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SULA STUDY REVISITED: 20-YEAR POST-FIRE REGENERATION IN THE SOUTHERN
BITTERROOT VALLEY, MONTANA.

By

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Bachelor of Science in Forestry, The University of Montana, Missoula, Montana, 2019

Thesis

presented in partial fulfillment of the requirements

for the degree of

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in Forestry

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Forestry

Sula study revisited: 20-year post-fire regeneration in the southern Bitterroot Valley, Montana.

Chairperson: Dr. Beth Dodson

Co-Chairperson: Dr. Peter Kolb

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ABSTRACT

In the summer of 2000, a number of large fires burned in the southern Bitterroot Valley near Sula, Montana. Research was conducted in 2001 and 2003 in the fire-affected areas of the French Basin and Larid Creek areas in order to investigate the effects of environmental variables, fire severity, and post-fire management on vegetation regeneration. In 2020 these areas were remeasured to understand trends over time by evaluating the impact of these same factors 20 years post fire. The results showed that the effects of environmental variables, fire severity, and post-fire management on vegetation regeneration were varied. The most influential environmental variable to affect vegetation regeneration for understory species and overstory species was aspect. Fire severity was influential, with differences in overstory and understory severity impacting the distribution, presence, and percent cover of vegetation species. The most influential post-fire management activity was seedling planting. Results suggest that study areas that were affected by high severity fire are unlikely to return to pre-fire conditions without tree planting or other management activities. Further research should be conducted on the survival rate of planted seedlings in managed areas over time. Comparisons should also be made between natural seedling regeneration and planted seedling viability in burned areas over time. More research should be conducted on fire severity's long-term effects on understory vegetation as these ecosystems return to a form of equilibrium over time.

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Introduction

In the summer of 2000, a number of large fires burned approximately 350,000 acres (550 square miles) across the southern Bitterroot Valley of Montana in national and state forest lands (Republic, 2014) (Figure 1). These fires, usually referred to as the Valley Complex or the Sula Complex, were ignited by lightning on July 31st, 2000 and continued burning until mid-September (Keegan et al. 2004). By October the 4th, cool cloudy weather stopped fire growth and immediate attention fell to the task of rehabilitating the burned areas.



Figure 1. Map of Montana with the site of the 2000 fires marked. Image source: Map of Montana [Online image]. https://www.nationsonline.org/oneworld/map/USA/montana_map.htm

Approximately 307,000 acres of the fire fell in the Bitterroot National Forest (BNF) which is under the jurisdiction of the United States Forest Service (USFS 2000a). This burned

acreage represented 20 percent of the BNF at the time, and while soil erosion mitigation work and watershed protection efforts had already been made by the Burned Area Emergency Rehabilitation (BAER) teams immediately after the fire, it was clear that extensive recovery work would be needed for the immediate future (USFS 2000a). The work proposed by the Forest Service included the reduction of fuels, the improvement of watersheds, revegetation, and improvements of forest health (Bitterroot 2001). While some restoration work was accomplished including extensive soil stabilization projects, seedling planting, and some salvage logging of standing dead trees, a majority of the plans were not completed due to public backlash and subsequent litigation over more extensive proposed salvage logging operations (Sienkiewicz 2006).

Over 15,000 acres of the fire fell in the Sula State Forest, which is under the jurisdiction of the Montana Department of Natural Resources and Conservation (DNRC) (Harrington 2003). Within 6 months of the fire, extensive salvage logging had taken place on 6,000 acres of the affected area and numerous fire rehabilitation projects, including erosion mitigation and culvert installation (Harrington 2003). The salvage logging conducted in this area was in response to the DNRC's explicit mandate to manage state trust lands for long-term revenue generation (Sienkiewicz 2006). Part of the rehabilitation projects involved the planting of over one million seedlings in areas that had been affected by high severity fire (Republic, 2014).

The rehabilitation work accomplished by both agencies affected large areas of the landscape. This, in combination with an unprecedented fire that occurred in a diverse forested ecosystem, raised the question of what possible long-term effects these management decisions and general fire characteristics could have on vegetative regeneration. This study will investigate

these long-term effects and trends by examining the effects of environmental variables, fire severity, and post-fire management on vegetation regeneration in the southern Bitterroot Valley.

It is important to understand the impacts of disturbance on a forest ecosystem's ability to regenerate when considering climate change. In particular, overstory tree cover provides a buffer to understory vegetation and tree seedlings from climate extremes (Davis et al. 2018). This buffer protects seedlings and understory vegetation from hotter temperatures and can help retain ground level moisture (Davis et al. 2018). The removal of the overstory due to uncharacteristically severe disturbances can have direct effects on the conditions that understory vegetation could experience, and tree regeneration may no longer be possible (Davies et al 2018). This could result in conditions that could affect forest resiliency to disturbance and could result in uncharacteristic ecosystem transitions (Davis et al. 2020). By examining post-fire ecosystems, we can determine to what extent severe disturbances have altered successional pathways and if mitigation efforts are effective in countering fire impacts.

An important fire characteristic is fire severity. Fire severity is defined as what happens to the dominant vegetation during a fire (Arno et al. 2000). In this case, the effects of fire on tree mortality determines the level of severity. If a majority of the overstory trees are killed, the fire is considered "high severity"; if the fire does not kill most of the overstory trees, it is considered "low severity"; and if a combination of the two occurs, the fire is considered "mixed severity" (Arno et al. 2000). Forest ecosystems in the northern Rocky Mountains, including the southern Bitterroot valley, historically operated under a mixed severity fire regime (Brown et al. 1994). However, the combination of fire suppression, livestock grazing, and the removal of Native Americans and their burning practices has caused a shift in the fire regime of the Douglas-fir (*Pseudotsuga mienzisia*) and ponderosa pine (*Pinus ponderosa*)-dominated dry forest ecosystem

types (Arno et al. 2000; Hessburg and Agee 2003). The historically low to mixed severity fires that were more frequent on this landscape have now shifted to high severity stand replacing fires that can have long-term ecological and social impacts (Arno et al. 2000). There are numerous examples of these ecological impacts, with one of them being a shift in overstory tree species composition from primarily ponderosa pine, western larch (*Larix occiedentalis*), and whitebark pine (*Pinus Albicaulis*) to Douglas-fir dominated stands in lower elevation areas (Arno et al. 2000; Arno et al. 1995). This, combined with an increase of dead and down woody debris and ladder fuels, increases the opportunity for the occurrence of unusually severe and extensive wildfires as seen with the Valley Complex and other large fire events in the last two decades (Keane et al. 2002; Arno et al. 2000).

Fire effects on tree mortality for ponderosa pine and Douglas-fir have been attributed to crown scorch volume and direct tissue damage (Fowler and Hull, 2004). Crown scorch volume is the proportion of foliage either consumed or damaged on a tree following a fire. A greater proportion of crown scorch volume was shown by Fowler and Hull (2004) to be an effective indicator of tree mortality, with 80-95 percent scorch volume for ponderosa pine and 70 to 95 percent for Douglas-fir indicating a high probability of tree death within two to three years post-fire. It could be expected that high severity fire would result in a high proportion of fire-scorched trees and subsequent mass mortality. This increase in tree mortality can have serious effects on forest regeneration and could induce uncharacteristic changes in forest ecosystems.

In conjunction with changes in the forest ecosystem due to fire severity, tree regeneration and understory vegetation is also affected. The effects of fire on understory vegetation regeneration are shown to be variable depending on severity and soil duff consumption (Armour et al. 1984). While short-term vegetational recovery in response to fire is variable depending on

severity, pre- and post-fire vegetation species composition, and overstory tree mortality, the eventual successional outcome is expected to be the same (Lyon and Stickney 1976; Armour et al. 1984). Succession generally follows the same path from herb-dominated, to shrub-dominated, to tree-dominated systems over time. However, this timeline is dependent on the severity of the disturbance and other site characteristics (Armour et al. 1984). While species richness and plant cover are shown to increase slightly post-fire, other contributing factors such as environmental variables most likely have a stronger influence on understory vegetation regeneration (Laughlin & Fule 2008). Environmental variables such as slope and aspect are important influences on vegetation regeneration due to their effects on solar and moisture availability (Laughlin & Fule 2008). Specifically, sunnier sites generally have higher soil temperature and shady sites have more soil moisture, both of which can greatly affect the species composition and percent cover of vegetation that is present (Xue et al. 2018).

Post-fire management can also have varying effects on tree regeneration and understory vegetation. In the case of the Valley Complex, the most prominent management activities for both agency-controlled areas were salvage logging and seedling planting. Studies that focused on salvage logging's effects on post-fire forest structure have found no significant long-term impacts on vegetation regeneration. Fifteen years after treatment, understory vegetation species composition and cover were not affected by logging activities when best management practices were followed (Peterson & Dodson, 2016). Salvage logging's effects were most noticeable on shrub cover, with higher salvage intensities resulting in lower cover due to the disturbance of underground rhizomes (Knapp & Ritchie, 2016). Rhizomes are characterized as horizontal underground stems that often can persist after severe disturbances. A common species that depends on rhizomes for regeneration is ninebark (*Physocarpus malvaceus*) which is known to

sprout vigorously after fires (Habeck 1992). Other studies found that immediate effects from post-fire logging were variable depending on a number of factors. The type of logging system had an effect on ground compaction and erosion, with ground-based skidding causing the most direct effects. In some cases, logging residue was shown to reduce overland flow and subsequently slow erosion. Logging has been shown to significantly reduce post-fire habitat for species that depend on standing dead snags for nesting habitat. However, logging also increased habitat for species that prefer non-boreal environments resulting in a change in overall species composition, but not richness (Mciver & Starr, 2000). McGinnis et al. (2010) showed that post-fire logging practices and herbicide treatments on shrub regeneration had differing effects on dead fuel amounts and understory species composition. Salvage logged areas showed greater amounts of dead fuel, however the predicted fire behavior of that area was not different from the untreated areas due to the persistence of shrub cover in both areas. Salvage logging is not without controversy, with some studies pointing to its possible negative effects on wildlife habitat and seedling regeneration. One study suggests that salvage logged areas could reduce seedling regeneration due to soil disturbance and excess woody debris, resulting in long-term effects on forest health (Donato et al., 2006).

Other post-fire treatments such as regeneration planting have shown to achieve their goal in increasing the number of saplings present in comparison to untreated stands (Donovan, et. al., 2019). However, spatially homogeneous planting methods have been found to be non-conducive to stand resilience to future disturbances such as fire (North, et al., 2019). Ouzts et al. (2015) showed that areas that had undergone post-fire seedling planting produced target amounts of mature trees over time as compared to non-planted areas which were not able to produce the desired density. While this study highlighted that only half of the areas planted would meet

desired tree densities over time, this was an improvement over non-planted areas that did not meet desired densities.

The areas affected by the fires of 2000 in the Southern Bitterroot valley display a unique combination of all of these factors. Wildland fires effects on forested ecosystems can have varied, extensive, and long-lasting consequences that need to be investigated in order to promote holistic land management in the future.

In 2001, twelve study transects were established in the fire-affected areas of the BNF and the Sula State Forest in order to document changes in vegetation recovery over time (Kolb & Thompson, 2001). Field research was conducted during the summers of 2001, 2003, and 2020 on these transects within the southern Bitterroot valley (Figure 2). The study area is split between the Laird Creek and French Basin areas located near Sula, MT. The Laird Creek study area, west of Sula, is located in the Bitterroot National Forest and managed by the US Forest Service. The French Basin area, north of Sula, is located in the Sula State Forest and is managed by the MT Department of Natural Resources and Conservation (Figure 2).

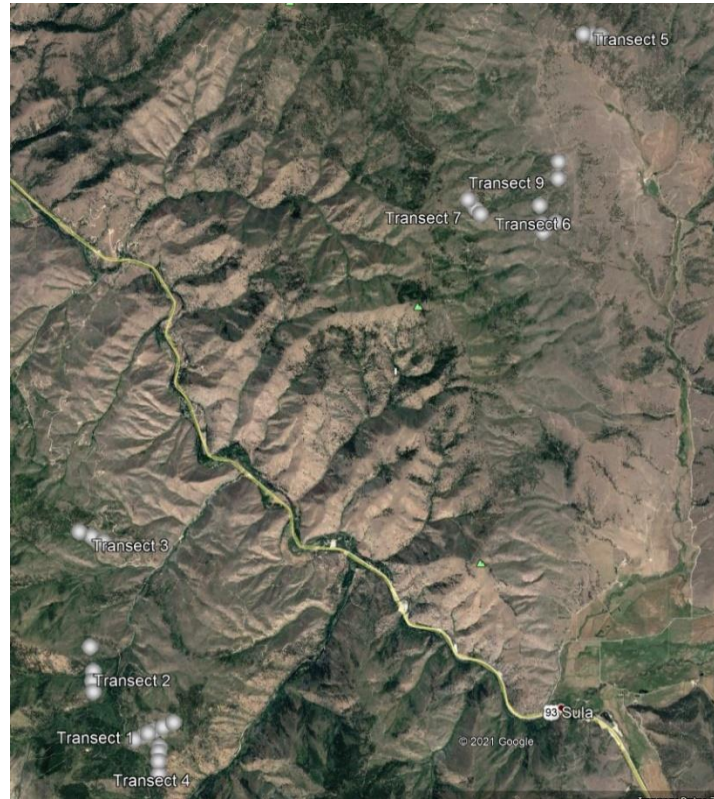


Figure 2. Overview map of study area. The Larid Creek study area consists of transects 1-4 in the far bottom left of the map. The French Basin area consists of transects 5-9 in the top right of the map.

These sites were sampled during the summer of 2001 and 2003 in order to investigate the influences of fire severity, environmental variables, and post-fire management on post-fire vegetation recovery. Data collected during these studies included tree counts, seedling counts, and understory vegetation cover. The results from the 2003 study indicated that the post-fire plant community was dominated by species that were resistant to fire. Salvage logged areas did not display any significant difference in vegetation as compared to non-salvage logged areas, although there was less vegetation variability in non-salvaged areas overall (Hollingsworth, 2005). Habitat type group, firegroup, understory vegetation cover, mean patch size in 2001, and overstory severity were shown to be the most influential indicators of understory cover in 2003 (Hollingsworth, 2005). In the summer of 2020, these transects were re-sampled in order to continue the research that had been previously conducted in 2001 and 2003.

The purpose of this study is to investigate the effects of environmental variables, fire severity, and post-fire management activities on vegetation regeneration in the areas affected by the fires of 2000 in the Southern Bitterroot Valley. Statistical analyses were conducted utilizing the data collected in 2001, 2003, and 2020 in order to answer the following questions:

1. What are the differences in overstory tree survival over 20 years, and what are the most significant explanatory variables that affect the number of live trees in 2020?
2. What are the changes in seedling survival between over time, and what are the most significant explanatory variables that affect the number of live seedlings in 2020?
3. What are the differences in understory vegetation patch size and composition over time, and what are the most significant explanatory variables that affect patch size, overall cover, and individual species cover over time? Additionally, what effects do overstory and understory fire severity have on the percent cover of major understory species over time?

Methods

Study Location and Transect Differences

Eight transects, which were established during the 2001 study (Figure 2), were remeasured during the summer of 2020. Four of these transects were located in the Larid Creek area (transects 1-4) and four were located in the French Basin area (transects 5-7 and 9). The topography was varied in the study area, with slopes ranging from 0 to 60 percent, with a majority of the nested plots located on 20% to 40% slopes. Study plots generally fell on east to southeastern facing aspects and were generally part of the warm/dry habitat type. The warm/dry Douglas-fir habitat type is generally found on the warmer slopes and benches in the area, and

generally consists of ponderosa pine and Douglas-fir, depending on the stage of succession the stand is in (Crane and Fischer 1986). These sites are generally more productive than the cool/dry sites that were also sampled within the study area. Warm/dry habitat types were sampled at transects 3, 5, 6, 7, and 9. The cool/dry habitat type is generally found on cooler facing slopes and is characterized as a less productive, Douglas-fir dominated stand structure (Crane and Fischer 1986). Within the study area, examples of this habitat type were sampled in transects 1, 2, and 4.

Transects were established across areas that experienced three kinds of post-fire salvage logging treatments after the fires, which are classified as a no salvage treatment, a delayed salvage treatment, and an immediate salvage treatment. Transects classified as a no salvage treatment were transects 1, 2, 3 and 5 and received no post-fire salvage logging of any kind. Transects 4 and 7 were classified as a delayed salvage treatment and were salvage logged during 2002 and 2003. Transects 6 & 9 were classified as an immediate salvage treatment and were salvage logged during the winter of 2000/2001 (Figure 3). Salvage classifications, overstory and understory fire severity rankings, aspect, and percent slope of each nested plot within these transects is listed in the following table (Table 1)

Table 1. Transect and plot data.

| Transect # Plot # | Aspect (degrees) | Slope (%) | Overstory Severity | Understory Severity | Salvage type |
|----------------------|---------------------|-----------|-----------------------|------------------------|-------------------|
| T01P01 | 68 | 32 | Mixed | Mixed | No Salvage |
| T01P02 | 200 | 39 | High | Mixed | No Salvage |
| T01P03 | 134 | 42 | High | High | No Salvage |
| T01P04 | 94 | 39 | High | High | No Salvage |
| T01P05 | 224 | 40 | High | High | No Salvage |
| T01P06 | 186 | 40 | High | High | No Salvage |
| T02P01 | 134 | 50 | Low | Mixed | No Salvage |
| T02P02 | 50 | 59 | Mixed | High | No Salvage |
| T02P03 | 360 | 65 | Low | High | No Salvage |
| T02P04 | 90 | 30 | High | High | No Salvage |
| T03P01 | 130 | 49 | Low | High | No Salvage |
| T03P02 | 136 | 35 | Mixed | High | No Salvage |
| T03P03 | 120 | 20 | High | High | No Salvage |
| T04P01 | 310 | 48 | Low | Mixed | Delayed Salvage |
| T04P02 | 296 | 48 | Mixed | Mixed | Delayed Salvage |
| T04P03 | 286 | 55 | High | High | Delayed Salvage |
| T04P04 | 250 | 55 | Low | Low | Delayed Salvage |
| T05P01 | 130 | 25 | Low | Mixed | No Salvage |
| T05P02 | 220 | 30 | Mixed | High | No Salvage |
| T05P03 | 290 | 30 | High | High | No Salvage |
| T06P01 | 81 | 2 | High | High | Immediate Salvage |
| T06P02 | 77 | 6 | Mixed | High | Immediate Salvage |
| T06P03 | 110 | 7.5 | Low | Low | Immediate Salvage |
| T07P01 | 72 | 27 | Low | High | Delayed Salvage |
| T07P02 | 348 | 50 | Mixed | High | Delayed Salvage |
| T07P03 | 44 | 40 | High | High | Delayed Salvage |
| T09P01 | 10 | 34 | Low | High | Immediate Salvage |
| T09P02 | 85 | 3 | Mixed | High | Immediate Salvage |
| T09P03 | 16 | 7 | High | High | Immediate Salvage |

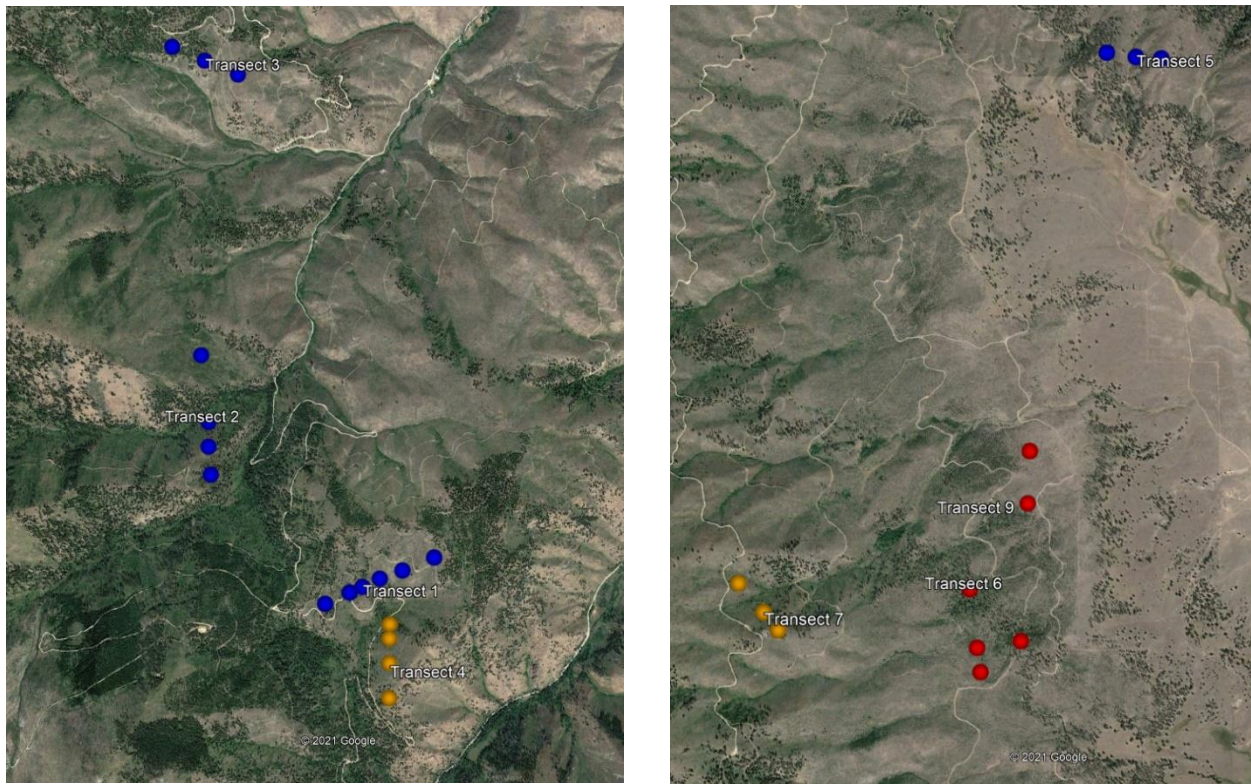


Figure 3. Maps of Laird creek (left) and French basin (right) study areas with transects marked with their corresponding salvage treatment. Blue indicates no salvage, orange delayed salvage, and red immediate salvage. (Retrieved from Google Earth, 4/8/2021).

Data Collection

Transects were approximately one kilometer in length and were located across areas where fire had created a mosaic of fire behavior. Transects had a minimum of three nested plots established along their length, with some transects having up to six nested plots. The fire severity of the area within the plots were classified in 2001 based on individual tree crown scorch and overstory tree mortality within the plot area (Table 2).

Table 2. Fire severity classes for overstory trees

| Fire Severity | Observed Individual Tree Effects (2001) |
|----------------------|--|
| Low | <50% Individual tree crown scorch <20% stand overstory tree mortality |
| Mixed | >50% Individual tree crown scorch 20-70% stand overstory mortality |
| High | >70% Stand overstory mortality |

In the summer of 2020, plot centers were located using GPS data points and photo reference points that had been established in 2001. Four photos were taken from each plot center, with one photo in each cardinal direction. Photos taken in 2001 and 2003 were used to locate plot centers and to observe changes in the surrounding area over time. A combination of a handheld recreation-grade GPS and the mobile map application Avenza were used to locate plot center. While plot centers had been temporarily monumented in 2001, these were not present in 2020.

Current live overstory tree data were recorded for a 37.2 ft radius, 1/10th acre (400-m²) circular plot (Figure 4). Overstory trees in 2020 were classified as living trees that were 6 ft tall and greater and were at least 3 inches DBH or greater. The species, an ocular estimate of diameter at breast height to the nearest inch (DBH), an ocular estimate of height in feet to the nearest 10 feet, and any types of observed defect for each tree were recorded for each stem with its pith (center) within the circular plot. Seedling counts were collected within the same 37.2 ft radius (400-m²) circular plot. Seedlings in 2020 were classified as being less than 6-feet tall and less than 3 inches DBH and were tallied by species.

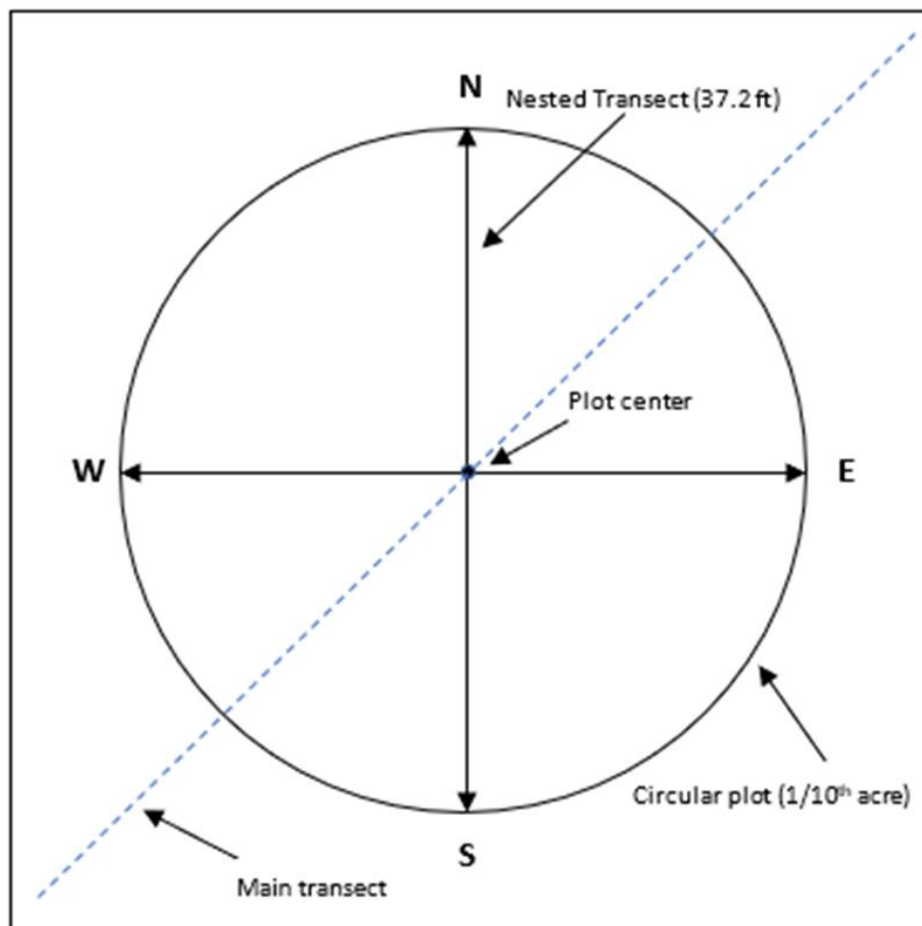


Figure 4. Diagram of plot measurements.

Each plot had four nested (37.2 ft) transects arranged in cardinal directions across the plot center (Figure 4). Vegetation data was collected along the length of each 37.2 ft nested transect within each plot. The goal of the data collection along the length of each transect was to record the length, overall percent vegetation cover, and percent individual species cover of each vegetation patch bisected by the transect. A vegetation patch was defined as a combination of understory species to include graminoids, forbs, mosses, and shrubs of a uniform density and composition. During the study conducted during 2001, direct fire effects on soil and vegetation cover were used to differentiate and identify patches. During the 2003 study, it was more difficult to differentiate these patches due to growth over time and greater reliance was placed on identifying understory vegetation cover and the presence of bare mineral soil (Hollingsworth, 2005). The data collection conventions of the 2003 remeasurements were carried over to the 2020 study.

The presence of bare ground for a minimum distance of 1 foot, a change in overall percent cover of at least 10 percent, or a change in species proportions of 10 percent would indicate the start of a new patch. The length in feet measured with a cloth tape, percent composition of individual species to the nearest 5 percent and overall percent cover of each patch to the nearest 5 percent were visually estimated and recorded along the length of each nested transect (Figure 4).

Data Analysis.

Data recorded in 2001, 2003, and 2020 was compiled in order to make statistical inferences between each of the study years. Environmental variables recorded and assigned in 2001 and 2003 to each of the nested plots were carried over to the 2020 data. These variables

include the following: aspect, slope, habitat type group, position on slope, vertical slope shape, horizontal slope shape, fire group, pre-fire (2000) percent estimated tree canopy cover, post-fire (2001) estimated percent tree canopy cover, stand crown burn severity, understory burn severity, and salvage type.

Aspect is defined as the compass direction that a slope is facing. Slope is defined as the rise or fall of the land surface and is measured as a percentage. Slope in the study plots ranged from 0 percent to 60 percent. Habitat type group is defined as the potential climax vegetation that can be supported on the landscape. The study area had two Douglas-fir habitat type groups: the warm/dry group, and the cool/dry group. Position on slope refers to the location of the plot center on the hillside. Plot centers were classified as either being on the bottom, middle, or top of the slope. Vertical and horizontal slope shape refers to the profile of the slope in reference to the plot center. This was classified as linear, convex, concave, or undulating. Fire group is defined as the grouping of habitat and community types as used by the Forest Service (Crane and Fischer 1986, Fischer and Bradley 1987). There are two fire groups in the study area: fire group four and six. Fire group four is described as a warm/dry Douglas-fir dominated habitat, and fire group six is described as a moist Douglas-fir habitat (Fischer and Bradley 1987). Pre- and post- fire estimated percent tree canopy cover refers to the estimated percent canopy cover of overstory trees in the plot area and were estimated in 2001.

Field data from 2001, 2003, and 2020 was manually entered into Microsoft Excel. These Excel databases were then read into R Studio in order to conduct statistical analyses.

Dependent variables were stratified by environmental variables in various ways utilizing tables and charts to show differences over the study years. These dependent variables include

mean vegetation patch size (average vegetation patch length in meters) and mean percent understory cover. Mean percent understory cover represents the average total percent cover of vegetation patches for each nested transect. Mean percent cover of individual species was also calculated to represent the average percent cover of understory vegetation for each nested transect. This was calculated by multiplying each vegetation patch length by the individual species cover for that patch. These values were summed and then divided by the total nested transect length and multiplied by 100.

Statistical analysis to test hypotheses were conducted in R Studio (v. 1.2.5033). Non-standard libraries used during analysis include doBY and MASS. Negative binomial models were used to explore the significance of variables for the presence of live seedlings and overstory tree counts in 2020. One-way ANOVA and Tukey HSD tests were used to determine the differences in the level of significance for dependent variables for mean patch size, mean percent cover, and individual species percent cover.

In order to investigate the significance of explanatory variables on the response using generalized linear modeling, the following procedure was used. The distribution of the response variables was investigated by using a paired panel R script. The distribution of the response and explanatory variables for overstory tree and seedling models can be found in Appendix A. After determining that the distributional assumptions of multiple linear regressions did not hold for the count data of live overstory trees and seedlings, a generalized linear model was chosen. Considering that the response variable is a positive count, a Poisson distribution and log link were used for each of the models. Model building procedures began with a fully saturated, additive model, and based on results from the summary tables of the fitted models, refinements were made by dropping explanatory terms that were shown to no longer be significant until a

final model was reached. Overdispersion for both models was investigated by calculating and examining the Pearson residuals. After determining that all models built here had Pearson values that exceeded one and clearly displayed overdispersion, a negative binomial distribution model fit was then used. Final models were created, and the model coefficients were evaluated to determine which explanatory variables had the most influence on the dependent variable.

Results

Overstory tree and seedling responses to environmental variables over time.

Overstory tree counts recorded from each of the study years were totaled and consolidated (Table 3) . Total tree counts for each of the years consisted of trees that were greater than 3 inches DBH and greater than 6 feet tall. These counted trees consisted of Douglas-fir and ponderosa pine. Total tree counts were then divided into three categories: dead, damaged, and live. Trees were considered dead if they displayed no live crown, trees were considered damaged if they had some live crown but had visible scorch or bole marks, and trees were considered live if they had a live crown and little to no visible damage. The overall trends in overstory trees per study for each study year can be seen in Appendix B (Figures B9-B14). The overall change in live overstory trees between each of the study years was significant, with 31 trees (24%) transitioning from live and growing in 2001 to either dead or damaged in 2003. Between 2003 and 2020, 45 (46%) live and growing trees either transitioned to damaged or down woody debris. The change from damaged to dead between the study years was also significant, with almost all (99%) of the counted damaged trees in 2001 (261) transitioning to dead in 2003. All of the counted dead trees in 2003 were no longer standing in 2020 and had transitioned to dead and down woody debris. Tree counts are also broken down by species. Table

3 shows the individual differences for ponderosa pine and Douglas-fir tree counts between the study years. While the 2001 data shows that there were more Douglas-fir present in the sites as compared to ponderosa pine, that difference is no longer present in 2020, with each species having almost the same total count. It is also interesting to note the continued die-off of mature Douglas-fir trees from 2003 to 2020, with 47 trees (70%) no longer being classified as live within the study area (Figures B12-B14). Out of twenty-nine sample plots, only seven plots (24%) had live Douglas-fir within the count areas which can be seen in Figure B14. In comparison, the number of live ponderosa pine trees increased slightly over that same period (Figures B9-B11).

Table 3. Overstory tree counts and Trees Per Acre for each study year by status and percent change over time for each classification and each year. Individual species totals are also shown.

| Mature Tree Summary Table | | | |
|----------------------------------|-------------------|-------------------|-------------------|
| Classification | Total 2001 | Total 2003 | Total 2020 |
| All | 477 | 412 | 80 |
| Dead | 88 | 313 | 0 |
| Damaged | 261 | 2 | 28 |
| Live | 128 | 97 | 52 |

| Ponderosa Pine Tree Summary Table | | | |
|--|-------------------|-------------------|-------------------|
| Classification | Total 2001 | Total 2003 | Total 2020 |
| All | 180 | 134 | 41 |
| Dead | 18 | 102 | 0 |
| Damaged | 131 | 2 | 9 |
| Live | 31 | 30 | 32 |

| Douglas-fir Tree Summary Table | | | |
|---------------------------------------|-------------------|-------------------|-------------------|
| Classification | Total 2001 | Total 2003 | Total 2020 |
| All | 297 | 278 | 39 |
| Dead | 70 | 211 | 0 |
| Damaged | 130 | 0 | 19 |
| Live | 67 | 67 | 20 |

| Ponderosa Pine Trees per acre | | | |
|--------------------------------------|-------------------|-------------------|-------------------|
| Classification | Total 2001 | Total 2003 | Total 2020 |
| All | 62 | 46 | 14 |
| Dead | 6 | 35 | 0 |
| Damaged | 45 | 1 | 3 |
| Live | 11 | 10 | 11 |

| Douglas-fir Trees per acre | | | |
|-----------------------------------|-------------------|-------------------|-------------------|
| Classification | Total 2001 | Total 2003 | Total 2020 |
| All | 102 | 96 | 13 |
| Dead | 24 | 73 | 0 |
| Damaged | 45 | 0 | 7 |
| Live | 33 | 23 | 7 |

Negative binomial models were fitted to these data in order to investigate the most significant explanatory variables that affected the number of live overstory trees in 2020. Models for the live presence of the two tree species, ponderosa pine and Douglas-fir, were fitted and the results were interpreted. Model results for the number of live ponderosa pine trees within the study area for 2020 indicate that an eastern facing aspect, a southeastern facing aspect, and a no salvage treatment option had significant influence. The final model fit, AIC value, and residual diagnostic plot of the model can be found in Appendix A. The model suggests that an eastern and southeastern facing aspect has a significant positive effect on the number of live trees as compared to the mean ($p = 0.27$, $p=0.29$), while a no salvage treatment option had a significantly negative effect on live tree count ($p = 0.0037$). Model results for the number of live Douglas-fir tree counts within the study area indicates that none of the of the variables had significant influence.

Seedling counts recorded from each of the study years were totaled and consolidated (Table 4). The overall trends in total seedling counts per species and plot by study year can be seen Appendix B (Figures B6-B8). The overall change in the number of live seedlings between each of the study years was significant, with Douglas-fir seedling numbers increasing by 1,241 (427 trees per acre) between 2001 and 2003 (Figures B6 and B7). Ponderosa pine seedling numbers also increased, although this may be attributed to seedling plantings that occurred in

some of the study transects. Douglas-fir seedlings from 2003 to 2020 decreased by 600 (206 trees per acre) while the number of ponderosa pine seedlings increased by 148 (51 trees per acre) (Figures B7 and B8).

Table 4. Live tree seedlings counts for each study year for ponderosa pine and Douglas-fir.

| Seedling Count Summary Table | | | |
|------------------------------|------------|------------|------------|
| Species | Total 2001 | Total 2003 | Total 2020 |
| Ponderosa pine | 11 | 68 | 216 |
| Douglas-fir | 37 | 1278 | 678 |

| Ponderosa Pine Seedling Count by salvage type | | | |
|---|------------|------------|------------|
| Salvage Type | Total 2001 | Total 2003 | Total 2020 |
| No salvage | 0 | 8 | 95 |
| Immediate Salvage | 0 | 25 | 64 |
| Delayed Salvage | 11 | 35 | 57 |

| Douglas-fir Seedling count by salvage type | | | |
|--|------------|------------|------------|
| Salvage Type | Total 2001 | Total 2003 | Total 2020 |
| No salvage | 1 | 489 | 325 |
| Immediate Salvage | 13 | 736 | 306 |
| Delayed Salvage | 23 | 53 | 47 |

| Ponderosa Pine Seedling count by understory severity | | | |
|--|------------|------------|------------|
| Understory Severity | Total 2001 | Total 2003 | Total 2020 |
| Low | 0 | 4 | 3 |
| Mixed | 11 | 7 | 64 |
| High | 0 | 57 | 149 |

| Douglas-fir Seedling count by understory severity | | | |
|---|------------|------------|------------|
| Understory Severity | Total 2001 | Total 2003 | Total 2020 |
| Low | 9 | 235 | 90 |
| Mixed | 28 | 980 | 515 |
| High | 0 | 63 | 73 |

| Ponderosa Pine Seedling count by overstory severity | | | |
|---|------------|------------|------------|
| Overstory Severity | Total 2001 | Total 2003 | Total 2020 |
| Low | 0 | 18 | 12 |
| Mixed | 11 | 33 | 87 |
| High | 0 | 17 | 117 |

| Douglas-fir Seedling count by overstory severity | | | |
|--|------------|------------|------------|
| Overstory Severity | Total 2001 | Total 2003 | Total 2020 |
| Low | 25 | 1062 | 341 |
| Mixed | 12 | 198 | 292 |
| High | 0 | 18 | 45 |

| Live TPA of Overstory and Understory Tree Species per year | | | | |
|--|---------|------|------|------|
| | Species | 2001 | 2003 | 2020 |
| Overstory TPA | DF | 33 | 23 | 7 |
| | PP | 11 | 10 | 11 |
| Seedling TPA | DF | 13 | 440 | 233 |
| | PP | 4 | 23 | 74 |

Stratifying each species' total seedling count by salvage type, understory severity, and overstory severity shows the differences in seedling counts between the levels of each variable over time (Table 4). For ponderosa pine seedlings, a no salvage and immediate salvage treatment resulted in higher seedling counts from 2001 to 2020 as compared to delayed salvage. A no salvage treatment had the highest number overall, with the greatest increase occurring between 2003 and 2020. A high and mixed understory severity had the highest number of counted ponderosa pine seedlings over time, with high understory severity having the largest number of ponderosa pine seedlings overall. Mixed and high overstory severity had the highest number of counted ponderosa pine seedlings over time, with high severity plots having the largest number of seedlings by 2020.

Douglas-fir seedlings stratified by salvage type displayed a similar trend as ponderosa pine seedlings with a no salvage and immediate salvage treatment having the highest seedling counts by 2020. It should be noted that in 2003, an immediate salvage treatment had higher seedling counts as compared to a no salvage treatment but by 2020, the opposite was true. Mixed understory severity had substantially higher counts over each of the study years as compared to low and high severity. Low severity areas did have higher counts in 2003, but this was no longer the case in 2020. Douglas-fir seedlings stratified by overstory severity showed that low and mixed severity had the highest counts over time. In 2003, there were substantially more Douglas-fir seedlings in low overstory severity areas overall, but by 2020 this difference had become less dramatic.

Model results for live ponderosa pine seedlings within the study year for 2020 indicate that an eastern facing aspect and a no salvage treatment were a significant influence. The final model fit, AIC value, and residual diagnostic plot of the model can be found in Appendix A. Model results suggest that an eastern facing aspect had a significantly positive effect on the number of live seedlings ($p = 0.028$) as compared to the mean, while a no salvage treatment option had a slight negative effect ($p = 0.05$). Model results for live Douglas-fir within the study area indicate that aspect, overstory fire severity, and understory fire severity had a significant influence. The final model fit, AIC value, and residual diagnostic plot of the model can be found in Appendix A. Model coefficients suggest that different aspect facings had negative effects on the number of seedlings, with a northeast, south, west, and northwest facing being the most significant ($p = 0.027$, $p = 0.003$, $p = 0.002$, $p = 0.004$). A southeastern facing aspect is suggested to have the greatest negative effect on live seedling numbers with a p-value of $2.55e-05$. Fire severity was also significant, with a high overstory severity having a significant positive effect ($p = 0.02$) and a high understory severity having a significant negative effect ($p = 0.003$). No significant correlation was found between aspect and fire severity in these data.

Mean Patch Size

Mean patch size refers to the average cross-sectional length of all patches intercepted in each nested transect. Comparisons of mean patch size were made across each of the study years stratified by independent variables and converted to meters (Table 5). In 2001, post fire tree cover, slope, and vertical slope shape were shown to be significant for mean patch size. In 2003, only post-fire tree cover and position on slope was shown to be significant (Hollingsworth 2005). In 2020, none of the variables tested were shown to be a significant influence for mean patch

size. The p-values from the one-way anova tests for 2020 were included in Table 4 to show the change in significance across each of the study years.

Table 5. Independent variables correlated with mean patch size for each study year.

| Variable | 2001 p | 2003 p | 2020 p |
|--|-------------------|-------------------|-------------------|
| Post-fire (2001) tree cover (%) | 0.001 | 0.001 | 0.669 |
| Slope (%) | 0.001 | - | 0.601 |
| Vertical slope shape | 0.046 | - | 0.623 |
| Position on slope | - | 0.021 | 0.697 |

Mean patch sizes stratified by each variable and category shows the shift to uniformity starting in 2003 and continuing through 2020 (Table 6).

Table 6. Mean patch length in meters stratified by variables for each study year. Numbers in bold and outlined in red were shown to be statistically significant ($p < 0.05$).

| Variable | Category | Number of observations | Mean Patch Size (m) | | |
|-------------------------------|-------------------|------------------------|---------------------|------------|------|
| | | | 2001 | 2003 | 2020 |
| Overstory severity | Low | 171 | 4.6 | 2.4 | 6.6 |
| | Mixed | 148 | 4.9 | 1.8 | 6.8 |
| | High | 212 | 7.6 | 1.8 | 6.5 |
| Understory severity | Low | 157 | 4.9 | 2.1 | 6.6 |
| | Mixed | 119 | 5.2 | 1.8 | 6.5 |
| | High | 255 | 8.5 | 1.8 | 6.7 |
| Habitat type group | Warm/dry | 383 | 6.1 | 1.8 | 6.5 |
| | Cool/dry | 148 | 5.2 | 2.1 | 6.6 |
| Aspect | North | 64 | 7.9 | 1.8 | 6.8 |
| | Northeast | 31 | 5.5 | 2.4 | 7.6 |
| | East | 151 | 4.9 | 1.8 | 6.7 |
| | Southeast | 118 | 4.6 | 1.8 | 6.4 |
| | South | 35 | 5.5 | 1.5 | 6.7 |
| | Southwest | 38 | 6.4 | 1.8 | 6.7 |
| | West | 58 | 6.4 | 1.5 | 6.1 |
| | Northwest | 36 | 6.1 | 1.8 | 6.6 |
| Slope (%) | 0-20 | 127 | 7.6 | 2.1 | 6.8 |
| | 21-40 | 229 | 6.1 | 2.1 | 6.6 |
| | 41-60 | 175 | 4.3 | 1.8 | 6.6 |
| Position on slope | Bottom | 110 | 5.2 | 2.1 | 6.8 |
| | Mid-slope | 222 | 4.9 | 1.8 | 6.5 |
| | Top | 199 | 7.3 | 1.8 | 6.6 |
| Vertical slope shape | Linear | 315 | 5.2 | 1.8 | 6.6 |
| | Convex | 136 | 8.5 | 1.8 | 6.4 |
| | Undulating | 80 | 4.9 | 2.1 | 6.9 |
| Horizontal slope shape | Linear | 91 | 4.6 | 2.1 | 6.3 |
| | Concave | 231 | 5.5 | 1.8 | 6.5 |
| | Convex | 209 | 6.7 | 1.8 | 6.8 |
| Firegroup | 4 | 106 | 5.8 | 2.1 | 6.7 |
| | 6 | 425 | 5.8 | 1.8 | 6.6 |
| Pre-fire (2000) tree cover % | 1-30% | 160 | 5.4 | 1.8 | 6.7 |
| | 31-45% | 179 | 6.3 | 2 | 6.6 |
| | >45% | 192 | 5.9 | 2.1 | 6.5 |
| Post-fire (2001) tree cover % | 0 | 237 | 7.1 | 1.6 | 6.5 |
| | 1-25% | 168 | 4.9 | 2.1 | 6.7 |
| | >25% | 126 | 4.3 | 2.5 | 6.7 |
| Salvage Type | Delayed Salvage | 135 | - | - | 6.5 |
| | Immediate Salvage | 102 | - | - | 6.9 |
| | No Salvage | 294 | - | - | 6.5 |

Understory Vegetation Cover

Understory vegetation cover refers to the mean percent cover of all understory species per transect. Comparisons of mean percent understory vegetation cover were made across each of the study years stratified by independent variables (Table 7). In 2001, post-fire tree cover, habitat type group, position on slope, and overstory severity were all shown to be significant for mean percent cover. In 2003, pre-fire tree cover, habitat type group, aspect, and fire group were shown to be significant (Hollingsworth 2005). In 2020, pre-fire tree cover, habitat type group, aspect, overstory severity, vertical slope shape, and horizontal slope shape were shown to have a significant influence on mean percent understory cover.

Table 7. Variables correlated with understory vegetation percent cover for each study year.

| Variable | 2001 | 2003 | 2020 |
|--------------------------------------|-------|-------|----------|
| | p | p | p |
| Pre-fire (2000) tree cover % | - | 0.008 | 3.39E-05 |
| Post fire (2001) tree cover % | 0.001 | - | - |
| Habitat type group | 0.015 | 0.001 | 1.44E-07 |
| Position on slope | 0.001 | - | - |
| Aspect | - | 0.01 | 3.00E-03 |
| Firegroup | - | 0.02 | - |
| Overstory severity | 0.008 | - | 0.0003 |
| Vertical slope shape | - | - | 0.0004 |
| Horizontal slope shape | - | - | 0.001 |

In 2020, there was a significant difference in mean percent understory cover for areas of high pre-fire tree cover (greater than 45%) and medium tree cover (31-45% , $p < 0.0001$). There was also a difference between high tree cover and low tree cover (1-30%, $p < 0.004$). Medium tree cover to low tree cover was not found to be statistically significant.

For habitat type group, a significant difference was shown for mean percent understory cover in each of the previous study years and 2020. Aspect also influenced mean percent understory cover in 2020, with the southeast aspect displaying the lowest overall mean percent understory cover. Overstory severity was influential in 2020, with the most significant difference in mean percent understory cover existing between high overstory severity and low overstory severity fire classifications ($p < 0.0001$). Vertical and horizontal slope shapes also were significant in 2020, with differences between linear and convex vertical slope shapes ($p < 0.01$), undulating and convex slope shapes ($p < 0.001$), and linear and concave horizontal slope shapes ($p < 0.001$) displaying the most significance for mean percent understory cover.

The summary table of mean percent understory cover shows the overall changes over the study years, with a significant increase in cover from 2001 to 2003 followed by an overall decrease in mean percent understory cover from 2003 to 2020 (Table 8).

| Category | Number of observations | Understory Cover (%) | | |
|-------------------|------------------------|----------------------|-----------|-----------|
| | | 2001 | 2003 | 2020 |
| Low | 171 | 39 | 41 | 22 |
| Mixed | 148 | 31 | 46 | 27 |
| High | 212 | 26 | 41 | 29 |
| Low | 157 | 39 | 45 | 26 |
| Mixed | 119 | 25 | 44 | 30 |
| High | 255 | 26 | 36 | 25 |
| Warm/dry | 383 | 28 | 38 | 24 |
| Cool/dry | 148 | 40 | 52 | 33 |
| North | 64 | 36 | 51 | 28 |
| Northeast | 31 | 36 | 45 | 30 |
| East | 151 | 38 | 51 | 29 |
| Southeast | 118 | 25 | 30 | 21 |
| South | 35 | 32 | 39 | 27 |
| Southwest | 38 | 40 | 26 | 30 |
| West | 58 | 14 | 35 | 23 |
| Northwest | 36 | 29 | 47 | 28 |
| 0-20 | 127 | 34 | 40 | 27 |
| 21-40 | 229 | 34 | 42 | 27 |
| 41-60 | 175 | 26 | 45 | 25 |
| Bottom | 110 | 45 | 41 | 28 |
| Mid-slope | 222 | 34 | 48 | 26 |
| Top | 199 | 22 | 37 | 25 |
| Linear | 315 | 31 | 42 | 27 |
| Convex | 136 | 24 | 36 | 21 |
| Undulating | 80 | 53 | 56 | 30 |
| Linear | 91 | 27 | 39 | 32 |
| Concave | 231 | 35 | 45 | 24 |
| Convex | 209 | 29 | 41 | 27 |
| 4 | 106 | 34 | 50 | 26 |
| 6 | 425 | 31 | 41 | 26 |
| 1-30% | 160 | 35 | 49 | 28 |
| 31-45% | 179 | 31 | 44 | 30 |
| >45% | 192 | 29 | 35 | 22 |
| 0 | 237 | 25 | 41 | 26 |
| 1-25% | 168 | 36 | 43 | 28 |
| >25% | 126 | 40 | 45 | 25 |
| Delayed Salvage | 135 | - | - | 25 |
| Immediate Salvage | 102 | - | - | 29 |
| No Salvage | 294 | - | - | 26 |

Table 8. Understory percent cover stratified by variables for each study year. Numbers in bold and outlined in red were shown to be statistically significant ($p < 0.05$).

Fire Severity effects on Major Understory Species

The effects of understory fire severity class was variable on mean percent cover of major understory species in 2020. Major understory species were those having greater than 5% cover in any of the severity classes in 2003 and are listed in the percent cover table (Table 9). While understory severity did not significantly influence overall mean percent vegetation cover in 2020, it did significantly influence cover of individual vegetation species, including ninebark (*Physocarpus malvaceus*), snowberry (*Symphoricarpos albus*), heartleaf arnica (*Arnica cordifolia*), spotted knapweed (*Centaurea micranthos*), and beargrass (*Xerophyllum tenax*) (Table 9) (Table 10).

Table 9. Percent cover of major understory vegetation species stratified by understory fire severity for each study year. A = accidental (species with mean cover <1% or species with only one occurrence)

| Lifeform | Species | 2001 Cover % | | | 2003 Cover % | | | 2020 Cover % | | |
|------------|--------------------------------|--------------|-------|------|--------------|-------|------|--------------|-------|------|
| | | Low | Mixed | High | Low | Mixed | High | Low | Mixed | High |
| Shrubs | <i>Arctostaphylox uva-ursi</i> | 6.0 | - | A | 7.4 | - | 2.1 | 5.6 | 1.2 | 2.4 |
| | <i>Linnaea borealis</i> | 10.8 | A | - | 6.4 | A | A | 2 | - | - |
| | <i>Physocarpus malvaceus</i> | 17.2 | 6.5 | - | 13.1 | 8.4 | - | - | 6.9 | 7 |
| | <i>Rubus parviflorus</i> | - | - | - | 9.6 | A | - | - | - | - |
| | <i>Spiraea betulifolia</i> | 5 | 6.6 | A | 3.9 | 7.7 | 3.4 | 2.2 | A | 1 |
| | <i>Symphoricarpos albus</i> | 4.2 | 6.1 | 3.5 | 4.5 | 8 | 4.8 | 4.8 | 17.2 | 7.9 |
| | <i>Vaccinium caespitosum</i> | - | - | - | 5.3 | - | 2.9 | - | - | - |
| | <i>Vaccinium globulare</i> | 7.3 | A | - | 6.3 | 3.1 | - | 1 | 1 | 1.1 |
| Forbs | <i>Arnica cordifolia</i> | 11 | 1.9 | 6.7 | 10.4 | A | 2.2 | 3.6 | 2.4 | A |
| | <i>Centaurea maculosa</i> | 14.6 | 5.5 | - | 6.8 | 7.8 | 3.5 | 9.1 | 3.3 | 6.8 |
| | <i>Cirsium arvense</i> | - | - | - | A | - | 9 | - | - | A |
| | <i>Epilobium angustifolium</i> | 14.3 | 10.2 | 15.9 | 6 | 6.8 | 7.3 | A | - | - |
| | <i>Erigeron spp.</i> | 21.1 | 10.3 | A | 13.5 | 9.8 | 3.7 | - | - | - |
| | <i>Verbascum thapsus</i> | - | - | - | 3.1 | - | 5.5 | - | - | - |
| | <i>Xerophyllum tenax</i> | 12 | 17.2 | 12.4 | 8.1 | 15.5 | 21.4 | 13.6 | A | 5 |
| Graminoids | <i>Calamagrostis rubescens</i> | 12.1 | 10.7 | 10.8 | 12.4 | 14.5 | 8.1 | 17.2 | 20.2 | 21.8 |
| Moss | Moss | - | - | - | 7.7 | 8.3 | 9.5 | - | - | A |

The differences in mean percent cover for each species between the levels of severity can be seen in Figure 5, with each graph representing a different species. The species that were

affected responded as expected due to their fire adaptations and no significant deviations from those trends were observed.

Table 10. List of understory species by common name that were significantly affected by understory severity in 2020 with corresponding p-values.

| Common Name | P-value |
|--------------------|----------------|
| ninebark | 0.02 |
| common snowberry | 1.42E-05 |
| heartleaf arnica | 0.05 |
| spotted knapweed | 0.05 |
| beargrass | 0.0008 |

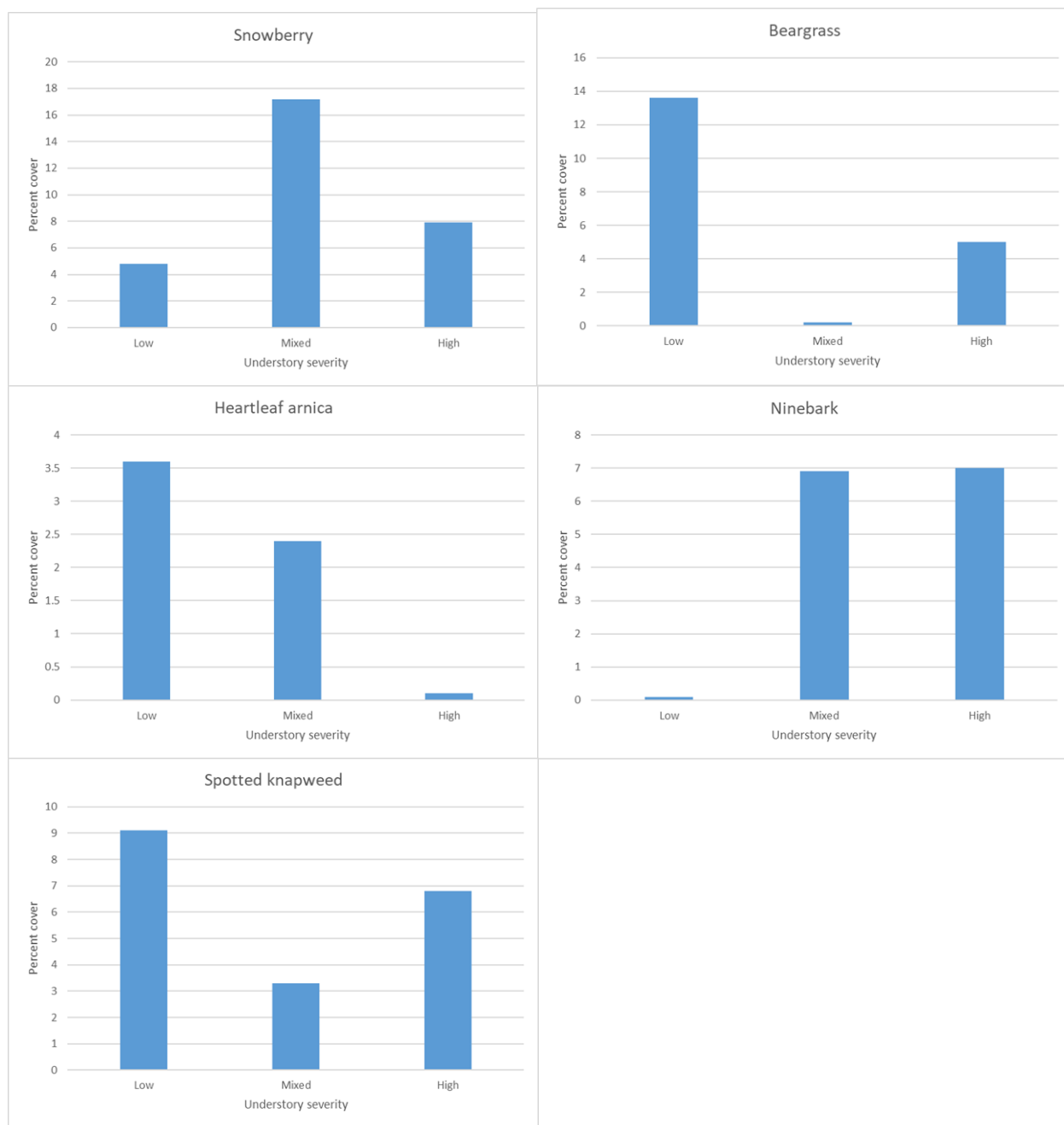


Figure 5. Mean percent cover of significantly affected understory species stratified by understory fire severity class.

The summary table of mean percent cover of major understory species for each study year stratified by understory severity class shows the overall changes in composition and the changes for each individual species by severity class. It is interesting to note the decrease in the

number of major vegetation species from 17 in 2003 to 10 in 2020. This can be seen in the absence of four forbs species and two shrub species from 2003 to 2020 (Table 9).

Overstory severity influenced overall mean percent understory vegetation cover for 2020 and it also had effects on individual mean percent cover of major species. Overall differences between the overstory fire severity classes are shown for each of the major species, with heartleaf arnica, spotted knapweed, and beargrass percent cover being influenced the most in 2020 (Table 11). Heartleaf arnica was influenced by overstory severity, with the differences between mixed and low severity ($p < 0.2$) and high and low severity ($p < 0.2$) being the greatest. Spotted knapweed was slightly influenced by overstory severity with the greatest difference existing between mixed and low severity ($p < 0.05$). Beargrass was significantly influenced by overstory severity, with the greatest difference existing between mixed and low severity ($p < 0.01$). The visual trend of these differences can be seen in Figure 6. Overall differences in percent cover for each of the major understory species stratified by overstory severity class are listed below (Table 12).

Table 11. Understory species significantly affected by overstory severity in 2020.

| Common Name | P-value |
|--------------------|----------------|
| heartleaf arnica | 0.01 |
| spotted knapweed | 0.06 |
| beargrass | 0.01 |

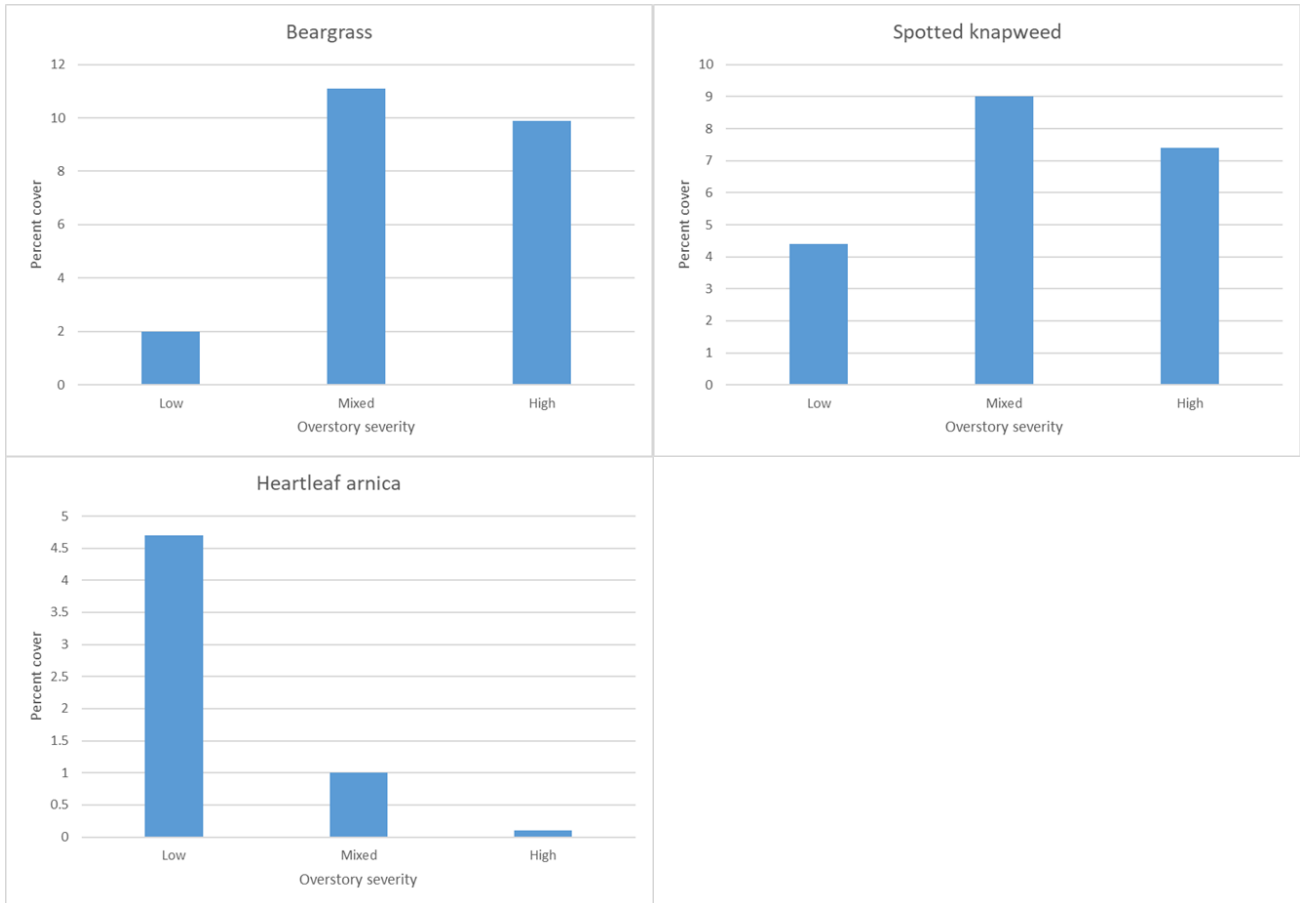


Figure 6. Mean percent cover of significantly affected understory species stratified by overstory severity class.

Table 12. Percent cover of major understory vegetation species stratified by overstory fire severity for 2020. A = accidental (species with mean cover <1% or species with only one occurrence)

| Lifeform | Species | 2020 Cover % | | |
|------------|--------------------------------|--------------|-------|-------|
| | | Low | Mixed | High |
| Shrubs | <i>Arctostaphylox uva-ursi</i> | 4.2 | 3.5 | 2.6 |
| | <i>Linnaea borealis</i> | 1.41 | 0.9 | - |
| | <i>Physocarpus malvaceus</i> | 2.8 | 2.2 | 8 |
| | <i>Rubus parviflorus</i> | - | - | - |
| | <i>Spiraea betulifolia</i> | A | 1.5 | 2.11 |
| | <i>Symphoricarpos albus</i> | 9.1 | 8.1 | 7.2 |
| | <i>Vaccinium caespitosum</i> | - | - | - |
| | <i>Vaccinium globulare</i> | A | 1.1 | 1.5 |
| Forbs | <i>Arnica cordifolia</i> | 4.7 | 1 | A |
| | <i>Centaurea maculosa</i> | 4.4 | 9 | 7.4 |
| | <i>Cirsium arvense</i> | - | A | - |
| | <i>Epilobium angustifolium</i> | - | A | - |
| | <i>Erigeron spp.</i> | - | - | - |
| | <i>Verbascum thapsus</i> | - | - | - |
| | <i>Xerophyllum tenax</i> | 2 | 11.1 | 9.9 |
| Graminoids | <i>Calamagrostis rubescens</i> | 15.9 | 21.3 | 21.16 |
| Moss | Moss | - | 0.1 | - |

Discussion

The most prominent changes for overstory tree counts for each of the study years are in the differences of dead and damaged trees between 2001 and 2003, and the differences in dead trees from 2003 to 2020. Almost all of the trees that were classified as damaged in 2001 had transitioned to standing dead by 2003, and by 2020 all of the standing dead trees within the plot areas had transitioned to down woody debris. The time it took to transition from standing dead to down woody debris is consistent with findings from other studies, with most disturbance effected areas taking between 10 and 12 years to lose half of their standing dead trees (Passovoy and Fule 2006). Another study concluded that salvaged logged areas would generally have fewer large, standing dead trees as compared to non-salvaged areas, and that dead tree persistence over time was shorter for salvaged areas (Russel et al, 2006). This would be expected given the nature of salvage logging, especially in the French Basin study area given the DNRC's mandate to manage the land for revenue purposes. Still, it is crucial to note that standing dead trees were observed in the study areas outside of the plots, indicating that some dead trees can persist well over 20 years post fire. Comparing site photographs shows the difference in standing dead numbers observed outside the study plot areas between the Larid Creek area and the French Basin area, with more standing dead being present in the Larid Creek area (Figure 7).



Figure 7. Examples of observed standing dead trees outside of plot areas. The picture on the left is located in the Larid Creek area (transect 4) and underwent a delayed salvage. The picture on the right is located in the French Basina area (transect 9) and underwent an immediate salvage.

Model results for the presence of ponderosa pine in 2020 suggest that a southeast and eastern facing aspect had a positive influence on the number of live trees in the study area. Ponderosa pine is able to persist on warm/dry sites as compared to Douglas-fir (Howard, 2003), and this is reinforced by the southeast facing aspect result of the model. Ponderosa pine is rated as a very fire-resistant species due to its thick bark and self-pruning attributes and would be more suited to survive on the more fire-prone south facing aspects in the study area (Howard, 2003). A no salvage treatment option was shown to have a significant negative effect on the number of live ponderosa pine in the study areas. This result is most likely attributed to some ponderosa pine seedlings that had been planted in 2000 that had transitioned to maturity since then, although without detailed planting data it is difficult to say how many actually did. With such low counts of ponderosa pine stems across the study area, even a few seedlings that had transitioned to maturity in non-salvaged areas over time could influence the model results. However, this model output could also be a result of the small sample size of the study and that

there are more non-salvaged plots overall. Further research could be conducted in order to fine tune this model.

Model results for the presence of Douglas-fir indicated that none of the variables were influential on live overstory tree counts in 2020. While this may indicate that no single variable had a significant effect on the distribution of live trees in the study area, it is more likely that the large number of zero count plots and the small sample size of the study contributed to this result. Out of twenty-nine sample plots, only 6 (20%) had live Douglas-fir within the count areas. A larger study plot radius may yield better results in future studies.

The live seedling counts for each study year per species shows the significant changes over time in number and composition of seedlings within the study areas (Table 3). Of particular note is the large increase in Douglas-fir seedlings between 2001 and 2003 with an overall increase of 1,241 (564 trees per acre). This increase can be attributed to some plots where survivor overstory Douglas-fir trees prolifically reseeded the immediate area (Steinberg 2002). This is followed by a large decrease in seedlings from 2003 to 2020, with almost half of seedlings found in 2003 no longer being present in the plot areas in 2020. A possible explanation for this decrease could be from competition from more robust Douglas-fir seedlings or a lack of shade in the fire effected areas, although it is difficult to say without data between 2003 and 2020 (Steinberg 2002). The increase in ponderosa pine seedlings over time is significant as well, with the largest increase occurring over the 2003-2020 time period. Again, this could be attributed to replanting efforts that took place in the study areas, but natural regeneration should not be dismissed. Model output for the presence of live ponderosa pine seedlings in the study areas suggest that aspect had a positive influence and a no-salvage treatment option had significant negative influence. An eastern facing aspect was shown to be a positive influence on the number

of seedlings in the study area. This may be attributed to a few plots located in the Laird Creek study area that may have been planted after the 2003 study. Reference pictures from the area show no viable seed source in the plot and in the immediate surroundings and it is assumed that they were planted (Figure 8). Other planting activities in the French Basin study area may also have influenced the model result as well. A no salvage treatment was shown to have a slightly negative affect on the number of live seedlings in the study area. This could be attributed to the lack of planting done in these areas; however, it is more likely that the lack of surviving overstory seed sources could be a contributing factor as alluded to in the overstory tree model.



Figure 8. Western facing view of transect 1, plot 5 in the Laird Creek study area. Suspected planting areas of ponderosa pine seedlings such as this could have influenced the model results for number of live seedlings in the study area. Note the lack of immediate live overstory trees within the area that could have provided a seed source.

The presence of more seedlings in treated areas vs non-treated areas does not definitively indicate that salvage logging was the most significant contributing factor to live seedling numbers in these results. Rather, assumed planting after salvage logging could be the most influential factor with logged areas more likely to have been planted. Observed natural regeneration in the study areas were present but low, which could be attributed to the die-off of overstory ponderosa pine between study years. Without a seed source, areas that underwent high severity fire would most likely be unable to return to a ponderosa pine/Douglas-fir dominated system without outside assistance. It should be noted that the number of ponderosa pine seedlings that were assumed to be planted in the French basin study area was almost identical between the 2003 and 2020. This indicates that these sites are capable of supporting tree seedlings regardless of what kind of salvage logging occurred there.

Model results for live Douglas-fir seedlings in the study area indicate that aspect, overstory, and understory fire severity had significant influence. Certain aspects were shown to have negative effects on the number of seedlings in the area. These include a northeast, northwest, west, southeast, and south facing aspect. The variability of this result could be attributed to survivor overstory trees that reseeded areas prolifically between the study years on various aspects. An example of this could be seen in transect 9 plot 1, where surviving overstory Douglas-fir in and around the plot area heavily reseeded the area (Figure 9). One study indicated that basal area was positively correlated with seedling density in that larger diameter trees resulted in more seedlings of the same species within the immediate plot area (Page et al., 2001). Future modeling should include average basal density as an explanatory variable to investigate if this relationship occurred within these study areas.

Seedlings also require partial shade on warm/dry facing aspects which further points to the importance of overstory survivors in the plot areas (Steinberg 2002). Douglas-fir seedlings in the Northern Rockies seem to prefer the cool, moist north facing aspect yet can also persist if adequate shade is provided (Steinberg 2002). The model output seems to suggest this, however it is more likely that the large number of zero count plots mixed with a small number of higher count plots on certain aspects could have influenced this result.

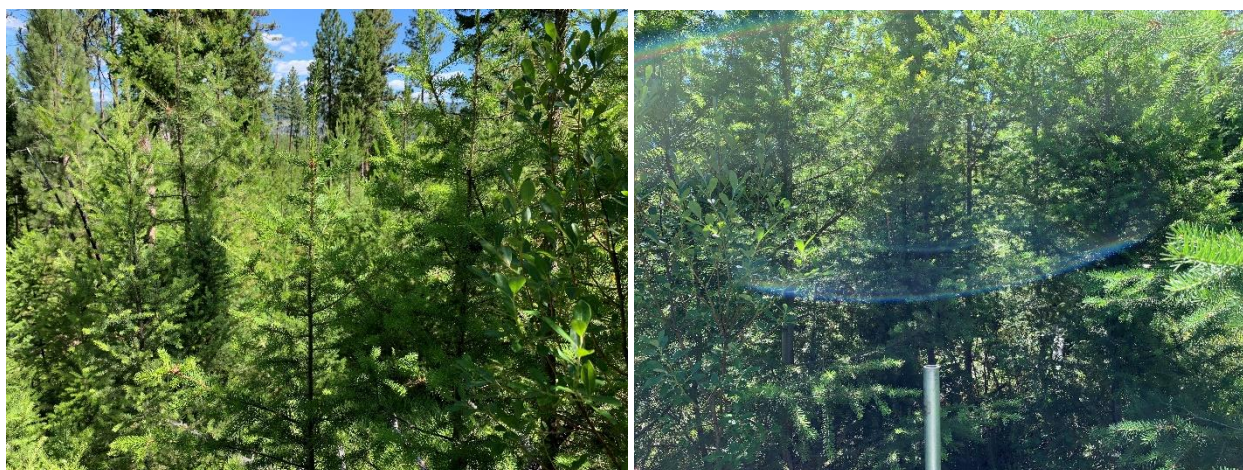


Figure 9. Northern and southern facing pictures of transect 9, plot 1 illustrates the ability of overstory Douglas-fir to prolifically reseed areas.

Fire severity was shown to have some influence on the number of live seedlings in the study area. In particular, high overstory fire severity was shown to have positive effect while high understory fire had a significantly negative effect. This may be attributed to plantings that took place on high overstory severity areas in both study areas, although natural regeneration from outside sources should not be discounted. Another possible explanation for this result is that areas that underwent high severity understory fire no longer had adequate shading to allow Douglas-fir seedlings to establish and persist (Steinberg 2002). Possible seed banks of Douglas-fir could also have been consumed during the fire in areas of high understory fire severity. This

result could also be attributed to the small sample size of the study and the small number of high seedling count plots encountered.

Mean understory vegetation patch size was not significantly affected by any variable in 2020. Previous studies conducted in the area pointed to the significance of post-fire tree cover, slope, and slope shapes on mean patch size. However, by 2020 these were no longer significant and overall mean patch size had become more uniform than previous study years. This result is consistent with the idea that successional pathways of understory species become less variable over time after disturbance (Armour 1984), and this can be seen in the decrease in variation through the study years. This result may point to a return to pre-fire understory patch size, although without pre-fire data it is difficult to say if that is what is occurring.

Mean understory percent vegetative cover was significantly influenced by a number of factors in 2020. These variables include pre-fire tree cover, habitat type group, aspect, overstory severity, vertical slope shape, and horizontal slope shape. In the case of pre-fire tree cover, 31-45% estimated overstory cover had the highest understory vegetation cover in 2020. A possible explanation for the significant differences in understory percent cover between the three classes of pre-fire tree cover could be the differences in fire severity between each of the cover classes (Arno et al. 2000). For example, an area with 31-45 % overstory cover may be more likely to have experienced a mixed severity fire which could result in a combination of high intensity and low intensity fire. This mosaic burn pattern could result in a broader range of understory survivor species within the area as compared to an area that only experiences a high intensity or low intensity fire. While most literature focuses on overstory response to different levels of severity,

it could be inferred that understory species may have a similar response and mixed severity areas could display more variety in species. More research should be conducted on this matter.

Habitat type group had a significant effect on understory percent cover, with the cool/dry group having significantly more understory vegetation cover than the warm/dry groups. The cool/dry habitat group generally has less evaporation and more available moisture which could contribute to more understory vegetation growth (Fischer et al. 1983). Aspect also had a significant effect on understory cover, with southern facing aspects generally having less overall cover than northern and eastern facing aspects. This can be attributed to more available sunlight on southern aspects which results in hotter, drier areas. Overstory severity had a significant effect on cover as well, with high severity areas having more overall understory cover than low and mixed severity areas. This can be attributed to the high overall percent cover of pinegrass (*Calamagrostis rubescens*) across all of the severity classes. Pine grass is known to be able to survive even the most severe wildfires and can sprout prolifically from its underground rhizomes (Matthews 2000). Vertical and horizontal slope shape also had an effect on vegetation cover, with an undulating vertical slope shape and a linear horizontal slope shape having the greatest percent cover overall. While there is little research done on how slope shape could affect vegetation cover, it is suspected that small variations in topography may give a small amount of cover that could allow varying levels of growth to occur and could also affect water retention.

Certain understory species were significantly affected by understory severity in 2020. The mean understory percent cover of ninebark was shown to be significantly affected by severity in that it was only present in mixed and high severity study areas. This species resprouts from underground rhizomes, surviving root crowns, and is known to be fire resistant (Habeck 1992). It also takes time to spread after disturbance and this is seen in the transition of the

species from low and mixed severity areas in 2001 and 2003 to mixed and high severity areas in 2020 (Habeck 1992).

Another species that was strongly influenced by severity was common snowberry with percent cover being significantly higher in mixed severity areas. The species is known to recolonize areas quickly post-fire that have experienced low to moderate soil disturbance (McWilliams 2000). This explains why areas of high understory fire severity do not display as much common snowberry cover due to higher soil consumption overall.

Heartleaf arnica was slightly affected by understory severity, with the greatest differences being between high and low severity areas. The species was mostly absent in high severity areas due to its small form that rarely persists above duff cover. This makes it highly susceptible to anything greater than a low severity fire, although its wind dispersed seeds allow it to easily resprout in such low severity areas or allow survivors to recolonize areas quickly as in the case of mixed-severity areas (Reed 1993).

Spotted knapweed was shown to be slightly affected by understory severity, with mixed severity sites having lower percent cover than low or high severity sites. Possible explanations for this are that the species has a perennial taproot that will survive most low severity fires which would allow it to repopulate quickly in the absence of other, more fire- susceptible species. In the study areas, it was noted that most instances of the species were in large, bare ground areas of the warm/dry habitat type areas, possibly indicating the effects of this survival mechanism (Zouhar 2001).

Beargrass was shown to be affected by understory severity with the most percent cover existing in areas of low severity. Previous study years indicated that the opposite was true, with

mixed and higher severity areas displaying more percent cover of the species. While the species can repopulate areas that have been cleared by recent disturbances, it is very sensitive to competition from other shrub species and will diminish over time (Crane 1990). It should be noted that most instances of the species occurred in the warm/dry habitat areas that generally had little to no overstory or shrub presence, which again affirms the species' preference for more open areas.

Overstory severity had some effects on the percent cover of certain species in 2020. Heartleaf arnica displayed the greatest percent cover in areas of low overstory severity, which again points to its susceptibility to higher severity fires. Spotted knapweed had the greatest percent cover in areas of mixed and high overstory fire which again could point to the species ability to reseed areas quickly from survivors or offsite colonizers. Beargrass had the greatest percent cover in areas of mixed and high severity, which could be attributed to some individuals that persisted in open areas of forest canopy on the site.

Conclusions

The effects of environmental variables, fire severity, and post-fire management on vegetation regeneration in the areas affected by the 2000 fires in the Southern Bitterroot valley were varied. The most influential environmental variable to affect vegetation regeneration for understory species and overstory species was aspect. Aspect was influential on the distribution of tree seedlings, overstory trees, and understory percent vegetative cover. Fire severity was also influential, with differences in overstory and understory severity having influence on the distribution, presence, and percent cover of vegetation species across the study areas. Areas of higher understory and overstory severity influenced the presence of certain fire-resistant species

while lower understory and overstory severity areas promoted more shade tolerant species. Overall, environmental variables and fire severity had the greatest long-term effects on vegetation regeneration, although as time continues to pass, these effects become less varied, especially in the understory. The most influential post-fire management activity was seedling planting. This resulted in a large increase of tree seedlings in areas that would most likely be unable to regenerate naturally. While natural regeneration was present on site, it was apparent that a majority of the study areas that had experienced high severity fire would be unable to naturally regenerate without outside influence.

These results indicate that without a targeted post-fire management plan, it will be difficult for severely affected ecosystems to return to their successional pathways over time. As more severe wildfires continue to persist on the landscape, it will be up to land managers to decide what comes back as overstory seed sources diminish. As the feedback loop of climate change and severe wildfires continue in the future, it will be extremely important to consider the implications of a delayed response when managing burned forested ecosystems.

There are some limitations of this study. The most prominent limitation is that the exact areas and times where tree seedling plantings occurred is unknown. While it was known that the DNRC study areas were planted in 2001, other study areas including some Forest Service plots displayed some evidence of plantings that occurred after 2003 and it was difficult to determine what could be considered natural regeneration or artificial. Having the planting data would significantly improve the model outputs and would help clarify the data overall.

Another limitation of this study is that in order to make comparisons across each of the study years, it was required that the study methods need to be kept uniform. However, these

methods could use some expansion. It is recommended that the overstory tree count plots are expanded in size beyond a 1/10th acre radius plot to gather further data on standing dead persistence and live tree seed sources in both study areas. It was also difficult to track some overstory tree counts and individual tree status through the study years as some of the data collected in the previous studies were unclear or too general in scope. Also, it would be useful to see if any rehabilitation work had been conducted in either study area between 2003 and 2020.

Further research should be conducted on the survival rate of planted seedlings in managed areas over time. Comparisons should also be made between natural seedling regeneration and planted seedling viability in burned areas over time. More research should be conducted on fire severity's long-term effects on understory vegetation as these ecosystems return to a form of equilibrium over time.

Overall, this research suggests that forested ecosystems that are affected by high severity wildfires will experience long-term changes to vegetation species composition and abundance. Areas that are managed immediately after fire with a clear goal in place could expect to meet their land management goals. Forested ecosystems that do not receive some form of post-fire management treatment could expect to take much longer to return to post-fire stand conditions or could possibly never return to that original state. Depending on agency goals or mandates, immediate action should take place in fire affected ecosystems if forest health and resiliency is to be maintained in the era of climate change.

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A18. T05P01

| | Distance | % Cover | Cov G | G | Cov F | F | Cov S | S |
|----|----------|---------|-------|------|-------|-------|-------|------|
| 1 | N | | | | | | | |
| 2 | 6 | 15 | 5 | CAGE | 5 | LUP | | |
| 3 | | | | | 5 | KNAP | | |
| 4 | 11 | 20 | 10 | CAGE | 5 | ACMI | | |
| 5 | | | | | 5 | LUP | | |
| 6 | | | | | 5 | KNAP | | |
| 7 | 14 | 0 | | | | | | |
| 8 | 16 | 20 | 15 | CAGE | 5 | KNAP | | |
| 9 | 20 | 5 | | | 5 | KNAP | | |
| 10 | 25 | 10 | 5 | CAGE | 5 | ACMI | | |
| 11 | 26 | 5 | | | 5 | KNAP | | |
| 12 | 30 | 5 | 5 | CAGE | | | | |
| 13 | 34 | 10 | | | 10 | ARNCA | | |
| 14 | 37 | 10 | 10 | CAGE | | | | |
| 15 | E | | | | | | | |
| 16 | 4 | 20 | | | 10 | ACMI | 5 | SYAL |
| 17 | | | | | 5 | HAWK | | |
| 18 | 7 | 10 | 5 | CAGE | | | 5 | SYAL |
| 19 | 14 | 10 | | | 5 | ACMI | | |
| 20 | | | | | 5 | KNAP | | |
| 21 | 34 | 40 | 25 | CARU | 5 | LUP | 5 | SYAL |
| 22 | | | | | 5 | KNAP | | |
| 23 | 37 | 10 | | | 5 | ACMI | | |
| 24 | | | | | 5 | KNAP | | |
| 25 | S | | | | | | | |
| 26 | 6 | 5 | 5 | CAGE | | | | |
| 27 | 10 | 10 | | | 5 | ACMI | | |
| 28 | | | | | 5 | KNAP | | |
| 29 | 19 | 15 | | | 10 | LUP | | |
| 30 | | | | | 5 | HOLLY | | |
| 31 | 29 | 15 | 10 | CARU | 5 | KNAP | | |
| 32 | 37 | 5 | | | 5 | HOLLY | | |
| 33 | W | | | | | | | |
| 34 | 9 | 10 | 5 | CAGE | 5 | LUP | | |
| 35 | 14 | 10 | 10 | CAGE | | | | |
| 36 | 18 | 5 | | | 5 | KNAP | | |
| 37 | 22 | 10 | 10 | CAGE | | | | |
| 38 | 37 | 20 | | | 10 | STAR | | |
| 39 | | | | | 5 | KNAP | | |
| 40 | | | | | 5 | HOLLY | | |

A30. Species key code table. (< in sheet indicates less than 5% cover).

| Lifeform | Species | Code (in sheet) |
|-----------------|--------------------------------|------------------------|
| Shrubs | <i>Arctostaphylox uva-ursi</i> | KNIC |
| | <i>Linnaea borealis</i> | LIBO |
| | <i>Physocarpus malvaceous</i> | NINE |
| | <i>Spiraea betulifolia</i> | SPIREA |
| | <i>Symphoricarpos albus</i> | SYAL |
| | <i>Vaccinium globulare</i> | VACCI |
| | | |
| Forbs | <i>Arnica cordifolia</i> | ARCO/ARNICA |
| | <i>Centaurea maculosa</i> | KNAP |
| | <i>Cirsium arvense</i> | THISTLE |
| | <i>Epilobium angustifolium</i> | FIREWEED |
| | <i>Xerophyllum tenax</i> | XETE |
| | | |
| Graminoids | <i>Calamagrostis rubescens</i> | CARU |
| | | |
| Moss | Moss | moss |

Appendix B

Table B1. Matrix plot of response and explanatory variables for seedling and overstory tree models for 2020. Distribution of variables are shown in the bottom left of the matrix plot. PP20 and DF20 represent tree seedling counts of each species for 2020. PP.over20 and DF.over20 represent overstory tree counts for each species for 2020.

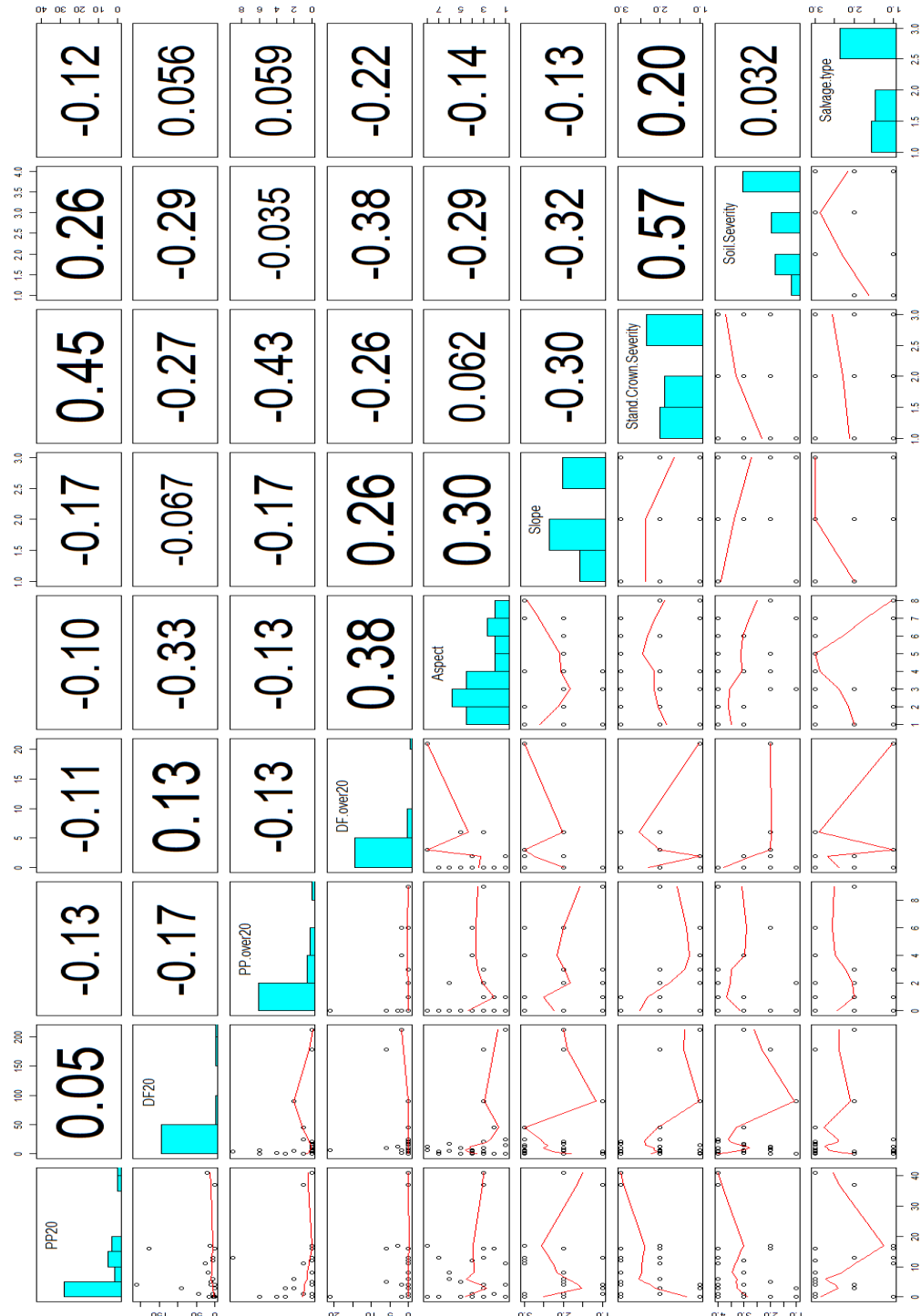


Table B2. Coefficient tables of ponderosa pine and Douglas-fir seedling models for 2020.

| ponderosa pine seedling model | | | | |
|--------------------------------------|-----------------|-----------------------|----------------|----------------|
| Coefficient | Estimate | Standard Error | z-value | p-value |
| Intercept | 2.39052 | 0.92746 | 2.577 | 0.00995 |
| Aspect NE | 0.03437 | 1.25666 | 0.027 | 0.97818 |
| Aspect E | 1.85111 | 0.84506 | 2.191 | <i>0.02849</i> |
| Aspect SE | 0.16289 | 1.0622 | 0.153 | 0.87812 |
| Aspect S | 0.23188 | 1.34093 | 0.173 | 0.86271 |
| Aspect SW | 1.2615 | 1.2921 | 0.976 | 0.3289 |
| Aspect W | -0.80554 | 1.17549 | -0.685 | 0.49317 |
| Aspect NW | -0.25046 | 1.31563 | -0.19 | 0.84902 |
| Immediate salvage | -1.38482 | 0.95965 | -1.443 | 0.14901 |
| No salvage | -1.70612 | 0.88412 | -1.93 | <i>0.05364</i> |
| AIC Value | 183.74 | | | |

| Douglas-fir seedling model | | | | |
|-----------------------------------|-----------------|-----------------------|----------------|-----------------|
| Coefficient | Estimate | Standard Error | z-value | p-value |
| Intercept | 5.365476 | 1.223282 | 4.386 | 1.15E-05 |
| Aspect NE | -2.43107 | 1.100663 | -2.209 | 0.02719 |
| Aspect E | -1.50977 | 0.84429 | -1.788 | 0.07374 |
| Aspect SE | -3.67561 | 0.873007 | -4.21 | <i>2.55E-05</i> |
| Aspect S | -3.57351 | 1.240993 | -2.88 | <i>0.00398</i> |
| Aspect SW | -1.85316 | 1.139706 | -1.626 | 0.10395 |
| Aspect W | -3.5596 | 1.169419 | -3.044 | <i>0.00234</i> |
| Aspect NW | -3.78914 | 1.330469 | -2.848 | <i>0.0044</i> |
| Mixed overstory severity | 1.171609 | 0.727191 | 1.611 | 0.10715 |
| High overstory severity | 1.954928 | 0.861554 | 2.269 | <i>0.02326</i> |
| Low understory severity | -0.00702 | 1.260726 | -0.006 | 0.99556 |
| Mixed understory severity | -0.91531 | 1.288121 | -0.711 | 0.47734 |
| High understory severity | -3.6187 | 1.25173 | -2.891 | <i>0.00384</i> |

| | |
|------------------|---------------|
| AIC Value | 213.19 |
|------------------|---------------|

Table B3. Coefficient table for overstory ponderosa pine trees in 2020.

| Overstory ponderosa pine model | | | | |
|---------------------------------------|-----------------|-----------------------|----------------|----------------|
| Coefficient | Estimate | Standard Error | z-value | p-value |
| Intercept | -1.06E+00 | 1.06E+00 | -0.999 | 0.31786 |
| Aspect NE | 1.15E+00 | 1.58E+00 | 0.728 | 0.46682 |
| Aspect E | 2.44E+00 | 1.11E+00 | 2.205 | 0.02747 |
| Aspect SE | 2.42E+00 | 1.11E+00 | 2.179 | 0.0293 |
| Aspect S | -3.28E+01 | 4.75E+07 | 0 | 1 |
| Aspect SW | 1.83E+00 | 1.41E+00 | 1.296 | 0.19484 |
| Aspect W | -3.41E+01 | 3.10E+07 | 0 | 1 |
| Aspect NW | -3.59E+01 | 4.75E+07 | 0 | 1 |
| Mixed overstory severity | -1.57E-01 | 5.27E-01 | -0.297 | 0.7662 |
| High overstory severity | -3.16E+00 | 1.09E+00 | -2.901 | 0.00372 |

| | |
|------------------|--------|
| AIC Value | 80.295 |
|------------------|--------|

Table B4. Transect and plot data.

| Transect # plot # | Aspect (degrees) | Slope (%) | Stand Crown Severity | Soil Severity | Salvage type |
|----------------------|---------------------|-----------|----------------------|---------------|-------------------|
| T01P01 | 68 | 32 | Mixed | Mixed | No Salvage |
| T01P02 | 200 | 39 | High | Mixed | No Salvage |
| T01P03 | 134 | 42 | High | High | No Salvage |
| T01P04 | 94 | 39 | High | High | No Salvage |
| T01P05 | 224 | 40 | High | High | No Salvage |
| T01P06 | 186 | 40 | High | High | No Salvage |
| T02P01 | 134 | 50 | Low | Mixed | No Salvage |
| T02P02 | 50 | 59 | Mixed | High | No Salvage |
| T02P03 | 360 | 65 | Low | High | No Salvage |
| T02P04 | 90 | 30 | High | High | No Salvage |
| T03P01 | 130 | 49 | Low | High | No Salvage |
| T03P02 | 136 | 35 | Mixed | High | No Salvage |
| T03P03 | 120 | 20 | High | High | No Salvage |
| T04P01 | 310 | 48 | Low | Mixed | Delayed Salvage |
| T04P02 | 296 | 48 | Mixed | Mixed | Delayed Salvage |
| T04P03 | 286 | 55 | High | High | Delayed Salvage |
| T04P04 | 250 | 55 | Low | Low | Delayed Salvage |
| T05P01 | 130 | 25 | Low | Mixed | No Salvage |
| T05P02 | 220 | 30 | Mixed | High | No Salvage |
| T05P03 | 290 | 30 | High | High | No Salvage |
| T06P01 | 81 | 2 | High | High | Immediate Salvage |
| T06P02 | 77 | 6 | Mixed | High | Immediate Salvage |
| T06P03 | 110 | 7.5 | Low | Low | Immediate Salvage |
| T07P01 | 72 | 27 | Low | High | Delayed Salvage |
| T07P02 | 348 | 50 | Mixed | High | Delayed Salvage |
| T07P03 | 44 | 40 | High | High | Delayed Salvage |
| T09P01 | 10 | 34 | Low | High | Immediate Salvage |
| T09P02 | 85 | 3 | Mixed | High | Immediate Salvage |
| T09P03 | 16 | 7 | High | High | Immediate Salvage |

Table B5. Total counts of tree seedlings in each study plot for each study year by transect and plot.

| Transect# Plot# | ponderosa pine seedlings 2001 | Douglas-fir seedlings 2001 | ponderosa pine seedlings 2003 | Douglas-fir seedlings 2003 | ponderosa pine seedlings 2020 | Douglas-fir seedlings 2020 |
|--------------------|-------------------------------------|----------------------------------|-------------------------------------|----------------------------------|-------------------------------------|----------------------------------|
| T01P01 | 0 | 0 | 1 | 47 | 16 | 178 |
| T01P02 | 0 | 0 | 1 | 4 | 5 | 10 |
| T01P03 | 0 | 0 | 0 | 0 | 12 | 5 |
| T01P04 | 0 | 0 | 0 | 0 | 41 | 21 |
| T01P05 | 0 | 0 | 0 | 0 | 8 | 18 |
| T01P06 | 0 | 0 | 0 | 0 | 0 | 0 |
| T02P01 | 0 | 1 | 0 | 32 | 0 | 6 |
| T02P02 | 0 | 0 | 0 | 99 | 1 | 45 |
| T02P03 | 0 | 0 | 1 | 2 | 0 | 15 |
| T02P04 | 0 | 0 | 0 | 0 | 4 | 10 |
| T03P01 | 0 | 0 | 2 | 292 | 0 | 2 |
| T03P02 | 0 | 0 | 1 | 5 | 1 | 6 |
| T03P03 | 0 | 0 | 1 | 5 | 1 | 4 |
| T04P01 | 0 | 11 | 0 | 23 | 0 | 6 |
| T04P02 | 11 | 12 | 1 | 4 | 17 | 12 |
| T04P03 | 0 | 0 | 0 | 3 | 13 | 4 |
| T04P04 | 0 | 0 | 0 | 0 | 0 | 0 |
| T05P01 | 0 | 0 | 0 | 0 | 0 | 0 |
| T05P02 | 0 | 0 | 1 | 3 | 6 | 5 |
| T05P03 | 0 | 0 | 0 | 0 | 0 | 0 |
| T06P01 | 0 | 0 | 1 | 0 | 37 | 0 |
| T06P02 | 0 | 0 | 12 | 29 | 13 | 4 |
| T06P03 | 0 | 9 | 4 | 235 | 3 | 90 |
| T07P01 | 0 | 0 | 10 | 2 | 0 | 0 |
| T07P02 | 0 | 0 | 10 | 16 | 11 | 25 |
| T07P03 | 0 | 0 | 14 | 5 | 16 | 0 |
| T09P01 | 0 | 4 | 0 | 472 | 4 | 212 |
| T09P02 | 0 | 0 | 8 | 0 | 3 | 0 |
| T09P03 | 0 | 0 | 0 | 0 | 4 | 0 |

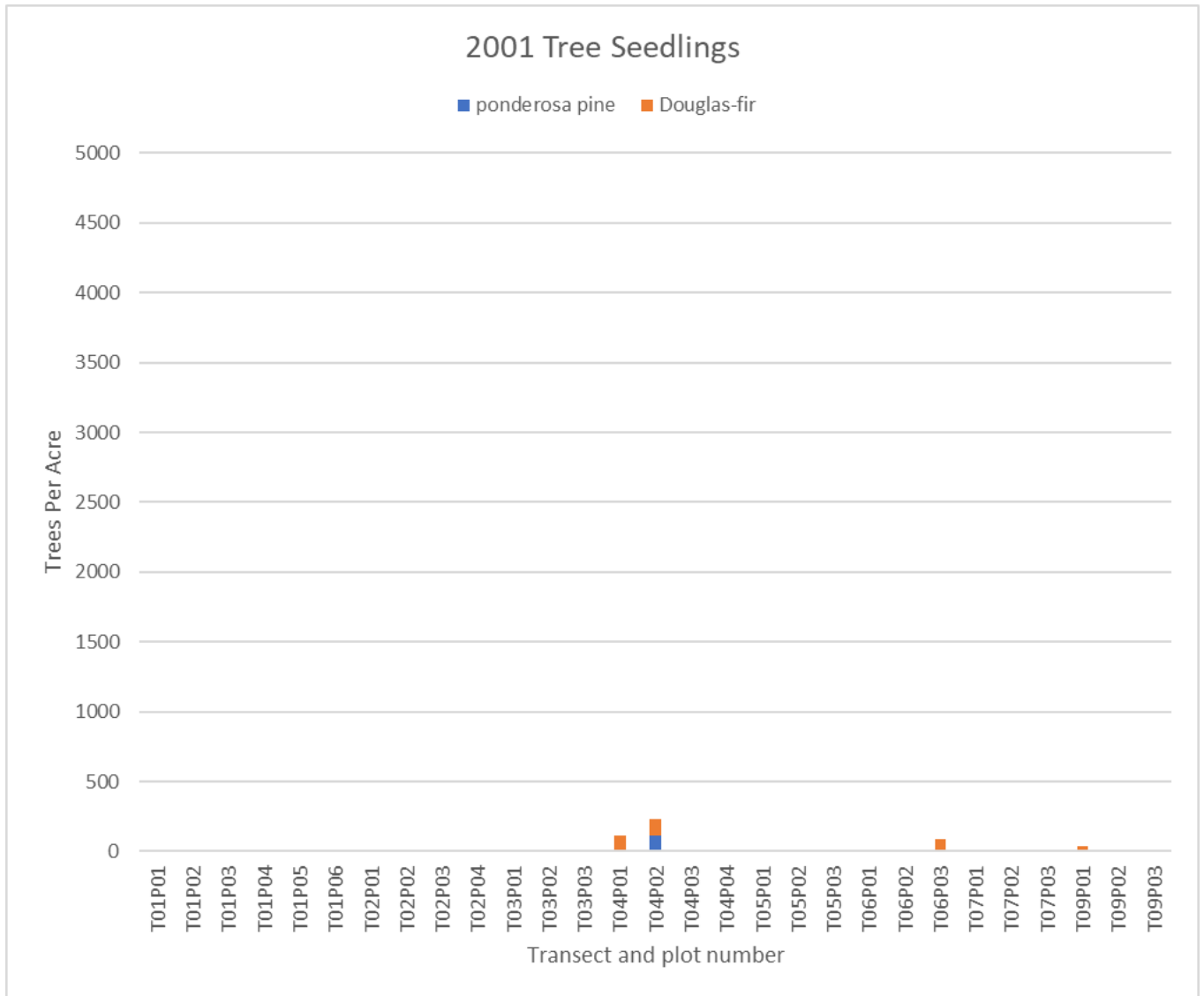


Figure B6. Number of tree seedlings per acre by species per study plot for 2001.

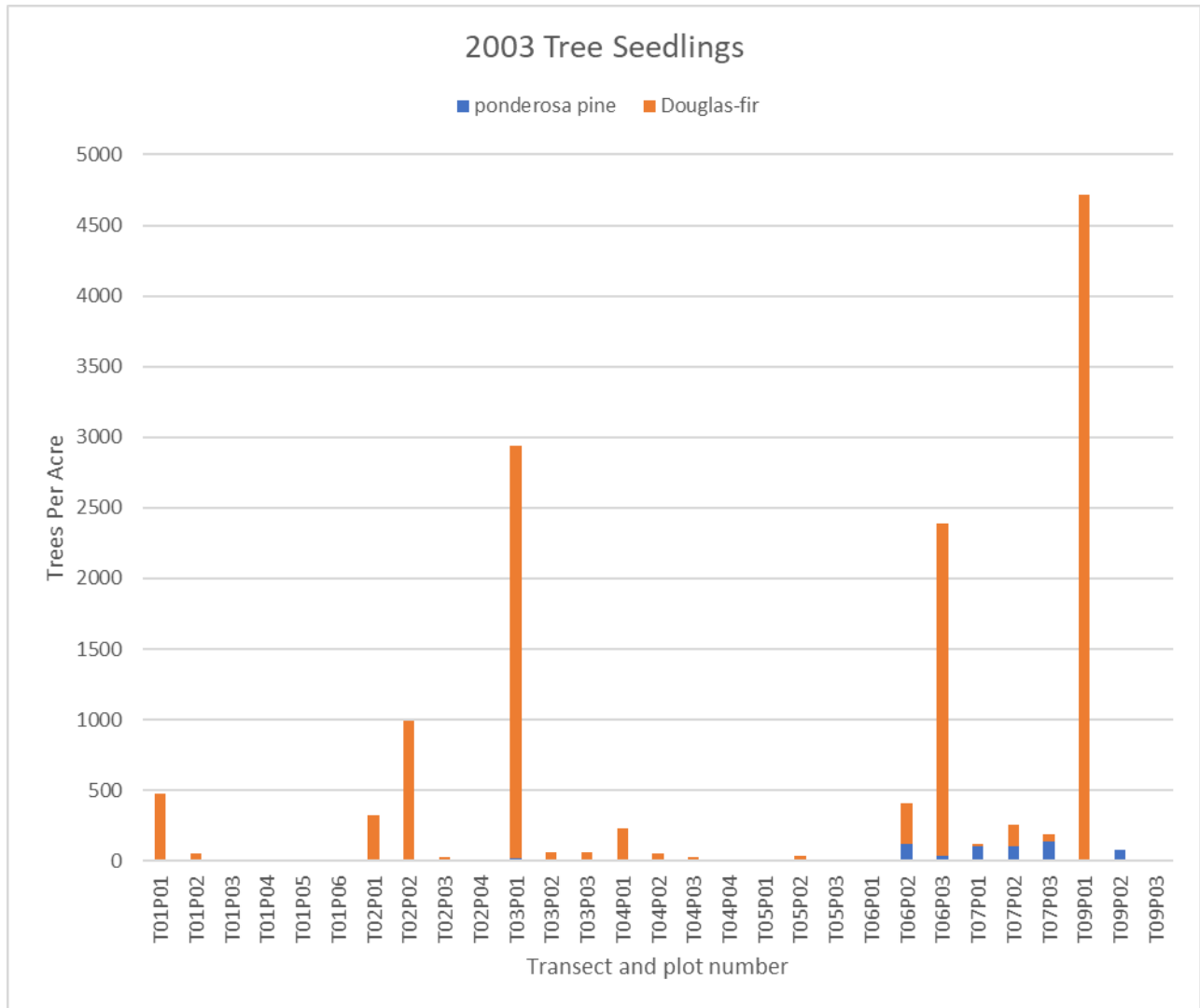


Figure B7. Number of tree seedlings per acre by species per study plot for 2003.

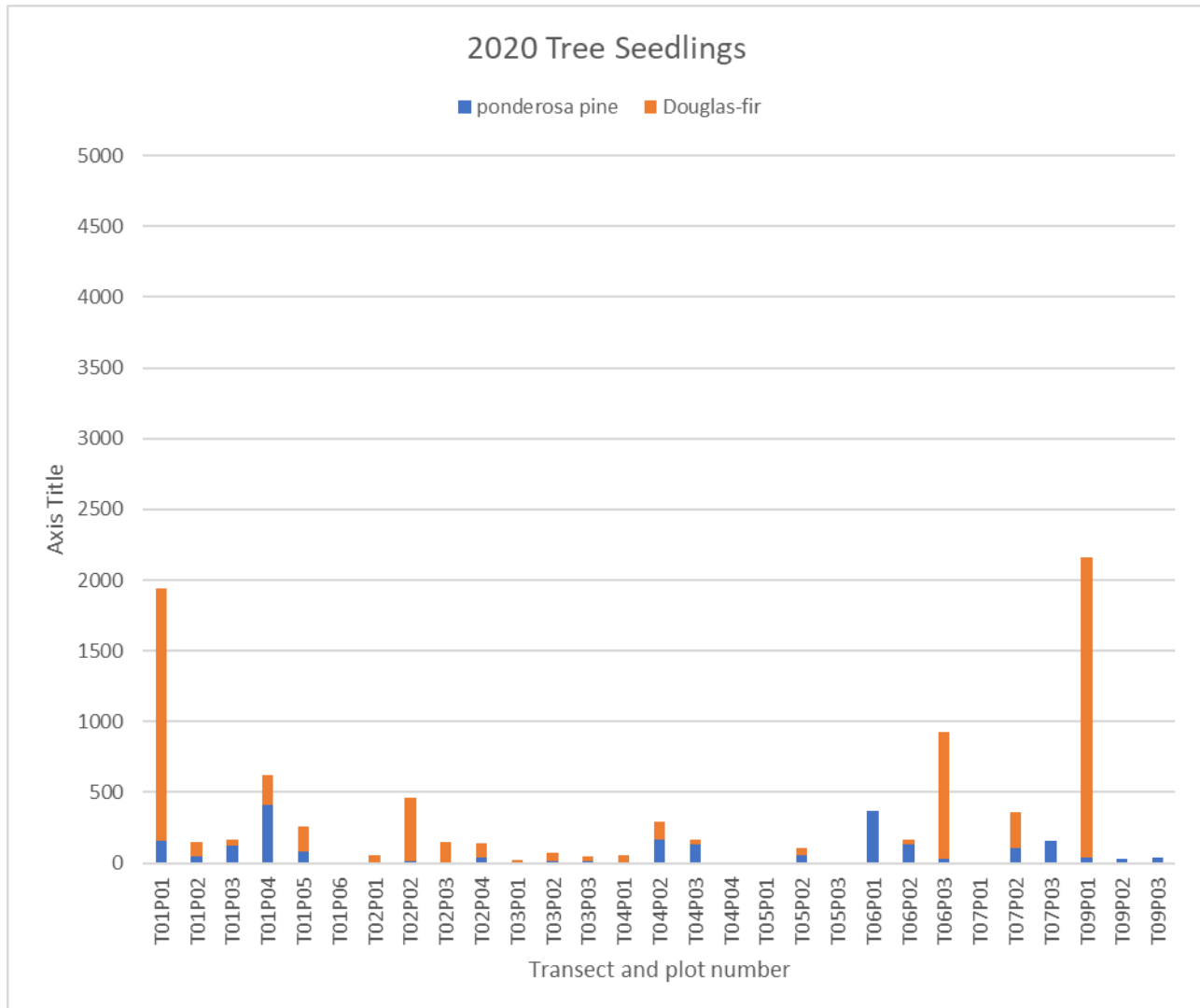


Figure B8. Number of tree seedlings per acre by species per study plot for 2020.

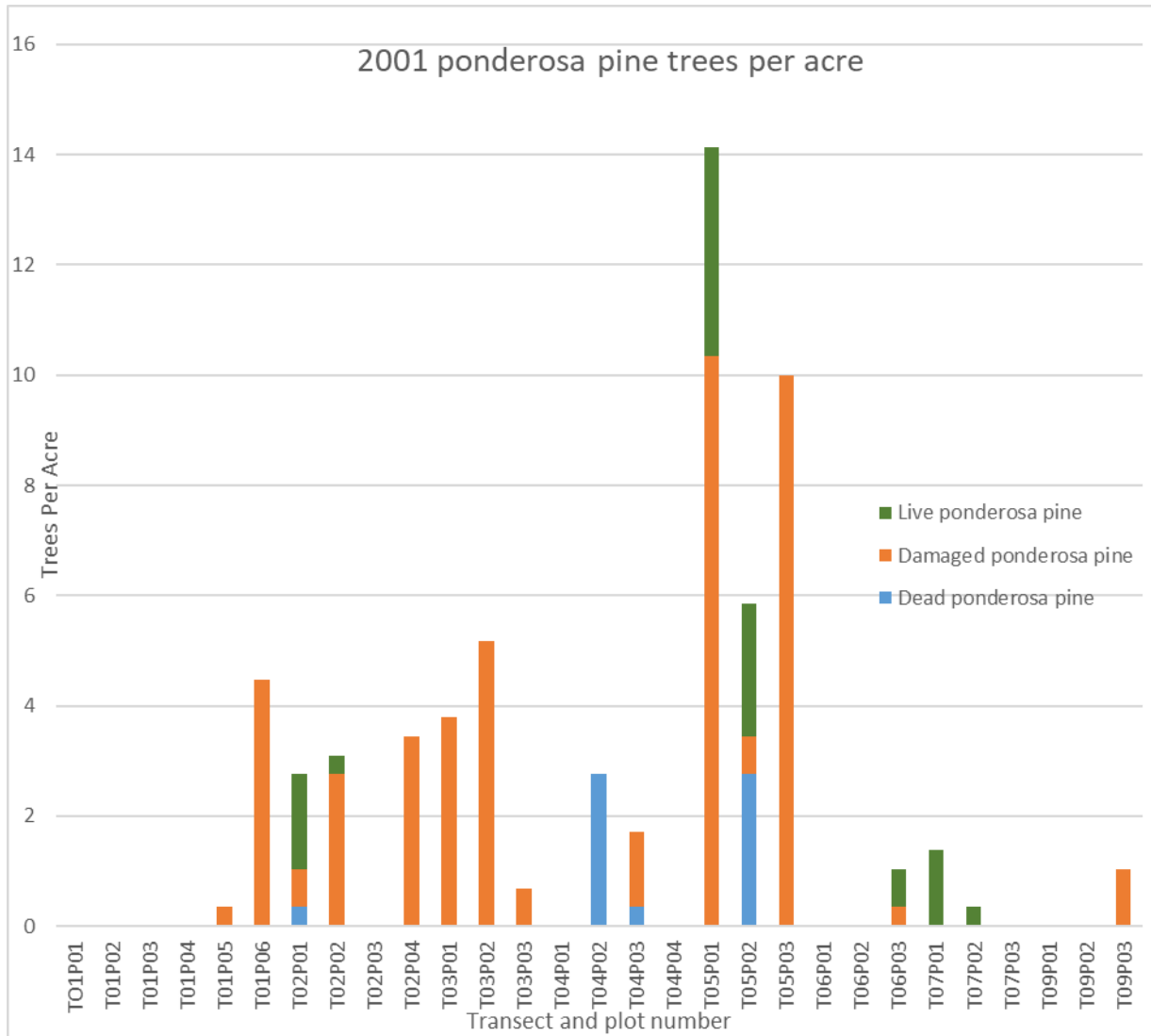


Figure B9. Number of ponderosa pine trees per acre per study plot for 2001.

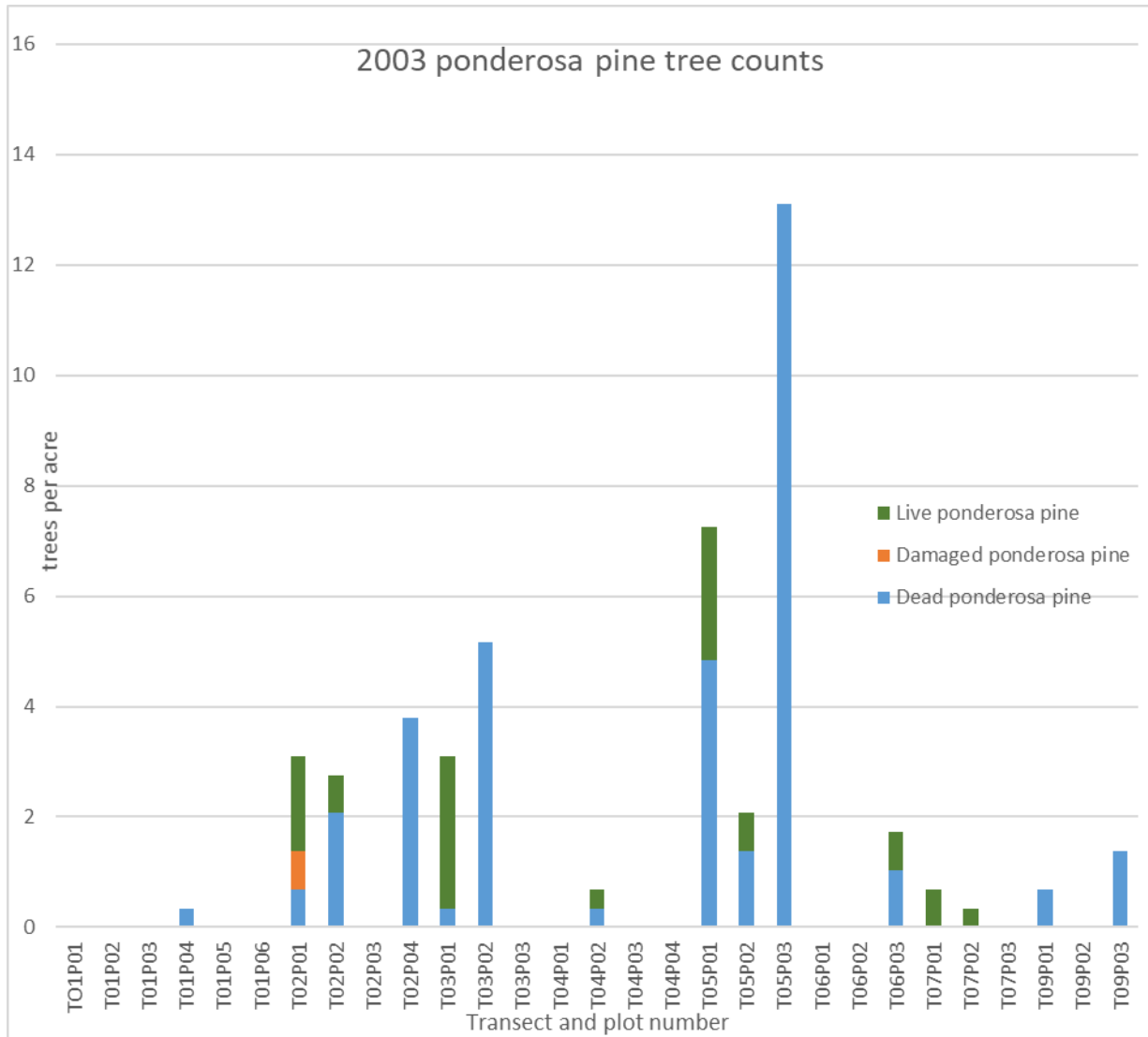


Figure B10. Number of ponderosa pine trees per acre per study plot for 2003.



Figure B11. Number of ponderosa pine trees per acre per study plot for 2020.

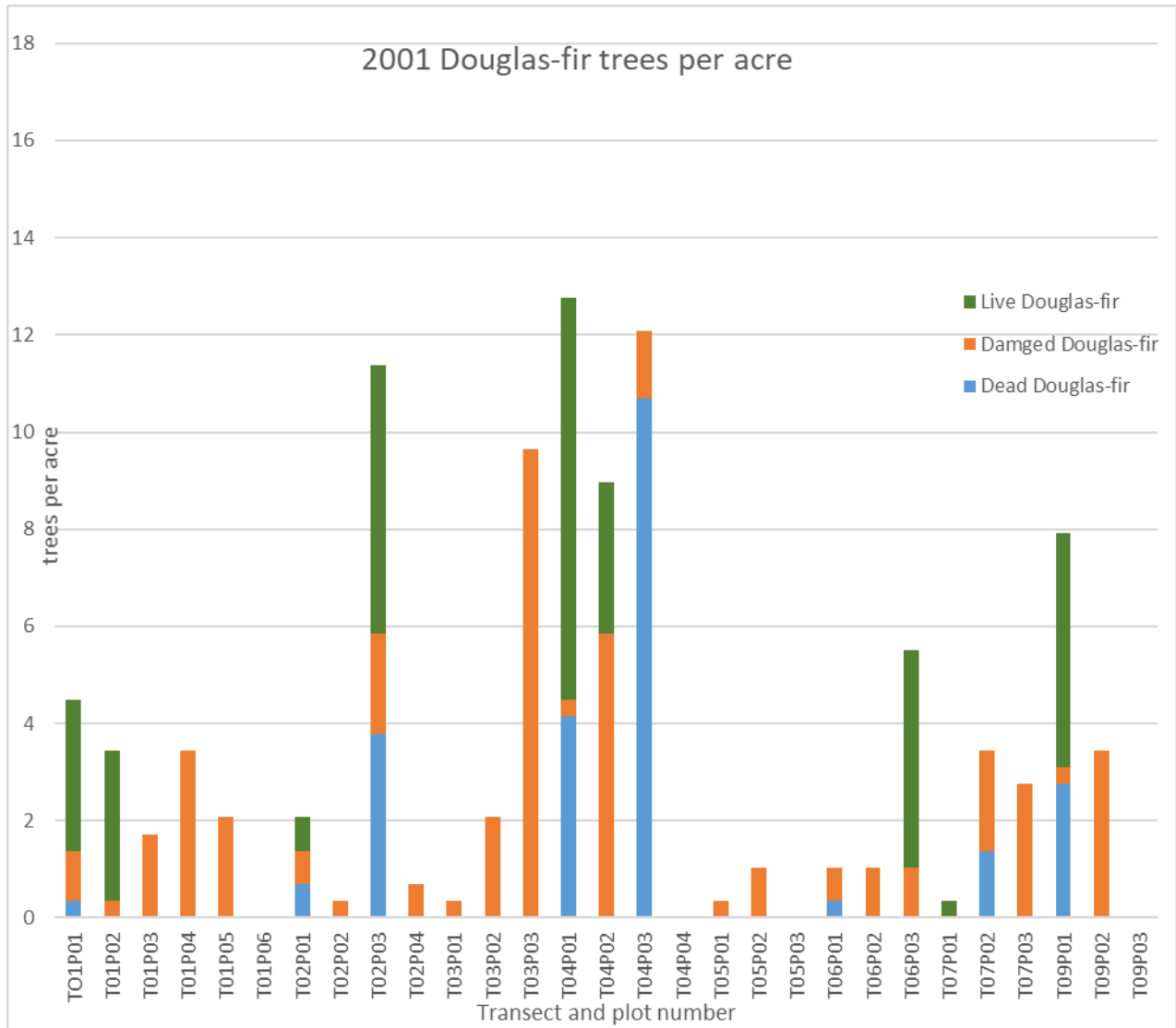


Figure B12. Number of Douglas-fir trees per acre per study plot for 2001.

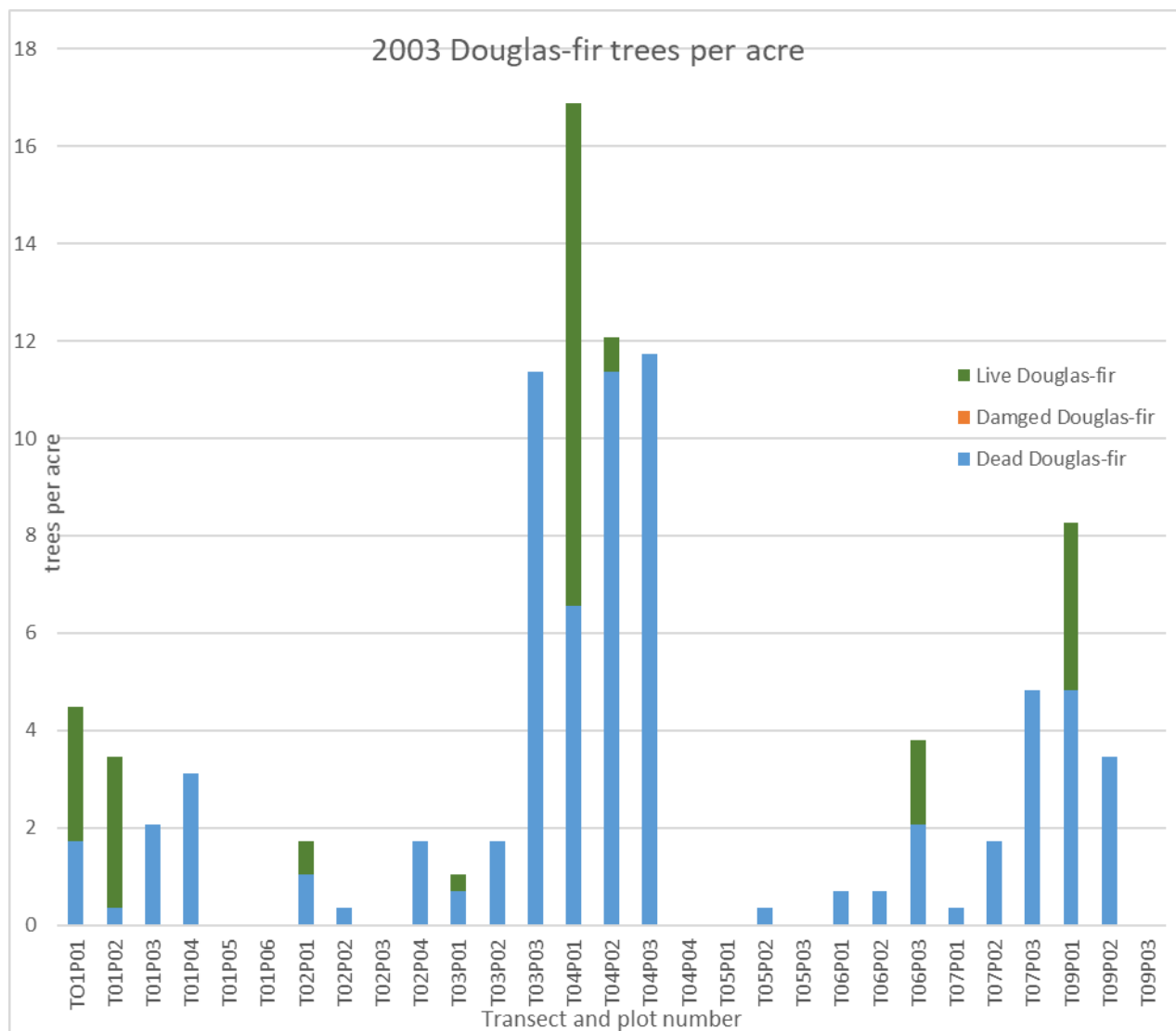


Figure B13. Number of Douglas-fir trees per acre per study plot for 2003.

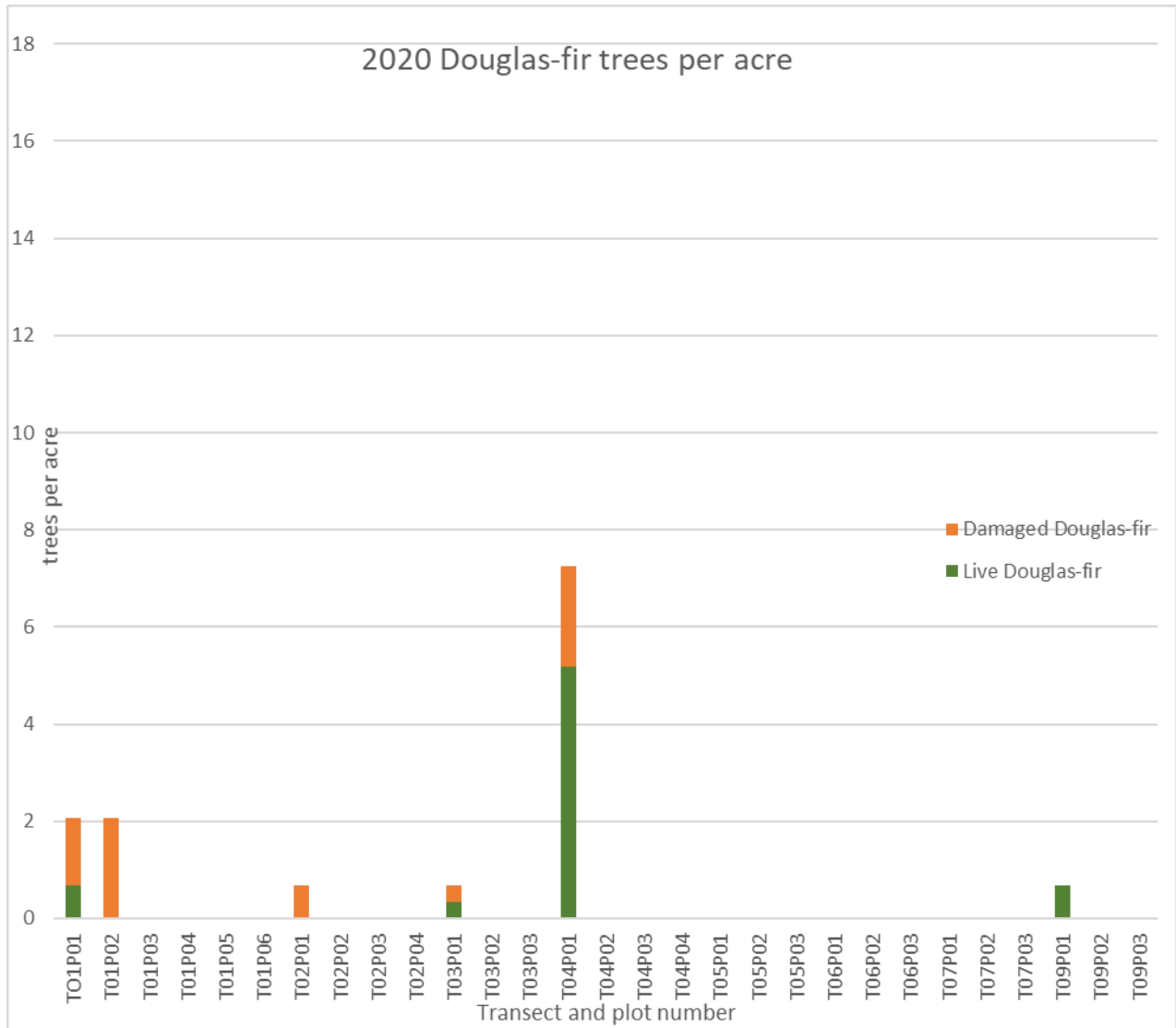


Figure B14. Number of Douglas-fir trees per acre per study plot for 2020.