Assessing the Effects of Manual Tree Removal as a Grassland Restoration Treatment in the Churn Creek Protected Area

by
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Abstract

High-elevation grasslands in the Churn Creek Protected Area of the British Columbia Interior are being lost to the expansion of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) forests. In 2015, trees were manually removed in four of six treatments of varying tree density, height and maturity. The two extreme vegetation treatments, open grassland and mature closed forest, did not undergo tree removal. Three plots were randomly assigned per treatment to assess grassland community recovery. Vegetation percent cover, stem density and height of new tree seedlings were recorded in fixed-radius plots pre- (2015) and post-treatment (2018 and 2023). In 2023, soil characteristics were also measured. There were declines in herbaceous cover in 2023, due to either observer differences or low total annual precipitation since 2021. Soils in the mature closed forest had lower soil water content, temperature and electrical conductivity. In 2023, stem density of Douglas-fir seedlings was greater than previous years, yet shorter in height.

Keywords: woody encroachment; grassland recovery; seedling recruitment; manual tree removal; cultural burning; soil dynamics

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List of Acronyms

ANOVA	Analysis of Variance
BC	British Columbia
CCPA	Churn Creek Protected Area
DBH	Diameter at breast height
ELO	Encroachment, low, open
ETD & ETM	Encroachment, tall, dense & Encroachment, tall, moderate
ETO	Encroachment, tall, open
FFCPAS	Friends of Churn Creek Protected Area Society
MC	Mature, closed
MO & ILD	Mature, open & Ingress, low, dense
N	No tree stems, reference grassland
SXFN	Stswecem'c Xget'tem First Nation

1. Introduction

Grassland ecosystems have been experiencing widespread degradation, in large part, due to habitat conversion for human use (Hoekstra et al. 2005). Grasslands have also seen a transition in species composition from herbaceous to woody-plant cover, by a process known as woody encroachment (Ratajczak et al. 2012; Belay & Moe 2015; Halpern et al. 2016). As such, grassland ecosystems are increasingly threatened. A variety of factors may be causing encroachment, including the climate crisis, overgrazing, and fire suppression (Ratajczak et al. 2012; Belay & Moe 2015). Woody encroachment impacts biodiversity, vegetation structure, hydrology, and other grassland ecosystem processes (Ratajczak et al. 2014; Halpern et al. 2016). The restoration of grassland ecosystems can be challenging particularly once the grassland has shifted into a hard to reverse ecosystem state (Ratajczak et al. 2012; Ratajczak et al. 2014). Resistances to grassland recovery can be caused by legacy effects and positive feedbacks that are induced by woody plants, such as changes to soil characteristics and new species recruitment (Halpern et al. 2016, Archer et al. 2017).

Grasslands are of ecological, cultural, and economic importance, as such, they have been managed by communities for centuries (Turner 2021; Hoffman et al. 2022). Cultural and prescribed burning is a management tool used by Indigenous and Western communities, respectively. Grassland maintenance through fire reduces fuel loads and attempts to mimic the frequency and intensity of a natural fire regime (Hoffman et al. 2022). Trees can be manually cut, piled, and burned, reducing burn area; or the entire area can be burned (broadcast burn) (Halpern et al. 2016). In cases where prescribed fire is not permissible, manual tree removal is the proposed strategy for restoration. However, it is a time- and labor-intensive process requiring continuous monitoring and management (Steen & Young 2019). Therefore, my research strives to assess the effectiveness of manual tree removal in the Churn Creek Protected Area (CCPA), British Columbia.

1.1 Site History

Within the Churn Creek Protected Area, British Columbia, Canada, grasslands provide valuable resources to the Stswecem'c Xget'tem First Nations (SXFN) community (SXFN n.d.; Steen & Young 2019). Community members not only observe wildlife (California bighorn sheep, *Ovis canadensis californiana*) and hunt in the area (e.g., moose, *Alces americanus* and mule deer, *Odocoileus hemionus*), they gather plants for both medicinal and nutritional purposes across the protected area (SXFN n.d.; Steen & Young 2019; H. Harry 2023, Cultural Heritage Stswecem'c Xget'tem First Nations, personal communication). Oral histories indicate that cultural burning was used by the First Nations of CCPA to maintain grasslands (Harvey et al. 2017; Turner 2021). The fires would be set during Spring to thin understory vegetation and prevent conifer encroachment. Around 1896 there appeared to be a reduction in wildfires in the CCPA based on tree dendrochronology (Harvey et al. 2017). At this time, ranching, fire suppression, and mineral extraction began in the region. The reduction in fire management led to increased forest density and lengthened intervals between fire events. In grasslands, wetter, cooler conditions supported plant growth, and the resulting increase in fuel loads have been correlated to grassland wildfires.

More recently, grasslands in the CCPA have been greatly reduced due to the encroachment of coniferous forest. Fire suppression has resulted in the spread of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees and other woody species (SXFN n.d.; Harvey et al. 2017; Steen & Young 2019). Throughout the year, resident and migratory herds of bighorn sheep and mule deer are dependent on these lower, middle, and upper grasslands for foraging (Ministry of Environment, Government of British Columbia n.d.; Ministry of Environment, Lands and Parks n.d.). Dense understory shrubs and low-lying branches within the upper elevation mature forests is problematic for resident ungulates. Encroachment reduces available herbaceous forage and makes refuge from deep snows under the forest canopy less accessible. Woody encroachment also increases the risk of high-intensity wildfire due to increased fuel loading (BC Parks 2000; Twidwell et al. 2013). With the potential for warmer and drier summers due to the climate crisis, wildfire frequencies are expected to increase in British Columbia in the coming years (ClimateReadyBC n.d.). Fire impacts a forest's ecological and functional aspects by altering soil hydrophobicity, species composition and regeneration, nutrient turnover, and biodiversity (Agbeshie et al. 2022). Soil responses to fire are dependent on fire severity and duration, fuel loads, and soil moisture. Therefore, the suppression of low-intensity fires in the CCPA has led to a loss of grasslands and associated ungulate habitat due to woody encroachment, while heightening future risks of high-intensity wildfire (Harvey et al. 2017; Steen & Young 2019).

High Lake, an area of approximately 103 ha in the southern section of CCPA, represents an upper elevation grassland ecosystem with very dry and mild conditions in the Interior Douglas Fir (IDFxm) biogeoclimatic zone of British Columbia (Steen & Young 2019). These grasslands have been categorized as an ecological benchmark area by BC Parks (2000), who define it as an area of limited human use and no cattle grazing. In the CCPA, the High Lake Benchmark Area is intended to support research and enhance knowledge of reference grassland ecosystems. Declines in the designated reference grassland area, due to woody encroachment, triggered a collaborative effort between BC Parks, SXFN, the Ministry of Forests and the Friends of Churn Creek Protected Area Society (FFCPAS) to restore the area.

1.2 Grassland Restoration at CCPA

The manual removal of encroaching coniferous trees via slashing, piling, and burning across nearly 750 hectares (High Lake included) was undertaken by FCCPAS and SXFN crews, with limited follow-up prescribed burns by BC Parks (CCPA n.d.). Due to the presence of many stems greater than 2 m in height across the High Lake Benchmark Area prescribed fire was likely to have minimal success in reducing tree encroachment (Steen & Young 2019). Other factors limiting prescribed burns included short burn windows, unpredictable weather, and insufficient staff. Therefore, prescribed burning alone was not a practical restoration technique and the preliminary treatment of manually slashing, piling, and burning of trees was implemented in 2015. Slashing, piling, and burning involved the felling of stems, removal of branches, cutting stems into manageable lengths (1 to 3 m), grouping cut stems, and ignition of piles.

1.3 Grassland Ecology

Grassland plant communities are governed by highly constrained environmental factors (Archer et al. 2017). Small, germinating woody plants can, at first be shaded out by taller, healthier grasses that compete for resources in the upper soil horizon. Grasses have shallower maximum rooting depths than mature trees and can take full advantage of the higher concentrations of resources found in the upper portion of the soil profile (20-30 cm). Therefore, fine-textured soils holding much of their water and nutrients near the surface favor grasses. If a tree survives the early stages of competition with grasses, the bulk of its root system develops deeper in the soil, giving the tree access to nutrients and water unavailable to grasses. Alternatively, in arid or semi-arid regions, root turnover and litter from grasses can facilitate tree recruitment as these processes improve soil organic matter and moisture retention. When environmental conditions favour grasses, intense competition can prevent woody encroachment. However, increased mean annual precipitation or carbon dioxide levels can alter species composition, growth, and survivability to maturity, potentially facilitating woody encroachment. The resulting encroachment has the potential to cause further changes to available sunlight, water, and nutrients for herbaceous species, often leading to declines in grassland species richness (Ratajczak et al. 2012; Archer et al. 2017). Woody plants impact hydrology by changing vegetation and soil structure. Woody plants, particularly coniferous species, intercept more rainwater and have longer periods of transpiration than grasslands. Water runoff is also reduced by the change in water infiltration rates into the soil and interception by high vegetation cover. An increase in woody-plant stem density can alter the intensity and frequency of the natural disturbance regime essential to maintaining and revitalizing grassland communities; thus, human intervention becomes necessary through manual tree removal.

Plant-community reassembly of a grassland after manual tree removal is dependent on dispersal ability, seedling development, precipitationm and community assembly history (Martin & Wilsey 2012; Halpern et al. 2016; Barber et al. 2019). Manual tree removal or management by fire can lead to an initial increase in species diversity; however, as restoration sites age, species richness or rate of community reassembly can decline (Halpern et al. 2016; Archer et al. 2017; Barber et al. 2019). Such declines over time could be due to species interactions negatively impacting inferior competitors or a high resilience of the encroached alternative stable state (Barber et al. 2019). If tree removal results in unoccupied spaces, highly competitive native and non-native species can quickly colonize the area and consume resources preventing the establishment of other desirable species (James et al. 2011; Halpern et al. 2016). Environmental conditions, such as drought, can limit new plant recruitment due to increased juvenile mortality of herbaceous plants, particularly in arid grasslands (James et al. 2011).

Soil characteristics can be impacted by changes to ecological conditions and the vegetation community (Evans et al. 2017). Regarding woody encroachment, soil responses can be varied depending on the pre-existing soil type, local climate, and the encroaching species. Variation in vegetation cover as a result of encroachment can increase soil heterogeneity yet reduce water infiltration, heightening the potential for erosion. On the other hand, encroachment can increase soil carbon, total and mineralizable nitrogen, available calcium, and aggregate stability, where grasslands are generally low in soil carbon and nutrients. Soil-litter mixing is slower in arid systems than in wetter systems. Furthermore, exposure of litter to UV radiation with lower percent cover of vegetation in grasslands results in the loss of carbon to the atmosphere and reduced storage in the soil. Conifer encroachment, in particular, can lead to the uptake of nutrients through the release of H+ ions into the soil, increasing acidity (Alfredsson et al. 1998). Acidic conditions induced by coniferous tree encroachment can persist even after tree removal. Physical soil properties and above-ground vegetation dictate what happens to the little water available in arid grasslands (Evans et al. 2017). In arid systems, soil surface water can be increased by encroaching woody plants through decreased evapotranspiration and the hydraulic lift of water from deeper soils. With increased soil porosity, water holding capacity increases, which can support enhanced electrical conductivity. However, the electrical conductivity of soils is dependent on increased organic matter and ion availability for cation exchange (USDA 2011). Increased root growth, organic matter inputs (i.e., litter decomposition), and vegetation ground cover aid in water infiltration and reduces runoff through decreasing compaction and encouraging soil aggregation (Evans et al. 2017). Therefore, above-ground patches of herbaceous vegetation can create a resource island with increased organic carbon, nitrogen, and water-holding capacity. Other factors, such as grazing, can lead to the compaction of soils increasing water runoff and reducing infiltration. Soil resilience can be negatively impacted by fire suppression, drought, and over grazing, often resulting in woody encroachment, increased bare soil cover, and erosion.

Biological soil crusts (biocrust) are found in areas where vegetation cover is discontinuous due to low water availability (Evans et al. 2017). Biocrust is an intermingled layer of lichens, mosses, fungi, cyanobacteria and algae that forms on top of surface soils. These crusts help to prevent erosion and runoff during rain events, while supporting water storage in small pools across the crust's surface. Biocrusts have the potential to support the formation of large soil aggregates through gelatinous sheath excretions; however, aggregation is less likely in dry conditions. In arid systems biocrusts can be the main contributor of nitrogen through nitrogen fixation. Disturbances to the biocrust layer, such as high-intensity fire or excessive trampling, can greatly alter nitrogen fixation, reducing soil fertility.

The legacies of removed trees after treatment must be considered, especially when there is a limited propagule source of grassland species or large amounts of exposed soil (Halpern et al. 2016). When grasses are predominant, woody recruitment from an existing seedbank or nearby mature trees can be prevented even when resources are abundant enough to support seedling development (Ratajczak et al. 2012; Halpern et al. 2016). However, increased precipitation or continued suppression of the natural fire regime can result in the release of woody species from constraining environmental factors. Furthermore, a lack of disturbance may not be the only mechanism supporting woody encroachment. Light availability, which can be altered by the remaining mature trees or invading shrubs can promote woody encroachment (Halpern et al. 2016; Archer et al. 2017). The resulting patches of encroachment can change vegetation structure, soil characteristics and create a microclimate facilitating further woody recruitment (Archer et al. 2017). As a restored grassland ages, the legacy of encroachment can be seen in the recruitment of tree seedlings. Woody species are most vulnerable when small; as a tree develops biomass and sufficient carbohydrate storage over time it becomes resilient to stressors and disturbance increasing its chances of survival to maturity (Archer et al. 2017). Faster growth to a mature life stage through increased carbon dioxide conditions could exacerbate encroachment and conflict with restoration efforts. Therefore, it would be ideal to prevent tree growth prior to the sapling stage while the seedlings are still susceptible to low-intensity disturbances such as fire and grazing.

Manual tree removal can involve burning of slashed and piled trees in confined patches across the restoration area (Steen & Young 2019). Burn-pile sites within the CCPA experienced different fire severity depending on the time of ignition since cutting, available fuel, and relative humidity within the burn-pile. When slashed trees were burned one year after piling, the lack of fine fuels (i.e., limbs lacking needles), large branches creating open air pockets, and a high relative humidity led to partial burning of a pile. Piles that still had an abundance of needles and thinner branches burned more effectively than other piles. The burn severity of a pile can alter plant reassembly order depending on the surviving root structures or seedbanks (Halpern et al. 2016). High-severity burn-pile sites would have reduced surviving vegetation roots and aboveground - cover and thus, be dependent on seed banks and dispersal ability for community reassembly (Martin & Wilsey 2012; Halpern et al. 2016). Increased burn severity may also facilitate the invasion of highly competitive native and non-native species resulting in reduced species diversity and increased invasive-species cover (Halpern et al. 2016; Barber et al. 2019). Monitoring for invasive species after treatment, particularly when bare ground is exposed, is essential to restoration. For example, in the High Lake Benchmark Area invasive species were actively monitored prior to and after treatment, and any plants that were identified were manually pulled, contained in plastic bags, and disposed of.

Moderate- to high-intensity fires tend to have detrimental impacts on the physical and biological characteristics of the soil (Agbeshie et al. 2022). The availability of potassium, calcium, and magnesium could increase or remain unaffected by fire, whereas, sulfur and nitrogen are typically lost to volatilization. Increased fire temperatures can also increase hydrophobicity in soils reducing water infiltration, and increasing surface soil erosion and pH (Diehl et al. 2010; Evans et al. 2017; Agbeshie et al. 2022). Old burn-pile sites experiencing lower burn severity would be expected to have greater root and rhizome survival (Halpern et al. 2016) and a potential increase in soil nutrient availability (Agbeshie et al. 2022).

Through consideration of the specific ecological components of grassland-community recovery, restoration strategies can be assessed and adapted as necessary. Therefore, my research of the High Lake Benchmark Area aims to provide insight into the effectiveness of manual tree removal on the grassland plant community's composition and structure, including new Douglas-fir seedling recruitment, eight years after treatment began. On a broader scale, the project furthers global knowledge of the mechanisms behind community recovery of high-elevation grasslands after the slashing, piling, and burning treatment to reduce woody encroachment.

1.4 Goal and Objectives

The goal of my research was to assess the effects of the slashing, piling, and burning on grassland recovery and Douglas-fir seedling recruitment in four encroachment categories (treatments) that varied in encroaching tree density, maturity and height. Control treatments consisted of open grassland and mature closed forest which remain untreated. Therefore, within a total of six encroachment categories (treatments) I addressed three main questions: (1) has plant structure and composition of the treated grassland become more or less similar to open grassland areas (reference condition) after treatment, (2) how did soil parameters differ between mature forest, encroached areas and reference grasslands and (3) how did the treatment affect new seedling recruitment of woody species? Data from the old burn-pile sites which resulted from the slashing, piling, and burning treatment was meant to pilot explorations on how a condensed burning area of varying intensity impacts grassland-community restoration.

My research objectives were:

1. Determine the species composition and percent cover (%) of the treated grassland community prior to treatment (2015) and post-treatment (2018 and 2023) in the control categories, treated categories (slashed, piled and burned), and old burn-pile sites across the High Lake Benchmark Area.
2. Determine soil pH, compaction (psi), water content (m^3/m^3), temperature (°C), electrical conductivity (bulk dS/m), total organic carbon (%) and nutrients (mg/kg) eight years after

treatment in untreated, treated (slashed, piled and burned) and old burn-pile sites across the High Lake Benchmark Area.

3. Determine the stem density (stems/50 m^2) and average height (cm) of Douglas-fir seedlings (< 1.3 m) eight years after treatment in untreated, treated (slashed, piled and burned) and old burn-pile sites across the High Lake Benchmark Area.

I predicted species composition and percent cover of graminoids, forbs, shrubs, mosses and lichens to be more similar to reference grasslands than mature forests when treatment resulted in a great reduction in tree stem density (James et al. 2011; Halpern et al. 2016; Barber et al. 2019). The mature closed forest and treated areas with greater remaining tree density were expected to have the least similar grassland plant cover and species richness compared to the reference grassland area (Gundale et al. 2008; Archer et al. 2017). Old burn-pile sites with low burn severity were expected to have greater percent cover and species richness of native vegetation than high-severity burn-pile sites. Whereas, old burn-pile sites of high fire severity were expected to have more exposed soil, early successional species (mosses and lichens) and possibly greater percent cover and species richness of non-native vegetation than low-severity burn-pile sites.

Soil pH was expected to be low (acidic) in areas with previous and current encroachment due to the release of H^+ ions and organic acids into soils by tree roots (Alfredsson et al. 1998). Increased root growth and organic matter were predicted to reduce compaction (psi) in areas of encroachment (Evans et al. 2017). I expected to see a higher soil water content in areas having experienced woody encroachment due to hydraulic lift and increased organic matter. In areas with high tree cover, I expected to see reduced soil temperatures (° C). Electrical conductivity (bulk dS/m) was expected to be greater in areas of woody encroachment, than areas with no or low encroachment, due to increased water holding capacity and organic matter (USDA 2011). I predicted the open grassland to have lower total organic carbon (%) and nutrient (mg/kg) levels than areas having experienced encroachment (Evans et al. 2017). Soil pH in old burn-pile sites was expected to vary with fire severity, where higher severity resulted in hydrophobic soils, pH was expected to increase (Diehl et al. 2010). Soil compaction was expected to be similar to the adjacent vegetation plot as they share similar tree encroachment levels. Old burn-pile sites were expected to have lower water content and electrical conductivity than unburned sites due to increased hydrophobicity (Evans et al. 2017; Agbeshie et al. 2022). In sites that experienced higher fire severity I expected low organic carbon and nutrient availability as a result of volatilization. Soil parameters were also anticipated to differ according to slope, aspect and elevation therefore, I expected differences not only between open grassland, treated encroached categories and mature closed forest but also between individual sampling plots.

I expected to see the lowest tree density (stems/50 m^2) within the untreated open grasslands. In treated encroached categories, I predicted the tree stem density of newly recruited seedlings to increase with mature tree density (Halpern et al. 2016). Average seedling height was expected to increase where a propagule source was available yet competition with other trees was low. Regeneration on old burn-piles sites was expected to follow the same pattern as their adjacent vegetation plots.

2. Methods

2.1 High Lake Benchmark Area

The Churn Creek Protected Area (CCPA) is located in the Central Interior Ecoprovince of British Columbia with a climate of cold winters and warm summers where most precipitation falls in late spring to early summer (Sinclair et al. 1999). Within this ecoprovince is the Fraser River Basin Ecosection, which, has the driest warmest climate of the Fraser River Plateau Ecoregion. Mean annual temperature, mean monthly precipitation and total precipitation between 2015 and 2023 at a high-elevation (1100 m) weather station in the CCPA was 6.3° C, 21 mm and 243 mm, respectively (Pacific Climate Impacts Consortium 2024) (Table 1). In 2015, mean (±Standard Deviation) annual temperature was 6.9±8.9° C, mean monthly precipitation was 23±13 mm and total annual precipitation was 276 mm. In 2018, mean (±Standard Deviation) annual temperature was 8.7±8.8° C, mean monthly precipitation was 28±25 mm and total annual precipitation was 250 mm. In 2023, mean (±Standard Deviation) annual temperature was 6.8±9.4° C, mean monthly precipitation was 17±15 mm and total annual precipitation was 199 mm. Climate data was taken hourly over the nine-year period however, in 2018, 2020, 2022 and 2023 consecutive days or months of data were not recorded over the winter or spring season resulting in skewed temperature and precipitation values.

Roughly 16,275 ha of the very dry and mild Interior Douglas Fir biogeoclimatic zone (IDFxm) can be found in the CCPA (BC Parks 2000) and consists of a mosaic of open grassland and forest. Forest patches are typically found on cool, wet sites and open grasslands on warm, dry sites. Typical grassland vegetation of the IDFxm zone covers nearly one hundred percent of the ground with a thick litter layer barring any recent wildfire events. Bluebunch wheatgrass (*Pseudoroegneria spicata*), short-awned porcupine grass (*Hesperostipa spartea*) and spreading needlegrass (*Achnatherum richardsonii*) are the predominant late-seral and climax species of the IDFxm zone. However, short-awned porcupine grass and spreading needlegrass are less common in the CCPA than areas further north, making bluebunch wheatgrass and a diverse array of forbs the predominant vegetation (Sinclair et al. 1999).

Grasslands within the High Lake Benchmark Area range between 1100 and 1200 m in elevation with level to gently rolling slopes (Steen & Young 2019). Soils were formed under fairly arid conditions due to the areas geographic location on the lee side of the coastal mountains (Sinclair et al. 1999). Silty or loamy aeolian materials contributed to developing a veneer (20 to 50 cm deep) over basal moraine. Vegetation is predominated by grasses and forbs atop a rapidly drained Brown Chernozemic soil on warmer slopes and well-drained Dark Brown Chernozemic soils are found on cooler aspects. Calcareous Chernozems can also be found within the CCPA. A typical example of Dark Brown Chernozem is an Ah horizon immediately atop a poorly developed Bm horizon which overlies the Cca horizon that is rich in precipitated calcium carbonate (Canadian Society of Soil Science 2020). An organic rich Ah horizon of 15- to 30-cm thick has developed as a result of leaf and root decomposition of herbaceous vegetation (Sinclair et al. 1999; BC Parks 2000). Brown Chernozems are typically found in dry conditions and contain little organic matter (Canadian Society of Soil Science 2020).

2.2 Experimental Design and Encroachment Categories

Encroaching tree density, height and maturity varied throughout the grasslands prior to treatment. To understand the effectiveness of manual tree removal, the Friends of Churn Creek Protected Area Society (FCCPAS), developed a complete random design to monitor grasslands experiencing different levels of tree encroachment. The area was stratified into six categories ranging from open grassland, encroached grasslands and mature forest (Table 2). In 2015, three 5.64 m fixed-radius plots (100 m^2) per encroachment category were randomly assigned using Google Earth imagery (Figure 1) and ground truthed prior to sampling (6 encroachment categories x 3 samples = 18 plots) (Steen & Young 2019). Between 2015 and 2016, the felling, bucking, piling and burning of trees was conducted across approximately 73 ha of encroached grassland at High Lake. Bucking is the removal of branches and cutting of trees to smaller lengths (approx. 3 m). The majority of trees with a diameter at breast height (DBH) of greater than 17.5 cm were not cut, due to the increased work demand and safety protocol required for cutting larger trees. Pile sites were burned in December, 2017 and, November and December, 2018, with lightly snow covered ground. Treatment was not carried out in grasslands deemed to be in reference condition (2.9 ha), or in areas of mature closed forest (3.9 ha).

Plot Characteristics

Across each plot, in 2023, I recorded slope grade (%) and aspect (°) using a clinometer (Suunto PM-5/360 PC, Finland) and compass (SILVA Expedition S, Canada), respectively. Measurements spanned a length of 11.28 m from the uppermost point of the plot to lowest point of the plot (Table 3). I determined the mesoslope position of each plot using the *Field Manual for Describing Terrestrial Ecosystems* (1998). From the centre of each plot I recorded elevation (m) and GPS coordinates (UTM 10, NAD83; Garmin, GPSMAP64st, Canada) as well as, percent open canopy (%) using a concave forest densiometer.

In 2023, site characteristics varied within and between encroachment categories (Figure 1, Table 3). Plot percent slope ranged from 0% (plot 11) to 19% (plot 2) and slope aspect was generally south facing, ranging between the east (90°, plot 8) and the southwest (219°, plot 17). Elevation was lowest in plot 3 (1201 m) and greatest in plot 12 (1265 m). In 2023, there was 100% open canopy within the N (open grassland), ELO (encroachment, low, open) and ETO (encroachment, tall open) categories. In the ETD & ETM (encroachment, tall, dense/moderate) category percent open canopy ranged from 18 to 100%. In the MO & ILD (mature open forest) category percent open canopy ranged from 79 to 100% and in the MC (mature closed forest) category percent open canopy ranged from 36 to 39%.

2.3 Vegetation Assessment

Baseline Vegetation Assessment and Monitoring (2015 and 2018)

Baseline data of graminoid, forb and shrub percent cover to the species level was collected prior to treatment, between July and August, 2015. Graminoid, forb and shrub percent cover to the species level was measured across all sites in September, 2018.

Vegetation Monitoring and Assessment (2023)

In June and August, 2023, I characterized the plant community at each fixed-radius plot, using the same methods as 2015 and 2018. I was trained by the same observer that conducted sampling in 2015 and 2018; however, he was not a sampler during the 2023 monitoring.

Plant Species Composition & Cover

In both June and August 2023, I sampled vegetation composition and percent cover. I measured percent cover to the species level of shrubs, forbs and graminoids within a fixed-radius plot of 5.64 m (100 m^2). Percent cover of mosses, lichens, plant litter, bare mineral soil, burned area and other objects (e.g., wood) was also measured. I identified all plants to the species or genus level when possible (Douglas et al. 1998a, 1998b, 1999a, 1999b, 2000, 2001a, 2001b, 2002; Parish et al. 2018). When identification was not possible a species was classified according its group, for example: "moss spp." or "lichen spp.". From the center of every plot I took one photo in each of the four cardinal directions.

Prior to sampling in 2023, I considered quadrats to be a more common method for measuring herbaceous vegetation cover and thought there was potential to be more efficient. Therefore, within a 100 m^2 fixed-radius plot, I placed four 1 m x 1 m subplots (1 m^2) to measure percent cover of shrubs, forbs, graminoids, mosses and lichens (Luttmerding & BCMELP 1990; Dickson & Foster 2008; Barber et al. 2019). Bare mineral soil, burned area and other objects (e.g., wood) were measured as well. I placed these subplots in each of the four cardinal directions, 2.82 m from the center of the larger 100 m^2 plot. Within the larger plots many species were found in trace amounts that were not captured in the smaller subplots. Due to the misrepresentation of species diversity in the smaller quadrats, I discontinued the subplot sampling and determined vegetation percent cover of the whole 100 m^2 fixed-radius plot.

Vegetation diversity and evenness was calculated for each plot and averaged per encroachment category per year using the Shannon-Wiener Diversity (equation 1) and Equitability (equation 2) Indices (Shannon 1948; Fedor & Zvarikova 2018).

Equation 1.

$$H = -\sum_{i=1}^{s} p_i \ln p_i$$

Where H is the index for vegetation species and p_i is the proportion of a given species' percent cover relative to the total percent vegetation cover of all species (s) of the plot.

Equation 2.

$$E_H = H/\ln S$$

Where E_H is the index for vegetation species evenness across a plot, H is the diversity index and S is the total number of species identified within the plot. Within the indices calculations, plants that were not consistently identified to the species level were grouped together according to the extent that they were identified (i.e., genus, family or group). For example, all species from the genus *Antennaria* were grouped together as "*Antennaria spp*", and their individual percent covers summed. The Shannon-Wiener diversity and equitability indices are hereinafter referred to as diversity and evenness, whereas species richness indicates the sum of species found within a given plot or encroachment category.

Plant Species Composition & Cover of Old Burn-pile sites

When an old burn-pile site was found within a few meters of a fixed-radius plot I recorded percent cover and species richness of shrubs, forbs, graminoids, mosses, lichens, litter and bare mineral soils using a 1 m x 1 m (1 m^2) quadrat. Not all treated encroachment categories had old burn-pile sites, either piling was not done in that area or pile sites were never burned. Therefore, there were an unequal number of burn-pile sites per encroachment category. All old burn-pile sites were sampled once between June and August, 2023. A photo of each old burn-pile site was taken from above to capture the entirety of the 1 m^2 plot. A qualitative description was used to determine the fire severity of each old burn-pile site.

2.4 Soil Characteristics

In June 2023, soil attributes were measured in the field in two locations within each of the 5.64 m fixed-radius plots. I chose two diagonally opposite sampling locations, one upslope, the other downslope, in areas of representative vegetation. Soil compaction was determined using a Dickey-John soil compaction tester (DjP/N:15585003D, S/N:1347-27602, D/C:201536, REV: E, United States of America). Water content (m^3/m^3), temperature (° C) and electrical conductivity (bulk dS/m) of the soil was measured using a Procheck soil probe (METER Group). When the probe could not provide an electrical conductivity reading due to dry soils approximately 15 ml of deionized water was added to the probe area after the water content and temperature measurements were recorded. Each measurement was taken 5- and 15-cm below the surface (e.g. 2 locations per plot x two depths per location = 4 measurements per plot). Due to the compaction and rocky components of the soil, pH was measured via a Kelway Soil Tester after an initial loosening of the soil via an auger. Soils were returned immediately to the soil pit to minimize disturbance.

Soils samples for laboratory testing were collected outside and immediately adjacent to selected plots, in an area of representative vegetation. Three soil samples were taken from each sampling location to a depth of 10 cm using the soil auger, then composited as one sample per location, sealed together in a Ziploc plastic bag and stored in a cooler with ice until sent for analysis. Due to limited funding I was only able to have six fixed-radius plot and two burn-pile site samples tested. Two samples each from the untreated open grassland (N) and mature closed forest (MC) plots were chosen for sampling to set a standard for the two extremes of the six encroachment categories in the High Lake Benchmark Area. One sample each from the low (ELO) and tall treed (ETO) open canopied categories was chosen to represent the treated areas. Samples from the ETD & ETM and MO & ILD categories were not selected. Plot selection was based on best covering a range of encroachment categories and elevations while trying to maintain two samples based on visual and tactile descriptions of the soil. Soil samples were tested at AGAT Laboratories for pH (pH meter), electrical conductivity (conductivity meter), soluble chloride (ion chromatograph), soluble calcium, potassium, magnesium and sodium (inductively coupled plasms – optical emission spectrometer), specific gravity (Baroid; mud balance), available nitrate, potassium and sulfur (inductively coupled plasms – optical emission spectrometer) and available phosphorus (discrete analyzer).

Soil Characteristics of Old Burn-pile Sites

Soil pH, water content (m^3/m^3), temperature (° C), electrical conductivity (bulk dS/m) and compaction (psi) of the 1-m^2 old burn-pile sites was measured using the same methods and equipment as in the 5.64 m fixed-radius plots. Two soil samples from the ETO category were sent to AGAT Laboratories and the same analyses conducted as those from the larger 5.64 m fixed-radius plots.

2.5 Douglas-fir Seedling Recruitment

Tree Seedling Recruitment Monitoring (2015, 2018 & 2023)

Monitoring of tree species, stem density and characteristics was carried out for stems shorter than 1.3 m within the treated encroachment categories before the slashing, piling, and burning treatment in 2015 and post-treatment in 2018 (Steen & Young 2019). In June 2023, I measured stem density, height (cm) and diameter at 30 cm (cm) of Douglas-fir seedlings within a fixed-radius plot of 3.99 m (50 m^2; Steen & Young 2019). A seedling was considered any Douglas-fir shorter than 1.3 m. Only the first 50 stems were recorded and an estimation of the proportion of plot counted was noted.

Douglas-fir Seedling Recruitment in Old Burn-pile Sites

In June and August 2023, the 1 m^2 old burn-pile sites found adjacent to the larger vegetation plots were also sampled for Douglas-fir seedlings (< 1.3 m height). Stems were counted and height measured using the same methodology as the larger 50 m^2 plots.

2.6 Data Analysis

Vegetation Assessment

Vegetation percent cover data from the 5.64 m fixed-radius baseline (2015) and post-treatment (2018 & 2023) monitoring samples was compiled for statistical analyses. Percent cover of the common bearberry (*Arctostaphylos uva-ursi*), a dwarf shrub that forms a low-lying mat with flexible branches, was included in the forb group as it is considered a part of the lower herb layer (NRCS Northeast Plant Materials Program 2006). Species richness, Shannon's diversity and equitability indices, and percent cover of vegetation were averaged according to encroachment category, per year (n = 3). As sampling in 2015 and 2018 occurred when herbaceous growth was at its seasonal prime (between July and September) only the August sampling from 2023 was used to compare encroachment categories statistically over time. Percent cover of vegetation from old burn-pile sites was averaged according to encroachment category. Some of the 5.64 m fixed-radius plots had more associated burn-piles sites than others resulting in an unequal number of replicates per encroachment category (ELO, n = 2, N = 4; ETO, n = 3, N = 6; ETD & ETM, n = 3, N = 6; and MO & ILD n = 2, N = 4; where n is the number of burn-piles sampled and N is the total number of burn-piles observed).

Soil Characteristics

Soil sampling from the Procheck soil probe showed no discernable difference between the two depth measurements (5 and 15 cm) therefore, all four samples per plot contributed to a single encroachment category average. In the old burn-pile sites plot 12 had only one Procheck soil probe measurement (5 cm depth) and no pH measurement due to excessively rocky soil.

Douglas-fir Seedling Recruitment

I weighed each Douglas-fir seedling count by the proportion of the plot that was measured. For example, plot 2 had a count of 50 seedlings covering three quarters of the plot. An estimated number of seedlings within the entire plot would then be 67. The estimated stem density per 50 m^2 plot was then converted to the number of stems per hectare for comparison with the 2015 and 2018 stem density values. Average seedling height (cm) was calculated per plot and averaged per encroachment category.

Statistical Analysis

All statistical analyses were conducted in the R programming language (R Core Team 2023). Shannon's diversity and equitability indices, vegetation percent cover, soil characteristics (pH, compaction, water content, temperature and electrical conductivity) and Douglas-fir seedling stem density and height were statistically analyzed using ANOVAs followed by Tukey's post-hoc test. When the assumption for normality was not met the Kruskal-Wallis and Wilcoxon signed-rank (rstatix package) tests were used. When the assumption for homoscedasticity of variances could not be met Welch's ANOVA (multcomp package) and Game-Howell post-hoc (rstatix package) tests were used. The assumptions for normality and homoscedasticity of variances were determined using the Shapiro-Wilk and Levene (car package) tests, respectively. The ggplot2 and patchwork packages were used to compute graphs and the count (plyr package) and wide (tidyverse package) functions were used to calculate mean species values and the diversity and equitability indices.

Statistical analyses were not conducted on the old burn-pile sites nor the soil laboratory analyses due to limited sample sizes. The methods for sampling tree seedling recruitment in 2015 and 2018 were very similar to 2023 sampling however, not identical, thus statistical analyses were not conducted to compare new seedling recruitment between years.

3. Results

The slashing, piling, and burning of Douglas-fir stems less than 17.5 cm in diameter at breast height (DBH) in the High Lake Benchmark Area resulted in the removal of thousands of stems from the treated encroachment categories (Steen & Young 2019). The untreated categories, open grassland (N) and mature closed forest (MC) had the same stand characteristics as pre-treatment (Table 2). The category with low tree height and open canopy (ELO) had a handful of remaining short stems. The tall treed and open canopy category (ETO) had a few short stems and uncut larger trees (DBH >17.5 cm). In the moderate to densely treed category (ETD & ETM) an estimated 1300 stems/ha were removed reducing the density of stems with a DBH greater than 1.5 cm by nearly 50%. In the mature open forest (MO & ILD) live stems greater than 1.3 m in height yet less than

17.5 cm in DBH were removed leaving behind a small regeneration of young seedlings and mature uncut trees.

3.1 Plant Community Assessment

Plant Species Composition & Cover

According to Steen and Young (2019), a total of 63 species were identified prior to treatment in 2015, increasing in 2018, with a total of 69 species observed across all sampling plots. In June and August, 2023, I observed 65 and 63 species, respectively. Pre-treatment (2015), mean species richness per plot was 23±4.6 species (Mean±Standard Deviation), where plot 10, within mature open forest (MO & ILD) had the highest species richness (32 species) and plot 17, in the open grassland (N) category had the lowest richness (15 species) (Table 4). Post-treatment, in 2018, the mean species richness per plot was 24±4.0 species where plot 6 in the tall treed and open canopy (ETO) category had the highest species richness (29 species) and plot 17 (N) had the lowest richness (14 species). In the early season sampling of 2023 (June) mean species richness per plot was 21±4.6 species where plot 13 with lower tree height and open canopy (ELO) had the greatest species richness (27 species) and plot 17 (N) had the lowest (10 species). In August, 2023, mean species per plot was 22±6.2 species, where plots 10 (MO & ILD), 13 (ELO) and 6 (ETO) shared the greatest species richness (30 species) and plot 17 (N) had the lowest (10 species).

<u>Shannon's Diversity and Equitability Indices</u>

Shannon-Wiener diversity (H) and equitability (E_H) indices changed over time and between encroachment categories (Table 5, Table 6). In 2015, diversity and evenness were lowest in the N and the mature closed forest (MC) categories. In 2015, the four treated categories ELO, ETO, ETD & ETM (moderate to dense tree cover) and MO & ILD had greater diversity and evenness than the untreated N and MC categories. One-way ANOVAs indicated significant differences in both diversity ($F(5, 12) = 3.80$, $p = 0.027$) and evenness ($F(5, 12) = 3.38$ $p = 0.039$) between encroachment categories in 2015 however, post-hoc testing was not significant.

In 2018, diversity and evenness remained similarly low in N and MC (Table 5, Table 6). Whereas, ELO, ETO and ETD & ETM showed a decline in diversity and evenness compared to 2015. The MO & ILD category remained stable in both diversity and evenness. There were no significant differences in diversity ($F(5, 12) = 1.38$, $p = 0.300$) or evenness ($F(5, 12) = 0.673$, $p = 0.652$) between encroachment categories in 2018.

Two encroachment categories were significantly different in diversity ($F(5, 12) = 4.56$, $p = 0.015$) in August 2023; ELO ($H = 2.3 \pm 0.62$) was significantly greater than N ($H = 0.86 \pm 0.50$; $p = 0.036$). Initial results comparing evenness across encroachment categories in 2023 indicated a significant difference between groups, however the post-hoc tests were insignificant.

In 2023, both indices showed a decrease in N) however, these declines were not significant for either diversity ($F(3, 8) = 0.289$, $p = 0.833$) or evenness ($F(3, 8) = 0.277$, $p = 0.840$) when compared to other sampling years (Table 5, Table 6). Both indices in MC were similar to 2018 and there were no significant differences over time in diversity ($F(3, 8) = 0.152$, $p = 0.926$) and evenness ($F(3, 8) = 0.319$, $p = 0.811$). In 2023, ELO showed a non-significant increase in diversity ($F(3, 8) = 2.77$, $p = 0.111$) and evenness ($F(3, 8) = 2.8$, $p = 0.109$). The ETO category also increased in diversity ($F(3, 8) = 0.364$, $p = 0.781$) and evenness ($F(3, 8) = 0.447$, $p = 0.726$) compared to 2018, however, it was not a significant change. The MO & ILD category increased in both diversity ($F(3, 8) = 2.39$, $p = 0.144$) and evenness ($F(3, 8) = 2.02$, $p = 0.189$) compared to 2015, however, this increase was not significant. Diversity ($F(3, 8) = 5.14$, $p = 0.029$) and evenness ($F(3, 8) = 4.99$, $p = 0.031$) in the ETD & ETM category was significantly lower in 2023 ($H = 1.1 \pm 0.40$, $p = 0.042$; $E_H = 0.36 \pm 0.10$, $p = 0.046$) than 2015.

<u>Graminoids</u>

A total of 15 graminoid species have been found since sampling began in 2015 (Table 8). The species having the greatest percent cover across multiple encroachment categories and over time were: Nelson's needlegrass (*Achnatherum nelsonii*), spreading needlegrass (*Acnatherum richardsonii*), Pinegrass (*Calamagrostis rubescens*), junegrass (*Koeleria macrantha*), Kentucky bluegrass (*Poa pratensis*) and bluebunch wheatgrass (*Pseudoroegneria spicata*). Pine grass was found in only three of the six encroachment categories.

In 2015, mean total graminoid percent cover between encroachment categories were not significantly different ($c^2(5, N = 18) = 5.19$, $p = 0.394$) (Figure 2, Table 7). Mean graminoid percent cover was greatest in the open grassland (N) and lowest in the tall and open canopied (ETO) and mature closed forest (MC) categories, in 2015. Pinegrass showed significant differences between encroachment categories in 2015 ($c^2(5, N = 18) = 13.5$, $p = 0.019$) however, post-hoc testing was not significant.

Mean total graminoid percent cover in 2018 was not significantly different between encroachment categories ($F(5, 12) = 1.352$, $p = 0.308$) (Figure 2, Table 7). The category with the greatest mean total graminoid percent cover in 2018 was N and the category with the lowest cover was MC, in 2018. Initially, there appeared to be significant differences in pinegrass between encroachment categories in 2018 ($c^2(5, N = 18) = 11.2, p = 0.048$); however, post-hoc testing was insignificant (Figure 4, Table 8).

In June, 2023, the mean total percent graminoid cover appeared to be significantly different between encroachment categories ($F(5, 12) = 3.24, p = 0.044$); however, post-hoc testing was insignificant (Figure 3, Table 7). The N category had the greatest mean total percent graminoid cover and the tall treed, open canopy (ETO) category was the lowest.

In August, 2023, mean total percent graminoid cover was significantly different between encroachment categories ($F(5, 12) = 6.46, p = 0.004$) (Figure 2, Table 7). The N (37±4.6%) category had significantly greater mean total graminoid percent cover than the ETD & ETM (moderate to densely treed; 8.8±10%, $p = 0.021$), ETO (4.5±1.8%, $p = 0.008$) and MC (5.8±8.0%, $p = 0.010$) categories.

Mean percent graminoid cover was not significantly different over time in the N, ELO (low treed encroachment), ETO, MO & ILD (mature trees with open canopy) and MC categories. Mean percent graminoid cover in the N category increased from 2015 to 2018, then declined in 2023 (Figure 2, Table 7). The ELO category had lower mean graminoid percent cover in 2015 than 2018 and the lowest cover in 2023. The ETO category had reduced graminoid percent cover in 2023 when compared to 2015 and 2018. Nelson's needlegrass in ETO had significantly lower percent cover in 2023 (0.07±0.11%) than 2018 (8.0±4.0%, $p = 0.044$) yet was not significantly different from 2015 (7.0±3.5%, $p = 0.072$) (Figure 4, Table 8). Mean total percent graminoid cover was significantly different between years in the ETD & ETM category ($F(2, 6) = 12.5, p = 0.007$); mean percent graminoid cover in 2023 (8.8±10%) was significantly lower than 2015 (42±17%, $p = 0.043$) and 2018 (61±9.9%, $p = 0.006$). Percent cover of Nelson's needlegrass was different between sampling years in the ETD & ETM ($F(2, 6) = 22.3, p = 0.001$) category with percent cover in 2023 (0.2±0.1%) being significantly lower than both 2015 (16±6.9%, $p = 0.010$ and 2018 (23±2.9%, $p = 0.001$). In the MO & ILD category mean percent graminoid cover was slightly lower in 2015 than 2018 and lower still in 2023. The MC category declined in mean percent graminoid cover over time.

<u>Forbs</u>

The forb group was the most species rich as a total of 59 forb species were identified in the High Lake Benchmark Area since 2015 (Table 9). Many of the forb species identified occurred in trace amounts; however, four species were found in the highest percent cover across multiple encroachment categories and over time: yarrow (*Achillea millefolium*), pussytoes (*Antennaria* spp.), silverweed (*Potentilla hippiana*) and common dandelion (*Taraxacum officinale*). Certain species of the same genus (e.g., *Antennaria spp.*) were identified separately in the field then grouped together for statistical analysis to keep sampling methodologies consistent across sampling years.

In 2015 total mean forb percent cover between encroachment categories was not significantly different ($F(5, 12) = 2.18$, $p = 0.125$) (Figure 5, Table 7). However, yarrow had significantly different mean percent cover ($F(5, 12 = 8.64$, $p = 0.001$) across encroachment categories in 2015. Yarrow percent cover in the low treed and open canopy category (ELO; 1.7±0.58%) was significantly greater than the open grassland (N; 0.27±0.24%, $p = 0.004$), tall treed and open canopy (ETO; 0.30±0.17%, $p = 0.004$), tall treed and moderate to dense canopy (ETD & ETM; 0.42±0.51%, $p = 0.008$), mature open forest (MO & ILD; 0.30±0.17%, $p = 0.004$) and mature closed forest (MC ;0.03±0.06, $p = 0.001$) categories (Figure 5, Table 7).

Total mean forb percent cover in 2018 was significantly different between encroachment categories ($F(5, 12 = 3.29$, $p = 0.042$) (Figure 5, Table 7). The MC (2.6±2.8%, $p = 0.028$) category had significantly lower total mean forb percent cover than the ETO (36±23%) category.

In June, 2023, mean total percent forb cover was significantly different between encroachment categories ($c^2(5, N = 18) = 11.9$, $p = 0.036$); however, post-hoc testing revealed no significant difference between groups (Figure 5, Table 7).

In August, 2023, mean total percent forb cover was significantly different between encroachment categories ($c^2(5, N = 18) = 12.3$, $p = 0.031$); however, post-hoc testing revealed no significant differences between pairs (Figure 5, Table 7). Yarrow percent cover initially appeared to be significantly different across encroachment categories in 2023 ($c^2(5, N = 18) = 12.7$, $p = 0.026$); however, post-hoc testing revealed no significant differences between groups (Figure 6, Table 9).

Mean forb percent cover changed over time in most encroachment categories; however, not all of these changes were significant. Mean forb percent cover in N was highest in 2015 followed by 2018 and lowest in 2023 (Figure 5, Table 7). Mean percent forb cover changed over time in the ELO category ($F(2, 6 = 5.81, p = 0.040)$, where 2023 (5.0±1.0%) was lower than 2018 (13± 1.9%) and significantly lower than 2015 (16±6.8%, $p = 0.040$). In the ETO category mean forb percent cover was lower in 2015 than 2018 and lowest in 2023. Percent cover of the common dandelion appeared significantly different between years in the ETO category ($c^2(2, N = 9) = 6.12$, $p = 0.047$); however, post-hoc tests indicated no significance (Figure 6, Table 9). Mean percent forb cover in the ETD & ETM category increased from 2015 to 2018 then decreased in 2023. In the ETD & ETM category there were significant differences in percent cover of the common dandelion ($F(2, 6 = 5.85, p = 0.039)$, where percent cover in 2023 (0.0±0.0%) was significantly lower than 2018 (0.73±0.46%, $p = 0.04$). In MO & ILD, mean percent forb cover was similar between 2015 and 2018 then declined in 2023. The mature closed forest (MC) had similar mean forb percent cover across all years.

Shrubs

A total of five shrub species were identified across the High Lake Benchmark Area since 2015 (Figure 8, Table 10). The common rabbitbrush (*Ericameria nauseosa*) and the common (*Juniperus communis*) and Rocky Mountain (*Juniperus scopulorum*) junipers were found in at least five of the six encroachment categories over time. Whereas, the prairie rose (*Rosa woodsii*) and western snowberry (*Symphoricarpos occidentalis*) were only found in the two mature treed categories (MO & ILD and MC).

Total mean percent shrub cover was not significantly different between encroachment categories in 2015 ($F(5, 12) = 1.18$, $p = 0.372$), 2018 ($F(5, 12) = 1.50$, $p = 0.262$), June 2023 ($F(5, 12) = 1.32$, $p = 0.320$) and August 2023 ($F(5, 12) = 1.48$, $p = 0.266$) (Figure 7, Table 7).

There were no significant differences in mean shrub percent cover nor individual species percent cover over time. Open grasslands (N) had similar mean shrub percent cover between 2015 and 2018 with a decline in mean shrub cover in 2023 (Figure 7, Table 7). Shrub percent cover in the N category consisted of only one species, the common rabbitbrush (Figure 8, Table 10). In the category with open canopy and lower tree height (ELO) mean percent shrub cover also declined over time. Both common rabbitbrush and Rocky Mountain juniper declined in percent cover over time. In 2015, mean shrub percent cover was lowest in the category with taller trees and open canopy (ETO) increasing in 2018, and highest in 2023. The Rocky Mountain juniper had the greatest change over time in ETO increasing between 2015 and 2023. The category with moderate to dense tree cover (ETD & ETM) had greater mean shrub percent cover in 2015 than in 2018.

Mean shrub percent cover was highest in 2023. The Rocky Mountain Juniper mean percent cover reflects these changes in shrub mean percent cover in ETD & ETM. In the mature, open forest (MO & ILD) mean shrub percent cover increased over time. Mean percent cover of the Rocky Mountain and common (*Juniperus communis*) junipers follow the increasing mean shrub percent cover trend in MO & ILD. The mature closed forest (MC) had stable mean shrub percent cover over time. Mean percent cover of the Rocky Mountain juniper increased while all other species mean percent covers remained constant or decreased over time in MC.

Mosses & Lichens

Mosses included feather mosses or simply moss spp., and lichens were grouped as *Cladonia* spp., *Peltigera* spp. or simply lichen spp. (Table 10). Mosses and lichens were found in every encroachment category in 2015 and 2023. In 2018, mosses were not found in the open grassland (N) and low encroachment, open canopied (ELO) categories, yet lichens were. Mosses and lichens were found across all remaining categories in 2018.

In 2015, total moss percent cover initially appeared significantly different across encroachment categories ($c^2(5, N = 18) = 12.6, p = 0.028$); however, post-hoc tests indicated no significance between groups (Figure 9, Table 10). Lichen percent cover, in 2015 was not significantly different across encroachment categories ($F(5, 12) = 2.30, p = 0.111$). Mean moss percent cover in 2018 was significantly different across encroachment categories ($c^2(5, N = 18) = 14.1, p = 0.015$). However, the post-hoc Wilcoxon signed-rank test revealed no significant differences between encroachment categories. Lichen percent cover in 2018, was not significantly different between encroachment categories ($c^2(5, N = 18) = 4.94, p = 0.422$). In August, 2023 mean percent cover between encroachment categories was not significantly different for moss ($F(5, 12) = 1.01, p = 0.452$) or lichen ($c^2(5, N = 18) = 9.18, p = 0.102$).There were no significant differences in total moss and lichen percent cover between sampling years in the same encroachment category.

Bare Mineral Soil, Burned Area & Litter

Bare mineral soil mean percent cover varied throughout time and space; however, these differences were not significant (Figure 10, Table 10). In 2015, percent cover of bare soil was greatest in the open grassland category (N) and lowest in the MC category where no bare soil percent cover was recorded. Bare mineral soil percent cover in 2018 was greatest in the N category and lowest in the ETD & ETM category. In 2023, bare soil percent cover was highest in the ETO category and lowest in the MC category.

Initially, there appeared to be significant differences in bare mineral soil between years in the ELO category ($F(2, 6) = 5.29$, $p = 0.047$) (Figure 10, Table 10). However, percent cover in 2023 (37±15%) was not significantly greater than 2015 (11±13%, $p = 0.082$) and 2018 (8.0±6.1, $p = 0.058$). There were no significant differences in percent cover of bare mineral soil between encroachment categories in 2015 ($F(5, 12) = 1.811$, $p = 0.185$), 2018 ($c^2(5, N = 18) = 5.61$, $p = 0.346$) and 2023 ($F(5, 12) = 2.16$, $p = 0.128$). There were no significant differences in percent cover of bare mineral soil across sampling years in the N ($F(2, 6) = 0.049$, $p = 0.953$), ETO ($F(2, 6) = 0.916$, $p = 0.450$), ETD & ETM ($c^2(2, N = 9) = 5.22$, $p = 0.074$), MO & ILD ($F(2, 6) = 0.402$, $p = 0.686$) and MC ($F(2, 6) = 1.85$, $p = 0.236$).

There was no burned area percent cover in pre-treatment (2015) sampling (Figure 10, Table 10). In 2018 and 2023, burned areas were not found in the untreated N and MC categories. In August 2023, plot 12 in the MO & ILD category was recorded as having 0.5% cover of burned area; however, upon further investigation with plot photos from both June and August 2023 the burned area measurement was removed. Burned areas were recorded in the ELO, ETO and ETD & ETM categories and ranged from 0.67±1.2% (ELO) to 8.7±6.0% (ETD & ETM) in 2018 and 0.67±1.2% (ELO) to 7.2±6.8% (ETD & ETM) in 2023.

Litter was found in all categories across all sampling years at the High Lake Benchmark Area (Figure 10, Table 10). In areas with few trees litter was predominantly made up of dead graminoids and forbs whereas in the mature open and closed forested areas Douglas-fir needles predominated the litter. Litter percent cover was not significantly different between encroachment categories in 2015 ($F(5, 12) = 0.866$, $p = 0.531$), 2018 ($F(5, 12) = 0.528$, $p = 0.751$) or 2023 ($F(5, 12) = 1.19$, $p = 0.0.372$). There were also no significant differences in litter percent cover between sampling years for each encroachment category. In 2015, MC had the greatest litter percent cover and ETO the least. In 2018, the MC category had the highest litter percent cover and the ELO category the lowest. Percent cover of litter in 2023 was greatest in the mature open forest (MO & ILD) and MC categories and lowest in ETO.

Plant Species Composition & Cover of Old Burn-pile sites

A total of 18 old burn-pile sites were sampled in the treated encroachment categories between June and August 2023. Graminoids were found in 16 of the 18 plots and a total of eight species were identified, including the six predominant species used to analyze the larger vegetation plots. Graminoid species percent cover ranged from 0.0 to 5.0%, with the exception of one burn-pile in MO & ILD having 15% cover of bluebunch wheatgrass. Forbs were found in 15 of the 18 plots and a total of 20 species were identified. Forb species percent cover within old burn-pile sites range from 0.0 to 5.0%. A single *Salix* spp. (shrub) with 2% cover was found in one old burn-pile site in the ETD & ETM category. Mosses were found in 15 of the 18 burn-piles ranging from 0.0 to 60% cover. Fire moss (*Ceratodon purpureus*) was the only moss identified to the species and was found in 5 burn-pile sites. *Peltigera* spp. (lichen) was found in one old burn-pile site with a 0.5% cover. Bare mineral soil was recorded in 12 burn-piles ranging from 5.0% cover in an ETO plot to 95% cover in an ETD & ETM plot. Litter and wood pieces were recorded in 17 of the 18 burn-pile sites. Litter percent cover ranged between 0.0 and 90% and wood percent cover ranged from0.0 to 50% cover.

3.2 Soil Assessment

Soil characteristics varied across encroachment categories. Soils in the open grassland category (N) were very fine to the touch and ranged from light brown to brown. In the treated categories with low to tall trees and open to dense canopy (ELO, ETO and ETD & ETM) soils ranged from light brown to brown and were very fine to fine in texture. In the N, ELO, ETO and ETD & ETM categories there was no discernable organic layer and many rocks were present (approx. 5-10 cm width). In the mature open and closed forests (MO & ILD and MC) there was a dark organic layer above a grey-brown granular layer.

Soil pH, Compaction, Dry Bulk Density, Water Content & Temperature

Soil pH was significantly different between encroachment categories ($c^2(5, N = 36) = 14.5$, $p = 0.013$); however, post-hoc Wilcoxon signed-rank test revealed no significant differences (Figure 11, Table 13). The MC, MO &ILD, and ETO categories had slightly higher pH than N, ELO, and ETD & ETM. Soil pH from laboratory testing was lowest in the ETO category (5.9) and highest in the N category (6.6 and 6.9). The MC (6.3 and 6.4) and ELO (6.7) categories had middling values.

Soil compaction (psi) was not significantly different between encroachment categories ($c^2(5, N = 36) = 8.75$, $p = 0.120$), where MC had the highest compaction (psi) followed by ETO, MO & ILD, ETD & ETM, ELO and N (Figure 11, Table 13). Dry bulk density was lowest in the

MC category (48 and 257 kg/m^3) and variable in the N category (161 and 835 kg/m^3). Dry bulk density was 466 and 385 kg/m^3 in the ELO and ETO categories, respectively.

Soil water content (m^3/m^3) was significantly different between encroachment categories ($F(5, 63) = 9.54$, $p < 0.001$) (Figure 11, Table 13). The MC category (0.07±0.02 m^3/m^3) had significantly lower water content than N (0.18±0.05 m^3/m^3, $p < 0.001$), ELO (0.18±0.05 m^3/m^3, $p < 0.001$), ETO (0.16±0.06 m^3/m^3, $p = 0.002$), ETD & ETM (0.20±0.07 m^3/m^3, $p < 0.001$) and MO & ILD (0.14±0.05 m^3/m^3, $p = 0.018$).

Soil temperature (° C) was significantly different across encroachment categories ($F(5, 28.7) = 49.0$, $p < 0.001$) (Figure 11, Table 13). The MC category (12±2.0 ° C) was significantly lower than N (21±2.5 ° C, $p < 0.001$), ELO (25±4.2 ° C, $p < 0.001$), ETO (24±2.2 ° C, $p < 0.001$), ETD & ETM (20±3.3 ° C, $p <0.001$), MO & ILD (27±5.6 ° C, $p <0.001$). The ETD & ETM category was significantly lower than ELO ($p = 0.026$), ETO ($p = 0.011$) and MO & ILD ($p = 0.033$).

<u>Soil Electrical Conductivity, Salinity & Nutrients</u>

Electrical conductivity (bulk dS/m) was significantly different in soils across encroachment categories ($F(5, 28.5) = 3.80$, $p < 0.009$) (Figure 11, Table 13). The MC category (0.03±0.029 bulk dS/m) had significantly lower electrical conductivity than ETD & ETM (0.10±0.06 bulk dS/m, $p = 0.026$). Electrical conductivity determined by laboratory testing was lowest in the ETO category (0.19 dS/m) followed by the ELO (0.23 dS/m), N (0.24 dS/m) and MC (0.24 and 0.28 dS/m) categories. Total organic carbon was less than 0.01% in the ETO category. The ELO (2.5%) and N (1.4 and 3.5%) categories had middling values and the MC category (3.5 and 6.7%) had the highest total organic carbon.

Sodium adsorption was less than 0.2 in the N and ELO soil samples. In the MC soil samples sodium adsorption was less than 0.2 and 0.32. In the ETO soil sample sodium adsorption was 0.76. Available nitrate was less than 5 mg/kg in all soil samples. Available phosphorus in the N soil samples was 47 and 61 mg/kg and in the MC category it was 90 and 72 mg/kg. The ELO and ETO categories had 57 and 56 mg/kg available phosphorus, respectively. Available potassium in the N soil samples was 311 and 810 mg/kg and in the MC soil samples, 330 and 468 mg/kg. Available potassium in the ELO category was 787 mg/kg and 554 mg/kg in the ETO category. Available Sulphur for the N soil samples was 4 and 5 mg/kg. In the MC category available sulfur was 8 and 10 mg/kg. In the ELO and ETO categories available sulfur was 6 and 7 mg/kg, respectively. In the

N soil samples there was less than 5 and 7 mg/l chlorine, 8 and 12 mg/l sulfur, 31 and 23 mg /l calcium, 6 and 26 mg/l potassium, 9 mg/l magnesium and 3 mg/l sodium. In the MC soil samples there was less than 5 and 8 mg/l chlorine, 15 and 16 mg/l sulfur, 35 and 39 mg /l calcium, 10 mg/l potassium, 16 and 12 mg/l magnesium and 9 and 5 mg/l sodium. In the ELO soil sample there was less than 5 mg/l chlorine, 9 mg/l sulfur, 23 mg /l calcium, 18 mg/l potassium, 10 mg/l magnesium and 2 mg/l sodium. In the ETO soil sample there was 6 mg/l chlorine, 20 mg/l sulfur, 16 mg /l calcium, 11 mg/l potassium, 6 mg/l magnesium and 14 mg/l sodium.

Soil Characteristics of Old Burn-pile Sites

Soil compaction (psi) in old burn-piles sites was lowest in the ELO category, similar between ETO and ETD & ETM and highest in MO & ILD (Table 14). Soil pH was lower in the ELO and ETD & ETM categories than in ETO and MO & ILD. Soil water content (m^3/m^3) was lowest the MO & ILD category followed by ETO, ELO and ETD & ETM. Soil temperature (° C) was lowest in ETD & ETM and highest in MO & ILD whereas, ELO and ETO had middling values. Electrical conductivity (bulk dS/m) was lowest in ELO, followed by MO & ILD, ETO and ETD & ETM.

Only two old burn-pile site soil samples were sent for laboratory testing and both belonged to the ETO category. Plot 4 had lower pH (7.2) and electrical conductivity (0.37 dS/m) than plot 6 (pH = 7.4, electrical conductivity = 0.43 dS/m). However, dry bulk density (562 kg/m^3) and total organic carbon (3.25%) was greater in plot 4 than in plot 6 (dry bulk density = 482 kg/m^3, total organic carbon = 2.35%). Sodium adsorption was less than 0.2 and available nitrate was less than 5 mg/kg in both burn site soil samples. Available phosphorus was 51 mg/kg in both samples and available potassium was 675 mg/kg in plot 6 and 1110 mg/kg in plot 4. In the plot 6 burn soil sample there was less than 5 mg/l chlorine, 11 mg/l sulfur, 44 mg /l calcium, 37 mg/l potassium, 20 mg/l magnesium and 4 mg/l sodium. In the plot 4 burn soil sample there was less than 5 mg/l chlorine, 7 mg/l sulfur, 39 mg /l calcium, 37 mg/l potassium, 15 mg/l magnesium and 2 mg/l sodium.

3.3 Douglas-fir Seedling Recruitment Assessment

Tree Seedling Recruitment Monitoring (2015, 2018 & 2023)

Prior to slashing, piling, and burning in 2015, no tree seedlings (< 1.3 m height) were found in the mature closed forest (MC) or tall encroachment with open canopy (ETD & ETM) categories (Steen & Young 2019) (Table 15). Only plots with trees were included therefore, the open grassland (N) category did not have a stem density count for 2015. The category with tall trees and open canopy (ETO) had the most seedlings (stems/ha) with the greatest average height (cm). The mature open forest (MO & ILD) category had middling stem density and average seedling height. The ELO category had the lowest mean stem density and average height.

Post-treatment in 2018, no tree seedlings (< 1.3 m height) were found in the MC or ETD & ETM categories, nor was there a stem count for the N category (Steen & Young 2019) (Table 15). The MO & ILD category had the greatest stem density (stems/ha) and average height (cm). Tree seedlings in MO & ILD were only found in one plot and the majority of the seedlings were young aspens (*Populus tremuloides*). The majority of tree seedlings in the ELO and ETO categories were sprouting from the cut stumps of the 2015 slashing, piling, and burning treatment. In November 2018, many of these offshoots were cut throughout the High Lake Benchmark Area.

In 2023, Douglas-fir seedlings were found in 12 of the 18 plots and across five of the six encroachment categories (Table A1). No seedlings were found in any of the plots within the N category (Figure 12, Table 15). There were significant differences in seedling stem density (stems/ha; $F(5, 12) = 5.03$, $p = 0.010$) and average seedling height (cm; $F(5, 12) = 4.72$, $p = 0.013$) between encroachment categories. The MC category ($31 \times 10^3 \pm 15 \times 10^3$ stems/ha) had significantly greater seedling stem density than the N (0.0±0.0 stems/ha, $p = 0.010$), ELO ($11 \times 10^2 \pm 23 \times 10^2$ stems/ha, $p = 0.013$) and ETO ($44 \times 10^2 \pm 77 \times 10^2$ stems/ha, $p = 0.029$) categories. The ETD & ETM and MO & ILD categories had lower stem density than MC; however, the difference was not significant. The N category (0.0±0.00 cm) had significantly shorter mean seedling height than ELO (11±2.6 cm, $p = 0.015$) and MO & ILD (9.4±3.7 cm, $p = 0.039$). The ETO, ETD & ETM and MC categories had greater mean seedling height than N; however, the difference was not significant. Quaking aspen was found in two plots, once in the MO & ILD category at 0.2% cover and once in the MC category at 0.05% cover.

Douglas-fir Seedling Recruitment in Old Burn-pile sites (2023)

In the treated encroachment categories where old burn-pile sites could be found the ELO and ETO categories had no seedlings. The ETD & ETM category had the highest mean stem density (15±22 stems/m^2) and average height (3.1±3.4 cm). The MO & ILD category had a lower mean stem density (1.7±2.9 stems/m^2) and average height (2.0±3.4 cm) than ETD & ETM. In one old burn-pile site in the MO & ILD category a 3% cover of quaking aspen was recorded in a 1 m^2 plot.

4. Discussion

4.1 Vegetation Assessment

Plant Species Composition and Percent Cover

Plant species, diversity, evenness and vegetation percent cover responded differently to the slashing piling and burning treatment over time and between encroachment categories within the High Lake Benchmark Area. Diversity was consistently lower in the untreated open grassland (N) and mature closed forest (MC) categories compared to the four treated encroachment categories (ELO, ETO, ETD & ETM and MO & ILD) as expected (Table 5). The N category decreased in diversity and evenness between 2018 and 2023, as this category was not treated it is necessary to determine what is causing the decline (Table 5, Table 6). Grasslands are dependent on disturbance such as fire and grazing (Ratajczak et al. 2012; Archer et al. 2017). For example, low-intensity fire has the potential to increase soil nutrient availability while promoting new plant growth from surviving roots and rhizomes (Halpern et al. 2016). As fire suppression in the CCPA began in the late 1800s it is possible that open grasslands are in need of a treatment that restores the historical disturbance regime. The reintroduction of fire would not only limit woody encroachment but help to revitalize grassland species recruitment, growth and heterogeneity (Turner 2021; Hoffman et al. 2022). Alternatively, the lower diversity in N and MC in relation to the treated encroachment categories could be due to the inhibition of woodland or grassland species recruitment, respectively. The untreated N and MC categories may be in alternative stable ecosystem states which is defined as the persistence of certain ecosystem parameters (i.e., plant-community assembly) while experiencing low level disturbance (Beisner et al. 2003). Whereas, the treated categories are still experiencing the effects of disturbance such as encroachment and modified conditions due to the slashing, piling, and burning treatment which leads to the recruitment of new species, increasing species richness (Archer et al. 2017).

The higher species diversity within the treated categories are partly the result of encroaching trees and shrubs as well as, associated woodland herbaceous species, increasing species richness (Archer et al. 2017). The treated ELO, ETO and MO & ILD categories (low, tall and mature trees with an open canopy) declined or remained stable in diversity and evenness in 2018 yet all three categories showed increased diversity and evenness in 2023 (Table 5, Table 6). Increased diversity in the treated categories are in part caused by the encroachment of woodland species (Figure 8, Figure 12). Modifications to vegetation structure, soil characteristics and microclimate are facilitating the establishment of other species (Ratajczak et al. 2012; Archer et al. 2017). Over time and without treatment, these modifications in areas of encroachment would shift to a forest plant community.

The significant decline in diversity and evenness in the encroachment category with moderate to dense canopy cover (ETD & ETM) over time follows suit with several other studies that have undergone manual tree removal or fire management. In these studies, the removal of trees initially increases diversity; however, as time since treatment increases, competitive species interactions or resilience of the encroached alternative stable state led to declines in diversity (Halpern et al. 2016; Archer et al. 2017; Barber et al. 2019). The variation in open canopy across the ETD & ETM category plots may be inhibiting shade intolerant species and lowering mean diversity and evenness values (Table 3, Table 5, Table 6). Furthermore, ETD & ETM had significantly lower forb percent cover in 2023, which could explain the lower diversity and evenness indices as forbs were the most species rich vegetation group across the High Lake Benchmark Area (Table 9). Compared to the ETD & ETM category, there was similarly low forb percent cover under the closed canopy of MC, whereas, the ETO category with a completely open canopy was higher in forb species richness and had greater mean total forb percent cover (Figure 5, Table 9).

Species richness changed across encroachment categories within the same season. In June 2023, ELO had the highest species richness but in August it was shared across three encroachment categories. This change was likely due to a changeover of forbs as they were the most species rich and certain species had died off (visually absent) by august while different species appeared. Monitoring of the vegetation community at different times in the season could provide insight into the plant communities ecosystem state and help to determine peak times for woodland species recruitment and predominance.

Graminoids

Graminoid percent cover varied with encroaching tree maturity and density (Figure 3, Table 7). Nearly every encroachment category (excluding MC) showed increases in graminoid percent cover between 2015 and 2018 followed by unexpected declines in 2023. Such a discrepancy between percent cover across years is likely due, in part, to observer differences as sampling was carried out by a different person in 2023 than in the first two sampling years (2015 and 2018). However, some of the decline in percent graminoid cover could be due to lower total annual precipitation from 2021 to 2023, relative to previous years (Table 1). Sampling was carried out twice in 2023 and graminoid percent cover estimates were consistently low for both sampling events (Figure 3).

The six species with greatest mean percent cover varied over time and across encroachment categories (Figure 4, Table 8); where open grassland (N) had the most even distribution and relatively high percent cover in four of the six species over time. In 2023, N was significantly higher in mean graminoid cover than three other categories (ETO, ETD & ETM and MC). The declines in graminoid percent cover the ETO and ETD & ETM categories in 2023, could be due to significant declines in Nelson's needlegrass as well as non-significant declines Kentucky Bluegrass and bluebunch wheatgrass (Figure 4, Table 8). The Nelson's and spreading needlegrasses showed declines in every encroachment category in 2023.

Bluebunch wheat grass appeared stable in percent cover over time in N, ELO and MO & ILD (Figure 4, Table 8). Nearly all plots within those three encroachment categories have 100% open canopies and steeper, well drained slopes which bluebunch is well suited to (Ogle et al. 2010). It is a drought tolerant species with an extensive root system that could indicate its ability to persist in the low annual precipitation values between 2021 and 2023. Declines in percent cover of bluebunch in other encroachment categories could be explained by increased canopy cover (ETD & ETM and MC) and acidic conditions (ETO).

Pinegrass is a shade tolerant species with a low resistance to drought, which could explain its relatively high percent cover in the mature treed categories where percent open canopy was low (USDA Natural Resources Conservation Service n.d.; Klinkenberg 2020). Slope grade was lower in these two categories with two plots having toe and depression mesoslope positions (Table 3). Reduced water runoff and the added potential of hydraulic lift from the mature trees could supply the wetter and cooler conditions that support the establishment of both pinegrass as well as Douglas-fir forests in the IDFxm zone (BC Parks 2000; Evans et al. 2017).

Kentucky bluegrass, a sod forming species, was most consistent in percent cover over time in the N category (Figure 4, Table 8) (Wennerberg 2004). The species prefers cool and humid sites with well drained soils. A burst of increased percent cover was observed in the ETO, ETD & ETM and MO & ILD categories in 2018 which is correlated to the higher annual precipitation for 2018 and previous years (Table 1). Kentucky bluegrass is a drought intolerant species which could explain its decline in percent cover across all categories in 2023 as total annual precipitation has been lower since 2021. Kentucky bluegrass has never been observed in any of the ELO category plots. As this species is a good competitor that can quickly dominate disturbed areas, determining what inhibits its growth in the ELO category could be valuable to future restoration efforts.

Junegrass, a small bunchgrass, prefers to establish in rocky, open areas. Otherwise it is cold, heat and drought resistant, which was consistent with its distribution in low percent cover across all encroachment categories over time, excluding the ETD & ETM category (Ogle et al. 2006). The ETD & ETM category covers a range of elevations, slopes and percent open canopy therefore, it is unclear what could be preventing junegrass establishment. The three plots of the ETD & ETM category are confined to one section of the Benchmark Area perhaps, a local site condition could be limiting junegrass recruitment (Figure 1).

The decline in average percent cover of Nelson's needlegrass over time in the N and MO & ILD categories seem to be driven by one plot thus, statistical analyses did not result in significant differences for either category. Such large variation across the three replicates within each category may be misrepresenting the changes in diversity and percent cover. Additional sampling plots could help to reduce this variation or provide insight on how different site characteristics (e.g., slope, aspect, elevation, etc…) drive plant-community composition.

<u>Forbs</u>

The forb group was the most species rich; however, due to the typically small size of an individual forb, mean percent cover was generally lower than other vegetation groups (Table 7, Table 9). In less common forbs, trends were apparent where a species was observed in encroachment categories sharing similar tree densities and maturity (Table 9). As with the graminoid group, mean percent cover in forbs was lower in 2023 than 2015 and 2018, which could be due to observer differences, or environmental factors such as reduced total annual precipitation (Figure 6, Table 1).

There were four forb species having the greatest relative percent cover across encroachment categories and over time (Figure 6, Table 9). The common yarrow, *Antennaria* spp. and the common dandelion can be found across a range of ecosystem types from grasslands to forests (Klinkenberg 2020). The dandelion has a particular affinity for disturbed sites and the woolly cinquefoil is typically found in grasslands or sagebrush scrublands. Forb percent cover of these species was generally greater and more evenly distributed across the four treated categories (ELO, ETO, ETD & ETM and MO & ILD) than in the open grassland (N) and mature closed forest

(MC) untreated categories. As forbs were the most species rich group across the study area the declining cover and lower diversity of forbs in the N and MC categories over time may be contributing to lower overall diversity and equitability values. In areas where disturbance has occurred forbs are able to establish more readily due to the decreased grass biomass creating available open space, which could be limiting forbs in the N category where graminoid percent cover was greater than all other categories across all sampling years (Evans et al. 2017). Increased precipitation could have contributed to greater forb percent cover in the treated ETO (tall treed and open canopy) and ETD & ETM (tall treed and moderate to dense canopy) categories in 2018 and subsequent decline in all categories following years of drought in 2023 (Figure 6, Table 1, Table 9). The increase in forb percent cover in 2018 for ETO and ETD & ETM could also be the initial release from competition with woody species followed by a loss of inferior competitors as time since restoration increases (Halpern et al. 2016; Barber et al. 2019). For the remaining four encroachment categories that saw a continual decline in species diversity over time (N; ELO, low treed, open canopy; MO & ILD, mature open forest; and MC) plots may have been lacking sufficient propagule sources or the environmental conditions required for herbaceous seedling dispersal and development (James et al. 2011; Martin & Wilsey 2012; Halpern et al. 2016). Species richness and forb percent cover within the mature closed forest (MC) was lower yet more stable over time compared to the treated encroachment categories (Table 9), which was consistent with diversity and evenness trends for MC.

The common yarrow was significantly greater in the ELO category than all other categories in 2015, which could be due to the consistently higher percent slopes across all plots increasing water drainage, a preferred trait of yarrow (Hurteau 2003). The ELO category was also predominated by bluebunch wheatgrass interspersed with patches of bare mineral soil which provides ideal unoccupied spaces for yarrow to establish. Yarrow had generally greater percent cover across three of the four treated categories (ELO, ETO and ETD & ETM) in 2018, which could be the result of increased precipitation. Its persistence in drier years may be attributed to its drought tolerance. Yarrow is often found in grasslands and open forests with poorly developed soils. As such, the greater canopy cover and visible organic layer within the MO & ILD and MC categories could explain its minimal cover and absence, respectively.

Shrubs

Shrub percent cover was generally greater in encroachment categories with taller or mature trees of mixed density across all sampling events (Figure 7, Table 10). However, shrub percent cover was not consistent within the individual plots of each encroachment category. The decline in shrub percent cover in the open grassland (N) in 2023, was due to the poor health of a pre-existing common rabbitbrush within one plot (Figure 8, Table 10). The common rabbitbrush, was not particularly high in percent cover in 2015 yet has declined over time across the open grassland and all treated categories. It is an early to mid-seral species that can quickly occupy space after a disturbance that is known to disappear as a site ages (Scheinost et al. 2010). It was not found in the MC category which was expected due to its affinity for dry, open areas.

The species responsible for increased shrub percent cover within all categories other than N and ELO appears to be the Rocky Mountain juniper (*Juniperus scopulorum*) (Figure 7, Figure 8, Table 10). Plots that had junipers in previous years showed an increase in percent cover over time and could be supporting the growth and recruitment of more junipers by modifying soil conditions, vegetation structure and microclimate (Halpern et al. 2016; Archer et al. 2017). A lack of disturbance such as fire may not be the only mechanism allowing for woody encroachment, light availability, which can be altered by the invading shrubs may be a contributing factor. The resulting encroachment patches can create nesting and protective vegetation structure for birds and juniper berries consumption by birds and other wildlife can result in further seed dispersal across the landscape (Stevens 2003).

In the ELO category Rocky Mountain juniper percent cover has slightly declined over time. The plots of the ELO category had a dominant percent cover of the desirable grassland species, bluebunch wheatgrass and to date, zero percent cover of the highly competitive and non-native Kentucky bluegrass. Further investigation is required to determine what has supported grassland recovery in this category and inform restoration strategies in other categories where shrub percent cover is still increasing.

Although shrub percent cover showed no significant changes over time it should be noted variance within categories increased. In other words, individual plots that had shrub cover in 2015 and 2018 have increased shrub cover in 2023. Whereas plots with minimal shrub cover in previous years remain low. The monitoring and management of shrub percent cover is recommended particularly in areas with Rocky Mountain juniper. Shrub percent cover showed less of a difference between years than graminoids and forbs (Figure 2, Figure 6, Figure 8). Perhaps shrubs, with a larger coverage within a 100 m^2 area, are less susceptible to observer differences.

Mosses & Lichens

Mosses and lichens were present in nearly all encroachment categories over time. Declines in moss and lichen percent cover were observed in the N, ETO and MO & ILD categories in 2018. However, interpretation of moss and lichen percent cover can be challenging, due to visual estimation being limited by increased graminoid or litter cover within a plot, perhaps accounting for large changes in percent cover between years. Trampling, by humans (i.e., restoration activities and monitoring) or wildlife, could have also contributed to these declines in percent cover (Steen & Young 2019). In 2023, mosses and lichen species generally increased in percent cover with the exception of the ELO and MC categories, and the *Cladonia* spp. group (Figure 9, Table 10) Historically, the mosses and lichens that form a biocrust network with fungi, cyanobacteria and algae were thought to have very slow recovery rates, particularly in arid environments (Belnap & Lange 2003; Kidron 2020). However, a recent literature review indicated that sites experiencing low-intensity disturbance recovered relatively quickly when there were surviving moss and lichen propagules (Kidron 2020). Additionally, the decreased graminoid and forb percent cover in these categories in 2023 could have increased exposure of the lichens and mosses resulting in greater visual percent cover estimates (Figure 2, Figure 5). Lichens and mosses had lower diversity and percent cover in 2018 and 2023 in the N category (Figure 9, Table 10). However, these lower values could be the result of increased graminoid and litter percent covers reducing visual estimations, as two of the plots within the N category had litter cover greater than 85% across all three sampling years (Figure 3, Figure 10). Biocrusts, a network of lichens, mosses, algae, fungi and cyanobacteria on the soil surface, support grassland vegetation and soils through nitrogen fixation, aggregate development, reduced erosion and water storage (Evans et al. 2017). More detailed sampling of biocrust species, percent cover and biomass could provide insight on biocrust response to disturbance across the High Lake Benchmark Area and the greater CCPA, particularly in the face of the climate crisis and increased wildfire risk (ClimateReadyBC n.d; Kidron 2020).

Bare Mineral Soil, Burned Area & Litter

Mean percent cover of bare mineral soil was higher in encroachment categories with fewer trees (N, ELO and ETO) and lower in categories with higher tree density or maturity (ETD & ETM, MO & ILD and MC) (Figure 10, Table 10). As grasslands are made up of a mixture of bare ground and vegetation cover these results were expected. However, in congruence with the declines in graminoid and forb percent cover in 2023, potentially caused by drought, these bare patches were larger (Figure 2, Figure 5). With greater areas of bare soil competition for water and soil resources is reduced and has the potential to promote woody species recruitment (Archer et al. 2017). Contrary to Archer et al. (2017), Douglas-fir stem density was lower in the three categories with the greatest bare soil cover; however, the seedlings that have established in the bare soil, particularly in the ELO category, are taller (Figure 12, Table 15). Indicating that seedling recruitment may not be facilitated with increased bare soil in the High Lake Benchmark area yet, seedling growth is. The minimal bare ground cover in the mature open and closed forest categories (MO & ILD and MC) could be the result of increased litter cover inputs from the higher percent cover of shrubs and density of Douglas-fir trees (Figure 7, Figure 12). The greater shrub cover (MO & ILD), less open canopy (MC) and interception of precipitation by trees in the mature forest categories could be reducing the litter layers' temperature and moisture levels (Figure 7, Figure 11, Table 3). Lower temperatures and available moisture slows the decay of Douglas-fir litter which could explain its greater visual percent cover in these categories (Moore 1986). As with vegetation percent cover there was a large amount of variation regarding bare soil and litter percent cover between plots. Additional monitoring plots could further delineate how different site characteristics impact both living and non-living structure and composition.

Burned area within a plot either remained constant or decreased between 2018 and 2023, which could be due to observer differences or vegetation colonization over time (Table 10). Burned areas may be slow to revegetate due to high-intensity fires that caused nutrient volatilization and hydrophobicity within the soils (Agbeshie et al. 2022). Additionally, a lack of propagule sources or means of dispersal could have limited plant recolonization (Martin & Wilsey 2012; Halpern et al. 2016).

Plant Species Composition and Percent Cover of Old Burn-pile Sites

The old burn-pile sites sampled as a pilot study during the 2023 monitoring season appeared to have experienced high-intensity fire for the most part. Within the burn-piles larger pieces of burnt wood remained and only a couple of the plots had patches with barked, unburnt wood pieces. Vegetation percent cover varied across the burn-piles; however, graminoid, forb and shrub cover remained low. As expected mosses, an early successional group, were predominating burn-pile site percent cover. These results are similar to a study conducted in the Oregon Cascades, in the United states of America (Halpern et al. 2014). Where richness and vegetation cover remained low, particularly towards the centre of the burn-pile site, yet was showing signs of grassland species recruitment seven years after treatment.

4.2 Soil Characteristics

The mature open and closed forest (MO & ILD and MC) categories had more neutral pH values than other categories (Figure 11, Table 13). Neutrality in the mature forest categories was unexpected as coniferous trees produce organic acids and release hydrogen ions into the soil during cation exchange, effectively reducing soil pH (Alfredsson et al. 1998). Soil pH field testing provided different results than laboratory testing, where the laboratory tests for MC were more acidic, as predicted. The Kelway Soil Tester used for the field sampling of pH may not have been ideal. Rocky soils made probe entry challenging and soils had to be loosened via an auger prior to sampling likely reducing soil contact with the probe. However, pH was sampled six times per encroachment category in the field, yet only one to two samples per encroachment category were sent to the lab. Therefore, certainty regarding laboratory sampling results was limited due to lack of replication.

Douglas-fir trees develop long taproots in deep soils and plate-like roots in shallow soils (Hermann & Lavender 1990). The consistently high compaction in the MC category could be due to an abundance of rope-like lateral roots near the soil surface (Figure 11, Table 13). Furthermore, the compaction measurements in both the MC and MO & ILD categories had a maximum depth of 9.8 cm before hitting a solid barrier. As with the field measurements for pH, the rocky soils across sampling sites in the High Lake Benchmark Area made it difficult to conclude whether soil was compact or rocks were impeding the probe's passage.

In field sampling, soil water content (m^3/m^3) and electrical conductivity were positively correlated across all encroachment categories (Figure 11, Table 13). However, deionized water was necessary for the majority of electrical conductivity sampling in the field due to very dry soils. The significantly lower soil water content in the MC category was unexpected, as hydraulic lift from woody plants and the presence of a dark organic soil layer was predicted to increase water availability and capacity in the upper portion of the soil horizon (USDA 2011, Evans et al. 2017). However, the lower water content could be due to increased interception of water from the low percent open canopy (Table 3), transpiration through vegetation or consecutive years of low total annual precipitation (Table 1) (Ratajczak et al. 2012; Archer et al. 2017). The lower temperatures

(° C) in the MC and ETD & ETM categories were expected as canopy cover is generally greater across these categories, reducing solar radiation on soil surfaces.

Laboratory testing showed variation in total organic carbon in the MC category; however, both samples were in the upper range compared to other categories. Increased total organic carbon (%) and lower dry bulk density (kg/m^3) in MC were expected as organic matter increases soil porosity (USDA 2011). Electrical conductivity measured in the laboratory was consistently greater than field results. Laboratory results for electrical conductivity (dS/m) were relatively higher in the MC category than other categories, which is congruent with greater soil porosity allowing for increased water holding capacity. Nutrient concentrations within the MC category were high-middling compared to the other categories; therefore, the higher concentrations of ions in the sample could have also contributed to increased electrical conductivity (Evans et al. 2017).

Laboratory results for the single ETO plot had the lowest total organic carbon, electrical conductivity and pH. There was uncertainty if the differences in ETO sampling compared to other categories was a plot or category specific trait as sampling was limited. Since 2023 was the first year of monitoring for soil parameters it remains unclear how soils have changed over time in response to woody encroachment and at what tree density or maturity this transition takes effect thus, requiring further study. It is also important to note that individual plots within each encroachment category were quite variable, covering a range of elevations, slopes and mesoslope positions. Additional soil monitoring plots and laboratory analyses could provide greater insight on how these soil characteristics are shaping the plant community.

Soil Characteristics of Old Burn-pile Sites

Field sampling of soils within the old burn-pile sites were generally less compact, yet similar in all other parameters compared to the larger vegetation plots. In laboratory testing both burn-pile sites had higher pH than their non-burned areas as was expected in plots with higher fire severity (Diehl et al. 2010). Total organic carbon, soluble ions and available nutrients were similar or higher in the burn-piles then the associated vegetation plot which is contrary to the expected loss of carbon and other nutrients through volatilization (Evans et al. 2017; Agbeshie et al. 2022).

4.3 Tree Seedling Recruitment

Tree seedling recruitment varied in species, density and height across encroachment categories and over time. In 2015, tree seedlings were generally taller and less dense than 2023, and were most prominent in the tall and open encroachment category (ETO) (Steen & Young

2019). In 2018, the most tree stems (mostly quaking aspens) were observed in one plot within the mature open forest (MO & ILD). Aspens that were recorded in 2018 may have been the result of a release from competition with Douglas-fir saplings; however, very little percent cover of aspens remained in 2023. The decline in aspen percent cover in 2023 could have been the result of grazing by wildlife, shade intolerance to encroachment categories with higher tree canopy cover or drought conditions from 2021 to 2023 (Table 1) (Nesom 2003). Douglas-fir seedlings observed in other plots in 2018, were the result of stump regeneration after the slashing, piling, and burning treatment in 2015. Many of the 2018 regenerative seedlings were removed that same year. In 2023, Douglas-fir seedlings were denser, yet shorter than 2015 and 2018 and were found in all four treated encroachment categories.

Stem density of Douglas-fir seedlings may have been greatest in the mature closed forest (MC) in 2023; however, average seedling height was greater in the more open canopied categories with low (ELO) or mature trees (MO & ILD) (Figure 12, Table 15). Indicating competition for nutrients and water may inhibit growth in a denser forest whereas, the open canopy (i.e., fewer competitors) might facilitate seedling development into viable trees. As woody-plant cover increases, resources and space become limited and a new seedling would be less likely to survive to maturity in spaces already occupied by other trees or seedling growth will not contribute to increased percent cover of woody species (Archer et al. 2017). Stem density and average height was not recorded in the untreated MC category in 2018 therefore, it remains uncertain whether the seedlings in 2023 are newly emerged or stunted by competition.

In 2023, the tall yet open canopied (ETO) encroachment category seedlings were on average shorter than the ELO and MO & ILD categories. One plot in the ETO category had soil with low total organic carbon (%). As Douglas-fir trees prefer high organic matter and total nitrogen, the soil conditions of this plot may be preventing seedling development (Hermann & Lavender 1990). Graminoid and forb percent cover was also lower in the ETO category than the ELO and MO & ILD categories (Figure 4, Figure 6). As first year Douglas-fir seedlings prefer light shading, the lack of vegetation cover could also be inhibiting seedling development.

Seedlings recorded in 2023 were on average shorter than 2015 and 2018. The small size of the seedlings measured in 2023 makes them vulnerable to stressors as they have not accumulated sufficient biomass and carbohydrate stores. If restoration efforts can be implemented quickly low-intensity burns or grazing has the potential to effectively reduce seedling densities. If these seedlings are not managed promptly or if site conditions change (i.e., greater annual precipitation or increased carbon dioxide levels due to the climate crisis), seedling growth rate could increase making management difficult (Ratajczak et al. 2012; Archer et al. 2017).

Douglas-fir Seedling Recruitment of Old Burn-pile Sites

On average fewer stems at shorter heights were found in the old burn-pile sites. The lack of vegetation in the old burn-pile sites could have exposed seedlings to full sun which can increase seedling mortality (Hermann & Lavender 1990). Grassland vegetation can also create resource islands by increasing water retention and nutrient levels in the upper portion of soils, if vegetation is lacking soil conditions may not be suitable for Douglas-fir recruitment (Archer et al. 2017). The lack of a propagule source in the ELO and ETO old burn-piles sites could be limiting Douglas-fir seedling recruitment (Figure 1) (Hermann & Lavender 1990). As tree encroachment continues to be managed in the High Lake Benchmark Area further study of tree recruitment in disturbed sites and their proximity to a viable tree may help identify more vulnerable sites.

4.4 Management Recommendations

The combined efforts of the Friends of Churn Creek Protected Area Society (FCCPAS) and the Stswecem'c Xget'tem First Nations (SXFN) has resulted in the removal of thousands of Douglas-fir stems across the High Lake Benchmark Area. Eight years after treatment grassland-community reassembly, soil characteristics and Douglas-fir seedling recruitment have responded differently across encroachment categories and over time. Therefore, support for the continued monitoring and management by FCCPAS and SXFN of the grasslands in the High Lake Benchmark Area within the Churn Creek Protected Area is recommended. Management strategies could benefit from diversification with a focus on:

Native grassland species such as, bluebunch wheatgrass and junegrass are viable options for reseeding grassland areas in need of revitalization. Both species are drought tolerant and very fire resistant (Ogle et al. 2006; Ogle et al. 2010) Ideally, local seeds would be collected across the High Lake Benchmark Area and spread in early spring or late fall. Bluebunch wheatgrass is not a strong competitor thus, management of other more competitive species like Kentucky bluegrass is recommended (Ogle et al. 2010). The common rabbitbrush can also be transplanted or seeded with the grasses as a means of erosion control after a disturbance or to fill empty spaces that may readily be occupied by other competitive species (Scheinost et al. 2010).

As 2023 was the first year for soil sampling it remains unclear how soils responded to woody encroachment and to slashing, piling, and burning over time across the High Lake Benchmark Area. Several of the measuring techniques conducted in the field proved challenging and reduced certainty in the resulting soil characteristics, alternative methods may be more effective.

The continued monitoring of Douglas-fir seedling recruitment and the creation of a management strategy for the Rocky Mountain juniper is recommended. The expansion of the Rocky Mountain Juniper and Douglas-fir seedlings across the High Lake Benchmark area was not consistent within and across encroachment categories. Greater focus could be placed on the open canopied categories in proximity to mature trees (i.e., propagule sources) to inhibit the development of viable trees. Reducing shrub cover could prevent the creation of suitable conditions for Douglas-fir seedling recruitment while creating open spaces for desirable grassland species (Halpern et al. 2016; Archer et al. 2017). However, the methods used to manage shrub cover could have detrimental effects such as increased bare soil and reduced leaf litter resulting in greater erosion after removal of shrubs through fire (Daryanto et al. 2019). Therefore, an adaptive management approach testing various removal techniques (e.g., burning, mechanical removal or chemical elimination) over small areas could guide the best cost-benefit tradeoff for the High Lake Benchmark Area. Such efforts could more effectively support grassland recovery while reducing fuel loads thus, the potential for high-intensity wildfire.

Perhaps further revitalization of both the treated and untreated grasslands could be achieved through a controlled burn, cultural or prescribed, using an adaptive management approach focusing on smaller, experimental areas. Historically, many First Nations communities would use cultural burning to create habitat heterogeneity and to recover native biodiversity and ecosystem function. A controlled burn would help to inhibit Douglas-fir seedlings in their currently small and vulnerable state (Ratajczak et al. 2012; Archer et al. 2017). Controlled burns would have to be repeated to effectively restore the natural disturbance regime that has maintained these grasslands for centuries (SXFN n.d.; Harvey et al. 2017; Steen & Young 2019). However, Douglas-fir seedlings can quickly occupy empty space after a disturbance once other competitors have been eliminated, particularly when mature trees are nearby providing a viable seed supply (Hermann & Lavender 1990). Therefore, a controlled burn should be followed by the prompt reestablishment of bare areas with desirable grassland and biocrust species through reseeding

(Martin & Wilsey 2012; Halpern et al. 2016; Kidron 2020). There is potential to share the task of a controlled fire between BC Parks, FCCPAS and SXFN, whose Guardian Program works extensively to protect the natural resources across the region. A combined effort would result in shared experience, blending both fire science and Traditional Ecological Knowledge.

Vegetation composition and structure, soil characteristics and Douglas-fir seedling stem density and height varied not only between but within encroachment categories. Such variation is in part, reducing certainty regarding the differences over time and between encroachment categories. Furthermore, individual plots within the majority of the encroachment categories span a variety of elevations, mesoslope positons, and slope aspects and grades. Additional monitoring plots could reduce uncertainty, providing greater insight on how site characteristics shape the grassland community. With a more diversified management strategy additional plots can be used to determine the success of any new treatments experimentally (i.e., adaptive management). For example, shrub responses to mechanical or fire based removal varies between techniques and location of the restoration project (Daryanto et al. 2019). Treatment by fire can be detrimental to litter and biocrust layers and has slower shrub regeneration than after mechanical removal. Through a smaller scale and adaptive management approach in the High Lake Benchmark Area, restoration efforts can support habitat heterogeneity and biodiversity of desirable grassland species on both the local and landscape scales (Archer et al. 2017).

5. Conclusion

Plant species, diversity, evenness and vegetation percent cover responded differently to the slashing piling and burning treatment over time across encroachment categories within the High Lake Benchmark Area at the CCPA. The species composition and diversity indices of the untreated open, open grassland (N) and mature closed forest (MC) indicates these categories have reached stable ecosystem states. However, recent declines in diversity and evenness (2023) in the N category highlights the need for reestablishment of low-intensity disturbance or reseeding to support biodiversity and plant growth within the existing grasslands. The grassland community in the four treated categories continues to be impacted by woody encroachment with exponentially higher stem densities of Douglas-fir seedlings in 2023 compared to 2015 and 2018. Variation in seedling height, and vegetation composition and structure between the treated encroachment categories is most correlated with percent open canopy. In particular, the ETD & ETM category with moderate to dense canopy cover has responded differently to manual tree removal over time compared to the three other open canopied, treated categories. Recent declines in diversity and evenness in the ETD & ETM category are similar to other studies where time since restoration increases competitive interactions between species, eliminating inferior competitors. Increased diversity in the ELO, ETO and MO &ILD categories could be the result of continued woody encroachment facilitating the establishment of forest associated species. Variation between sampling years was in part due to observer differences; however, declines in graminoid and forb percent cover in 2023 can also be attributed to prolonged low annual precipitation. Based on the data acquired in 2023 soil conditions do not seem to be facilitating or inhibiting woody encroachment. Quaking aspens showed an initial release from competitive interactions after manual tree removal; however, their presence was short-lived. Douglas-fir seedling recruitment is most likely facilitated by proximity to mature trees providing propagule sources, and seedling growth supported by reduced competition in the less densely treed encroachment categories. Prompt treatment of these young and vulnerable seedlings would be most effective. The large variation across sampling plots within each category may be misrepresenting vegetation community composition and structure, soil characteristics and Douglas-fir seedling recruitment. Additional sampling plots would reduce uncertainty and provide insight on how the different site characteristics (e.g., slope, aspect, elevation, etc…) drive grassland recovery.

The slashing, piling, and burning of thousands of stems across the High Lake Benchmark Area acted as an initial and significant step in the removal of encroaching trees. Now, eight years after treatment began, the data indicates there is a need for diversified monitoring and management. The knowledge acquired throughout this restoration project has the potential to inform more specific and effective treatment across different levels of encroachment while facilitating renewal of the existing grasslands.

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Tables

Table 1. Mean and standard deviation of minimum, maximum and average daily temperature (° C) as well as, average daily and total precipitation (mm). Data was recorded hourly, 365 days a year in the Churn Creek Protected Area, British Columbia at 1100 m.a.s.l (Pacific Climate Impacts Consortium 2024).

	Temperature (° C)						Precipitation (mm)		
	Minimum		**Maximum**		**Average**		**Monthly**		**Total**
Year	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	
2015	2.5	7.8	12	10	6.9	8.9	23	13	276
2016	2.0	8.0	11	9.9	6.1	8.9	31	33	376
2017	1.0	9.6	10	12	5.3	11	16	10	188
2018*	4.3	7.5	14	10	8.7	8.8	28	25	250
2019	0.97	9.1	9.9	11	5.1	9.7	25	24	298
2020**	3.2	7.0	12	9.5	7.3	8.2	27	23	319
2021	1.2	10	11	12	5.7	11	13	9.0	152
2022‡	0.01	11	9.5	13	4.5	12	11	7.9	129
2023	2.6	8.2	11	11	6.8	9.4	17	15	199

Note. Climate data was contributed by a weather station managed by the Wildfire Management Branch of the British Columbia Ministry of Forests, Lands, Natural Resource Operations and Rural Development.
* Consecutive hourly recordings of temperature and precipitation are missing from January to May.
** Consecutive hourly recordings of temperature and precipitation are missing for some periods in February, March and April.
‡ Consecutive hourly recordings of temperature and precipitation are missing for some periods from May through June

Table 2. Description of the six encroachment categories in the High Lake Benchmark Area, Churn Creek Benchmark Area, British Columbia. Descriptions are before (2015) and after (2016) the slashing piling and burning treatment, determined by tree stem density, height and maturity.

Code	Encroachment Category	Pre-treatment	Post-treatment
N	No encroachment stems	No tree species of any size within open grassland	-
ELO	Encroachment, low, open	Low stem density of short (2-4 m) trees with a grassland understory	A few small stems (< 1.3 m) remaining
ETO	Encroachment, tall, open	Low stem density of taller trees (> 4 m) with a grassland understory	Low stem density of taller trees (≥ 17.5 cm dbh) & few small stems (< 1.3 m)
ETD & ETM	Encroachment, tall, dense & encroachment, tall, mature	Moderate to high stem density of tall & mature trees (> 4 m)	Lower stem density of tall & mature trees (≥ 17.5 cm dbh) and significant removal of smaller trees
MO & ILD	Mature, open & ingress, low, dense	Low stem density of mature trees & patches of short, yet numerous trees (< 4 m)	Low stem density of mature trees (≥ 17.5 cm dbh) & few regenerating aspens (< 1.3 m)
MC	Mature, closed	High stem density of tall (> 6 m), mature trees	-

Note. Adapted from Steen & Young (2019).

Table 3. Site elevation (m), slope aspect (°), slope grade (%), slope position, and open canopy (%) for each plot across six encroachment categories in 2023, across the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

Plot	Encroachment Category*	Elevation	Slope Aspect	Slope Grade	Slope Position	Open Canopy**
1	ETD & ETM	1223	192(SSW)	12	mid	18
2	ETD & ETM	1246	182(S)	19	upper-mid	78
3	ETD & ETM	1201	197(SSW)	15	lower-mid	100
4	ETO	1228	207(SSW)	12	upper-mid	100
5	ETO	1210	178(S)	9	mid	100
6	ETO	1204	173(SSE)	12	lower-mid	100
7	MC	1242	129(SE)	11	mid	36
8	MC	1227	90(E)	1	toe	38
9	MC	1232	175(S)	8	mid	39
10	MO & ILD	1220	130(SE)	10	upper	79
11	MO & ILD	1216	176(SSE)	0	depression	100
12	MO & ILD	1265	167(SSE)	11	upper-mid	100
13	ELO	1270	183(S)	15	upper-mid	100
14	ELO	1242	157(SSE)	18	lower-mid	100
15	ELO	1214	195(SSW)	18	upper-mid	100
16	N	1250	120(ESE)	16	mid	100
17	N	1211	219(SW)	1	depression	100
18	N	1225	152(SE)	21	mid	100

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).
**Canopy cover was recorded in June, 2023.

Table 4. Mean species richness and standard deviation per sampling event across six tree encroachment categories (n=3). A sampling event is distinguished by year or month and consists of the completed recording of vegetation percent cover within all 18 plots in 2015, 2018 and 2023 in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	2015		**2018**		**2023 (June)**		**2023 (August)**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
N	17	2.1	19	4.6	17	6.2	14	5. 3
ELO	25	4.5	25	1.5	25	1.5	28	2.1
ETO	25	2.1	28	0.58	24	1.5	28	2.0
ETD & ETM	25	2.6	24	2.9	21	1.5	21	5.5
MO & ILD	27	5.0	26	1.5	21	2.9	26	3.6
MC	20	3.1	19	2.1	15	3.5	18	1.0

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Table 5. Mean and standard deviation of the Shannon-Wiener Diversity Index (H) for 2015, 2018 and August 2023, across six tree encroachment categories (n=3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	2015		**2018**		**2023**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
N	1.1	0.25	1.2	0.09	0.86	0.50
ELO	1.8	0.28	1.3	0.37	2.3	0.62
ETO	2.1	0.21	1.7	0.23	2.0	0.51
ETD & ETM	2.0	0.34	1.2	0.33	1. 1	0.40
MO & ILD	1.4	0.57	1.4	0.40	1.9	0.46
MC	1.2	0.51	1.1	0.24	1.0	0.33

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Table 6. Mean and standard deviation of the Shannon's Equitability index (E_H) in 2015, 2018 and August 2023, across six tree encroachment categories (n = 3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	2015		2018		2023	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
N	0.39	0.08	0.40	0.03	0.32	0.14
ELO	0.56	0.07	0.41	0.11	0.68	0.18
ETO	0.64	0.05	0.50	0.07	0.60	0.15
ETD & ETM	0.63	0.09	0.39	0.11	0.36	0.10
MO & ILD	0.43	0.15	0.43	0.13	0.59	0.16
MC	0.39	0.18	0.37	0.07	0.36	0.12

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Table 7. Mean and standard deviation of graminoid, forb and shrub total percent cover per sampling event across six encroachment categories. Mean and standard deviation were derived from the summed percent cover of grouped species within a plot then averaged by encroachment category (n=3). A sampling event is distinguished by year or month and consists of the completed recording of percent cover within all 18 plots in 2015, 2018 and 2023 in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	2015		**2018**		**2023 (June)**		**2023 (August)**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
Graminoids								
N	61	25	71	32	27	7.5	37	4.6
ELO	34	5.5	43	5.1	30	13	27	11
ETO	29	22	46	35	4.0	2.3	4.5	1.8
ETD & ETM	42	17	61	9.9	6.3	2.6	8.8	10
MO & ILD	44	25	47	21	26	15	20	13
MC	29	18	26	20	11	16	5.8	8.0
Forbs								
N	12	13	7.9	5.4	2.3	0.46	1.9	0.83
ELO	16	6.8	13	1.9	5.1	1.3	5.0	1.0
ETO	29	20	36	23	12	9.6	12	9.6
ETD & ETM	7.9	7.2	14	12	4.1	2.9	2.5	1.9
MO & ILD	10	2.5	9.9	4.7	2.8	0.67	3.3	0.96
MC	2.6	3.1	2.6	2.8	1.1	0.56	0.40	0.46
Shrubs								
N	0.50	0.87	0. 67	1.2	0.33	0.58	0.17	0.29
ELO	3.7	3.8	3.1	3.9	3.9	5.6	2. 3	2.6
ETO	5.6	4.6	6.7	3.5	8.7	5.7	7.6	7.0
ETD & ETM	14	14	12	11	14	12	17	14
MO & ILD	11	9.8	14	11	14	12	15	12
MC	8.1	5.1	8.1	5.1	8.0	7.3	8.2	11

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Table 8. Mean percent cover (%) of graminoid species in 2015, 2018 and August, 2023, across six tree encroachment categories (n=3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	N*			**ELO**			**ETO**			**ETD & ETM**			**MO & ILD**			**MC**		
Graminoid Species	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023
Achnatherum nelsonii	15	9.0	2.9	0.33	1.1	ǂ	7.0	8.0	ǂ	16	23	0.20	10	7.1	ǂ	1.1	0.80	ǂ
Acnatherum richardsonii	17	23	10	ǂ		ǂ	3.2	8.4	1.6	0.67		ǂ	12	12	1.4	0.35		
Bromus anomalus										0.73	0.33							
Calamagrostis rubescens										1.5	0.67	ǂ	8.7	13	10	17	17	15
Carex petasata		ǂ	ǂ		ǂ		0.23	ǂ	ǂ	ǂ				ǂ				
Carex spp.									ǂ				ǂ		ǂ	0.43	0.43	ǂ
Danthonia intermedia													0.20	0.33				
Elymus trachycaulus			ǂ							0.67	1.7	ǂ	ǂ			ǂ	ǂ	
Festuca saximontana	ǂ	ǂ		ǂ			ǂ			ǂ			ǂ		ǂ	ǂ	ǂ	ǂ
Koeleria macrantha	ǂ	0.70	0.33	0.23	0.60	0.38	0.20	ǂ	ǂ	0.10	0.13	ǂ	0.53	0.37	ǂ	0.25	0.25	
Poa spp.				ǂ			0.33	0.13		0.10	1.1		ǂ			0.17	1.5	
Poa pratensis	15	25	10		0.10		1.7	25	1.7	1.0	17	0.67	0.73	2.7	ǂ	8.0	ǂ	ǂ
Poa Secunda	ǂ	ǂ		0.33	1.8			ǂ				ǂ		1.1	ǂ			
Poaceae spp.									ǂ			ǂ			ǂ		3.3	ǂ
Pseudoroegneria spicata	14	13	14	33	39	27	17	5.0	1.1	21	17	7.8	12	11	8.0	2.0	2.0	ǂ
Total Percent Cover	**61**	**71**	**37**	**34**	**43**	**27**	**29**	**46**	**4.5**	**42**	**61**	**8.8**	**44**	**47**	**20**	**29**	**26**	**16**
Total Species Richness	**7**	**8**	**7**	**7**	**6**	**4**	**8**	**8**	**8**	**11**	**8**	**9**	**11**	**9**	**10**	**10**	**10**	**7**

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).
Note. Species and total mean values have been rounded to two significant figures, cells containing ǂ indicate a species presence; however, in trace amounts (<0.10%). The total values include these trace amounts.

Table 9. Mean percent cover (%) of forb species in 2015, 2018 and 2023, across six tree encroachment categories (n=3) High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	N*			**ELO**			**ETO**			**ETD & ETM**			**MO & ILD**			**MC**		
Forb Species	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023
Achillea millefolium	0.27	0.56	0.60	1.7	2.2	0.83	0.30	1.7	ǂ	0.42	3.7	0.17	0.3	0.80	0.42	ǂ		ǂ
Agoseris glauca		ǂ												ǂ				
Allium cernuum	ǂ	ǂ	ǂ	ǂ	ǂ	ǂ	ǂ	0.13	ǂ	ǂ	0.13	ǂ	ǂ	0.10	ǂ	ǂ	ǂ	ǂ
Anemone multifida			ǂ	ǂ		ǂ	0.10	ǂ	ǂ			ǂ	0.17	ǂ	ǂ			
Antennaria neglecta						ǂ			ǂ									
Antennaria umbrinella			0.50						3.4			0.67			0.33			ǂ
Antennaria spp.	7.7	2.7	ǂ	6.3	3.5	1.7	18	20	6.7	3.4	5.4	1.0	1.3	1.3	0.30	1.0	1.0	0.20
Arabis holboellii	ǂ				ǂ				ǂ	ǂ				ǂ				
Arctostaphylos uva-ursi													0.67	0.13	0.67			ǂ
Artemisia campestris	0.23	0.10	0.10				ǂ		ǂ				1.0					
Artemisia frigida	1.0	0.33	0.10	0.73	0.30	0.15		0.13	ǂ	ǂ	0.2	ǂ	0.17	0.17	ǂ			
Aster spp.			ǂ															ǂ
Aster campestris var. campestris			ǂ			ǂ									ǂ			ǂ
Aster conspicuus																0.17	0.33	ǂ
Aster ericoides ssp. pansus														1.3				
Astragalus agrestis		0.20		0.17	0.23	ǂ	ǂ	0.13	ǂ			ǂ	ǂ	0.13	ǂ			
Astragalus miser		ǂ	ǂ	0.73	1.5	0.20		0.20	ǂ	0.33	0.27	ǂ	0.50	0.43	ǂ			ǂ
Astragalus tenellus				ǂ	0.33	ǂ			ǂ				ǂ					
Calochortus macrocarpus	ǂ	ǂ					ǂ	0.10			ǂ		ǂ					
Campanula rotundifolia	ǂ																	
Cerastium arvense	1.5	0.67		1.2	1.2	0.45	0.40	0.37	0.18	0.13	ǂ	ǂ	ǂ	0.10	ǂ			
Chenopodiastrum spp.		ǂ									ǂ						ǂ	
Cirsium hookerianum						ǂ			ǂ			ǂ						
Cirsium undulatum				0.13	0.13	ǂ		0.20	ǂ					ǂ				
Cirsium spp.						ǂ						ǂ			ǂ			
Comandra umbellata	0.33	ǂ	0.10	1.7	1.7	0.67	1.3	1.3	0.67									
Cynoglossum officinale											ǂ							
Erigeron compositus				ǂ														
Erigeron flagellaris var. flagellaris	ǂ	0.17	ǂ	ǂ	0.27			ǂ		ǂ	0.33	ǂ	0.10	0.16	0.67			
Erigeron speciosus var. speciosus				ǂ	ǂ	ǂ	ǂ	0.17	ǂ	1.0	2.0	0.17	1.3	2.4	0.14	0.87	0.87	ǂ
Erigeron spp.																		ǂ
Eriogonum heracleoides	0.37	1.0	ǂ	ǂ			0.20	1.1	ǂ		ǂ		ǂ	ǂ				
Erysimum inconspicuum	ǂ	ǂ																
Fragaria virginiana				0.40	ǂ	0.17	1.1	1.0	0.18	0.37	ǂ	0.10	ǂ	ǂ	ǂ	ǂ		ǂ
Galium boreale										ǂ	0.20		0.10	0.43	ǂ	0.10	0.10	ǂ
Geum triflorum													ǂ					
Goodyera oblongifolia																		ǂ
Hieracium albiflorum													ǂ					
Lathyrus ochroleucus													0.33	0.17				
Linum perenne ssp. Lewisii			ǂ	0.20	ǂ	ǂ												
Lithospermum ruderale		ǂ	ǂ	ǂ	ǂ	0.18	ǂ	0.40	ǂ				ǂ	ǂ	ǂ	ǂ	ǂ	ǂ
Lomatium macrocarpum		ǂ			ǂ			ǂ						ǂ				

	N*			ELO			ETO			ETD & ETM			MO & ILD			MC		
Forb Species Continued	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023
Opuntia fragilis		ǂ	ǂ															
Orthocarpus luteus	ǂ	ǂ	ǂ	0.17	0.13	ǂ	0.23	1.1	0.19		ǂ	ǂ	0.17	0.10	ǂ			
Penstemon procerus	ǂ													ǂ	ǂ			ǂ
Polygonum spp.									ǂ									
Potentilla hippiana	0.1	0.10	ǂ	1.8	1.1	0.42	7.0	7.4	0.37	1.0	0.27	ǂ	1.1	0.35	0.17	ǂ	ǂ	ǂ
Potentilla pensylvanica	ǂ	0.13	ǂ	ǂ	ǂ	ǂ	0.13		ǂ	0.33	ǂ		ǂ		ǂ			
Prunella vulgaris		ǂ																
Rhinanthus minor														ǂ				
Selaginella densa													0.67	0.23				
Silene drummondii		ǂ	ǂ		0.13	ǂ		ǂ	ǂ		ǂ				ǂ			
Solidago spathulata	ǂ			ǂ	ǂ	ǂ	0.10	0.17	ǂ	0.23	ǂ	ǂ	0.33	0.20	ǂ	0.20	0.21	0.10
Taraxacum officinale	ǂ	1.1	0.10	0.12	0.23	ǂ	ǂ	0.17		0.11	0.73		0.27	0.27	ǂ	ǂ	ǂ	ǂ
Tragopogon dubius	0.17	0.17	ǂ	0.10	0.10	ǂ	0.13	0.17	0.17		0.13	ǂ	ǂ	ǂ				
Tragopogon pratensis			ǂ			ǂ			ǂ			ǂ			ǂ			
Tragopogon spp.		ǂ	ǂ	ǂ	ǂ	ǂ	ǂ	0.10	ǂ			ǂ	ǂ	0.10	ǂ			ǂ
Vicia americana		ǂ								0.35			1.5	0.67	ǂ	ǂ		ǂ
Zygadenus venenosus						ǂ			ǂ									
Total Percent Cover	**12**	**7.9**	**1.9**	**16**	**13**	**5.0**	**29**	**36**	**12**	**7.9**	**14**	**2.5**	**11**	**10**	**3.3**	**2.6**	**2.6**	**0.88**
Total Species Richness	**20**	**27**	**24**	**25**	**25**	**29**	**20**	**24**	**30**	**16**	**21**	**20**	**30**	**31**	**27**	**12**	**10**	**21**

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).
Note. Species and total mean values have been rounded to two significant figures, cells containing ǂ indicate a species presence; however, in trace amounts (<0.10%). The total values include these trace amounts.

Table 10. Mean percent cover (%) of shrub species, mosses & lichens and other (i.e., non-living) groups in 2015, 2018 and 2023, across six tree encroachment categories (n=3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	N*			ELO			ETO			ETD & ETM			MO & ILD			MC		
Shrub Species	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023	2015	2018	2023
Ericameria nauseosa	0.50	0.67	0.17	1.3	0.77	0.4	0.27	0.17	ǂ	0.10	ǂ	ǂ	0.83	1.0	0.30			
Juniperus communis				ǂ		0.22	1.0	1.4	1.4	ǂ	2.7	0.18	1.7	2.3	2.7	0.67		ǂ
Juniperus scopulorum				2.3	2.3	1.7	4.3	5.2	6.2	13	9.1	17	7.3	9.8	11	6.7	7.3	12
Rosa woodsii													1.0	0.57	0.20	0.57	0.57	0.13
Symphoricarpos occidentalis													0.33	0.23	0.18	0.23	0.23	0.23
Total Percent Cover	**0.50**	**0.67**	**0.17**	**3.7**	**3.1**	**2.3**	**5.6**	**6.7**	**7.6**	**14**	**12**	**17**	**11**	**14**	**15**	**8.1**	**8.1**	**12**
Total Species Richness	**1**	**1**	**1**	**3**	**2**	**3**	**3**	**3**	**3**	**3**	**3**	**3**	**5**	**5**	**5**	**4**	**3**	**4**
Mosses & Lichens																		
Cladonia spp.	19	0.33		32	15	ǂ	33	5.4	ǂ	1.7	0.33	ǂ	12	10	ǂ	0.23	0.23	ǂ
Feather moss spp.									0.17	8.7	3.7				ǂ	12	12	8.7
Lichen spp.	1.3		1.7	1.7	17	4.3	3.0		28.4	0.37		ǂ	3.0	0.67	5.2	0.10	0.10	ǂ
Moss spp.	0.77		0.33	0.80		2.0	3.0	0.73	10.8	8.3	8.7	14	1.7	0.33	3.4	5.0	4.7	0.13
Peltigera spp.		ǂ			0.67	ǂ		0.17	0.17	0.57	0.33	ǂ	4.3	1.0	0.10	1.3	1.3	0.17
Total Percent Cover	**21**	**0.34**	**2.0**	**34**	**32**	**6.4**	**39**	**6.3**	**40**	**20**	**13**	**14**	**21**	**12**	**8.8**	**18**	**18**	**9.0**
Other																		
Bare mineral soil	22	13	19	11	8.0	37	6.7	4.7	16	0.10	0.10	6.8	5.1	6.7	0.85		0.67	0.20
Burned area					0.67	0.67		3.0	2		8.7	7.2						
Litter	71	74	59	63	55	35	51	56	33	80	78	67	75	79	75	90	88	75
Total	**93**	**87**	**77**	**74**	**64**	**72**	**57**	**64**	**51.0**	**80**	**86**	**81**	**80**	**86**	**76**	**90**	**89**	**75**

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).
Note. Species and total mean values have been rounded to two significant figures, cells containing ǂ indicate a species presence; however, in trace amounts (<0.10%). The total values include these trace amounts.

Table 11. Mean and standard deviation of graminoid, forb and shrub total percent cover in old burn-pile sites across four slashed, piled and burned encroachment categories in 2023. Mean and standard deviation were derived from the summed percent cover of grouped species within a plot then averaged by encroachment category (n = 3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	Graminoids		**Forbs**		**Shrubs**		**Mosses**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
ELO**	8.4	6.1	4.5	3.1	0.0	-	15	13
ETO	2.9	1.4	3.5	2.3	0.0	-	22	26
ETD & ETM	0.97	1.5	2.2	1.8	0.33	0.82	17	25
MO & ILD**	0.7	0.55	0.37	0.64	0.0	-	11	12

*ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate) and MO & ILD (mature, open & ingress, low, dense).
** (n = 2)

Table 12. Mean and standard deviation of bare mineral soil, litter and wood total percent cover in old burn-pile sites across four slashed, piled and burned encroachment categories, in 2023. Mean and standard deviation were derived from the summed percent cover of grouped species within a plot then averaged by encroachment category (n = 3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia.

	Bare Mineral Soil		**Litter**		**Wood**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
ELO**	49	17	8.3	10	15	8.7
ETO	32	34	11	17	24	14
ETD & ETM	27	43	41	38	14	7.8
MO & ILD**	28	49	30	23	32	28

*ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate) and MO & ILD (mature, open & ingress, low, dense).
** (n = 2)

Table 13. Mean and standard deviation of soil compaction (psi), pH, water content (m^3/m^3), temperature (° C) and electrical conductivity (bulk dS/m) across six encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia, in 2023 (n = 3).

	Compaction (psi)		**pH**		**Water Content (m^3/m^3)**		**Temperature (° C)**		**Electrical Conductivity (bulk dS/m)**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
N	233	94	6.7	0.22	0.18	0.05	21	2.5	0.08	0.06
ELO	233	94	6.7	0.26	0.18	0.05	25	4.2	0.05	0.04
ETO	317	49	7.0	0.08	0.16	0.06	24	2.2	0.07	0.04
ETD & ETM	267	94	6.6	0.29	0.20	0.07	20	3.3	0.10	0.06
MO & ILD	300	80	6.9	0.33	0.14	0.05	27	5.6	0.04	0.02
MC	350	0.0	7.0	0.04	0.07	0.02	12	2.0	0.03	0.03

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Table 14. Mean and standard deviation of soil compaction (psi), pH, water content (m^3/m^3), temperature (° C) and electrical conductivity (bulk dS/m) in old burn-pile sites across the four treated encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia, in 2023 (n = 3).

	Compaction (psi)		**pH**		**Water Content (m^3/m^3)**		**Temperature (° C)**		**Electrical Conductivity (bulk dS/m)**	
Encroachment Category*	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation	Mean	Standard Deviation
ELO	150	0.0	6.5	0.12	0.20	0.03	25	4.7	0.05	0.03
ETO**	283	103	6.9	0.10	0.15	0.07	26	4.7	0.09	0.11
ETD & ETM	283	103	6.5	0.42	0.23	0.04	22	4.4	0.20	0.08
MO & ILD**	250	115	6.9	-	0.12	0.10	29	2.9	0.06	0.02

* ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate) and MO & ILD (mature, open & ingress, low, dense).
** (n = 2)

Table 15. Mean and standard deviation of estimated Douglas-fir stem density (stems/ha) and height (cm) for each tree encroachment category prior to slashing piling and burning in 2015 and post-treatment in 2018 and 2023 in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n = 3). In 2023, mean stem density was determined using the estimated stem density from individual plots and averaged per encroachment category and mean height was calculated using the height of each recorded seedling of individual plots and averaged per encroachment category.

	2015		**2018**		**2023**			
	Stems per Hectare	**Height (cm)**	**Stems per Hectare**	**Height (cm)**	**Estimated Stems per Hectare**		**Height (cm)**	
Encroachment Category*	Mean	Mean	Mean	Mean	Mean	Standard Deviation	Mean	Standard Deviation
N	-	-	-	-	0.0	0.0	0.0	0.0
ELO	67	50	33	15	11×10^2	23×10^2	11	2.6
ETO	17 x 10	100	67	37	44×10^2	77×10^2	3.3	5.8
ETD & ETM	0.0	-	0.0	-	74×10^2	68×10^2	3.6	3.2
MO & ILD	13 x 10	60	60 x 10	30	69×10^2	11×10^3	9.4	3.7
MC	0.0	-	-	-	31×10^3	15×10^3	4.0	0.65

Note. Mean values for 2015 and 2018 adapted from Steen & Young (2019).
*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

Figures

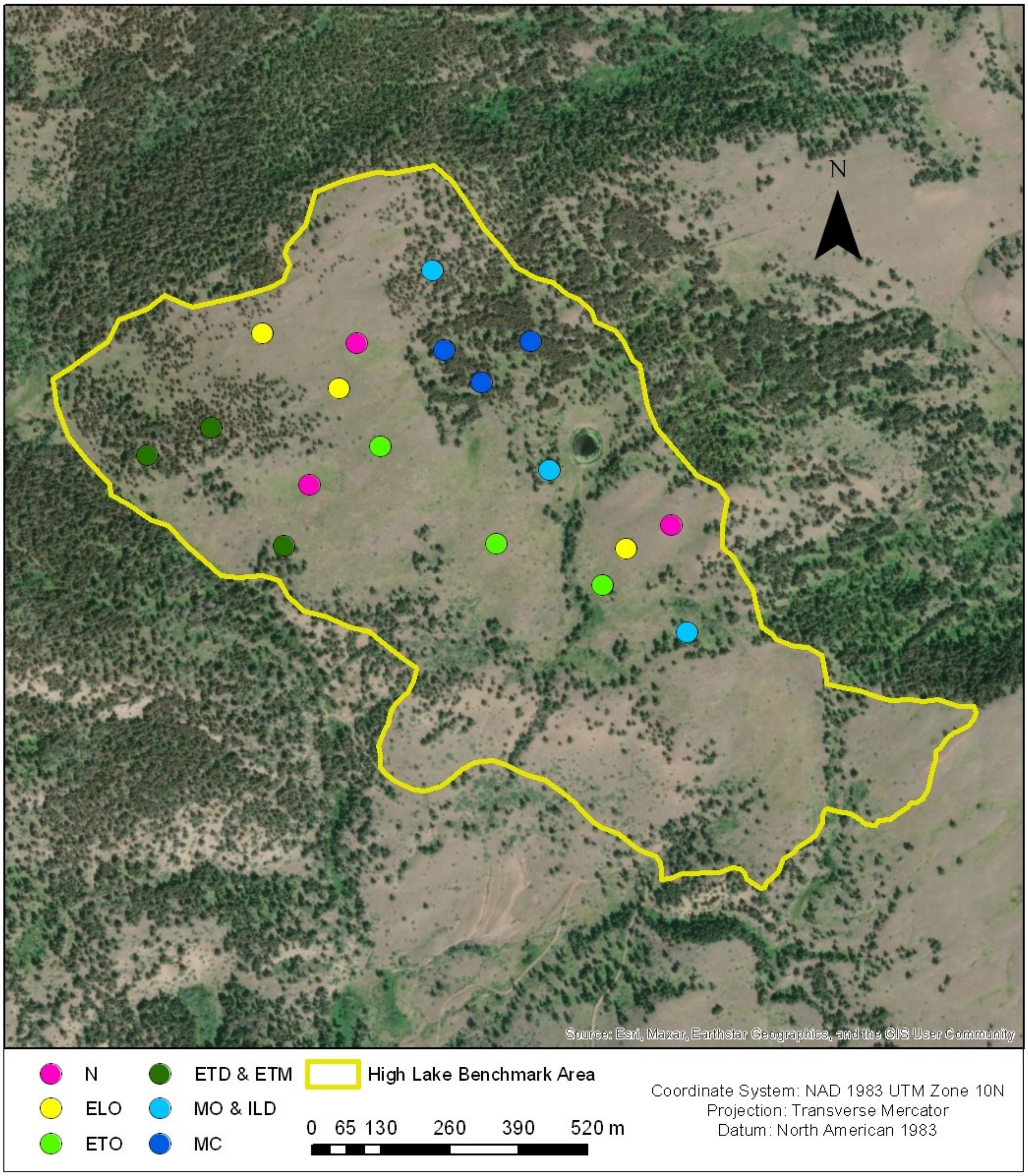

Figure 1. Map of the High Lake Benchmark Area within the Churn Creek Protected Area, British Columbia. The coloured dots represent fixed-radius sampling plots stratified by encroachment category: N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Both Douglas-fir seedling stem density and vegetation percent cover were measured at the indicated locations, in 50 m^2 and 100 m^2 plots, respectively.

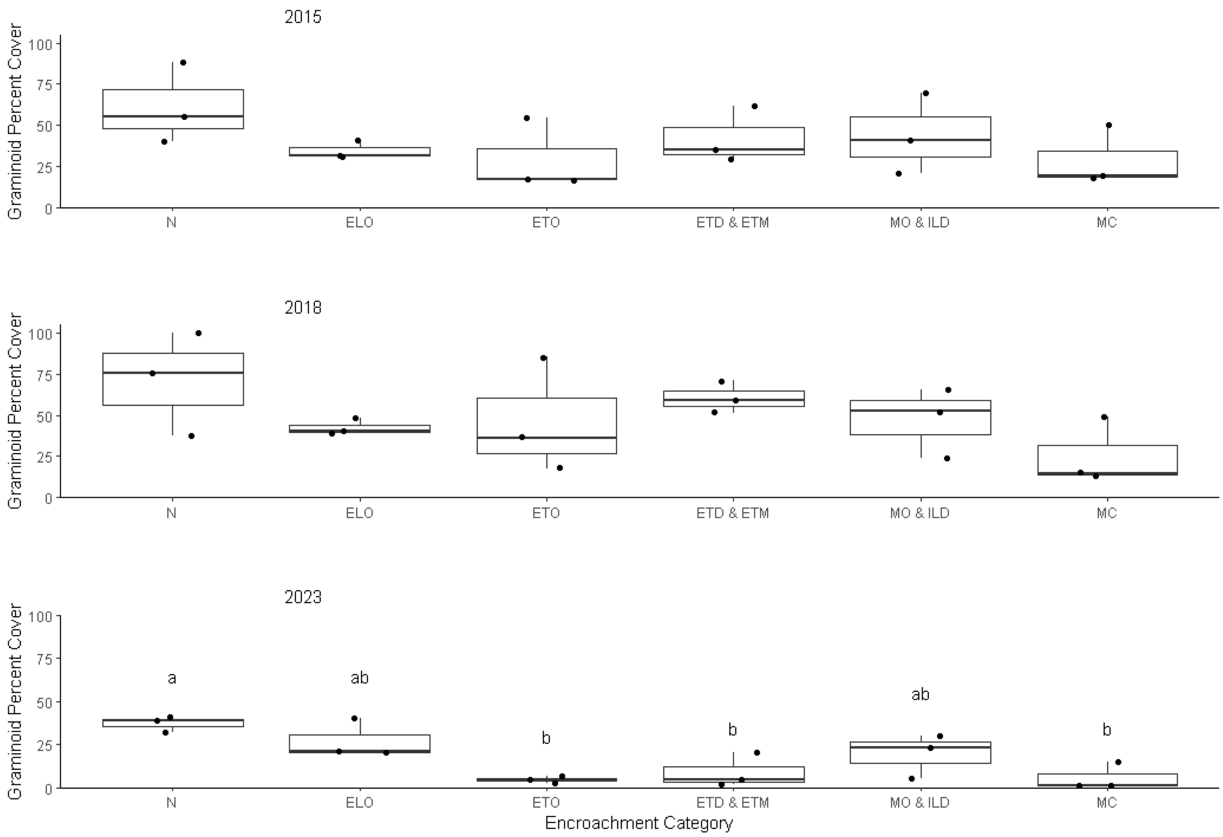

Figure 2. Mean total graminoid percent cover prior to the slashing, piling, and burning treatment in 2015 and after treatment in 2018 and August, 2023 across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Data points represent the summed proportion of graminoid species per individual plot (n=3). Letters (a,b) group encroachment categories by significance (α = 0.05), boxplots sharing the same letter are not significantly different.

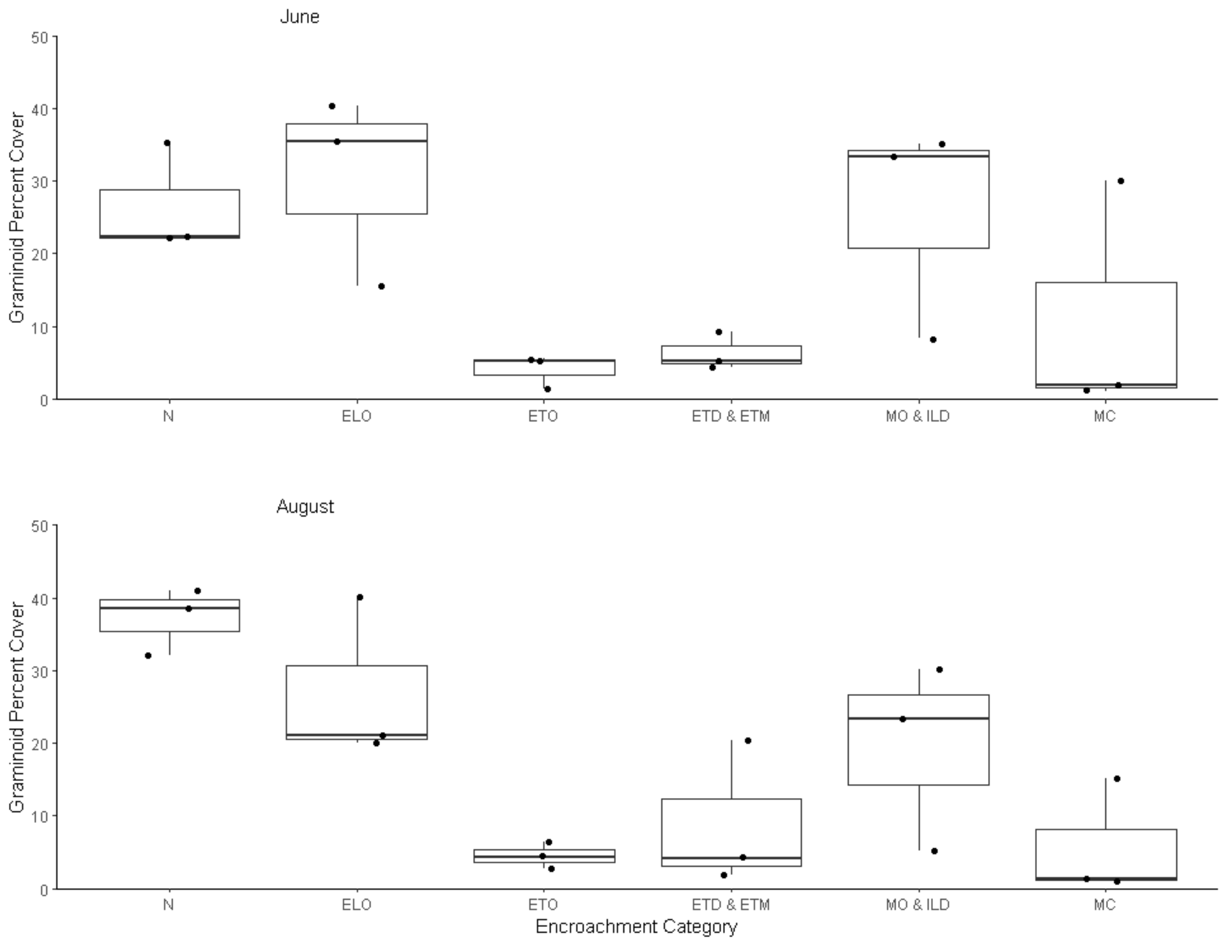

Figure 3. Mean total graminoid percent cover in June and August, 2023, across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Data points represent the summed proportion of graminoid species per individual plot (n=3).

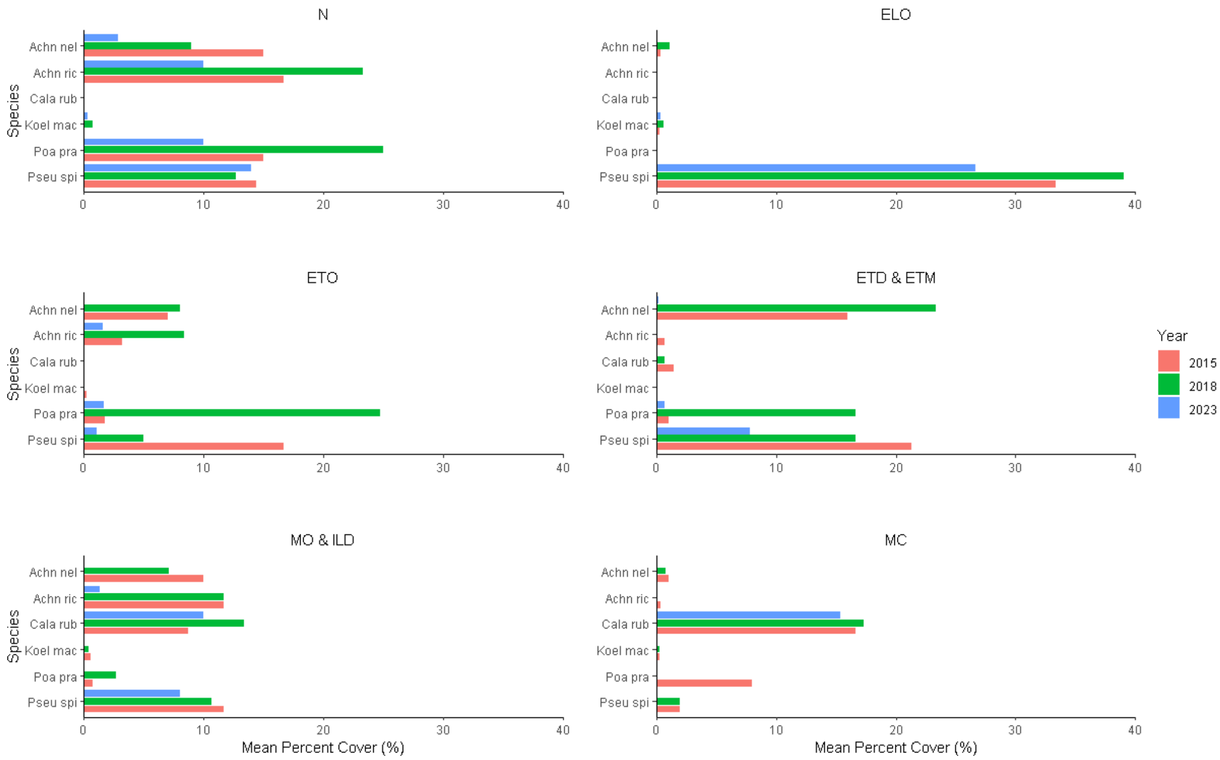

Figure 4. Mean percent cover of six graminoid species prior to the slashing, piling, and burning treatment in 2015 (peach) and after treatment in 2018 (green) and 2023 (blue) across six tree encroachment categories (n=3) in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

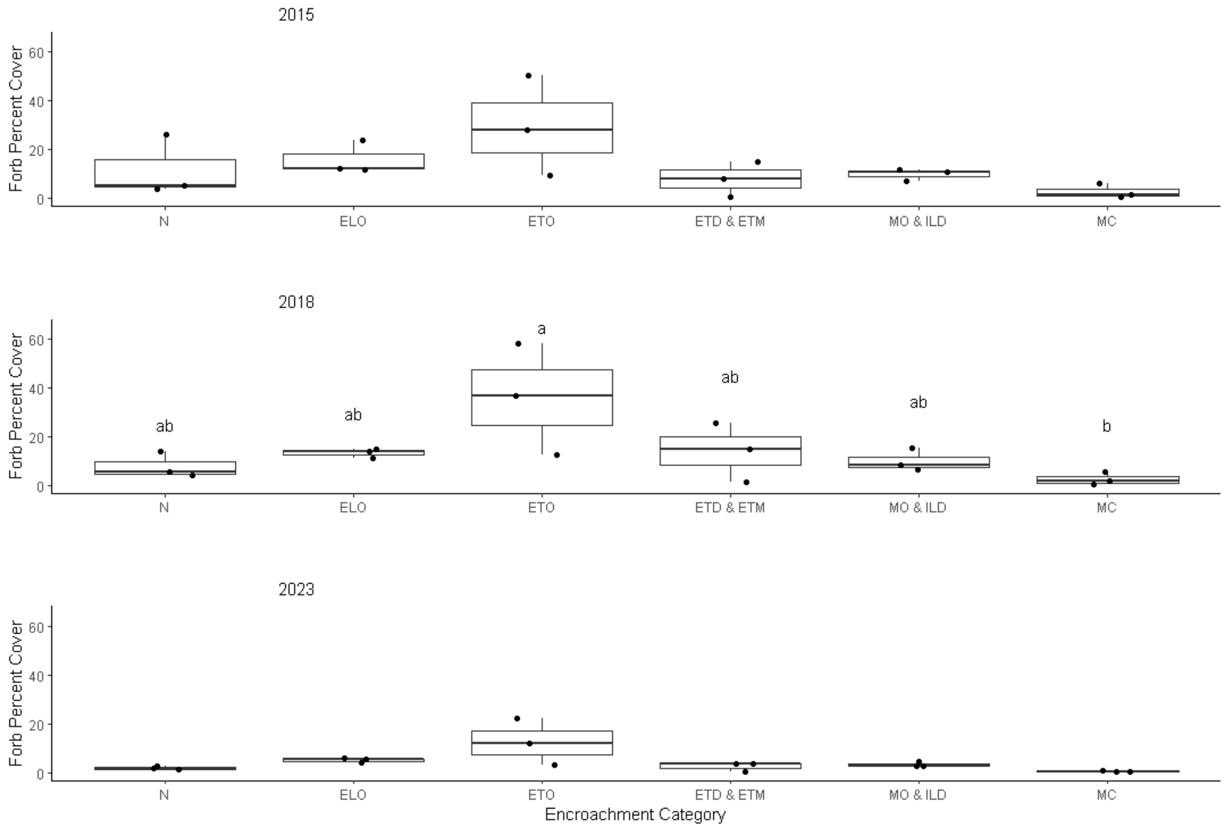

Figure 5. Mean total forb percent cover prior to the slashing, piling, and burning treatment in 2015 and after treatment in 2018 and 2023 across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Data points represent the summed proportion of forb species per individual plot (n=3). Letters (a,b) group encroachment categories by significance (α = 0.05), boxplots sharing the same letter are not significantly different.

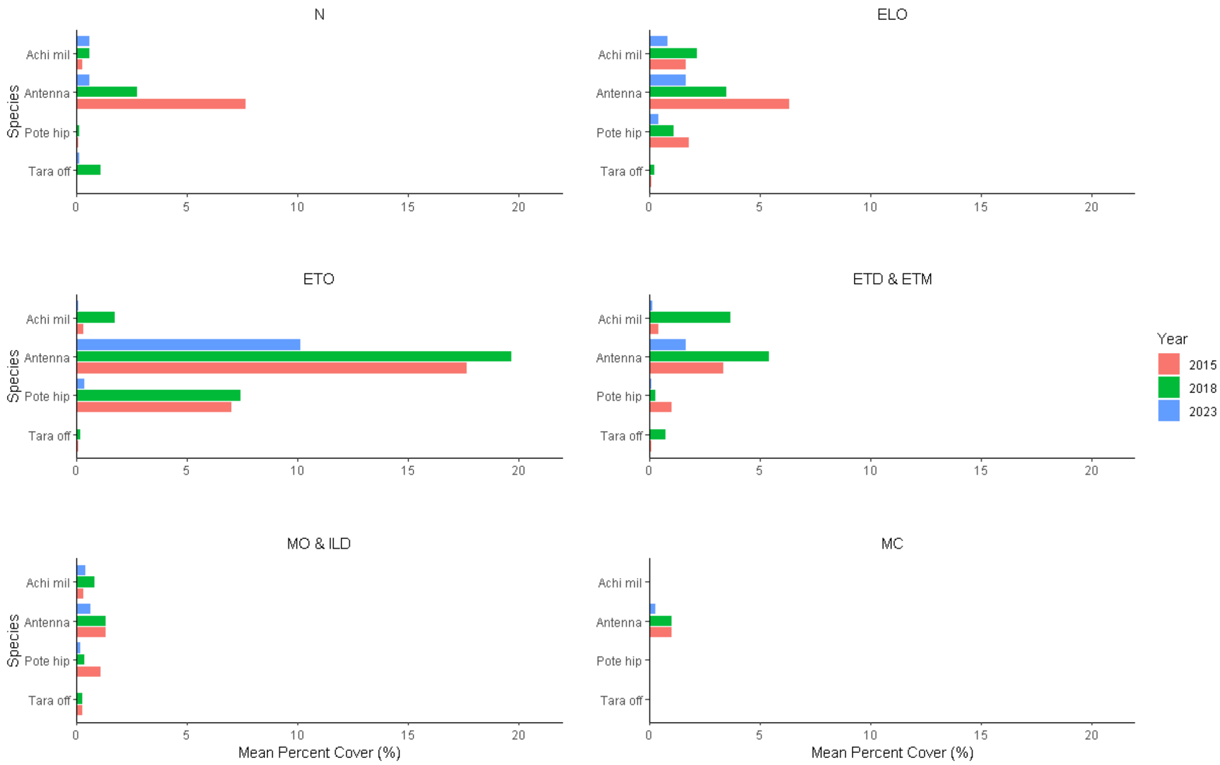

Figure 6. Mean percent cover of four forb species prior to the slashing, piling, and burning treatment in 2015 (peach) and after treatment in 2018 (green) and 2023 (blue) across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n=3). N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

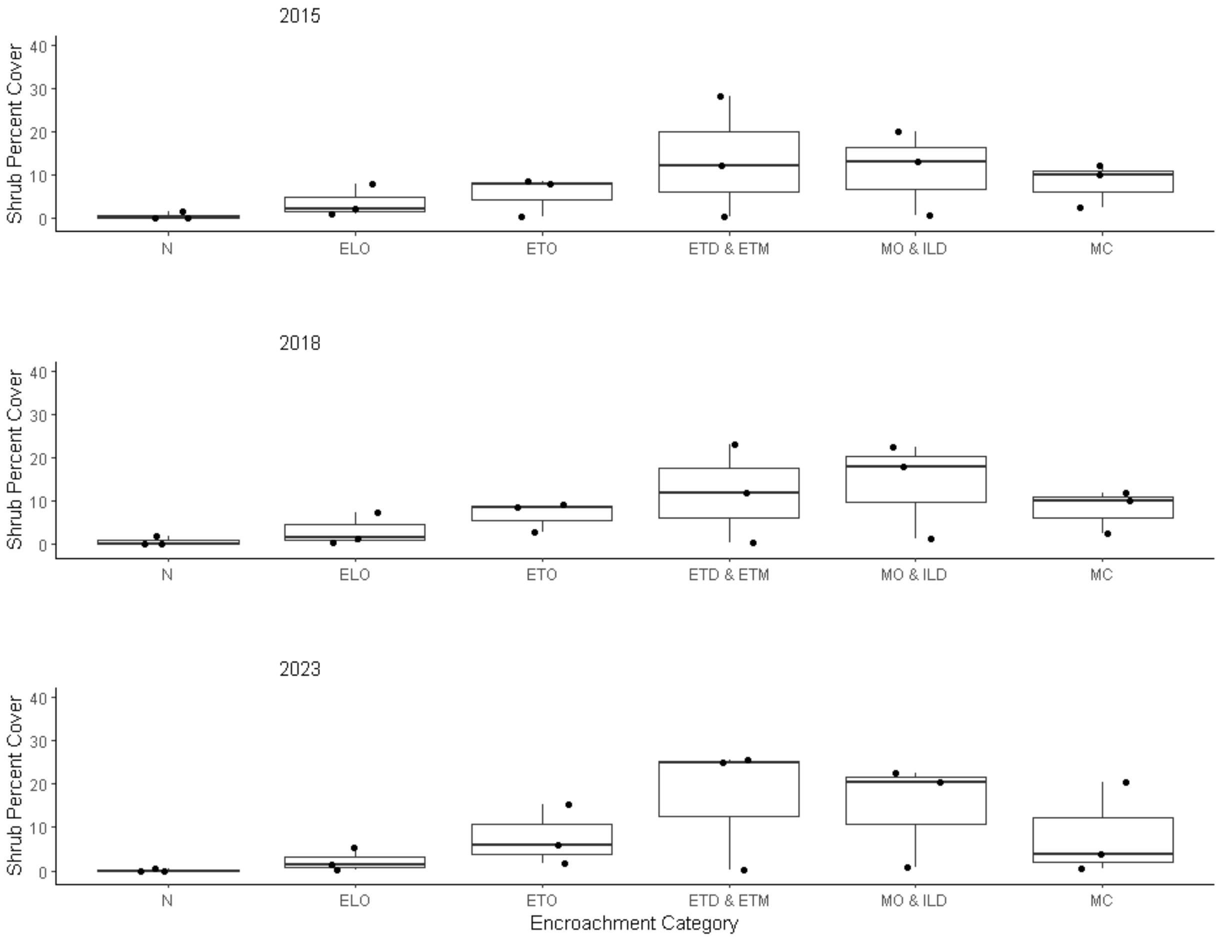

Figure 7. Mean total shrub percent cover prior to the slashing, piling, and burning treatment in 2015 and after treatment in 2018 and 2023 across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Data points represent the summed proportion of shrub species per individual plot (n=3).

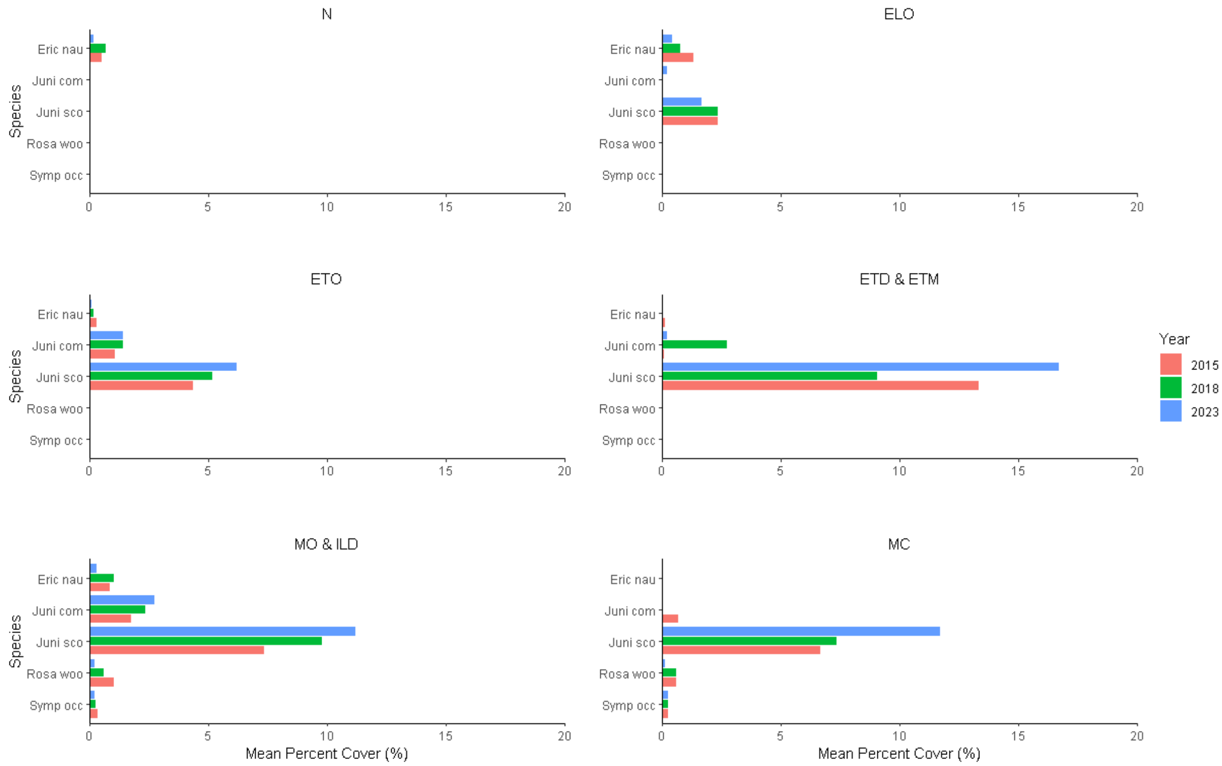

Figure 8. Mean percent cover of all shrub species prior to the slashing, piling, and burning treatment in 2015 (peach) and after treatment in 2018 (green) and 2023 (blue) across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n=3). N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

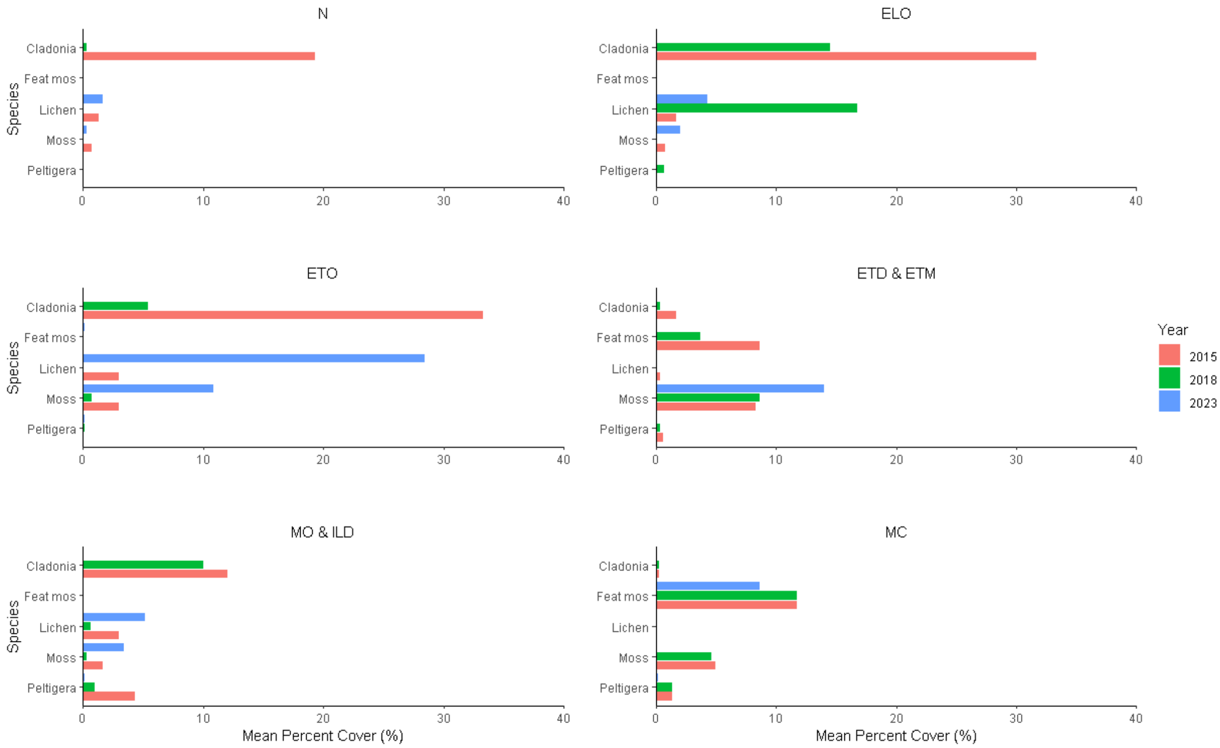

Figure 9. Mean percent cover of all mosses and lichens prior to the slashing, piling, and burning treatment in 2015 (peach) and after treatment in 2018 (green) and 2023 (blue) across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n=3). N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

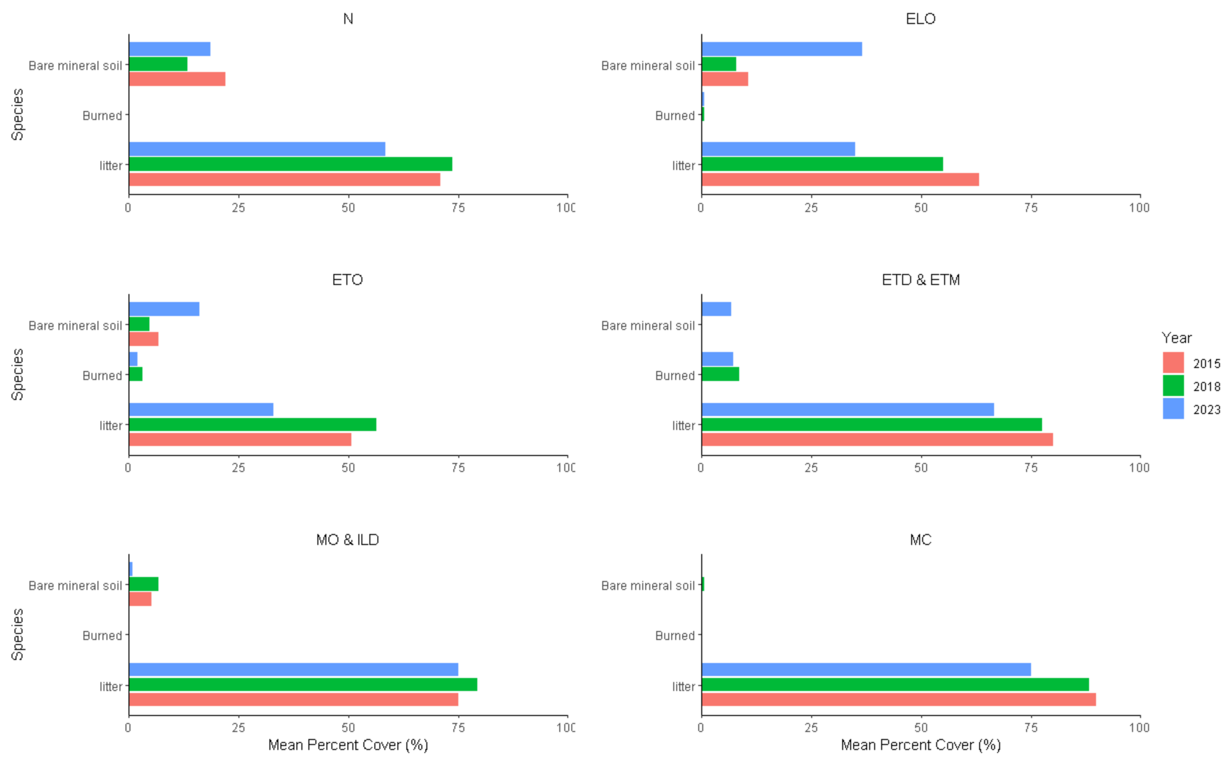

Figure 10. Mean percent cover of bare mineral soil, burned area and litter prior to the slashing, piling, and burning treatment in 2015 (peach) and after treatment in 2018 (green) and 2023 (blue) across six tree encroachment categories in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n=3). N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).

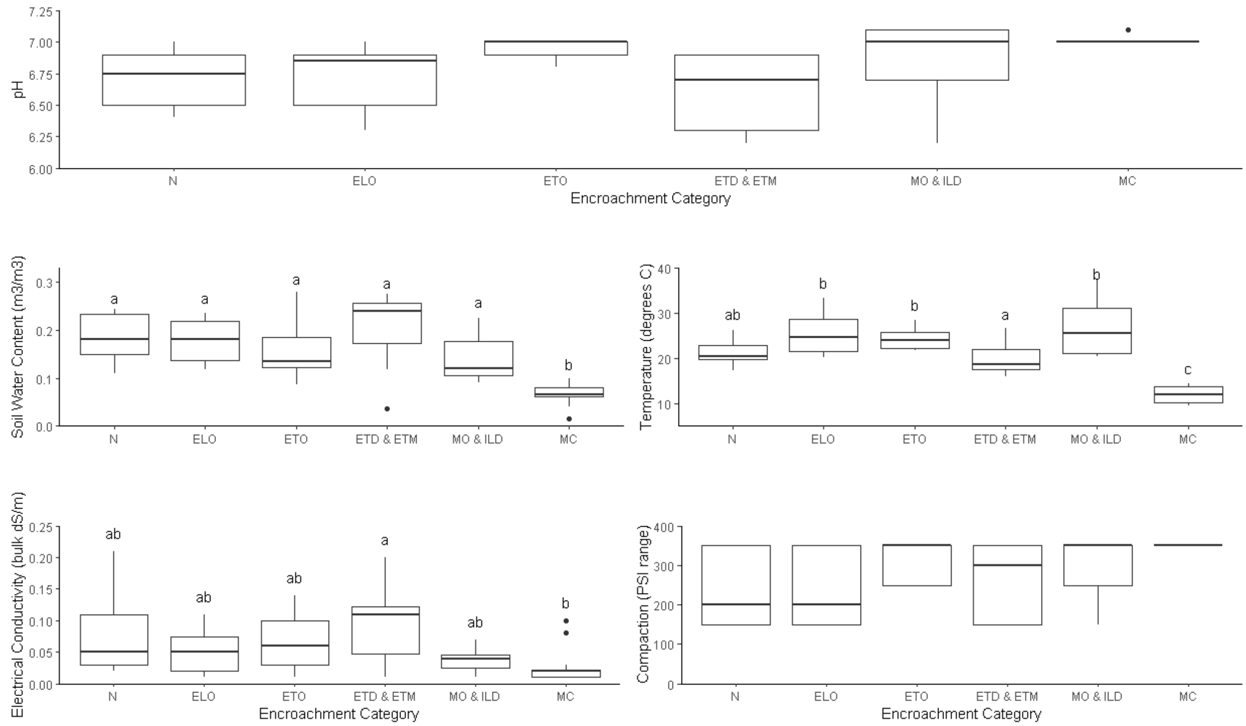

Figure 11. Soil pH, water content (m^3/m^3), temperature (°C), electrical conductivity (bulk dS/m) and compaction (psi) across six tree encroachment categories in 2023, in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia (n = 3). N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Letters (a, b, c) group encroachment categories by significance (α = 0.05), boxplots sharing the same letter are not significantly different.

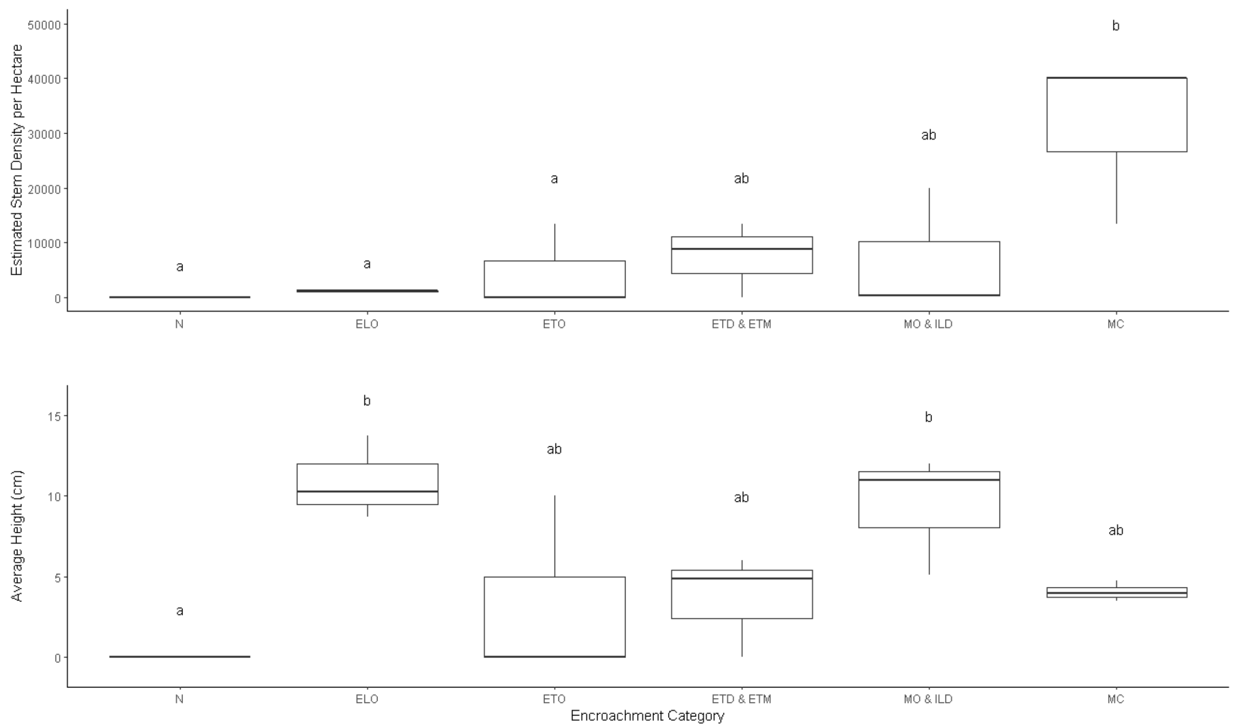

Figure 12. (a) Estimated stem density (stems/ha) and (b) average height (cm) of Douglas-fir seedlings across six tree encroachment categories in 2023, in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed). Data points represent the (a) estimated stem density and (b) average height (cm) of seedlings measured within each plot (n=3). Letters (a, b) group encroachment categories by significance (α = 0.05), boxplots sharing the same letter are not significantly different.

Appendix – Douglas-fir Seedling Recruitment per Vegetation Plot

Table A1. Douglas-fir stem density (stems/50 m^2), proportion of plot measured, estimated stem density (stems/50 m^2), mean height (cm) and standard deviation (SD) in 2023, in the High Lake Benchmark Area, Churn Creek Protected Area, British Columbia. Estimated Douglas-fir stem density is the expected value if the whole plot had been measured. Mean height (cm) was calculated using the heights of all recorded seedlings within a plot.

Plot	**Encroachment Category***	**Observed Stem Density**	**Plot Proportion**	**Estimated Stem Density**	**Height**	
					Mean	Standard Deviation
1	ETD & ETM	44	1	44	6.0	1.7
2	ETD & ETM	50	0.75	67	4.8	2.1
3	ETD & ETM	0	1	0	0.0	-
4	ETO	0	1	0	0.0	-
5	ETO	0	1	0	0.0	-
6	ETO	50	0.75	67	10	6.7
7	MC	50	0.25	200	3.9	1.1
8	MC	50	0.25	200	4.7	1.1
9	MC	50	0.75	67	3.5	0.82
10	MO & ILD	2	1	2	11	0.0
11	MO & ILD	50	0.5	100	12	4.8
12	MO & ILD	1	1	1	5.1	-
13	ELO	6	1	6	10	5.1
14	ELO	6	1	6	8.7	1.6
15	ELO	4	1	4	14	4.1
16	N	0	1	0	0.00	-
17	N	0	1	0	0.00	-
18	N	0	1	0	0.00	-

*N (no encroachment), ELO (encroachment, low, open), ETO (encroachment, tall, open), ETD & ETM (encroachment, tall, dense/moderate), MO & ILD (mature, open & ingress, low, dense) and MC (mature, closed).