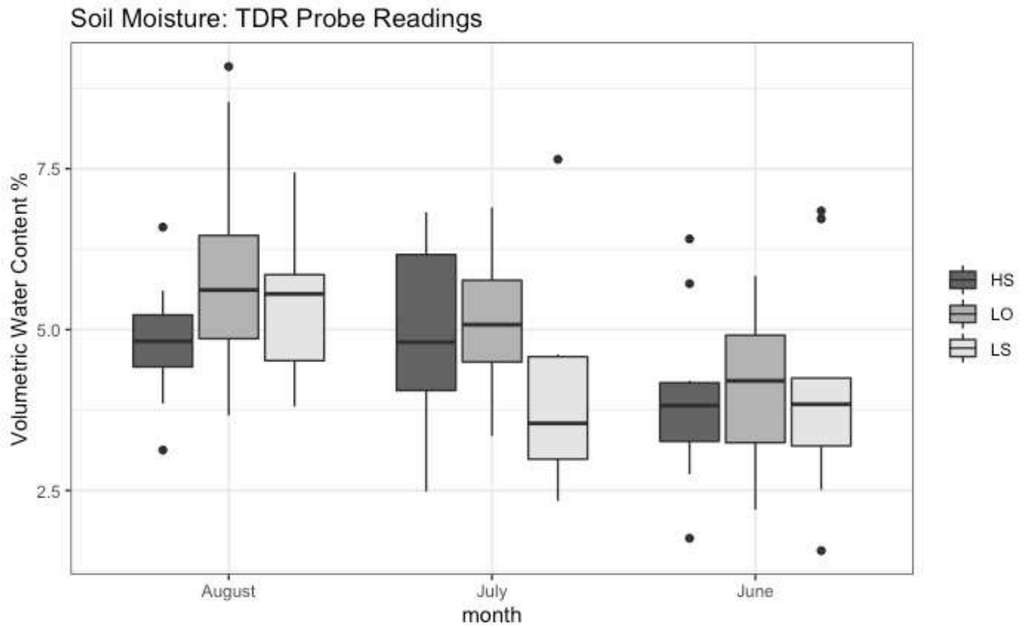

Figure 8: Soil moisture readings collected with Spectrum TDR FieldScout Probe. No significant differences between site categories on any of the three collection periods (F = 1.515, p = 0.225).

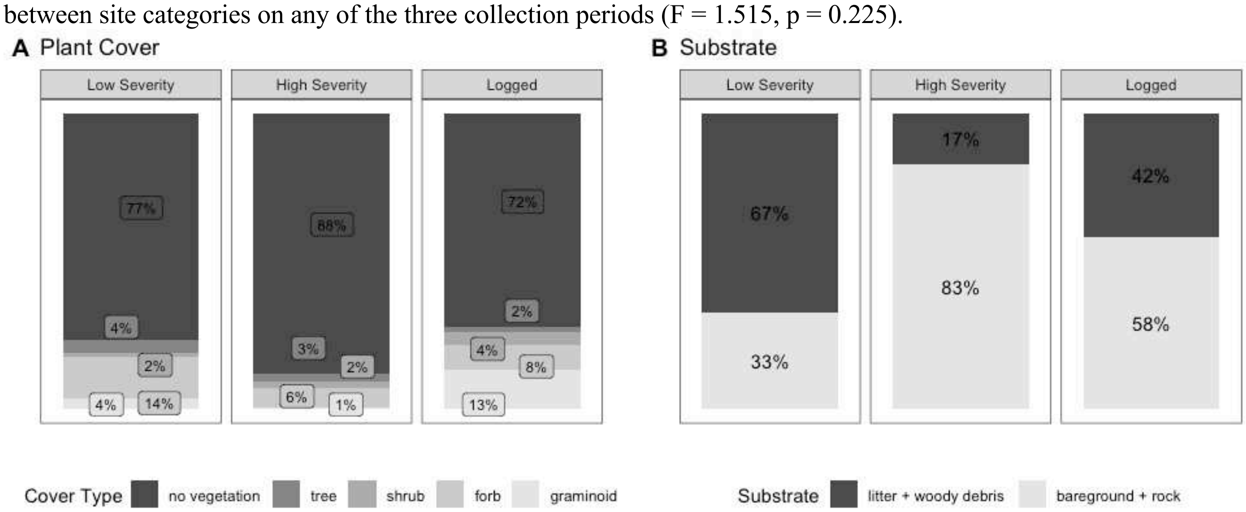

Figure 9: (A) Proportion of plant cover in treatments in 2020. No significant differences for any functional group except graminoids, which were significantly higher in LS sites than in HS or LO (p = 0.0016). (B) Surface substrate compared between treatments. LS sites have significant greater cover of litter and woody debris than LO sites (p < 0.0009), which in turn have greater cover than HS sites (p = 0.0008).

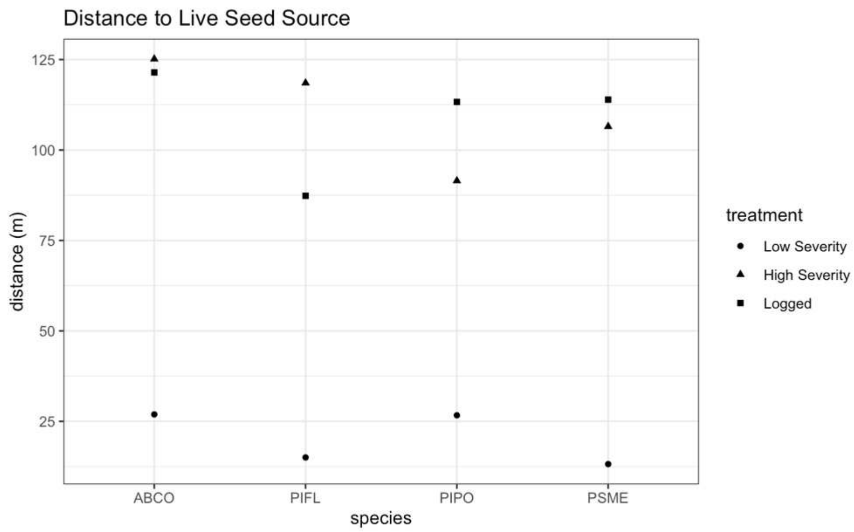

Figure 10: Distance to seed source for non-serotinous conifer species. ABCO = Abies concolor, PIFL = Pinus flexilis, PIPO = Pinus ponderosa, PSME = Psuedotsuga menziesii. No significant difference in distances between HS and LO sites (F = 0.897, p = 0.897), but both significantly greater than LS sites (F = 47.7, p <0.0000). Note: 150 m is upper limit of rangefinder, and distances beyond this were also recorded as 150 meters.

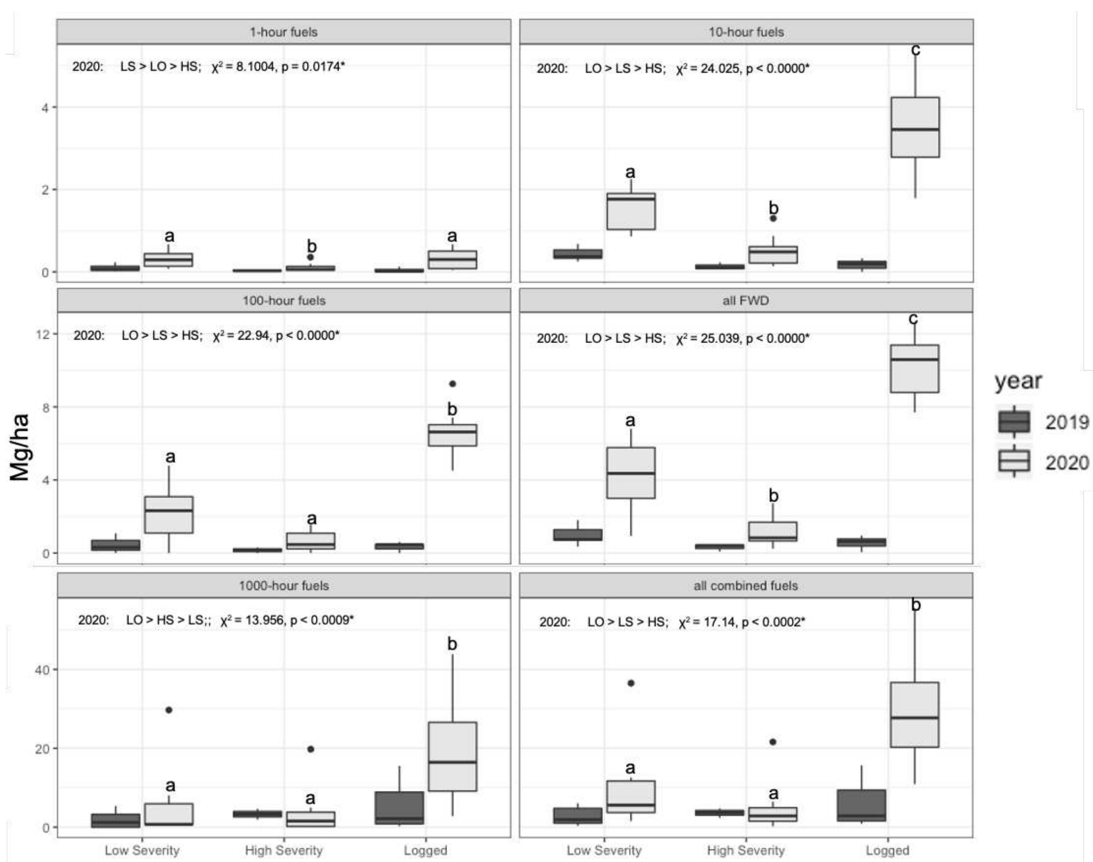

Figure 11: Surface fuel loading in sites, with significance indicated by lowercase letters above boxplots. Note: for LO sites, 2019 boxplots represent fuel loading prior to logging.

Table 1: Site categories with canopy cover, stand density density (trees ha^{-1}), and mortality means (standard deviations).

Table 1

Site Code	Disturbance	Canopy Cover	Trees ha ^{1}	Tree Mortality
LS	Low - moderate severity fire	61.5% (15.0%)	565 (220)	57.5% (13.7%)
HS	High severity fire	40.5% (17.6%)	500 (241)	99.8% (0.7%)
LO	High severity fire + logging	0.3% (0.9%)	9 (30)	100% (0.0%)

Table 2: Topographical and forest type data for all blocks across study area, with mean and standard deviation of elevation, slope, aspect, and maximum distance between sites within a block. Block 8 and 9 contained no LS treatment due to high severity fire in lodgepole / aspen forest type. All other blocks contain one each LS, HS and LO site.

Table 2		Elevation (m)		Slope (°)		Aspect (°)		max distance
Block	Forest Type	mean	sd	mean	sd	mean	sd	between sites (m)
1	mixed - PIPO	2665.7	39.1	5.0	1.0	67.3	37.4	654.3
2	mixed - PIPO	2666.1	39.8	7.0	2.6	114.3	14.6	513.3
3	mixed - PIPO	2694.1	45.6	10.0	4.4	133.3	105.6	251.7
4	mixed - PIPO	2720.3	31.2	5.7	3.2	113.3	111.3	263.9
5	mixed - PICO	2880.1	7.7	6.3	1.5	256.7	35.1	213.1
6	mixed - PICO	2890.9	12.3	5.3	0.6	233.3	41.6	294.9
7	mixed - PIFL	2980.9	21.7	15.7	1.5	56.7	11.5	171.5
8	PICO / POTR	2992.1	4.1	4.5	0.7	295.0	7.1	208.0
9	PICO / POTR	3003.0	6.7	3.0	0.0	270.0	0.0	89.2
10	mixed - PIFL	3020.9	5.0	3.3	1.5	71.3	111.7	252.6
11	mixed - PIFL	3029.1	2.1	8.0	1.7	223.3	20.8	167.6
mean:	-	-	19.6	-	1.7	-	45.2	280.0

Table 3: Surface fuel loadings two years post-fire (measured in 2020). Note: logging occurred in 2019.

Table 3

Site Code	1-hour	10-hour	100-hour	all FWD	1000-hour	all combined fuels
LS	0.32	1.56	2.28	4.17	5.33	9.50
HS	0.32	3.40	0.63	1.23	3.45	4.68
LO	0.10	0.49	6.53	10.24	19.76	30.00

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CHAPTER 2:

CONCLUSION

Changing climate conditions threaten forest regeneration while climate-driven increases in wildfire are removing more forests from the landscape. Compared to adult trees, seedlings tolerate a narrower range of suitable climate conditions and are more sensitive to temperature extremes and moisture deficits. While the ability of forests to regenerate under changing climate conditions is uncertain, the necessity of regeneration to maintain forested landscapes after high severity disturbances is not. At the local level, land managers cannot hold back climate change nor cancel fire season, but when fire does occur, they can choose how to manage burned landscapes.

Our findings show that post-fire logging increased the quantity and continuity of surface fuels, although loadings are still within the acceptable range identified by Brown et al. (2003) for balancing fire risk with ecological benefits of downed materials. Proportions of fuel classes may change over time as CWD decays into smaller sections that align more with classifications for FWD. Logging effectively removed aerial fuels, and therefore fuel loadings at logged sites will likely converge with unlogged sites as canopy fuels are transferred to the surface. However, two years post-fire, logged sites do have considerably greater surface fuel loadings than unlogged sites and could be at greater risk of reburning or altered fire behavior in the short term.

Following fire, successful regeneration requires seed availability and climate conditions conducive to recruitment and seedling survival. Two years post-fire, we observed non-serotinous conifer seedlings at fewer than half of sites where they were before the fire. Given large distances to seed source at some sites, the episodic nature of masting events for some species and the short time elapsed since fire, this is not necessarily surprising. However, if climate change is leading to reduced soil moisture at the landscape level in these forests, eventually moisture will fail to meet necessary minimums for successful recruitment. Our data indicates lower daily average and daily minimum soil moisture levels at logged sites. If sufficient soil moisture is a bottleneck to successful recruitment, this suggests that logged sites may experience regeneration failures at a greater frequency than similar sites where logging was not conducted. Additionally, we observed higher maximum daily temperatures in logged sites during the early growing season, which could result in increased seedling mortality for early germinants.

Our research was subject to limitations and uncertainties which should be considered for future study design and also suggests future research opportunities. We did not take soil type into account when examining differences in soil moisture, temperature and disturbance, but within blocks we can assume similar soil conditions. Understanding the drivers of soil moisture reduction in logged areas could provide valuable insight to land managers planning post-fire logging or thinning operations. Moisture deficits in logged sites may be a result of reduced canopy cover but could also be related to increases in compaction or changes to soil structure if logging negatively affected water infiltration within management areas. If moisture changes are due to soil disturbance rather than canopy reduction, results may differ here from a study examining these factors following alternate logging methods. More research is needed to identify why maximum daily temperatures were higher at logged sites during late May and early June, and then overtaken by temperatures at high severity sites for most of the next ten weeks. Additionally, research exploring the seasonal or time-since-germination importance of high temperatures on regeneration success would be useful in determining the practical effect of soil temperature differences between treatments. Greater understanding of the role of facilitation versus competition between conifers and deciduous resprouting species could help identify the probability of ecosystem conversion versus extended recovery time to pre-fire forest structure. Understanding regeneration tradeoffs between reduced soil moisture, increased soil disturbance, and greater microsite variability resulting from soil disturbance and increased woody debris, e.g. surface fuels, in logged sites would be useful for modeling regeneration trends following post-fire logging. Research at greater intervals following logging will help illuminate if changes in soil temperature, moisture and disturbance are persistent or ephemeral. As climate change and wildfire continue to reshape western landscapes, the role of post-fire management in hastening or delaying changes to forests merits careful consideration.

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