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Evaluating Aspen Seedling Outplanting Success Following High Severity Wildfire in the Southwest

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EVALUATING ASPEN SEEDLING OUTPLANTING SUCCESS FOLLOWING HIGH SEVERITY WILDFIRE IN THE SOUTHWEST

by

Sarah M. Kapel

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ABSTRACT

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by

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Utah State University, 2024

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Aspen stands (*Populus tremuloides* Michx.) have been declining in western U.S. forests due to drought, changing climate, and altered disturbance regimes. Management of aspen has focused on promoting vegetative suckering, which requires viable rootstock. Outplanting with nursery-grown aspen stock is not a common practice in this region, though this is an active restoration approach that increases genetic variation and allows managers to introduce the species. Limitations for survival in past experiments have been desiccation and herbivory. Additionally, with increasing wildfire frequency and severity in the West, there are opportunities for aspen reforestation post-fire. We tested how deadwood in fire footprints can be used as treatments for nursery-grown seedlings. We outplanted 1,140 aspen seedlings in three recently burned areas in experimental treatments: 1) pockets of snags, 2) alongside logs, and 3) in open spaces. We also explored the use of Vexar® tree shelters to reduce mortality due to herbivory. We conducted a separate experiment to directly measure volumetric soil moisture content by treatment. Survival and relative growth were monitored after one growing season. We observed survival rates between 41-76% depending on the site. Survival was highest for seedlings by snags, likely due to increased soil moisture, in all sites. Growth was not influenced by treatment. Tree shelters reduced probability of herbivory-caused mortality and thereby increased seedling survival across all treatments and sites. Soil moisture varied based on seasonal precipitation events, with probes by snags yielding the greatest soil moisture content during the dry season. The results from this research suggest that aspen seedling survival in post-fire settings can be improved by strategically planting in the pockets of snags where soil moisture is more plentiful during the dry season and reducing herbivory by ungulates. This thesis contributes to the development of best practices in future aspen seedling restoration projects and suggests a reason for snag retention with post-fire reforestation.

(74 pages)

PUBLIC ABSTRACT

Evaluating Aspen Seedling Outplanting Success Following High Severity Wildfire in the Southwest

Sarah M. Kapel

Quaking aspen (*Populus tremuloides* Michx.) is an ecologically important forest species in the western U.S. Aspen forests host a variety of understory species, are critical wildlife habitat, and are considered a “natural fuel break” since they are less likely to support crown fires than conifers. Because of climate change and altered disturbance regimes, populations are declining, and innovative strategies are needed to restore aspen. Planting aspen seedlings is a solution, though not a common practice in the West and has been met with high mortality in past experiments. For aspen planting to be more broadly implemented, managers need guidance to increase probability of seedling success. We developed a planting experiment to determine whether specific types of planting sites and conditions in the field could increase the survival and growth of nursery-grown aspen seedlings. The increasing occurrence and severity of wildfire presented an opportunity to plant seedlings in high severity fire footprints. Aspen trees thrive in severely burned landscapes, where conifer regeneration may be limited. We planted 1,140 seedlings, using remnant standing dead trees, snags, and downed logs as nurse structure treatments. We also tested individual tree shelters on half of the seedlings to determine if this is an effective tool to prevent herbivory. After one growing season we assessed seedling survival, causes of mortality, growth, and local competition. In a separate field experiment we also installed soil moisture-sensing probes to monitor soil moisture content at each type of planting site (snag, log, open). My results show high survival overall compared to past experiments, notably in seedlings next to snags. This could be because soil moisture content was high near snags during dry periods. Tree shelters increased the probability of seedling survival but had mixed effects on growth within the one season. The results from this research will contribute to the development of best practices in future aspen seedling restoration projects and suggest a need for snag retention on the landscape after fire.

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CHAPTER 1

INTRODUCTION

Quaking aspen (*Populus tremuloides* Michx.) is the most broadly distributed tree species in North America (DeByle, 1990). As a foundational forest species, aspen play a significant role in community structuring and richness due to high levels of understory biodiversity supported in aspen-dominated forests (Ellison et al., 2005; Kouki et al., 2004; Stohlgren et al., 1997). Many native wildlife species depend on aspen habitat, and young trees are a preferred diet for foraging ungulates (Collins and Urness, 1983; Martin and Maron, 2012). Aspen trees also provide aesthetic, recreational, and cultural significance for humans (McCool, 1998), and are a commercially viable source of fiber and wood products in some portions of its range. Aspen trees have the ability to regenerate clonally, or asexually, through vegetative suckering. Clonal stands, which are common throughout the distribution of aspen, contain the same genetic material with each new expansion (Schier et al., 1985). Aspen can also reproduce sexually from seed and although rare compared to suckering, this reproductive strategy is more common than once was perceived (Mock et al., 2008). Aspen regenerate by both reproductive strategies following natural disturbance events such as wildfires (Kreider and Yocom, 2021). The species is considered fire-resilient since dense regeneration of suckers is common following wildfire (DeByle, Norbert V., 1990) and post-wildfire environments provide ideal habitat for the germination of seeds (Landhäusser et al., 2019). Aspen have also been thought to be less likely to support crown fire than conifer forests (DeByle et al., 1987). Though this relationship is complex, the bulk of evidence shows that aspen stands can reduce both fire occurrence and severity (Nesbit et al., 2023).

In the western United States (hereafter the West), aspen populations are vulnerable to climate change, drought, and altered disturbance regimes (Singer et al., 2019). Aspen stands are projected to decline 6-41% by the year 2030 due to climate change (Rehfeldt et al., 2009) and extensive mortality events linked to drought in western forests have been described (Huang and Anderegg, 2012). Aspen trees rely on surface water and shallow soil moisture sources which are easily affected by high temperatures and altered precipitation (Anderegg et al., 2013). This is especially concerning as climate models of the West predict drier and more unfavorable conditions as climate change progresses (Mankin, 2021). Long term fire suppression and land use changes have also played a role in decline since aspen stand regeneration is stimulated by severe disturbance (Bretfeld et al., 2016; Romme et al., 1995; Shinneman et al., 2013).

Aspen stand establishment by naturally dispersed seed has been documented in the West (Fairweather et al., 2014) and is suspected to occur in other western landscapes based on patterns of genetic diversity (Mock et al., 2008). In subsequent observations in the Intermountain West, seedlings were found predominately in high severity burned areas using fire footprints as seed beds (Kreider and Yocom, 2021). Seedlings may succeed in these burned areas as there is more light availability and soil moisture due to reduced overstory and ground cover (Calder et al., 2011; Landhäusser et al., 2019). Seedlings observed in post-fire areas also have been associated with coarse woody debris (Fairweather et al., 2014) and clustered by deadwood, such as snags or logs, acting as nurse structures (Kreider and Yocom, 2021). It is unclear if these microsites aid in seedling survival, or whether the pattern of occurrence is influenced by wind deposition of seeds.

Restoration of aspen can be implemented through passive approaches, such as promoting vegetative suckering, or active approaches, such as outplanting (Landhäusser et al., 2019). Management for aspen regeneration in the West largely focuses on established clones, targeting asexual reproduction by overstory clear-felling, or coppicing, and prescription burns (DeByle and Winokur, 1985; Long and Mock, 2012). These approaches work well to regenerate existing clones if the stand roots are healthy enough to sucker and herbivory pressure is low (Britton et al., 2016). However, there are drawbacks to relying solely on asexual reproduction (Long and Mock, 2012). Asexual reproduction limits genetic variation, which is necessary for adaptation (Mock et al., 2008) and is reliant on existing populations. The use of nursery-grown seedlings on the other hand, has several benefits. Planting seedlings could be the basis for an assisted migration program where seed is sourced from more arid landscapes (Gray et al., 2011). Research suggests that tree adaptation may not be keeping up with current climate projections and assisted migration has been suggested as a measure to prevent forest loss (Williams and Dumroese, 2013). Outplanting also allows managers to have a more active role deciding where aspen seedlings are introduced as a desired species, which can be a tool to build forest resiliency. Because fire activity and severity have increased throughout the Southwest in recent years (Abatzoglou and Williams, 2016), managers need adaptive strategies to reforest burned areas (Stevens et al., 2021). Especially as high severity wildfire leaves more forests vulnerable to type conversions (Coop, 2023; Guiterman et al., 2022), outplanting aspen seedlings might be a tactic to direct type conversion. Further, because of aspen’s relationship with milder wildfire activity, planting has been suggested around high-value communities and resources. This approach has already been adopted in some Colorado communities (LaConte, 2020; Sienkiewicz, 2020).

While active restoration of aspen through the outplanting of nursery-grown seedlings is well studied in boreal forest systems (King and Landhäusser, 2018), it has rarely been employed in the West and the few experiments with outplanting nursery-grown seedlings have yielded limited success. One experimental planting in southern Utah on plots located in burned open meadows, yielded high mortality associated with desiccation and rodent herbivory (Howe et al., 2020). In an another outplanting experiment in southern Utah, aspen seedlings were planted in an area that had been burned by wildfire and salvage logged. Treatments included planting next to logs, in open areas, and with biochar added to some seedlings. Overall survival of seedlings on the burned plots in the first year was high, but survival was lowest next to logs, as logs provided cover for small herbivores and resulted in high mortality of seedlings in those locations (Yocom and Mock, unpublished).

Since desiccation has been an obstacle in past experiments and drought stress has resulted in aspen forest decline at a landscape scale, we aimed to identify outplanting microsites with high soil moisture. Stemming from observations of natural aspen seedling establishment, we evaluated the utility of remnant wildfire deadwood as nurse structures. These structures, such as snags and logs, naturally occur in high severity fire footprints, and could facilitate greater soil moisture content. Application of deadwood and coarse woody debris has been shown to increase soil water content in boreal forests (Dhar et al., 2022) and temperate Australian woodlands (Goldin and Brookhouse, 2015). The root pockets of snags, which we define as the depressed area between the trunk of a tree and the roots, are concave microsites, which collect precipitation or run-off and are associated with increased soil moisture (Filazzola and Lortie, 2014). Moisture assessments of various deadwood has been conducted (Boulanger and Sirois, 2006; Green et al., 2022), yet how soil moisture relates to these structures has not been explored. Furthermore, targeted planting in the root pocket/concave microsite of snags has also not been tested. Using downed logs as nurse structures had varying success in past aspen outplanting, but is a useful reforestation technique in conifer experiments (Castro et al., 2011). North-facing seedlings in particular have a higher potential for survival since shade-tolerant conifers are protected from intense afternoon heat (Marsh et al., 2022). Knowledge gaps remain on the effectiveness of different nurse structures regarding aspen, as well as if the azimuth from nurse structures influences aspen seedlings. Planting seedlings strategically where they are more likely to benefit from increased soil moisture, like concave microsites or in north-facing aspects, could reduce desiccation-related mortality (Marshall et al., 2023), but might also influence seedling growth. Overstory growth is an important metric determining seedling success as it relates to the potential for root system growth and future vegetative suckering (Martens et al., 2007).

Herbivory also remains a driver of seedling mortality in natural regeneration and in outplanting experiments. Browsing pressure from ungulates impedes early establishment and stem growth (DeByle and Winokur, 1985; Seager et al., 2013). To minimize these effects, fenced enclosures have been installed in other aspen experiments (Kay and Bartos, 2000), though construction can be costly to managers. Therefore, fencing is not realistic for large scale reforestation efforts. Individual tree shelters, such as Vexar®, can be a more cost-effective and practical product to deter ungulate herbivory

(Engeman et al., 1997; Marsh et al., 1990). While Vexar has been successful in conifer outplanting experiments, it has not been used regularly for a more palatable species like aspen.

The overall objective of this project was to investigate factors influencing the success of nursery grown aspen seedlings planted in high severity fire footprints in the southwestern US. Our goal was to provide meaningful recommendations to improve post-fire planting programs. We explored this by posing four questions:

1) Does planting next to post-fire deadwood nurse structures (snags and logs) aid in seedling survival and growth? We hypothesized that nurse structures create beneficial microsites for aspen, and seedling survival and growth would be higher compared to those in open spaces.

2) Does volumetric soil moisture content differ between planting sites in the open, alongside logs, and root pockets of snags? We predicted higher soil moisture content in sites with nurse structures compared to the open.

3) How effective is individual tree sheltering (Vexar) in reducing ungulate herbivory of aspen seedlings? Tree sheltering reduced browsing pressure in conifers, and we predicted that herbivory of aspen seedlings would be reduced with Vexar.

4) Does the azimuth from nurse structures influence aspen seedling survival and growth? Based on observations in conifer outplanting, we expected to see greater survival and growth rates for north-facing seedlings.

To answer these questions, we developed outplanting experiments in three recent fire footprints using 1,140 aspen seedlings. We measured seedling success, survival and growth, as well as volumetric soil moisture, during the following growing season. The results from these experiments fill gaps in aspen reforestation research, specifically following wildfire, and provide guidance to forest managers on how to increase survival and growth when outplanting nursery-grown seedlings.

CHAPTER 2

METHODS

Study Area

This project was conducted in four planting sites across three wildfire footprints in the Colorado Plateau of the southwestern United States: the Horton fire (Arizona; 33.66, -109.321); the Luna fire (New Mexico; 36.26, -105.37); and two sites in the Pack Creek fire (Utah; Site 1: 38.493, -109.233, Site 2: 38.465, -109.264). Fire perimeters were accessed through National Interagency Fire Center's Incident Information System ("InciWeb," 2023) and selected because they occurred in late 2020 or 2021, were in close proximity to road access, burned in high-elevation locations with the potential to support aspen, and included public land. Grazing and browsing by livestock and wild ungulates such as elk and deer were present across all sites.

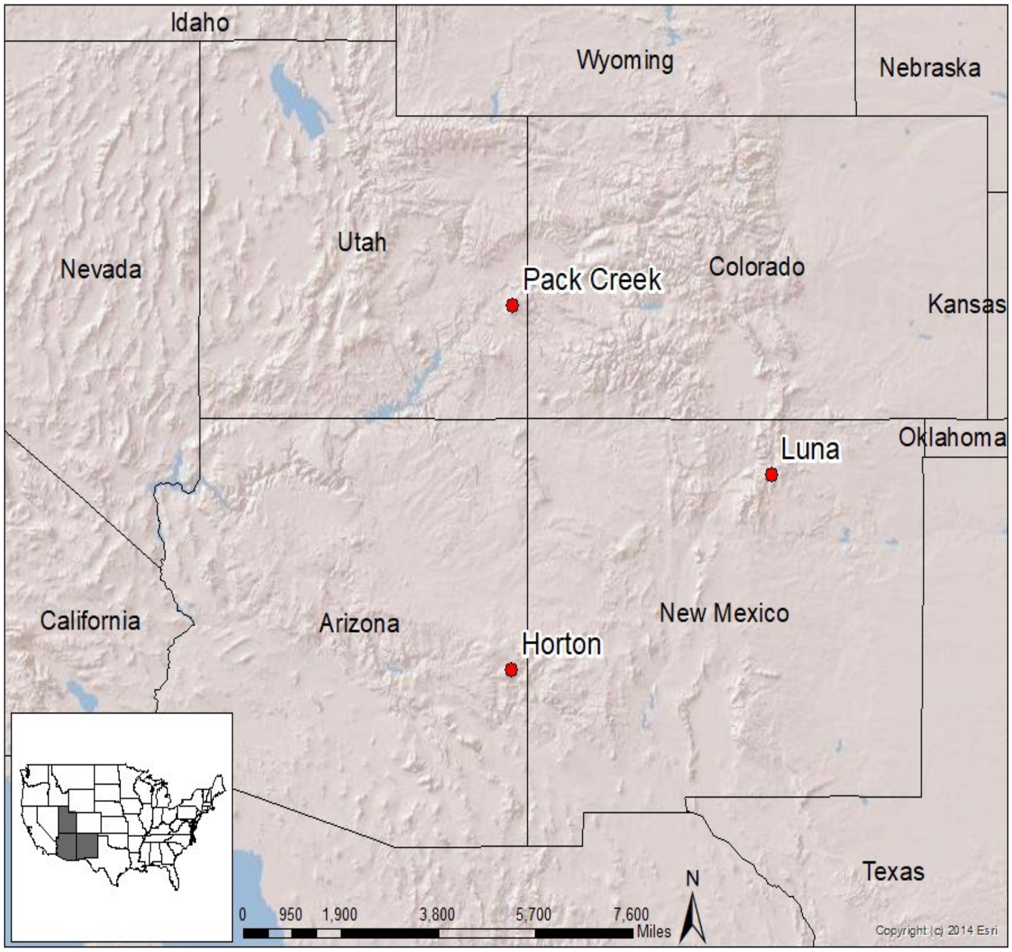

Figure 1. Map of the study sites at three fire footprints across the Southwest.

Horton

The Horton fire reburned 4,963 hectares of the Wallow Fire burn scar in the Apache-Sitgreaves National Forest in June 2021. The Wallow Fire occurred in May 2011 and consumed 217,741 hectares of land. At this site, the thirty-year mean annual temperature (1991-2020) was 7°C with variation by season: mean annual temperature in January was -1°C while in July it was 16°C. Mean annual precipitation was 727 mm and the total precipitation in 2022 was 770 mm. Precipitation has a bimodal distribution, peaking in winter from westerly storms and in summer from North American monsoons ("PRISM Climate Data," 2023). The elevation at the planting site is 2738 m. Dominant tree species include white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr) and ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson). A USDA soil survey was available for the nearby town of Alpine and the soil consists of predominately Bushvalley cobbly sandy loam ("Web Soil Survey," 2023).

Luna

The Luna fire burned 4,102 hectares of the Carson National Forest and occurred in October 2020. Mean annual temperature was 4°C with variation by season: thirty-year mean annual temperature in January was -5°C while in July it was 14°C. Thirty-year mean annual precipitation was 737 mm and the annual precipitation in 2022 was 673 mm. Precipitation has a bimodal distribution, peaking in winter from westerly storms and summer from North American monsoons ("PRISM Climate Data," 2023). Elevation at the planting site is 3,170 m. Dominant tree species include subalpine fir (*Abies lasiocarpa* [Hook], Nutt.), white fir (*Abies concolor* [Gord. & Glend.] Lindl. ex Hildebr), Engelmann spruce (*Picea engelmannii* Parry ex. Engelm.) and blue spruce (*Picea pungens* Engelm.). The soil consists of predominantly Presa cobbly loam and Maes cobbly loam ("Web Soil Survey," 2023).

Pack Creek

The Pack Creek fire burned 3,755 hectares of the La Sal National Forest in June 2021. Thirty-year mean annual temperature was 7°C with variation by season: thirty-year mean annual temperature in January was -3°C while in July it was 19°C. Thirty-year mean annual precipitation was 657 mm and the annual precipitation in 2022 was 671 mm. Precipitation has a bimodal distribution, peaking in winter and summer, though this site is not consistently driven by monsoon systems as the locations of the previous fires ("PRISM Climate Data," 2023). There are two planting units at the Pack Creek fire. Unit 1 is higher, at 3,288 m in elevation and unit 2 is lower, at 3,066 m in elevation. The dominant tree species in both sites were Engelmann spruce (*Picea engelmannii* Parry ex. Engelm.) and subalpine fir (*Abies lasiocarpa* [Hook], Nutt.). Both sites' soil consists of loam, primarily Namon gravelly loam and Richens-Herd complex in unit 1 and Leighcan cobbly loam in unit 2 ("Web Soil Survey," 2023).

Table 1. Description of each high severity wildfire footprint used as study sites in this project.

Fire	**Total hectares burned**	**Year of fire**	**National Forest**	**Elevation (m)**	**Dominant tree species**
Horton	4,963	2021	Apache-Sitgreaves National Forest	2,737.50	Ponderosa pine, White fir
Luna	4,102	2020	Carson National Forest	3,170	Blue spruce, Engelmann spruce, white fir, sub-alpine fir
Pack Creek ***2 sites***	3,755	2021	La Sal National Forest	3,066 – 3,288	Engelmann spruce, sub-alpine fir

Planting site selection

Within each fire footprint, planting sites were identified within a 0.40 km distance of a road for accessibility during planting. The sites selected were flat, with less than a 20% slope. Sites had minimal to no live aspen trees present to reduce the likelihood of aspen sucker competition.

Seedling stock

Aspen seedlings were grown by the JTH Forestry Research Center in Mora, New Mexico with New Mexico State University. Seed was provided from collections in Cedar Mountain (UT), Gallinas Canyon (NM), and Taos (NM). Germination tests were conducted in February 2022 on the seed sourced from Utah. Two hundred seeds from each collection were placed evenly in 10 x 10 rows on damp paper towels in petri dishes. The paper towels were moistened twice a day and signs of germination were recorded after two days and again after one week. The Utah seed had an average germination rate of 95.15%. Similar germination tests were conducted by the J.T. Harrington Forestry Research Center with the seed sourced from New Mexico, yielding an average germination rate of 87%. Only seed sourced from New Mexico was permitted to be used in the Luna fire footprint per Forest Service request and seed from the Utah collection was used in the Pack Creek and Horton fire footprints.

Sowing took place in late March 2022. Seeds were sown in SC10 RayLeach containers in a mixture of 2-parts peat moss, 1-part vermiculite, and 1-part perlite. The seeds were misted 3-5 times a day. After four weeks, cells were thinned to one seedling per container, and the irrigation schedule changed to a volumetric weight-based schedule with a target of 85% dry-down of media field capacity. Fertilizer treatments began weekly using a 21-5-20 (N-P-K) general-user water-soluble fertilizer. Seedlings were moved to an outdoor sub-irrigation house in June. Once moved, irrigation continued to be based on volumetric weights with fertilizer applications occurring weekly. In August, seedlings were top pruned, which is a technique to prepare seedlings to cope with stressful growing conditions by increasing the root: shoot ratio (South and Blake, 1994).

Seedlings were also treated for spider mites. Shade was added over the top of the aspen seedlings in the sub-irrigation house until seedlings were transported for planting in September and October 2022.

Treatment design

We looked for naturally occurring snags, logs, and open spaces as treatment structures for the seedlings at each fire. In the Luna fire, logs were created by felling snags in the site. All snags and logs selected for treatment were conifers. If the site had aspen, seedlings were planted at least 2 m from any regeneration. Areas with large shrubs, bunchgrasses, or saplings were avoided to reduce vegetative competition, though small forbs and grasses were present in all sites. Data on these were captured in ground cover assessments.

To test whether nurse structures remnant after fire increase survivorship and growth for seedlings, we tested three different treatments. These treatments involved planting: 1) in the root pockets of snags, 2) alongside logs, and 3) in open spaces void of any structures, acting as our control. A minimum of thirty snags, logs, and open spaces were identified at each fire.

For snag treatments, conifer snags within a similar range of diameters at breast height (DBH) were selected. Snag were classified at a decay class of 2, meaning the tops were intact and ~50% of the coarse branches were intact (Vanderwel et al., 2006). The average conifer snag DBH varied by site and species. Snags in the Horton fire were primarily ponderosa pine and averaged 35 cm. Snags in the Luna fire were primarily white fir, sub-alpine fir, Engelmann spruce, or blue spruce and averaged 31 cm. Snags in the Pack Creek fire were primarily sub-alpine fir or Engelmann spruce and averaged 33 cm DBH. Seedlings were planted within a snag "pocket" which we define as the concave spot between roots and as close to the base of the snag as possible. Two to four seedlings were planted around one structure depending on how many distinct pockets were present and one seedling was planted per pocket. The azimuth and distance from seedling to snag were recorded.

For the log treatments, 4-8 seedlings were planted alongside a single log with 2-4 seedlings on either side of the log. Logs were chosen that were in contact with the soil surface, and seedlings were planted as close to that contact area as possible. Seedlings were distanced ≥0.5 m from each other on the same side of the log. As with snags, we selected logs within a similar range of height. Log height was on average 25 cm in the Horton fire, 29 cm in the Luna fire, and 23 cm in the Pack Creek fire. Logs were classified as a decay class of 2 or 3, meaning bark may be missing but the wood was hard when kicked (Vanderwel et al., 2006). At the Luna fire footprint, naturally occurring logs were not present. Ten snags were felled and limbed in order to create logs and these logs were dispersed at random and in various directions to ensure a variety of potential azimuths for seedlings.

For open space treatments, seedlings were planted without nurse structures (logs or snags) and at least 2 m from the nearest structure. Four to eight seedlings were clustered in a single open space distanced ≥0.5 m from one another.

We installed rigid seedling 5" by 36" Mesh Vexar Seedling Protector Tubes reinforced with one bamboo stake on roughly half of the total seedlings to test effectiveness of herbivory protection. Once all the seedlings were planted in a site, half of the structures and open spaces were randomly selected using the Randomizer App©.

Protection was evenly distributed across treatments and all the adjacent seedlings associated with one structure or open space were protected. For example, if a snag was selected, all seedlings around that one snag would receive protection.

Figure 2. Unprotected seedlings planted in (a) the root pockets of snags, (b) adjacent to logs, and (c) Vexar treated seedlings clustered in open spaces in Pack Creek Unit 2.

Outplanting

Planting began in the fall of 2022. Planters used Leonard Planting Bars and each planter rotated between treatments to disperse any planter quality effects. Planters abandoned nurse structures if the area was too difficult to plant due to rocks, roots, or other obstacles. Once in the ground, each seedling was given a unique identification tag. In the Horton fire, 329 seedlings were planted. In the Luna fire, 335 seedlings were planted. In the Pack Creek fire, 476 seedlings were planted across the two sites (Table 2).

Table 2. Distribution of seedlings at each fire footprint by treatment.

		Horton		Luna		Pack Creek		
Treatment		Number of		Number of		Number of		
	Vexar?	Structures	Seedlings	Structures	Seedlings	Structures	Seedlings	*Total*
Snag	No	14	46	16	64	21	68	
	Yes	15	56	12	48	24	84	
	Total		*102*		*112*		*152*	*366*
Log	No	11	41	18	66	22	88	
	Yes	16	60	12	47	15	69	
	Total		*101*		*113*		*157*	*371*
Open Space	No	18	69	14	51	19	82	
	Yes	15	57	15	59	17	84	
	Total		*126*		*110*		*167*	*403*
Overall Total			**329**		**335**		**476**	**1,140**

Measurements

To capture baseline seedling characteristics, height measurements and groundline diameter (GLD) of each seedling were recorded immediately after outplanting. We measured height as the distance from the ground to terminal bud. If we could not determine the terminal bud in the field, we measured the axillary bud on the longest branch. For snag and log seedlings, we recorded azimuth to the nearest 0.5° in the field and these values were categorized into four distinct cardinal directions after. The distance to the structure, as well as the species/genus and decay class of the structure were also recorded for each snag and log. Snag DBH and log height were also recorded. We measured local competition by estimating ocular ground cover around each seedling using a 0.44 m^2 circular quadrat centered at the seedling. Cover classes included grass, shrub, forb, snag/root, coarse woody debris, bare soil, conifer litter, and other. Cover classes were measured to the nearest whole percentage. For the seedlings near snags or logs, ocular cover was estimated in 50% of the quadrat while seedlings in open spaces were estimated in 100% of the quadrat.

First year measurements were completed near the end of the following growing season (August 2023) and seedling survivorship status, growth, and ground cover were recorded. For living seedlings, we remeasured height and GLD. Signs of desiccation were noted based on a scale we developed. A "one" was assigned to seedlings with mild desiccation in which less than 25% of the leaves had visible signs of drying. A "two" was assigned to seedlings with moderate desiccation in which 25-50% of the leaves had visible sign of drying. A "three" was assigned to seedlings with severe desiccation in which 50+% of the leaves had visible sign of drying.

Another scale was developed to quantify herbivory (Table 3). Herbivory was first identified by either ungulate, insect, or rodent and damage was categorized. Vexar was also repaired for living seedlings if needed. We noted other seedling specific conditions, including obvious signs of poor planting such as exposed roots.

Table 3. Scale of herbivory damage of living seedlings

Damage		**Defining Qualities**
0	None	No sign of herbivory, leaves are green, intact, and whole
1	Light	<10% of the leaves show damage from herbivory from either ungulate or insect
2	Mild	10-25% of the leaves show damage from herbivory from either ungulate or insect
3	Moderate	25-50% of the leaves show damage from insect or the seedling was top pruned from ungulate
4	Heavy	50-75% of the leaves show damage and the seedling was top pruned from ungulate
5	Severe	>75% of the leaves show damage and the seedling was top pruned from ungulate

Ocular ground cover measurements were repeated for all seedlings, dead or alive.

If the seedling and tag was missing, the seedling was marked as dead and ground cover measurements were not retaken.

For dead seedlings, a likely cause for mortality was attributed based on observations. Herbivory was assigned as the cause of death when the stem was clearly bitten at or above ground level. Desiccation was assigned if dried leaves and/or stem were present. If the seedling was missing or an obvious cause of death was not identified, the cause of death was undetermined.

Soil moisture

In June 2023 at the Luna fire footprint, we deployed HOBO USB Micro Station Data Loggers and HS Soil Moisture Smart Sensors to monitor soil moisture throughout the summer growing season. We installed 30 data logging stations with 10 stations at each treatment. Each station had four ports for probe sensors which we deployed at each treatment (snag, log and open space) to mimic where seedlings were planted. We identified new nurse structures throughout the planting site that did not have seedlings previously planted, and we installed probes ≥1 m from any nearby seedlings. We noted species, diameter at breast height, and decay classes for the ten stations at snags. We also noted species, height, and decay classes for the ten stations at logs. We placed probes 8 inches under the soil surface, oriented horizontally into the soil. This distance was selected to match seedling root depth from the container pots. Once installed, we dug probe sensor wires into the soil and staked them down. Tag IDs were assigned to each probe and probes logged at 1-hour intervals throughout the growing season (2023/06/16 – 2023/09/30). In October 2023, data was downloaded from these stations and the systems were reset. The measurement range for the HS Soil Moisture Smart Sensors is 0-0.570

(m^3/m^3) and we excluded values outside of this range as outliers.

Analysis

We conducted all data analysis with R version 4.3.1 (R, 2023). Statistical tests were conducted for each fire separately and statistical significance was determined at the alpha level of $p < 0.05$, unless otherwise noted.

Seedling survival

We examined predictors of one-year aspen seedling survival using a generalized linear mixed model (GLM) with a binomial distribution and a logit link in R package lme4 (Bates et al., 2015). Independent predictor variables include structure type, presence of Vexar, cardinal direction, and biotic ground cover. Cardinal direction was determined based on azimuth from the structure. Azimuth values > 315° or < 45° were considered North. Values > 45° and < 135° were considered East. Values > 135° and < 225° were considered South. Values > 225° and < 315° were considered West. Biotic ground cover was the summation of all the living cover classes: grass, forb, fern, and shrub (maximum for open spaces was 100%, maximum for snags and logs was 50% due to the structure presence). Biotic cover was treated as a random effect. Our global model for Luna and Pack Creek fire footprints is as follows:

$$\begin{aligned}\text{Survival} \sim\ & \alpha + \beta_1\ (\text{Structure}) + \beta_2\ (\text{Vexar}) + \beta_3\ (\text{Direction}) + \\ & \beta_1\ (\text{Structure}){:}\ \beta_2\ (\text{Vexar}) + \beta_1\ (\text{Structure}){:}\ \beta_3\ (\text{Direction}) + \\ & \beta_2\ (\text{Vexar}){:}\ \beta_3\ (\text{Direction}) + \beta_1\ (\text{Structure}){:}\ \beta_2\ (\text{Vexar}){:}\ \beta_3\ (\text{Direction}) + \\ & \alpha_i\ (1|\ \text{Biotic})\end{aligned}$$

We observed significant variance in planting quality in the Horton fire and included planters as fixed effect for this model only. For the model to converge, we simplified the global model for the Horton fire by removing one interaction term, displayed here:

$$\text{Survival} \sim \alpha + \beta_1 \text{ (Structure)} + \beta_2 \text{ (Vexar)} + \beta_3 \text{ (Direction)} + \beta_4 \text{ (Planter)} + \beta_1 \text{ (Structure)}{:}\ \beta_2 \text{ (Vexar)} + \alpha_i \text{ (1| Biotic)}$$

Model selection was conducted using the dredge function of the Multi-model inference package (Bartoń, 2013). Akaike's information criterion (AIC) (Akaike, 1974) was assessed to select the best GLM model for survival from all measured parameters for each fire footprint. Model significance can be assessed by using AIC index values and Δ AIC, defined as the difference (Δ) between the AIC index of each model relative to the lowest AIC value. A null model was also included to determine a baseline AIC index value against which all other models could be compared. Models with AIC scores greater than the null model were excluded.

Seedling growth

We examined predictors of one-year seedling growth using a linear regression analysis in R package lme4 (Bates et al., 2015). Growth was defined as the relative change in surviving seedling GLD (mm) from the baseline measurements and remeasurements. The same independent predictor variables were used for this model as the survival model: structure type, presence of Vexar, cardinal direction, and included biotic ground cover as a random effect. Negative values of relative growth were present, yet normally distributed across all fires. Our global model for all fire footprints is as follows:

$$\text{Growth} \sim \alpha + \beta_1 \text{ (Structure)} + \beta_2 \text{ (Vexar)} + \beta_3 \text{ (Direction)} +$$

β_1 (Structure): β_2 (Vexar) + β_1 (Structure): β_3 (Direction) +

β_2 (Vexar): β_3 (Direction) + β_1 (Structure): β_2 (Vexar): β_3 (Direction) +

α_i (1| Biotic)

Similar to the survival model, AIC model selection was used to determine the best fit model for growth.

Herbivory

To determine probability of herbivory given the presence of Vexar, we created a generalized linear mixed model (GLM) with a binomial distribution and a logit link in R package lme4 (Bates et al., 2015). Our response variable was the presence (1) or absence (0) of ungulate herbivory. Presence included dead seedlings with ungulate herbivory as the cause of death and surviving seedlings with herbivory damage attributed to ungulates. Since Vexar is designed to reduce ungulate herbivory, insect herbivory was excluded from the model. Independent predictor variables were structure type interacted with the presence of Vexar.

Soil moisture

Volumetric soil moisture content was measured from 2023-06-18 to 2023-09-23 in the Luna fire footprint. To assess differences in weekly mean of soil moisture among the three structure treatments, we analyzed response variables with a two-way analysis of variance (ANOVA; Chambers, 1992), and then used a post-hoc Tukey's test to evaluate pairwise comparisons for statistical significance. Response variables were treatment structure interacted with week. We also wanted to see if there were differences in cardinal direction of the probes by snags and logs. For this, we filtered seedlings by snags and logs, running the same ANOVA model to compare variance of mean weekly volumetric soil moisture.

CHAPTER 3

RESULTS

Seedling survival

AIC Model Selection

Candidate models by site were ranked on the basis of AIC index values and change of AIC values (Δ). The model with the lowest AIC value was considered the best fit statistically, though we identified any models with Δ AIC < 2 to have substantial support as well (Burnham and Anderson, 2004). We also considered models on the basis of ecological importance and consistency across fire footprints. For the Horton fire, the best fit model had the lowest AIC value and included parameters of structure type, Vexar, and planter. For the Luna fire, the model with the lowest AIC value excluded structure type as a parameter, which was important to our research. Therefore, we selected a model within our Δ AIC range, which included the parameters of structure type and Vexar, as best fitting. For the Pack Creek fire, the model with the lowest AIC was also the best fit model and included parameters of structure type and Vexar, the same parameters for the Luna fire model.

Seedling Survival

Seedling survival varied widely by fire, with 41% survival in the Horton fire (142 of 329 seedlings), 47% in the Luna fire (157 of 335 seedlings) and 76% in the Pack Creek fire across both planting units (360 of 476 seedlings). In all three fires, mortality was primarily from unknown causes followed by desiccation (Fig. 3).

Seedling survival was highest by snags and lowest alongside logs in each fire (Fig 3). In the Horton fire, planting in open spaces significantly decreased survival probability compared to planting by snag structures, consistent with my hypothesis ($p < 0.05$) and 38% percent of the surviving seedlings were planted near snags (n = 60). Planting alongside logs significantly increased probability of mortality (odds ratio = 0.290; $p < 0.001$). In the Luna fire, 39% (n = 62) of surviving seedlings were planted near snags. Seedlings in open spaces had a survival rate of 31% (n = 49) and logs had a survival rate of 29% (n = 46). The Pack Creek fire has the highest overall survival and 35% (n = 128) of surviving seedlings were planted by snags. Planting near logs significantly increased mortality probability than those to snags ($p < 0.01$). Seedlings planted in open spaces outperformed seedlings planted by logs.

Vexar increased survival probability for each treatment and across all fires (Fig. 4). Survival probability was significantly higher for seedlings with Vexar than seedlings without in the Luna fire ($p < 0.05$) and Pack Creek fire ($p < 0.001$).

Planting personnel were included as a fixed effect in the Horton fire due to numerous seedlings identified as poorly planted (n = 37). This was due to a single planter having a significantly lower probability of survival compared to the other planters ($p < 0.001$). Poor planter quality was not detected in the other fires and therefore not included in their models.

Table 4. Seedling survival and mortality by treatment and sites.

Status after 1 year		**Horton** (n = 329)		**Luna** (n = 335)		**Pack Creek** (n = 476)	
Living	Snag	60	59%	62	55%	128	84%
	Log	26	13%	46	41%	105	67%
	Open	50	40%	49	45%	127	76%
	Total	136	41%	157	47%	360	76%
Dead	Herbivory	10	5%	15	8%	2	2%
	Desiccation	77	40%	72	40%	47	41%
	Unknown	100	52%	84	47%	61	53%
	Total	193	59%	178	53%	116	24%
Survival Rate		**41%**		**47%**		**76%**	

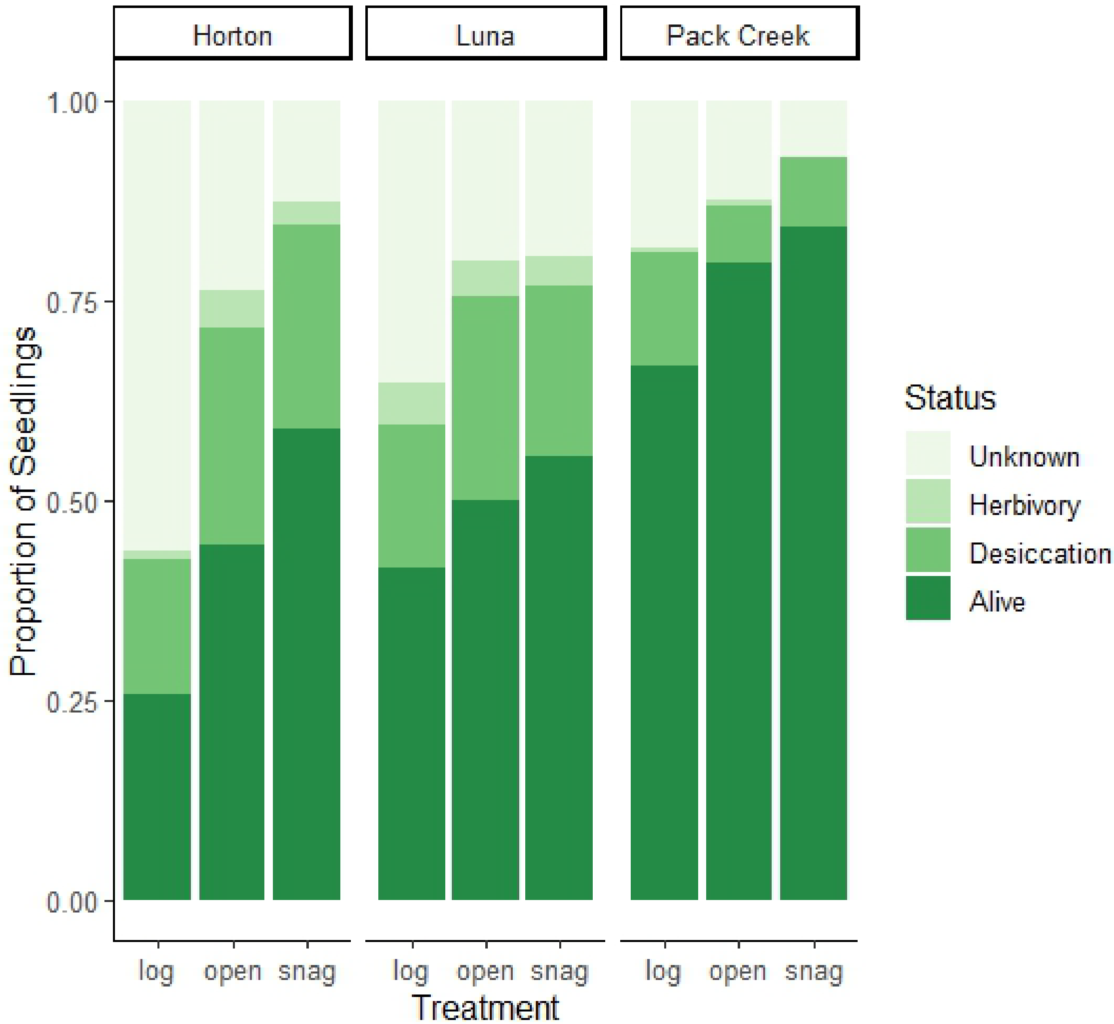

Figure 3. Proportion of seedling survival and cause of death across sites and treatments from observed data.

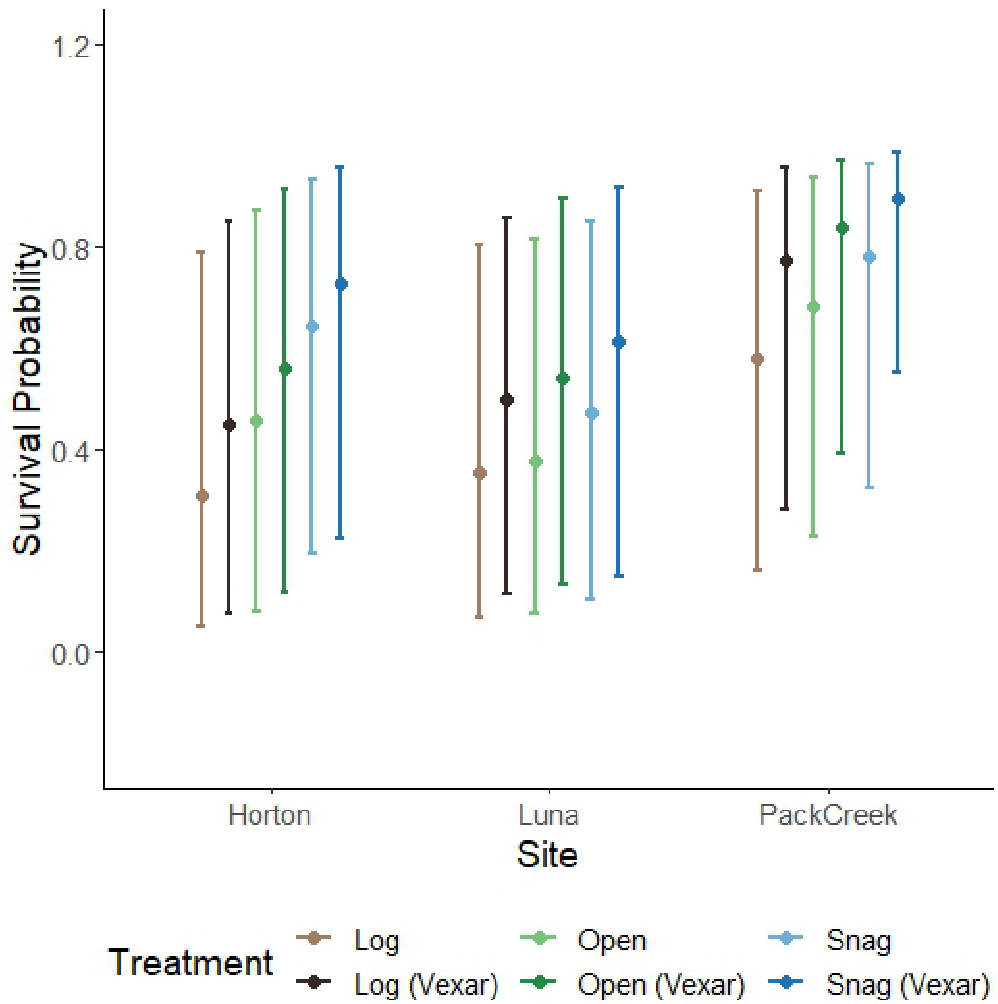

Figure 4. Predicted survival probability results from the GLM models for each treatment and with Vexar across sites, displayed at a confidence level of 95%.

Seedling growth

AIC Model Selection

Like model selection for survival, candidate growth models by site were ranked based on AIC index values and Δ values. Just like in the survival model selection, we identified any models with Δ AIC < 2 to have substantial support (Burnham and Anderson, 2004), and considered ecological importance and consistency across fire footprints. For both the Horton fire and Luna fire sites, the models with the lowest AIC score were our null models (i.e., no explanatory parameters). For the Horton fire, we selected the best fit model within the Δ AIC range which included Vexar as the only explanatory parameter. For the Luna fire, all models were outside of our Δ AIC range limit of the null model so the best fit model for this fire included no parameters. For the Pack Creek fire, the best fit model had the lowest AIC score and included Vexar as a parameter. In all the models, structure type and cardinal direction were not included as parameters, nor any interaction between parameters, and biotic was included as a random effect.

Seedling Growth

Model selection and linear regression revealed that no single parameter from our study design significantly influenced relative growth. Local competition, indicated by presence of biotic cover, had a slight effect on seedling growth as a random effect. Generally, as the percentage of live ground cover increased, relative growth decreased.

Seedlings grew the least in the Luna fire, with an average actual growth increase of 0.20 mm (SE ± 0.53). The remaining sites had similar increases, with an average actual growth of 0.73 mm (SE ± 0.97) in the Horton fire and 0.71 mm (SE ± 0.762) in the

Pack Creek fire. For our models, we considered relative growth of ground line diameter of living seedlings.

Effects from Vexar on seedling relative growth was not uniform and there were no clear patterns across fires (Table 5; Fig. 7). Vexar did not significantly affect relative growth. In the Horton and Luna fire footprints, seedlings by snags with Vexar had less relative growth than those without sheltering, while protected seedlings in open spaces had more relative growth. In the Pack Creek fire, seedlings by snags and logs with Vexar had more relative growth, while sheltered seedlings in the open spaces had less relative growth.

Table 5. Distribution of seedling mean relative growth by treatment across fire footprints.

		Horton		**Luna**		**Pack Creek**	
Treatment	**Vexar?**	Mean relative growth (mm)	SE	Mean relative growth (mm)	SE	Mean relative growth (mm)	SE
Snag	No	1.040	1.180	0.301	0.444	0.488	0.622
	Yes	0.499	1.080	0.214	0.505	0.818	0.717
	Total	0.714	1.140	0.253	0.477	0.685	0.697
Log	No	1.200	0.922	0.127	0.596	0.452	0.701
	Yes	0.494	0.958	0.129	0.386	0.825	0.801
	Total	0.691	0.983	0.127	0.525	0.633	0.771
Open Space	No	0.719	0.746	0.182	0.631	0.830	0.710
	Yes	0.830	0.748	0.220	0.590	0.768	0.964
	Total	0.767	0.741	0.205	0.600	0.796	0.856
Overall Total		**0.729**	**0.972**	**0.203**	**0.531**	**0.709**	**0.778**

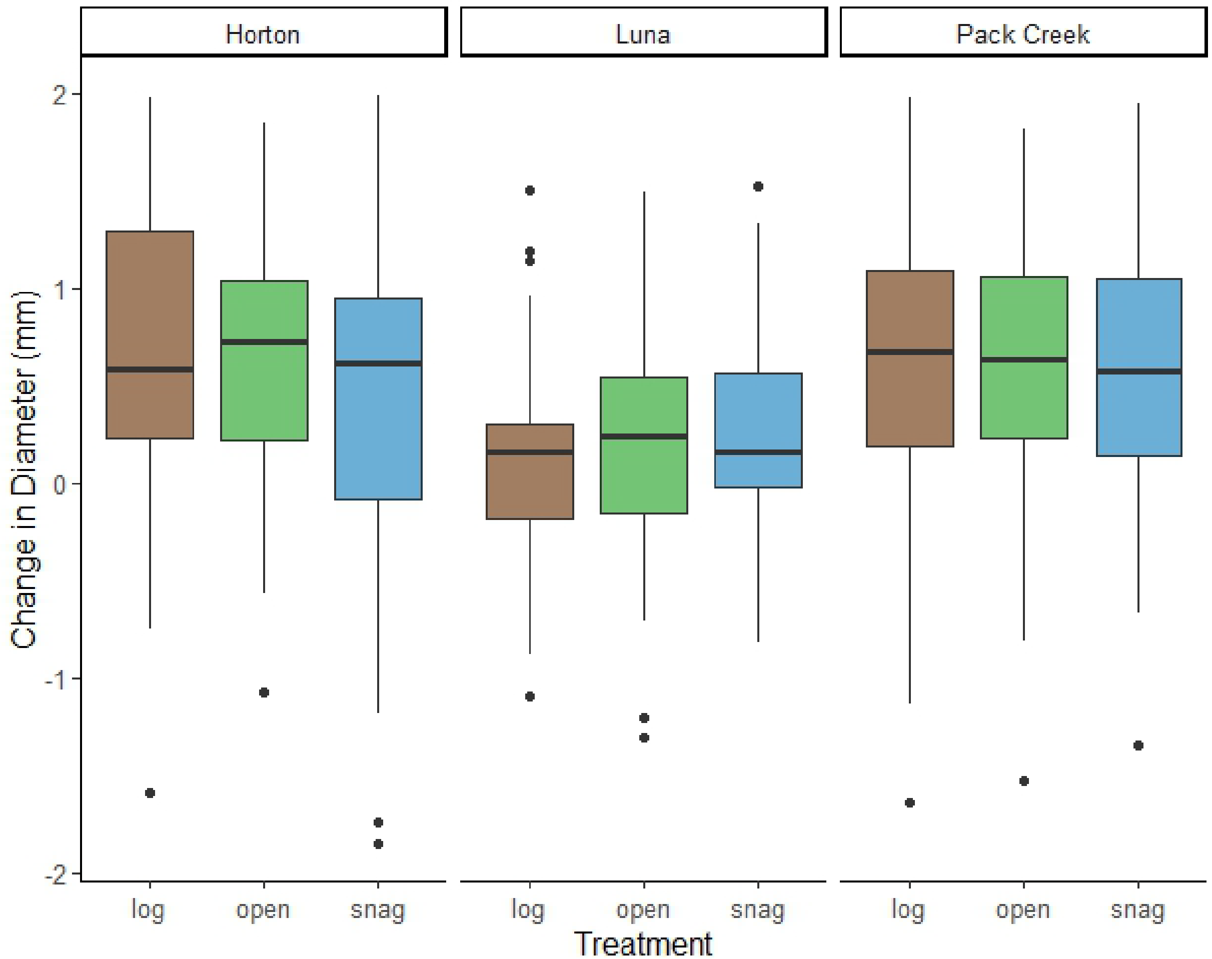

Figure 5. Distribution of living seedling changes in diameter actual growth across sites and treatments from observed data.

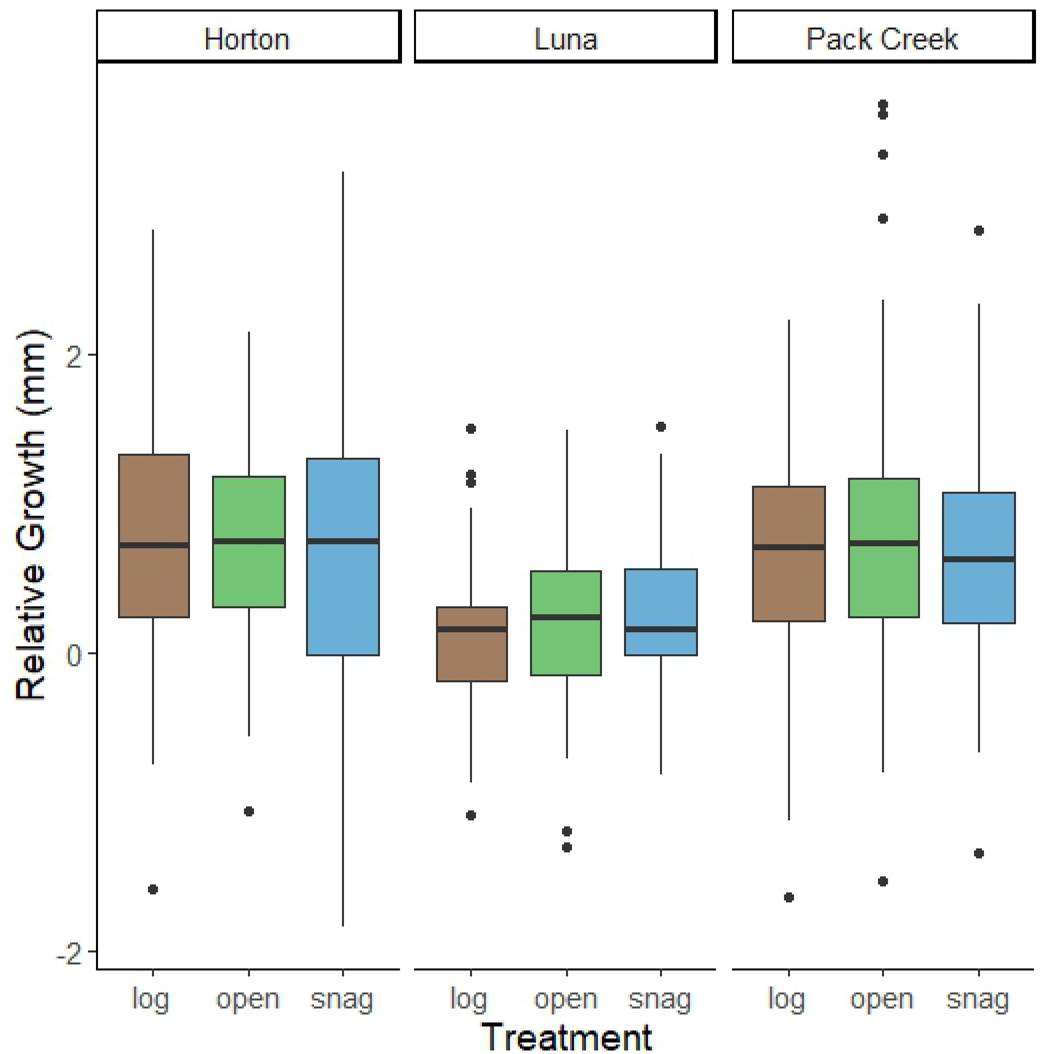

Figure 6. Distribution of living seedling changes in relative growth across sites and treatments from observed data.

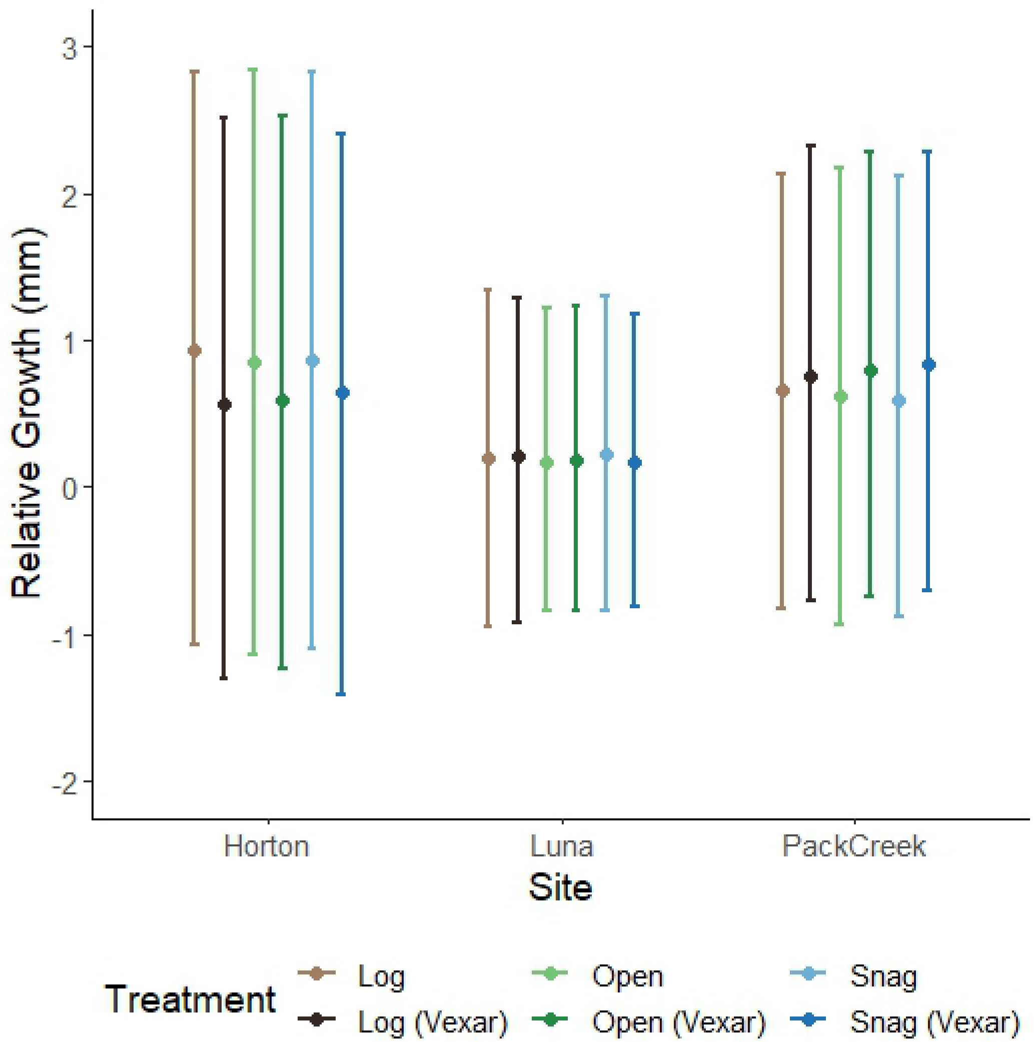

Figure 7. Predicted relative growth for each treatment and with Vexar across sites, displayed at a 95% confidence interval.

Herbivory

In general, tree sheltering reduced the likelihood of ungulate browsing across all three fire footprints (Fig. 8). In the Horton fire, the use of Vexar reduced probability of herbivory with seedlings near snags and in open spaces. In the Luna and Pack Creek fires, the use of Vexar with respect to all treatments reduced probability of herbivory. Vexar was significant at reducing likelihood of herbivory of seedlings by logs in the Pack Creek fire ($p < 0.001$).

Table 6. Predicted probability of ungulate herbivory across treatments for each fire footprint.

Probability of Herbivory

		Horton	**Luna**	**Pack Creek**
Vexar	Snag	7.1%	16.7%	16.3%
	Log	8.3%	21.3%	7.7%
	Open	8.8%	22.0%	49.1%
No Vexar	Snag	10.9%	29.7%	38.9%
	Log	7.3%	33.3%	52.4%
	Open	11.6%	31.4%	51.1%

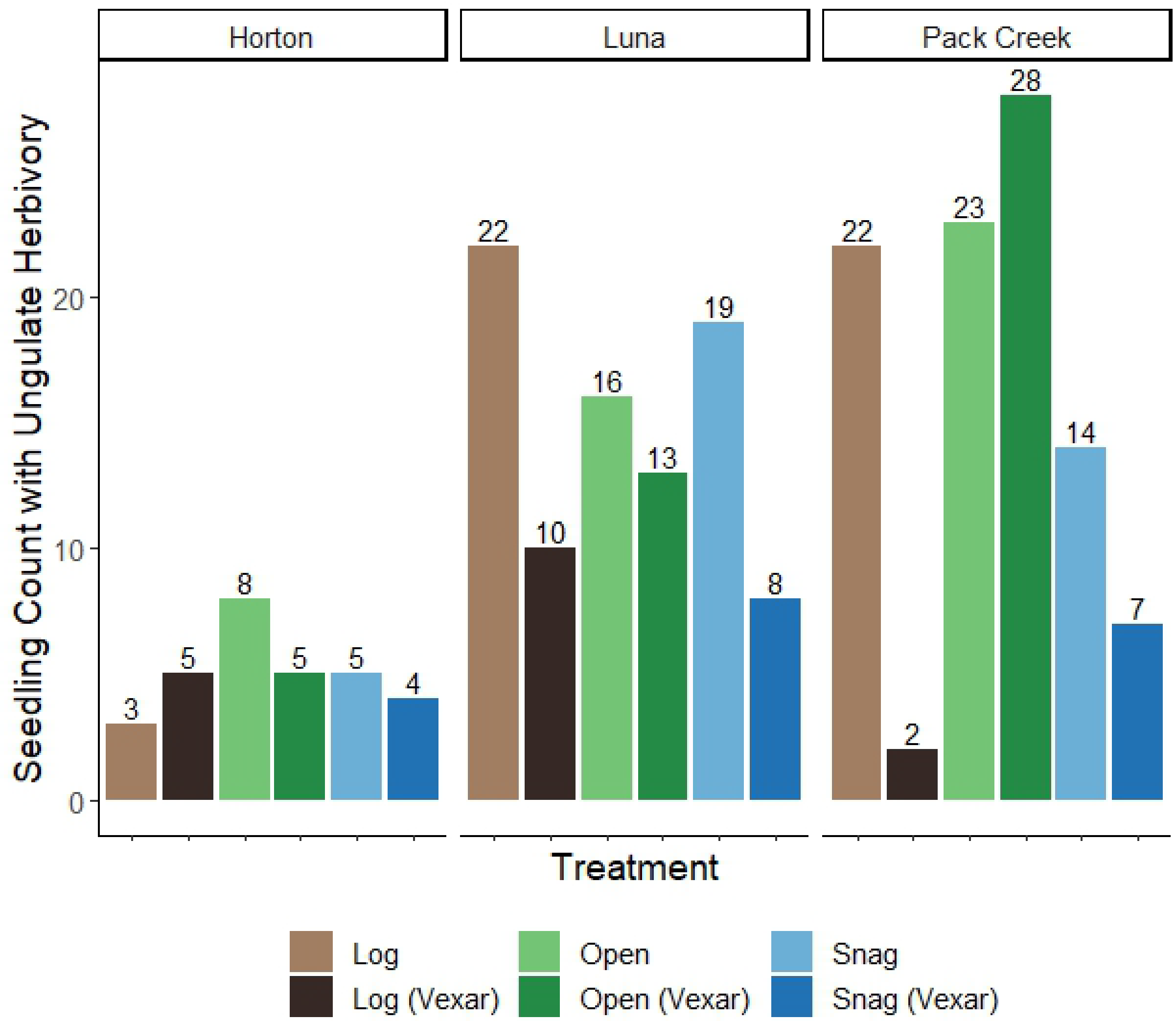

Figure 8. Herbivory by treatment and the presence of Vexar by fire footprint from observed data.

Soil moisture

In a separate but related study I installed soil moisture sensing stations and probes that logged hourly throughout one growing season to compare differences among the treatment types and in various cardinal directions. This soil moisture data was only collected in the Luna fire. Across weeks and treatments, average volumetric soil moisture content varied significantly depending on whether the area was experiencing drying or wetting. Significance was only detected in the interaction between week and treatment, therefore treatment alone as a parameter did not significantly affect average soil moisture content. The observation period was from 19-06-2023 to 18-09-2023 (Table 7; Figure 9). Snags led with the highest soil moisture from deployment in June during the pre-monsoonal growing season. This pattern continued with significantly more soil moisture content in July while open spaces had the lowest soil moisture content. August brought monsoonal precipitation which resulted in increasing in content across all probes. Logs then had the highest soil moisture throughout the rest of the observational period while snags had the lowest soil moisture content.

In addition to seasonality being an influential parameter, direction was also significant according to the model. When observing direction for probes by snags and logs, north and south-facing probes had access to significantly greater volumetric soil moisture content compared to the east and west-facing probes of the same treatment ($p < 0.001$).

Table 7. Mean volumetric soil moisture content measured across treatments by week in 2023. $\overline{\theta v}$ is measured in m3/m3.

Week No.	Snag		Log		Open Space	
	$\overline{\theta v}$	SE	$\overline{\theta v}$	SE	$\overline{\theta v}$	SE
19 June	0.299	0.006	0.298	0.006	0.302	0.006
26 June	0.294	0.006	0.290	0.006	0.291	0.006
03 July	0.285	0.006	0.274	0.006	0.276	0.006
10 July	0.270	0.006	0.255	0.006	0.254	0.006
17 July	0.250	0.006	0.231	0.006	0.234	0.006
24 July	0.237	0.006	0.218	0.006	0.227	0.006
31 July	0.230	0.006	0.212	0.006	0.224	0.006
07 Aug.	0.223	0.006	0.205	0.006	0.219	0.006
14 Aug.	0.270	0.006	0.288	0.006	0.299	0.006
21 Aug.	0.268	0.006	0.284	0.006	0.294	0.006
28 Aug.	0.278	0.006	0.300	0.006	0.310	0.006
04 Sept.	0.267	0.006	0.278	0.006	0.284	0.007
11 Sept.	0.267	0.006	0.275	0.006	0.276	0.007
18 Sept.	0.275	0.006	0.292	0.006	0.295	0.007

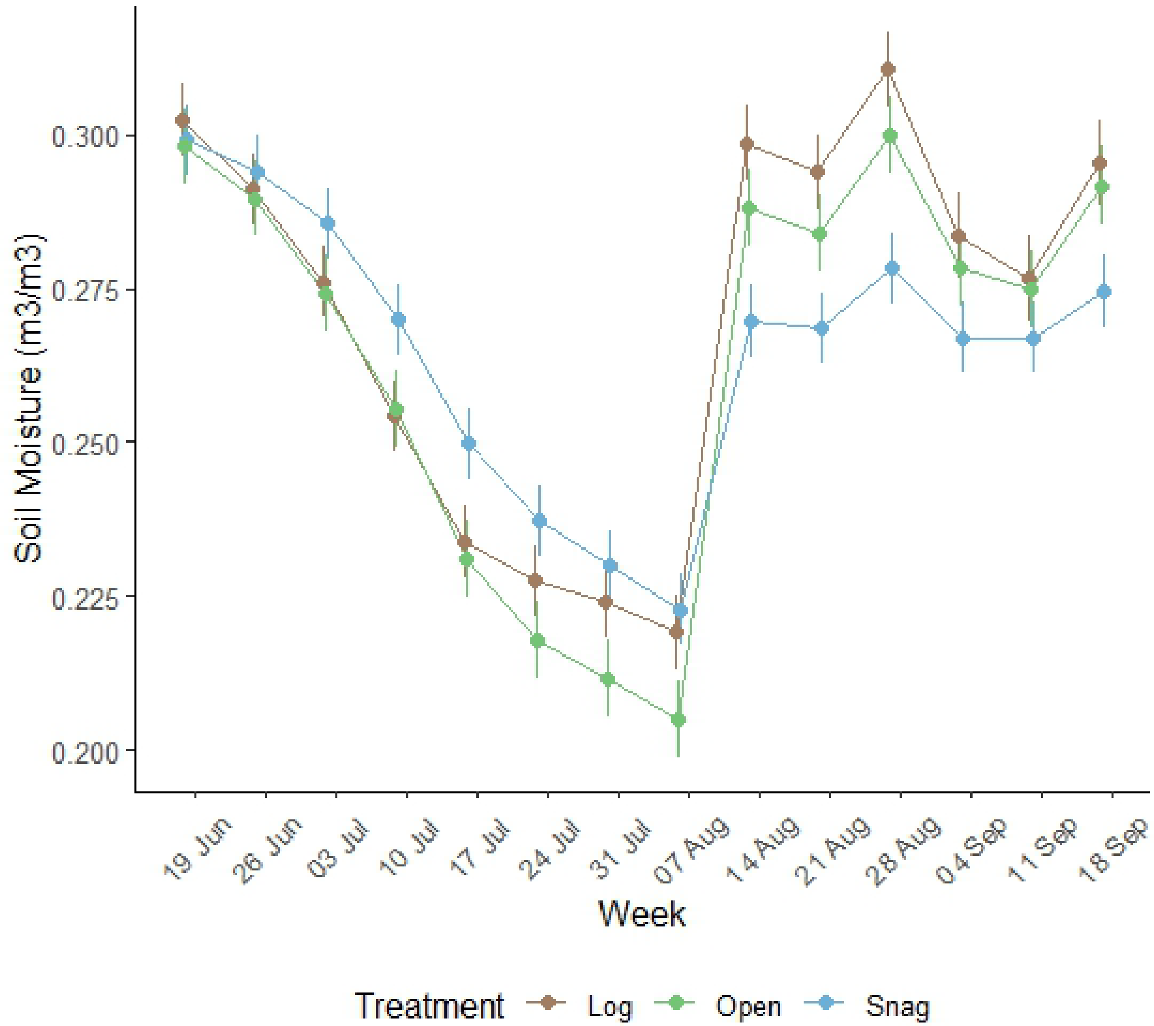

Figure 9. Distribution of mean volumetric soil moisture across treatments during observational period (19-06-2023 through 18-09-2023).

CHAPTER 4

DISCUSSION

We conducted this study with a larger goal to improve aspen seedling outplanting performance and provide managers with guidance regarding aspen planting in post-fire environments. Two key take-aways emerged from this study: 1) snags remaining after wildfire have utility as nurse structures for planted seedlings, likely owing to the high soil moisture in snag root pockets; 2) planted aspen seedlings perform well in high severity fire footprints across microsite types. This research may be useful for managers who wish to use nursery-grown aspen seedlings to accomplish strategic reforestation, assisted migration, and infusion of genetic diversity.

Using deadwood as nurse structures

Survival

Primary limitations to seedling survival are lack of soil moisture and herbivory (Howe et al., 2020), but planting in snag microsites seems to have mitigated these threats. Seedlings planted in the root pockets of snags had higher survival probability across all experimental fire footprints (Fig. 3), likely due to greater soil moisture and reduced ungulate browsing at these microsites compared to open spaces. Snag pockets had more volumetric soil moisture content during the pre-monsoon growing season, which is a critical period for seedlings (Fig. 9). The higher soil moisture is likely due to the concavity of root pockets, which allowed moisture to be collected from the surrounding environment and/or wicked downward by snags. The diameter of snags, which generally was larger than log height across the sites, may also explain the variations in soil moisture content. Goldin and Brookhouse (2015) found a positive relationship between diameter of deadwood and adjacent temperate woodland plant moisture content, suggesting that larger structures protect understory plants from extreme moisture loss and increase survival during drought. Interestingly, patterns were reversed in our study following monsoonal precipitation when probes near snags logged lower soil moisture content than at other treatments, suggesting soil moisture access is more critical during the dry season than during wetter periods. Microsites at structures were also less attractive to browsing ungulates (Fig. 10), which select for areas of high forage availability (Ordway and Krausman, 1986) and selectively browse for plants high in protein and energy (Burney and Jacobs, 2013). Ungulates may restrict their search to more resource dense patches, which would be open areas in our study sites, where more rewarding forage is found and decrease search effort in resource sparse areas, such as by snags or logs where less forage grows.

Contrary to our hypothesis that other nurse structures would aid in seedling success, downed logs did not provide beneficial microsites for aspen seedlings. This may be because soil moisture content near logs was low and similar to open areas during the pre-monsoonal growing season. Dhar et al. explains that logs can act as a barrier between the snow and the soil surface, thereby reducing infiltration during the spring melt (Dhar et al., 2022). We also speculate that the blackened logs could have attracted heat which would cause soil to dry, offsetting gains in soil moisture due to the shade of the log. More research would be needed to explain this further. The use of nurse logs by forest managers has been a practice in many conifer plantings (Castro et al., 2011; Haffey et al., 2018) and in many cases of natural establishment, deadwood facilitates regeneration, reducing competition and providing shade (Birch and Lutz, 2023). However, for the planted aspen seedlings in this study, logs were associated with lower survival than both snags and open areas, meaning the type of nurse structure matters.

Our research identified snag pockets as microsites with greater seedling survival. This demonstrates the importance of snag retention in post-fire landscapes, beyond how deadwood is currently managed. Following high severity wildfire, management options include removing standing deadwood by salvage logging (Innes et al., 2006), or leaving snags on the landscape to foster wildlife habitat (Bagne et al., 2008; Hutto, 2006). However, the utility of post-fire snags as nurse structures benefited aspen seedlings and could be extended to other restoration plantings.

Growth

Survival is an initial indicator of seedling success, but growth is required for long term survival and persistence. According to our model selection, the only parameter considered in our Horton and Pack Creek models is the presence of Vexar, and for the Luna fire, the null model was selected. This result could be due to a short observational period. Our study only looked at relative growth of aboveground vegetation within one year, and long-term assessments are needed to accurately predict trends across fires and treatment types. Treatment structures could have an influence as time passes. Aspen growth is related to the availability of light as well as water and nutrients, which combined, determines photosynthetic rates, thus more light results in larger seedlings (Hemming and Lindroth, 1999). Direct light access of seedlings in open spaces might be advantageous once the seedlings are well established as increased light can also promote asexual reproduction, meaning more potential for vegetative suckering in the future (Bates et al., 2006). Our research also did not consider root growth within the first growing season, or extension of roots beyond the potting medium, and we recommend that future projects include elements to further understand and predict seedling development.

We expected azimuth from nurse structures to influence aspen seedling growth. Instead, cardinal direction was not included as a parameter in any of our relative growth models. This finding is in contrast to the importance of azimuth for conifer seedlings, which often require north-facing aspects and shade objects for outplanting success (Marshall et al., 2023). In fact, when observing soil moisture across direction we see that north and south-facing azimuths yield significantly more volumetric soil moisture content than east or west-facing azimuths. This finding could indicate that reforestation can be expanded at sites with the use of aspen seedlings, that might have once been confined to certain aspects if outplanting with conifer seedlings.

Tree sheltering & herbivory

Herbivory by ungulates inhibits early seedling development (Barton and Hanley, 2013) and stunts seedling vigor (Burney and Jacobs, 2013) which can challenge forest restoration projects. Although the primary sources of mortality in our research were unknown or desiccation, ungulate herbivory has greatly influenced success of past aspen restoration efforts and is an important consideration for planting prescriptions (Britton et al., 2016). We explored one option to help limit ungulate related damage when outplanting: protecting seedlings with individual tree sheltering. The presence of Vexar reduced the likelihood of ungulate herbivory across all fires and most treatments and is a successful tool where browse occurs. This aligns with past observations by Thyroff et al. (2022) in which sheltered seedling had significantly less ungulate browse compared to unsheltered seedlings. Additionally, outplanting in recent fire footprints may have deterred browsing. Timing planting efforts immediately following a disturbance is a strategy to reduce ungulate browse since animals are not quite accustomed to foraging in the area (Burney and Jacobs, 2013).

Before application of Vexar, management should consider specific project goals, cost, time, and labor. Vexar did require regular maintenance. From our field observations, 63% of the shelters required reinstallation when visiting sites one year later. In some cases, shelters collapsed from winter snowfall and surviving seedlings exhibited altered growth forms, such as deformed or crooked stems. We used a single bamboo stake per mesh tube and using two bamboo stakes per mesh tubing may prevent collapse and reduce maintenance. Solid-wall tubing has also been documented to outperform mesh tubing at preventing browse in oak species (Thyroff et al., 2022). Additionally, Vexar, as expected, does not prevent insect herbivory.

Planting in high severity fire footprints.

Increasing occurrence and severity of wildfire in the Southwest (Abatzoglou et al., 2017) has produced fire footprints in need of reforestation and these sites provided ideal planting opportunities for aspen seedlings. Aspen regeneration and establishment are often constrained by the presence of dense conifer canopies, and seedlings perform well where overstory species are removed (Berrill et al., 2017). Additionally, managers might be motivated to reforest high severity fire footprints out of concern for forest loss and potential type conversion (Coop, 2023; Coop et al., 2020). Planting aspen in previously conifer-dominated areas would be an example of directing type conversion to ensure that forest ecosystem services are maintained, and successful aspen establishment could influence future fire behavior. Because of its unique relationship to wildfire, aspen has been labeled a natural fuel break (Fechner and Barrows, 1976) since aspen are less likely to support crown fire (Nesbit et al., 2023). Introducing or promoting aspen as a desired species can be a strategic way to limit the risk of severe crown fire.

Limitations

Many of the guidelines from our research can be deployed in outplanting projects, although uncertainties still exist. Notably, this research looked at success of treatments within a short window of one growing season. Longer assessments would refine our understanding of seedling survival and growth by treatment and under varying climatic conditions. For example, planting near snags benefits young seedlings, but snags have the potential to be blown down and uproot seedlings in the future. As seedlings establish and sucker, snags might also limit root expansion. Since we witnessed the most growth of seedlings in open spaces, these sites might be more advantageous for vegetative suckering. Our research also did not account for any abnormal growth in root systems due to Vexar which has been studied in conifer species (Engeman et al., 1997) and long-term assessments should account for this.

Additionally, this research does not encompass climatic variation such as years with drought or variations to winter snowpack which affect aspen stand health (Love et al., 2019). Our focused study area of the Colorado Plateau is also strongly influenced by monsoonal trends and does not reflect precipitation patterns found in other forest systems. A thorough monitoring of this project for several years and in different ecoregions is required to make temporal and spatial inferences.

Planting quality

This project was dependent on volunteer planters and for many, this was their first experience tree planting. Inconsistency and poor planting quality across sites resulted in some seedlings having exposed roots. It is important to control planting quality by ensuring sufficient training and instruction to all planters and checking planting quality.

Soil moisture

Soil moisture measurements were only conducted in the Luna fire. Seedling remeasurements in the same site were conducted in early August, at the beginning of the monsoonal growing season. Therefore, seedling assessments do not represent soil moisture data captured from mid-August to the end of the soil moisture observational period (September 2023). Installing data logging stations at all sites would help further explain the relationships between soil moisture, treatment, seedling survival, and growth.

Future research

Since western forests are vulnerable to changing climates and species may not be able to efficiently adapt, assisted population migration, range expansion, and species migration have been suggested as restoration tools (Handler et al., 2018). The outplanting techniques illustrated in this research can be applied to movement of aspen and given its wide distribution across North America, aspen seed can be sourced from maternal trees with more drought-tolerant characteristics (Gray et al., 2011). The distance to the nearest established aspen stand from the first planting unit in the Pack Creek fire was 0.5 km, which was the furthest from an established stand. This unit yielded the most seedling survival, which indicates that migration of aspen beyond its current range might be successful. To fully understand how nursery-grown aspen stock might perform in the context of assisted migration, further experiments are needed and are an important area of research.

Conclusion

Recent changes in wildfire frequency and severity pose challenges for restoration practitioners to explore new reforestation methods. In this study, we demonstrated how elements of post-fire landscapes, notably snags as nurse structures, can be applied to benefit aspen seedlings and outplanting efforts. While the use of nursery grown aspen seedlings has been rare in the West, this research provides an example of active restoration. Western aspen populations are declining, which could drastically change understory species composition, biodiversity, and native wildlife foraging habitat. Aspen forest loss can also impact experiences for humans, especially since the species is considered a natural fuel break. Outplanting seedlings is one restoration method that can both strengthen existing populations for changing conditions and allow managers to promote aspen in novel locations.

REFERENCES

Abatzoglou, J.T., Kolden, C.A., Williams, A.P., Lutz, J.A., Smith, A.M.S., 2017. Climatic influences on interannual variability in regional burn severity across western US forests. Int. J. Wildland Fire 26, 269. https://doi.org/10.1071/WF16165

Abatzoglou, J.T., Williams, A.P., 2016. Impact of anthropogenic climate change on wildfire across western US forests. Proc. Natl. Acad. Sci. U.S.A. 113, 11770–11775. https://doi.org/10.1073/pnas.1607171113

Akaike, H., 1974. A new look at the statistical model identification. IEEE Trans. Automat. Contr. 19, 716–723. https://doi.org/10.1109/TAC.1974.1100705

Anderegg, L.D.L., Anderegg, W.R.L., Abatzoglou, J., Hausladen, A.M., Berry, J.A., 2013. Drought characteristics' role in widespread aspen forest mortality across Colorado, USA. Glob Change Biol 19, 1526–1537. https://doi.org/10.1111/gcb.12146

Bagne, K.E., Purcell, K.L., Rotenberry, J.T., 2008. Prescribed fire, snag population dynamics, and avian nest site selection. Forest Ecology and Management 255, 99–105. https://doi.org/10.1016/j.foreco.2007.08.024

Bartoń, K., 2013. MuMIn: Multi-model inference, R package version 1.10.0.

Barton, K.E., Hanley, M.E., 2013. Seedling–herbivore interactions: insights into plant defence and regeneration patterns. Annals of Botany 112, 643–650. https://doi.org/10.1093/aob/mct139

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting Linear Mixed-Effects Models Using **lme4**. J. Stat. Soft. 67. https://doi.org/10.18637/jss.v067.i01

Bates, J.D., Miller, R.F., Davies, K.W., 2006. Restoration of Quaking Aspen Woodlands Invaded by Western Juniper. Rangeland Ecology & Management 59, 88–97. https://doi.org/10.2111/04-162R2.1

Berrill, J.-P., Dagley, C.M., Coppeto, S.A., Gross, S.E., 2017. Curtailing succession: Removing conifers enhances understory light and growth of young aspen in mixed stands around Lake Tahoe, California and Nevada, USA. Forest Ecology and Management 400, 511–522. https://doi.org/10.1016/j.foreco.2017.06.001

Birch, J.D., Lutz, J.A., 2023. Spatial patterns of seedlings dominated by proximity to deadwood and adult trees for Pinus flexilis and Pinus longaeva. Forest Ecology and Management 540, 121049. https://doi.org/10.1016/j.foreco.2023.121049

Boulanger, Y., Sirois, L., 2006. Postfire dynamics of black spruce coarse woody debris in northern boreal forest of Quebec. Can. J. For. Res. 36, 1770–1780. https://doi.org/10.1139/x06-070

Bretfeld, M., Franklin, S.B., Peet, R.K., 2016. A multiple-scale assessment of long-term aspen persistence and elevational range shifts in the Colorado Front Range. Ecol Monogr 86, 244–260. https://doi.org/10.1890/15-1195.1

Britton, J.M., DeRose, R.J., Mock, K.E., Long, J.N., 2016. Herbivory and advance reproduction influence quaking aspen regeneration response to management in southern Utah, USA. Can. J. For. Res. 46, 674–682. https://doi.org/10.1139/cjfr-2016-0010

Burney, O.T., Jacobs, D.F., 2013. Ungulate herbivory of boreal and temperate forest regeneration in relation to seedling mineral nutrition and secondary metabolites. New Forests 44, 753–768. https://doi.org/10.1007/s11056-013-9381-9

Burnham, K.P., Anderson, D.R. (Eds.), 2004. Model Selection and Multimodel Inference. Springer New York, New York, NY. https://doi.org/10.1007/b97636

Calder, W.J., Horn, K.J., St. Clair, S.B., 2011. Conifer expansion reduces the competitive ability and herbivore defense of aspen by modifying light environment and soil chemistry. Tree Physiology 31, 582–591. https://doi.org/10.1093/treephys/tpr041

Castro, J., Allen, C.D., Molina-Morales, M., Marañón-Jiménez, S., Sánchez-Miranda, Á., Zamora, R., 2011. Salvage Logging Versus the Use of Burnt Wood as a Nurse Object to Promote Post-Fire Tree Seedling Establishment. Restoration Ecology 19, 537–544. https://doi.org/10.1111/j.1526-100X.2009.00619.x

Chambers, J.M. (Ed.), 1992. Statistical models in S. Wadsworth & Brooks, Pacific Grove, Calif.

Collins, W.B., Urness, P.J., 1983. Feeding Behavior and Habitat Selection of Mule Deer and Elk on Northern Utah Summer Range. The Journal of Wildlife Management 47, 646. https://doi.org/10.2307/3808601

Coop, J.D., 2023. Postfire futures in southwestern forests: Climate and landscape influences on trajectories of recovery and conversion. Ecological Applications 33, e2725. https://doi.org/10.1002/eap.2725

Coop, J.D., Parks, S.A., Stevens-Rumann, C.S., Crausbay, S.D., Higuera, P.E., Hurteau, M.D., Tepley, A., Whitman, E., Assal, T., Collins, B.M., Davis, K.T., Dobrowski, S., Falk, D.A., Fornwalt, P.J., Fulé, P.Z., Harvey, B.J., Kane, V.R., Littlefield, C.E., Margolis, E.Q., North, M., Parisien, M.-A., Prichard, S., Rodman, K.C., 2020. Wildfire-Driven Forest Conversion in Western North American Landscapes. BioScience 70, 659–673. https://doi.org/10.1093/biosci/biaa061

DeByle, Norbert V., 1990. Aspen ecology and management in the western United States.

DeByle, N.V., Bevins, C.D., Fischer, W.C., 1987. Wildfire Occurrence in Aspen in the Interior Western United States. Western Journal of Applied Forestry 2, 73–76. https://doi.org/10.1093/wjaf/2.3.73

DeByle, N.V., Winokur, R.P., 1985. Aspen: Ecology and management in the western United States (No. RM-GTR-119). U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station, Ft. Collins, CO. https://doi.org/10.2737/RM-GTR-119

Dhar, A., Forsch, K.B.C., Naeth, M.A., 2022. Effects of Coarse Woody Debris on Soil Temperature and Water Content in Two Reconstructed Soils in Reclaimed Boreal Forest. Soil Systems 6, 62. https://doi.org/10.3390/soilsystems6030062

Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliott, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B., Webster, J.R., 2005. Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 3, 479–486. https://doi.org/10.1890/1540-9295(2005)003[0479:LOFSCF]2.0.CO;2

Engeman, R.M., Anthony, R.M., Krupa, H.W., Evans, J., 1997. The effects of Vexar® seedling protectors on the growth and development of lodgepole pine roots. Crop Protection 16, 57–61. https://doi.org/10.1016/S0261-2194(96)00064-6

Fairweather, M.L., Rokala, E.A., Mock, K.E., 2014. Aspen Seedling Establishment and Growth after Wildfire in Central Arizona: An Instructive Case History. Forest Science 60, 703–712. https://doi.org/10.5849/forsci.13-048

Fechner, G.H., Barrows, J.S., 1976. Aspen Stands as Wildfire Fuel Breaks.

Filazzola, A., Lortie, C.J., 2014. A systematic review and conceptual framework for the mechanistic pathways of nurse plants. Global Ecology and Biogeography 23, 1335–1345. https://doi.org/10.1111/geb.12202

Goldin, S.R., Brookhouse, M.T., 2015. Effects of coarse woody debris on understorey plants in a temperate Australian woodland. Applied Vegetation Science 18, 134–142. https://doi.org/10.1111/avsc.12120

Gray, L.K., Gylander, T., Mbogga, M.S., Chen, P., Hamann, A., 2011. Assisted migration to address climate change: recommendations for aspen reforestation in western Canada. Ecological Applications 21, 1591–1603. https://doi.org/10.1890/10-1054.1

Green, M.B., Fraver, S., Lutz, D.A., Woodall, C.W., D'Amato, A.W., Evans, D.M., 2022. Does deadwood moisture vary jointly with surface soil water content? Soil Science Soc of Amer J 86, 1113–1121. https://doi.org/10.1002/saj2.20413

Guiterman, C.H., Gregg, R.M., Marshall, L.A.E., Beckmann, J.J., Van Mantgem, P.J., Falk, D.A., Keeley, J.E., Caprio, A.C., Coop, J.D., Fornwalt, P.J., Haffey, C., Hagmann, R.K., Jackson, S.T., Lynch, A.M., Margolis, E.Q., Marks, C., Meyer, M.D., Safford, H., Syphard, A.D., Taylor, A., Wilcox, C., Carril, D., Enquist, C.A.F., Huffman, D., Iniguez, J., Molinari, N.A., Restaino, C., Stevens, J.T., 2022. Vegetation type conversion in the US Southwest: frontline observations and management responses. fire ecol 18, 6, s42408-022-00131-w. https://doi.org/10.1186/s42408-022-00131-w

Haffey, C., Sisk, T.D., Allen, C.D., Thode, A.E., Margolis, E.Q., 2018. Limits to Ponderosa Pine Regeneration following Large High-Severity Forest Fires in the United States Southwest. fire ecol 14, 143–163. https://doi.org/10.4996/fireecology.140114316

Handler, S., Pike, C., Clair, B., 2018. Assisted Migration. USDA Forest Service Climate Change Resource Center.

Hemming, J.D.C., Lindroth, R.L., 1999. [No title found]. Journal of Chemical Ecology 25, 1687–1714. https://doi.org/10.1023/A:1020805420160

Howe, A.A., Landhäusser, S.M., Burney, O.T., Long, J.N., Violett, R.D., Mock, K.E., 2020. Exploring seedling-based aspen (Populus tremuloides) restoration near range limits in the Intermountain West, USA. Forest Ecology and Management 476, 118470. https://doi.org/10.1016/j.foreco.2020.118470

Huang, C., Anderegg, W.R.L., 2012. Large drought-induced aboveground live biomass losses in southern R ocky M ountain aspen forests. Glob Change Biol 18, 1016–1027. https://doi.org/10.1111/j.1365-2486.2011.02592.x

Hutto, R.L., 2006. Toward Meaningful Snag-Management Guidelines for Postfire Salvage Logging in North American Conifer Forests. Conservation Biology 20, 984–993. https://doi.org/10.1111/j.1523-1739.2006.00494.x

InciWeb, 2023.
Innes, J.C., North, M.P., Williamson, N., 2006. Effect of thinning and prescribed fire restoration treatments on woody debris and snag dynamics in a Sierran old-growth, mixed-conifer forest. Can. J. For. Res. 36, 3183–3193. https://doi.org/10.1139/x06-184
Kay, C.E., Bartos, D.L., 2000. Ungulate Herbivory on Utah Aspen: Assessment of Long-Term Exclosures. Journal of Range Management 53, 145. https://doi.org/10.2307/4003274
King, C.M., Landhäusser, S.M., 2018. Regeneration dynamics of planted seedling-origin aspen (Populus tremuloides Michx.). New Forests 49, 215–229. https://doi.org/10.1007/s11056-017-9614-4
Kouki, J., Arnold, K., Martikainen, P., 2004. Long-term persistence of aspen – a key host for many threatened species – is endangered in old-growth conservation areas in Finland. Journal for Nature Conservation 12, 41–52. https://doi.org/10.1016/j.jnc.2003.08.002
Kreider, M.R., Yocom, L.L., 2021. Aspen seedling establishment, survival, and growth following a high-severity wildfire. Forest Ecology and Management 493, 119248. https://doi.org/10.1016/j.foreco.2021.119248
LaConte, J., 2020. Should aspens replace lodgepole in local forests? The Aspen Times.
Landhäusser, S.M., Pinno, B.D., Mock, K.E., 2019. Tamm Review: Seedling-based ecology, management, and restoration in aspen (Populus tremuloides). Forest Ecology and Management 432, 231–245. https://doi.org/10.1016/j.foreco.2018.09.024
Long, J.N., Mock, K., 2012. Changing perspectives on regeneration ecology and genetic diversity in western quaking aspen: implications for silviculture. Can. J. For. Res. 42, 2011–2021. https://doi.org/10.1139/x2012-143
Love, D.M., Venturas, M.D., Sperry, J.S., Brooks, P.D., Pettit, J.L., Wang, Y., Anderegg, W.R.L., Tai, X., Mackay, D.S., 2019. Dependence of Aspen Stands on a Subsurface Water Subsidy: Implications for Climate Change Impacts. Water Resources Research 55, 1833–1848. https://doi.org/10.1029/2018WR023468
Mankin, J.S., 2021. NOAA Drought Task Force Report on the 2020–2021 Southwestern U.S. Drought. https://doi.org/10.25923/J0RJ-AN85
Marsh, C., Crockett, J.L., Krofcheck, D., Keyser, A., Allen, C.D., Litvak, M., Hurteau, M.D., 2022. Planted seedling survival in a post-wildfire landscape: From experimental planting to predictive probabilistic surfaces. Forest Ecology and Management 525, 120524. https://doi.org/10.1016/j.foreco.2022.120524
Marsh, R.E., Koehler, A.E., Salmon, T., 1990. Exclusionary methods and materials to protect plants from pest mammals--A Review, in: Proceedings of the Fourteenth Vertebrate Pest Conference 1990. Presented at the VERTEBRATE PEST CONFERENCE, Univ. of Calif., Davis, pp. 174–180.
Marshall, L.A.E., Fornwalt, P.J., Stevens-Rumann, C.S., Rodman, K.C., Rhoades, C.C., Zimlinghaus, K., Chapman, T.B., Schloegel, C.A., 2023. North-facing aspects, shade objects, and microtopographic depressions promote the survival and growth of tree seedlings planted after wildfire. fire ecol 19, 26. https://doi.org/10.1186/s42408-023-00181-8

Martens, L.A., Landhäusser, S.M., Lieffers, V.J., 2007. First-year growth response of cold-stored, nursery-grown aspen planting stock. New Forests 33, 281–295. https://doi.org/10.1007/s11056-006-9027-2

Martin, T.E., Maron, J.L., 2012. Climate impacts on bird and plant communities from altered animal–plant interactions. Nature Clim Change 2, 195–200. https://doi.org/10.1038/nclimate1348

McCool, S.F., 1998. Quaking aspen and the human experience: dimensions, issues, and challenges, in: Sustaining Aspen in Western Landscapes: Symposium Proceedings. RMRS.

Mock, K.E., Rowe, C.A., Hooten, M.B., Dewoody, J., Hipkins, V.D., 2008. Clonal dynamics in western North American aspen (*Populus tremuloides*). Molecular Ecology 17, 4827–4844. https://doi.org/10.1111/j.1365-294X.2008.03963.x

Nesbit, K.A., Yocom, L.L., Trudgeon, A.M., DeRose, R.J., Rogers, P.C., 2023. Tamm review: Quaking aspen's influence on fire occurrence, behavior, and severity. Forest Ecology and Management 531, 120752. https://doi.org/10.1016/j.foreco.2022.120752

Ordway, L.L., Krausman, P.R., 1986. Habitat Use by Desert Mule Deer. The Journal of Wildlife Management 50, 677. https://doi.org/10.2307/3800980

PRISM Climate Data, 2023.

R, 2023. R Core Team. RStudio: Integrated Development for R.

Rehfeldt, G.E., Ferguson, D.E., Crookston, N.L., 2009. Aspen, climate, and sudden decline in western USA. Forest Ecology and Management 258, 2353–2364. https://doi.org/10.1016/j.foreco.2009.06.005

Romme, W.H., Turner, M.G., Wallace, L.L., Walker, J.S., 1995. Aspen, Elk, and Fire in Northern Yellowstone Park. Ecology 76, 2097–2106. https://doi.org/10.2307/1941684

Schier, G.A., Jones, J.R., Winokur, R.P., 1985. Vegetative regeneration in Aspen: Ecology and Management in the Western United States (General Technical Report No. RM-119). USDA Forest Service, Fort Collins, CO.

Seager, S.T., Eisenberg, C., St. Clair, S.B., 2013. Patterns and consequences of ungulate herbivory on aspen in western North America. Forest Ecology and Management 299, 81–90. https://doi.org/10.1016/j.foreco.2013.02.017

Shinneman, D.J., Baker, W.L., Rogers, P.C., Kulakowski, D., 2013. Fire regimes of quaking aspen in the Mountain West. Forest Ecology and Management 299, 22–34. https://doi.org/10.1016/j.foreco.2012.11.032

Sienkiewicz, T., 2020. Aspens to be planted at Barney Ford open space site for wildfire protection and tree diversity. Summit Daily.

Singer, J., Turnbull, R., Foster, M., Bettigole, C., Frey, B., Downey, M., Covey, K., Ashton, M., 2019. Sudden Aspen Decline: A Review of Pattern and Process in a Changing Climate. Forests 10, 671. https://doi.org/10.3390/f10080671

South, D.B., Blake, J.I., 1994. Top-pruning increases survival of pine seedlings. Presented at the Highlights of Agricultural Research, Alabama Agricultural Experiment Station.

Stevens, J.T., Haffey, C.M., Coop, J.D., Fornwalt, P.J., Yocom, L., Allen, C.D., Bradley, A., Burney, O.T., Carril, D., Chambers, M.E., Chapman, T.B., Haire, S.L., Hurteau, M.D., Iniguez, J.M., Margolis, E.Q., Marks, C., Marshall, L.A.E., Rodman, K.C., Stevens-Rumann, C.S., Thode, A.E., Walker, J.J., 2021. Tamm Review: Postfire landscape management in frequent-fire conifer forests of the southwestern United States. Forest Ecology and Management 502, 119678. https://doi.org/10.1016/j.foreco.2021.119678

Stohlgren, T.J., Coughenour, M.B., Chong, G.W., Binkley, D., Kalkhan, M.A., Schell, L.D., Buckley, D.J., Berry, J.K., 1997. Landscape analysis of plant diversity. Landscape Ecology 12, 155–170. https://doi.org/10.1023/A:1007986502230

Thyroff, E.C., Burney, O.T., Oliet, J.A., Redick, C.H., Jacobs, D.F., 2022. Toward Identifying Alternatives to Fencing for Forest Restoration: Tube Shelters Outperform Mesh Shelters for Deer Browse Protection of Live Oak, Quercus virginiana. Land 11, 966. https://doi.org/10.3390/land11070966

Vanderwel, M.C., Malcolm, J.R., Smith, S.M., 2006. An integrated model for snag and downed woody debris decay class transitions. Forest Ecology and Management 234, 48–59. https://doi.org/10.1016/j.foreco.2006.06.020

Web Soil Survey, 2023.

Williams, M.I., Dumroese, R.K., 2013. Preparing for Climate Change: Forestry and Assisted Migration. Journal of Forestry 111, 287–297. https://doi.org/10.5849/jof.13-016

APPENDICES

Appendix A.

Parameter estimates of aspen seedling survival model for each fire; confidence intervals display (lower, upper) bounds; 95%; ***p < 0.001, ** p < 0.01, *p < 0.05

Horton								
Independent Predictor Variables	Estimate	Standard Error	95% Confidence Interval				Pr(>|z|)	
Open (intercept)	0.013	0.493	(	0.218	1.645	)	0.978	
Log	-0.547	0.343	(	0.889	3.439	)	0.110	
Snag	0.689	0.309	(	1.711	6.553	)	0.026	*
Vexar	0.387	0.263	(	0.912	2.548	)	0.141	
Planter 1	-1.776	0.526	(	0.054	0.433	)	0.001	***
Planter 2	-0.337	0.470	(	0.281	1.765	)	0.473	
Planter 3	-1.377	0.742	(	0.053	1.010	)	0.064	
Planter 4	-1.207	0.732	(	0.063	1.167	)	0.099	
Planter 5	-0.155	0.556	(	0.286	2.538	)	0.781	
Planter 6	-0.280	0.650	(	0.206	2.659	)	0.667	
Luna								
Independent Predictor Variables	Estimate	Standard Error	95% Confidence Interval				Pr(>|z|)	
Open (intercept)	-0.463	0.265	(	0.373	1.072	)	0.080	
Log	-0.171	0.295	(	0.468	1.497	)	0.562	
Snag	0.308	0.309	(	0.734	2.479	)	0.320	
Vexar	0.594	0.237	(	1.143	2.898	)	0.012	*
Pack Creek								
Independent Predictor Variables	Estimate	Standard Error	95% Confidence Interval				Pr(>|z|)	
Open (intercept)	0.763	0.209	(	1.440	3.339	)	0.000	***
Log	-0.401	0.265	(	0.393	1.106	)	0.130	
Snag	0.503	0.308	(	0.898	2.984	)	0.103	
Vexar	0.831	0.226	(	1.480	3.602	)	0.000	***

Appendix B.

Survival model parameters selection ranked by AIC index values. ΔAIC represents the difference between the statistically best fit AIC model and the selected model.

Horton

Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
β_1 (Structure) + β_2 (Vexar) + β_4 (Planter)	**0.0133**	**11**	**410.328**	**0.000**	**0.389**
β_1 (Structure) + β_4 (Planter)	0.1464	10	410.378	0.050	0.379
β_1 (Structure) + β_2 (Vexar) + β_4 (Planter) + β_1 (Structure): β_2 (Vexar)	0.1461	13	412.423	2.095	0.136
β_1 (Structure) + β_2 (Vexar) + β_3 (Direction) + β_4 (Planter)	0.7388	14	414.919	4.591	0.039
β_1 (Structure) + β_3 (Direction) + β_4 (Planter)	0.9018	13	415.252	4.925	0.033
β_1 (Structure) + β_2 (Vexar) + β_3 (Direction) + β_4 (Planter) + β_1 (Structure): β_2 (Vexar)	0.5546	16	416.723	6.395	0.016
β_2 (Vexar) + β_4 (Planter)	0.1317	9	419.944	9.616	0.003
β_4 (Planter)	0.3082	8	420.193	9.865	0.003
β_1 (Structure)	-0.4620	4	423.474	13.146	0.001
β_1 (Structure) + β_2 (Vexar)	-0.6098	5	423.836	13.508	0.000
β_1 (Structure) + β_2 (Vexar) + β_1 (Structure): β_2 (Vexar)	-0.4453	7	426.444	16.116	0.000
β_3 (Direction) + β_4 (Planter)	0.2482	12	427.582	17.254	0.000
β_2 (Vexar) + β_3 (Direction) + β_4 (Planter)	0.0865	13	427.607	17.279	0.000
β_1 (Structure) + β_3 (Direction)	0.4761	7	428.505	18.177	0.000
β_1 (Structure) + β_2 (Vexar) + β_3 (Direction)	0.3083	8	428.654	18.326	0.000

β_1 (Structure) + β_2 (Vexar) + β_3 (Direction) + β_1 (Structure): β_2 (Vexar)	0.2266	10	431.096	20.769	0.000
Null	-0.3919	2	444.070	33.742	0.000
Luna					
Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
β2 (Vexar)	-0.419	3	452.190	0.000	0.227
β1 (Structure) + β2 (Vexar) + β3 (Direction)	-0.046	8	452.566	0.375	0.188
β1 (Structure) + β2 (Vexar)	**-0.463**	**5**	**453.580**	**1.390**	**0.113**
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar)	-0.223	10	453.847	1.657	0.099
β2 (Vexar) + β3 (Direction)	-0.337	7	453.996	1.806	0.092
β1 (Structure) + β2 (Vexar) + β1 (Structure): β2 (Vexar)	-0.496	7	454.710	2.519	0.064
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β2 (Vexar): β3 (Direction)	-0.486	13	454.837	2.646	0.060
β1 (Structure) + β3 (Direction)	0.200	7	456.133	3.943	0.032
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β2 (Vexar): β3 (Direction)	-0.282	12	456.467	4.277	0.027
Null	-0.135	2	456.581	4.391	0.025
Pack Creek					
Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
β_1 (Structure) + β_2 (Vexar)	**0.763**	**5**	**510.287**	**0.000**	**0.653**
β_1 (Structure) + β_2 (Vexar) + β_1 (Structure): β_2 (Vexar)	0.774	7	512.900	2.614	0.177
β_1 (Structure) + β_2 (Vexar) + β_3 (Direction)	1.170	8	514.143	3.856	0.095
β_1 (Structure) + β_2 (Vexar) + β_3 (Direction) + β_1 (Structure): β_2 (Vexar)	0.970	10	516.568	6.282	0.028
β_2 (Vexar)	0.730	3	516.678	6.392	0.027

β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β3 (Direction)	1.263	11	519.467	9.180	0.007
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β2 (Vexar): β3 (Direction)	1.146	12	520.316	10.029	0.004
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β1 (Structure): β3 (Direction)	0.963	13	520.626	10.339	0.004
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β2 (Vexar): β3 (Direction)	1.022	13	522.051	11.764	0.002
β1 (Structure)	1.138	4	522.194	11.908	0.002
β2 (Vexar) + β3 (Direction)	0.628	7	522.423	12.136	0.002
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β1 (Structure): β3 (Direction)	1.226	15	525.740	15.453	0.000
β1 (Structure) + β3 (Direction)	1.647	7	525.845	15.558	0.000
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β1 (Structure): β3 (Direction) + β2 (Vexar): β3 (Direction)	1.011	16	526.404	16.118	0.000
β1 (Structure) + β2 (Vexar) + β3 (Direction) + β1 (Structure): β2 (Vexar) + β1 (Structure): β3 (Direction) + β2 (Vexar): β3 (Direction) + β1 (Structure): β2 (Vexar): β3 (Direction)	0.956	19	528.758	18.472	0.000
β2 (Vexar) + β3 (Direction) + β2 (Vexar): β3 (Direction)	0.584	11	528.910	18.624	0.000
Null	1.112	2	530.503	20.216	0.000

Appendix C.

Parameter estimates for aspen seedling growth model; confidence intervals display (lower, upper) bounds; 95%; ***p < 0.001, ** p < 0.01, *p < 0.05

Independent Predictor Variables	Mean	Standard Error	95% Confidence Interval				Pr(>|t|)	
Horton								
(Intercept)	0.904	0.127	(	1.927	3.164	)	0.000	***
Vexar	-0.312	0.169	(	0.526	1.019	)	0.067	
Luna								
(Intercept)	0.197	0.048	(	1.090	1.337	)	0.002	
Pack Creek								
(Intercept)	0.605	0.066	(	1.609	2.087	)	0.000	***
Vexar	0.205	0.082	(	1.046	1.441	)	0.013	*

Appendix D.

Relative growth model parameters selection ranked by AIC index values. ΔAIC represents the difference between the statistically best fit AIC model and the selected model.

Horton					
Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
Null	0.729	3	375.437	0.000	0.523
β_2 (Vexar)	**0.904**	**4**	**375.890**	**0.453**	**0.417**
Luna					
Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
Null	**0.197**	**3**	**246.680**	**0.000**	**0.912**
β_2 (Vexar)	0.196	4	251.849	5.169	0.069
Pack Creek					
Parameters	Intercept	Degrees of Freedom	AIC	ΔAIC	Weight
β_2 (Vexar)	**0.605**	**4**	**838.305**	**0.000**	**0.609**
Null	0.72	3	839.33	1.02	0.37

Appendix E.

Parameter estimates for herbivory model; confidence intervals display (lower, upper) bounds; 95%; $^{***}p < 0.001$, $^{**} p < 0.01$, $^{*}p < 0.05$

Horton					
Independent Predictor Variables	Mean	Standard Error	95% Confidence Interval	Pr(>|t|)	
Open (Intercept)	-2.031	0.376	(0.058 0.258)	0.000	***
Log	-0.508	0.708	(0.126 2.226)	0.473	
Snag	-0.246	0.646	(0.198 2.661)	0.703	
Vexar	-0.310	0.601	(0.210 2.335)	0.605	
Log: Vexar	0.451	0.969	(0.243 11.619)	0.641	
Snag: Vexar	0.023	0.952	(0.156 6.922)	0.981	
Luna					
Independent Predictor Variables	Mean	Standard Error	95% Confidence Interval	Pr(>|t|)	
Open (Intercept)	-0.783	0.302	(0.246 0.812)	0.009	**
Log	0.090	0.399	(0.502 2.415)	0.822	
Snag	-0.079	0.407	(0.415 2.067)	0.845	
Vexar	-0.481	0.436	(0.260 1.448)	0.270	
Log: Vexar	-0.134	0.620	(0.256 2.937)	0.829	
Snag: Vexar	-0.161	0.646	(0.235 2.991)	0.803	
Pack Creek					
Independent Predictor Variables	Mean	Standard Error	95% Confidence Interval	Pr(>|t|)	
Open (Intercept)	0.044	0.298	(0.581 1.886)	0.882	
Log	0.051	0.429	(0.452 2.451)	0.906	
Snag	-0.496	0.454	(0.247 1.472)	0.274	
Vexar	-0.080	0.399	(0.421 2.022)	0.842	
Log: Vexar	-2.501	0.892	(0.011 0.406)	0.005	**
Snag: Vexar	-1.106	0.668	(0.087 1.205)	0.098	

Appendix F.

Parameter estimates for soil moisture ANOVA model (top) comparing treatment interacted with week for all probes and Tukey's HSD pairwise comparison by treatment (bottom); confidence intervals display (lower, upper) bounds; 95%; ***p < 0.001, ** p < 0.01, *p < 0.05

Model variable	Mean	Degrees of Freedom	Pr(>F)	
Treatment	0.003	2	0.231	
Week	0.003	1	0.246	
Treatment: Week	0.017	2	0.000	***
Residuals	0.002	1542		

Pairwise Comparisons	Difference of Means	95% CI		Adjusted p-value
Log-Open	-0.005	(-0.011	0.002)	0.257
Snag-Open	-0.004	(-0.011	0.003)	0.352
Snag-Log	0.001	(-0.006	0.007)	0.965

Appendix G.

Parameter estimates for soil moisture ANOVA model (top) comparing direction for probes by snag and log treatments and Tukey's HSD pairwise comparison results (bottom); confidence intervals display (lower, upper) bounds; 95%; ***p < 0.001, ** p < 0.01, *p < 0.05

Model variable	**Mean**	**Degrees of Freedom**	**Pr(>F)**	
Treatment	0.000	1	0.802	
Direction	0.041	3	0.000	***
Residuals	0.002	1035		
Pairwise Comparisons	**Difference of Means**	**95% CI**		**Adjusted p-value**
North-East	0.024	(0.013	0.035)	0.000
South-East	0.017	(0.006	0.028)	0.000
West-East	-0.001	(-0.011	0.009)	0.993
South-North	-0.007	(-0.018	0.004)	0.326
West-North	-0.025	(-0.035	-0.015)	0.000
West-South	-0.018	(-0.028	-0.007)	0.000