# Title: Limited conifer regeneration, but widespread regeneration of aspen seedlings following the Cameron Peak Fire, northwestern Colorado

## Authors: Sarah V. Carter1 and Sarah J. Hart1

## Affiliations:

1 - Department of Forest and Rangeland Stewardship, Colorado State University, 1472 Campus Delivery, Fort Collins, CO 80523, USA

Corresponding author: Sarah J. Hart (email: sarah.hart@colostate.edu)

# Abstract [350 words]

## Background

Changing fire regimes and climate conditions have decreased postfire conifer regeneration densities in many ecosystems across western North America. In the Southern Rocky Mountains, sparse postfire regeneration of conifers has often been accompanied with abundant regeneration of quaking aspen, much of which is from vegetative resprouting. While aspen also produce abundant lightweight seed that can disperse great distances, we know considerably less about the patterns and drivers of postfire establishment of aspen from seed. To understand the potential for severe fire to catalyze shifts to aspen dominated forests, we studied recently burned conifer-dominated forests and asked (1) how does postfire regeneration of conifers and aspen vary across sites? (2) how are aspen and conifer seedlings distributed within sites? and (3) how do site and microsite conditions influence aspen seedling establishment? To answer these questions, we conducted field surveys at 34 sites within the burn scar of the Cameron Peak Fire, northwestern Colorado.

## Results

Two years following fire, we found widespread but highly variable establishment of aspen seedlings, despite the absence of aspen in the pre-fire stand. Aspen seedlings were more likely to occur at higher elevations, where they were often clustered within concave microsites with bryophyte substrates and near large coarse woody debris. Further we found aspen seedlings often outnumbered conifer seedlings and occurred within the same local neighborhood as conifer seedlings.

## Conclusions

Increased fire frequency, severity, and extent in western North America may exceed the capacity of some formerly conifer-dominant forests to maintain historical distributions, while conditions following high severity burns may allow establishment of aspen seedlings both within and outside of pre-fire aspen populations.

# Keywords

aspen, postfire tree regeneration, seedling, *Populus tremuliodes,* sexual establishment

# Background

Over the past two decades, fires have increased in frequency, size, and severity across the western United States (the West) ([Parks and Abatzoglou 2020](#ref-parks2020WarmerDrierFire), [Iglesias et al. 2022](#ref-iglesias2022USFiresBecame)), largely due to climate warming ([Abatzoglou and Williams 2016](#Xb2c9e850c0ec1283a06a871949844aa875af530)), more frequent human ignitions ([Balch et al. 2017](#ref-balch2017HumanstartedWildfiresExpand)), and past land management practices ([Allen et al. 2002](#X38019930a7963793e8077c52e50c6cb3b6ad2bd)). Not only is fire activity changing, numerous studies across the West have shown low postfire regeneration densities, and even complete regeneration failure, is becoming increasingly common ([Stevens-Rumann and Morgan 2019](#X621eff21b20d26397ca583d3c14a7b6bd8eb0d1), [Coop et al. 2020](#X11a688d84acc973314350de60810e0a8ca4666e)). Low postfire regeneration has largely been attributed to more extreme fire behavior and severity. However, the effects of changing fire regimes and climate on postfire recovery are mediated by species-specific traits that drive seed production, germination, and seedling survival and growth, as well as biotic interactions in early seral communities ([Lloret et al. 2005](#ref-lloret2005FireRegenerativeSyndromes), [Hansen et al. 2016](#ref-hansen2016ShiftingEcologicalFilters), [Davis et al. 2023](#ref-davis2023ReducedFireSeverity)). Understanding the complexity of these ecological relationships is not only a central challenge for fire ecology but also has huge implications for society.

There are three primary mechanisms that drive postfire tree regeneration: (1) vegetative resprouting, (2) germination of seed from live tree residuals or the unburned forest, or (3) germination of seeds from canopy seed banks. Because an individual’s regeneration traits in part reflect the system’s history of wildfire, alternations to fire regimes may compromise regeneration mechanisms that previously conferred resilience to wildfire ([Johnstone et al. 2010](#ref-johnstone2010ChangesFireRegime)). For example in lodgepole pine forests, short-intervals (i.e., <30 years) may lead to low post-fire regeneration when trees in the young pre-fire stand have not had enough time to develop serotinous cones ([Turner et al. 2019](#ref-turner2019ShortintervalSevereFire)). However, species rely upon multiple strategies may be more resilient to changing disturbance regimes. For example, quaking aspen (*Populus tremuloides*) may colonize burned areas through rapid suckering from root systems that survive fire or germination of seed from live trees ([Baker 1925](#ref-baker1925AspenCentralRocky)). Retaining both these strategies may allow quaking aspen to exist in a broader range of conditions and increase genetic diversity ([Gill et al. 2017b](#ref-gill2017PopulusTremuloidesSeedling), [Landhäusser et al. 2019](#Xbbcc5e66aa8cf65dff2ed6af463e6939df30e2a)), thereby increasing its adaptive capacity to changing climate and disturbance regimes.

Concurrent with increases in fire frequency and severity, aspen is increasingly playing an important role in the post-fire ecology of many ecosystems across North America ([Turner et al. 2003](#ref-turner2003PostfireAspenSeedling), [Johnstone et al. 2010](#ref-johnstone2010ChangesFireRegime), [Buma and Wessman 2011](#ref-buma2011DisturbanceInteractionsCan), [Kulakowski et al. 2013](#X84239c661d99f8cf6ec0289043ac2ca4f8080d1), [Gill et al. 2017b](#ref-gill2017PopulusTremuloidesSeedling), [Andrus et al. 2021](#ref-andrus2021FutureDominanceQuaking), [Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling), [Nigro et al. 2022](#ref-nigro2022WildfireCatalyzesUpward), [Dawe et al. 2025](#ref-dawe2025SexualVegetativeRecruitment)). In the West most post-fire regeneration has historically been understood to be from root suckering, while in other parts of aspen’s distribution seedling establishment is more widespread ([Long and Mock 2012](#Xe66e7d65f2fb2b8ce2d36bfb5f130522b2bcb04)). These patterns have been hypothesized to occur as a result of differences in climate and the ensuing implications for trade-offs among growth, defense, and reproduction ([Mock et al. 2012](#ref-mock2012WidespreadTriploidyWestern), [DeRose et al. 2015](#ref-derose2015CytotypeDifferencesRadial), [2022](#ref-derose2022PolyploidyGrowthDefense)). However, recent research suggests that post-fire aspen seedling establishment is more common across the West ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling)). Because aspen produce abundant lightweight seed that can be transported several kilometers by strong winds ([Landhäusser et al. 2019](#Xbbcc5e66aa8cf65dff2ed6af463e6939df30e2a)), a greater role of seedling establishment profoundly increases the area across the West where aspen may establish following fire. This has important implications for post-fire ecosystem dynamics, particularly if aspen seedling establishment occurs in areas where conifer regeneration is limited ([Andrus et al. 2021](#ref-andrus2021FutureDominanceQuaking)). Further, once established within a post-fire stand, aspen colonization may trigger change in plant-soil feedbacks that create fuel conditions conduce to lower intensity fire ([Johnstone et al. 2020](#ref-johnstone2020FactorsShapingAlternate), [Nesbit et al. 2023](#ref-nesbit2023TammReviewQuaking), [Harris et al. 2025](#ref-harris2025AspenImpedesWildfire)). These changes to the fire regime combined with aspen’s propensity for rapid post-fire vegetative regeneration can limit regeneration opportunities for conifers and promote the persistence of an aspen-dominated community ([Johnstone et al. 2020](#ref-johnstone2020FactorsShapingAlternate), [Walker et al. 2023](#ref-walker2023ShiftsEcologicalLegacies)).

The potential for aspen to colonize new areas through aspen seedling recruitment is broadly understood to depend on several factors. First, seed must disperse from the unburned edge or fire refugia. While aspen seedling establishment has been observed several kilometers from a live population, most seedling establishment occurs relative close to a parent tree ([Kreider and Yocom 2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)). Given seed arrives, successful germination occurs on moist substrates, which are often located within microdepressions where water collects or near coarse woody debris that provides shade ([Kreider and Yocom 2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1), [Carter et al. 2024](#X518c5e11f1226b7f0686c896ea7fda75925f525)). In addition to adequate soil moisture, germinants require partial or direct sunlight to survive because aspen seeds lack an endosperm ([Landhäusser et al. 2019](#Xbbcc5e66aa8cf65dff2ed6af463e6939df30e2a)). Thus seedling establishment is far more likely to occur at sites that were recently severely burned, where competition for light is limited ([McIlroy and Shinneman 2020](#ref-mcilroy2020PostfireAspenPopulus)). While the root systems of young seedlings are developing and capable of accessing water close to the soil surface they are highly vulnerable to drought ([Landhäusser et al. 2019](#Xbbcc5e66aa8cf65dff2ed6af463e6939df30e2a)) and so survival of young seedlings is highly dependent on post-fire climate conditions ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling)). Notably many of these environmental filters are also limiting to successful post-fire establishment of conifers ([Stevens-Rumann and Morgan 2019](#X621eff21b20d26397ca583d3c14a7b6bd8eb0d1)), although most conifer seeds disperse far shorter distances ([Davis et al. 2023](#ref-davis2023ReducedFireSeverity)). Thus, understanding how seed availability and environmental factors intersect to influence patterns of both aspen and seedling establishment is essential to predicting where fires may catalyze shifts from conifer to aspen dominance.

In this study, we make use of a large wildfire that burned across multiple forest types and environmental conditions to better understand patterns and drivers of conifer and aspen seedling establishment in the West. Specifically, we asked we asked (1) how does postfire regeneration of conifers and aspen vary across sites? (2) how are aspen and conifer seedlings distributed within sites? and (3) how do site and microsite conditions influence aspen seedling establishment? For question 1, we hypothesized that tree regeneration patterns would be driven by regeneration traits. Specifically, we expected that regeneration of most conifers would be limited two years following fire, but lodgepole pine seedlings would be relatively abundant in stands where it was present pre-fire due to the species’ capacity to produce serotinous cones. We also expected that aspen seedlings would be widespread, but at relatively low densities due to the species’ capacity to produce abundant light-weight seeds. For question 2, we expected that both aspen and conifer seedlings would be spatially clustered within sites, due to local variation in microclimate, soils, and seed availability. For question 3, we expected that aspen seedlings would be more likely to occur at high elevations, cooler and wetter aspects, and within sites with conifer seedlings, which likely indicate conditions were moisture availability is conducive to successful germination and initial survival. At the local scale, we hypothesized that aspen seedlings would be associated with microsite features associated with increased moisture availability, including the presence of CWD, concave topography, and substrates with high proportions of bryophyte cover.

# Methods

## Study area

We surveyed tree regeneration across upper montane and subalpine forests that burned in the Cameron Peak Fire located in northern central Colorado (Fig. ). Over the period from August to December 2020, the fire burned across 845 square kilometers ([MTBS Project 2024a](#ref-mtbsproject2024MTBSDataAccessa)). Approximately 39% of the fire burned at high severity ([MTBS Project 2024b](#ref-mtbsproject2024MTBSDataAccess)), including a wide gradient of elevation ranging from about 1645 to 3544 meters above sea level. Generally, winters are cold (1991-2020 mean January daily minimum temperature: -0.2°C) and summers are hot (1991-2020 mean July daily maximum temperature: 23.4°C), but across this elevation gradient there is considerable variation in climate. Mean annual daily temperatures range from -1.4 to 9.3°C and total annual precipitation ranges from 361 to 1145 mm (1991-2020 means) ([PRISM Climate Group 2021](#Xc6a61ae9e57c85d16f010348d89f94dac75b6d1)).

![](data:application/octet-stream;base64,)

**Figure** **:** The study area, study sites (triangles), and the distribution of live and burned aspen. The inset map displays the study area relative to western United States.

Pre-fire forest composition and structure within the burn scar of the Cameron Peak Fire scar also varied with elevation. Lower elevation zones below 2800 m were characterized by dry and mesic montane forests, dominated by ponderosa pine (*Pinus ponderosa*), with lesser components of Douglas-fir (*Pseudotsuga menziesii*). Moderate and higher elevation stands between approximately 2800 – 3300 m were made up of subalpine forest, dominated by Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*). In the two decades prior to the fire, outbreaks of mountain pine beetles (*Dendroctonus ponderosae*), spruce beetles (*Dendroctonus rufipennis)*, and western balsam bark beetles have caused mortality of pine, Engelmann spruce, and subalpine fir, respectively, across about 40% of the burned area ([Rodman et al. 2021](#ref-rodman2021EffectsBarkBeetle)). Only about 3% of the area burned was dominated by aspen prior to fire, most (~80%) of which occurred between 2235 and 2874 m in elevation. About 37% of the aspen forest burned at high severity.

## Study sites.C

We surveyed 34 sites in the summer of 2022 (Fig. ). Sites were randomly located using geospatial data describing the extent and severity of the Cameron Peak Fire from the Monitoring Trends in Burn Severity (MTBS) Project ([2024b](#ref-mtbsproject2024MTBSDataAccess), [2024a](#ref-mtbsproject2024MTBSDataAccessa)) and the extent of pre-fire aspen cover from Cook et al. ([2024](#ref-cook2024MappingQuakingAspen)). We combined these data sources to identify areas that: (a) burned at high or moderate severity and (b) were located >50 m and <1000 m away from a pre-fire patch of aspen. Sites were further limited to accessible areas, defined as public land within 1000 m of a road or 500 m away from a trail and no more than 1 km up the trail. To reduce the potential effects of psuedoreplication, all sites were located >400 meters apart, or about twice as far as the minimum sampling distance used by Kreider et al. 2021 ([2021a](#ref-kreider2021LowdensityAspenSeedling)). Despite the extensive area burned, the accessible area was notably restricted by a limited road network and postfire treatments (e.g., aerial mulching) and damage to infrastructure. Nonthelesss our sample sites span broad gradients in elevation (2462 - 3255 m), mean annual daily temperature (-0.2 - 5.6°C), and annual total precipitation (431 - 1069 mm) (1991-2020 means) (Table S).

## Field data collection

At each site, we recorded the coordinates, elevation, slope, aspect, and pre-fire tree-species composition. We then established two perpendicular 2 x 50 m transects that intersected at the midpoint ([Andrus et al. 2021](#ref-andrus2021FutureDominanceQuaking)). We tallied the presence of all conifer seedlings, aspen seedlings, and aspen root suckers within each transect and recorded their location along the transect within 2 x 2m subplots. Aspen seedlings were differentiated from root suckers using methods outlined by Kreider et al. ([2020](#X44bfc23e7c771928f103170a5120f8dbe5a4826)), which have been demonstrated to be more than 95% accurate ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling)) and were shown to be effective in our study area ([Carter et al. 2024](#X518c5e11f1226b7f0686c896ea7fda75925f525)). Along each transect, we also collected information on percent herbaceous cover in three 1 x 1 m quadrats located every 20 m along each transect.

As a proxy for aspen seed availability, we also recorded the distance from plot center to live aspen trees. Where live aspen were not visible in the field, we estimated the distance to live aspen using a GIS. Specifically, we overlaid the MTBS map of fire severity ([MTBS Project 2024b](#ref-mtbsproject2024MTBSDataAccess)) on Cook et al.’s ([Cook et al. 2024](#ref-cook2024MappingQuakingAspen)) map of pre-fire aspen occurrence and assumed that any aspen stem that burned at moderate or high severity was likely killed in the fire. We then calculated the distance from plot center to the nearest live aspen pixel in R ([R Core Team 2024](#X4bd034c52731e178b4de125c2c778bdc533803c)) using the *terra* package ([Hijmans 2022](#ref-hijmans2022TerraSpatialData)).

To test the hypothesis that microsite conditions influence aspen seedling establishment, we additionally recorded information on proximity to coarse woody debris, substrate conditions, and microtopographic condition for each aspen seedling. We characterized the microtopography as flat, concave, or convex within 2.5 cm of seedling (hereafter ‘small microtopography’) and within 50 cm of seedling (hereafter ‘large microtopography’). We also recorded the presence or absence of small (diameter = 2.5-10 cm) and large (diameter >10 cm) coarse woody debris (CWD) within 10 cm of seedling. We also collected microsite conditions at 5 meter intervals along each transect to serve as a baseline of the relative abundance each condition across the site.

## Data analysis

### Patterns of postfire regeneration

We characterized patterns of post-fire conifer and aspen seedling regeneration using basic summary statistics, including the proportion of plots with seedlings and the mean, median, minimum, and maximum seedling densities in plots with seedlings. Summary statistics were calculated for each individual species and pooled for all conifer seedlings. To understand how pre-fire forest composition affects initial species composition, we also quantified the number of plots where each species was present pre- and postfire.

### Clustering of seedlings

We tested if seedlings were clustered using a Chi-squared goodness-of-fit-test where the null hypothesis was complete spatial randomness (i.e., that seedlings were equally distributed across the subplots). Because clustering is difficult to detect at low densities, we only tested for clustering of seedlings in plots with at least 1000 seedlings per ha (i.e., or on average 2 seedlings per subplot). To quantify statistical significance, we used a bootstrapping approach with 1000 replications. We performed this analysis separately for each site and tested for spatial randomness among aspen seedlings and then among all conifer seedlings combined. We also tested if conifer and aspen seedlings were more likely to co-occur using a Chi-squared goodness-of-fit-test where the null hypothesis was that conifer seedlings were equally likely to occur in subplots with and without aspen seedlings. As above, we performed this analysis separately for each site and used a bootstrapping approach with 1000 replication to determine the statistical significance. All tests were performed in R ([R Core Team 2024](#X4bd034c52731e178b4de125c2c778bdc533803c)).

### Site-level drivers of postfire aspen seedling regeneration

To understand the site factors that affect postfire aspen seedling establishment, we used a zero-inflated generalized linear modeling approach with a negative binomial distribution to model the total number of seedlings observed at a site. Zero-inflated models are useful for modeling an abundance of zeros ([Zuur et al. 2009](#ref-zuur2009Chapter11ZeroTruncated)), such as the high number of sites without postfire aspen seedlings that we observed here. As predictor variables we tested elevation and heat load, which we calculated using site-level information on slope, aspect, and latitude ([McCune and Keon 2002](#ref-mccune2002EquationsPotentialAnnual)). As an indicator of seed availability, we included distance to live aspen. Finally, as indicators of site quality and competition, we tested for effects of herbaceous cover and conifer seeding density, which was log transformed to the align with the log transformation applied to the aspen seedling densities. Prior to building the model, all variables were standardized to allow for easier comparison of coefficients. We trimmed the full model by dropping each predictor variable individually and using likelihood ratio tests to compare the full model with the reduced model ([Zuur et al. 2009](#ref-zuur2009Chapter11ZeroTruncated)). Variables were trimmed first from the zero part of the model and then the count component. As a measure of model performance, we calculated a pseudo R2, defined as the squared correlation between the model’s actual and predicted response. To visualize the effects of each significant predictor variables, we calculated the conditional predicted aspen seedling density as a function of the predictor, holding all other predictor variables to their mean value. Analyses were performed in R ([R Core Team 2024](#X4bd034c52731e178b4de125c2c778bdc533803c)) using the *glmmTMB* package ([Brooks et al. 2017](#ref-glmmTMB)) to fit models, the *lmtest* package ([Zeileis and Hothorn 2002](#ref-lmtest)) to perform likelihood ratio tests, the *performance* package ([Lüdecke et al. 2021](#ref-performance)) to calculate pseudo R2 and *marginaleffects* ([Arel-Bundock et al. 2024](#ref-marginaleffects)) and *ggplot2* packages ([Wickham 2016](#ref-ggplot2)) to construct and plot conditional predictions.

### Microsite drivers of aspen seedling establishment

To understand if aspen seedlings preferentially established in different microsite conditions, we followed the approach outlined by Kreider and Yocom ([2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)). Briefly, we first modeled the probability that aspen seedlings occur in each category of the predictor variable using a mixed effect multinomial logistic regression that included a random intercept term of site identity. We then modeled the probability that systematically surveyed points occurred in each category of the variable, also using a using a mixed effect multinomial logistic regression that included a random intercept term of site identity. We used these two models to quantify microsite preference following methods outlined by Krieder and Yocom ([2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)), where:

and *P(occurrence)* is the probability of seedling occurring in a given microsite condition and *P(availability*) is the probability of a given microsite condition is available. Here, preference values greater than zero indicate seedlings established in that microsite condition more often than expected based on the systematic sample. Multinomial regression was performed in R ([R Core Team 2024](#X4bd034c52731e178b4de125c2c778bdc533803c)) using the *mclogit* package ([Elff 2023](#ref-mclogit)).

# Results

## Postfire seedling establishment within the Cameron Peak Fire

Across sites that were dominated by conifers prior to fire and then severely burned, we found limited seedling regeneration two years following fire (Table ). Aspen seedlings were present at 62% of sites, with a mean density of 1155 seedlings per hectare. Conifer seedlings were present at a similar proportion of sites (59% of plots), but at a lower density on average when present (Table ).

**Table** **:** The number of sites with postfire seedlings and measures of density trees per ha) for sites with seedlings present.

| species | n | mean | median | min | max |
| --- | --- | --- | --- | --- | --- |
| subalpine fir | 1 | 150 | 150 | 150 | 150 |
| lodgepole pine | 16 | 384 | 250 | 50 | 1,350 |
| Engelmann spruce | 3 | 200 | 150 | 50 | 400 |
| ponderosa pine | 3 | 133 | 100 | 50 | 250 |
| quaking aspen | 21 | 1,155 | 300 | 50 | 7,850 |

Generally, the conifer species present in the postfire community reflected the pre-fire community. Lodgepole pine seedlings were present at 55.6% of sites were lodgepole was present prior to fire (Fig. ). In contrast, Engelmann spruce and subalpine fir seedlings were only present at 9.5% and 4.8% of sites where they occurred prior to fire. Ponderosa pine occurred at 42.9% of sites and Douglas fir seedlings were entirely absent postfire; however, the sample size for both species was small (nponderosa pine=7 sites; nDouglas fir=2 sites). postfire seedlings of lodgepole pine and Engelmann spruce were both observed at one site where they were absent in the pre-fire community (Fig. ).

![](data:application/octet-stream;base64,)

**Figure** **:** Patterns of postfire species presence by pre-fire species presence.

## Clustering

Within sites with aspen seedlings, on average 12.1% of the fifty 2 x 2 m subplots contained aspen seedlings (mean=6 subplots; range=1 - 30 subplots). About 56% of aspen seedlings occurred in subplots with another aspen seedling, with densities ranging from 0.25 to 6.5 seedlings per square meter (mean=0.95 seedlings per square meter). Across the 6 sites with at least 1,000 aspen seedlings per hectare, aspen seedlings were significantly (p<0.05) clustered at all sites (Table S).

Within sites with conifer seedlings, on average only 9.2% of the fifty 2 x 2 m subplots contained seedlings (mean=4.6 subplots; range=1 - 21 subplots). About 33% of conifer seedlings occurred in subplots with another conifer seedling and densities for subplots with conifer seedlings varied between 0.25 to 2 seedlings per square meter (mean=0.4 seedlings per square meter). We found multiple ponderosa pine seedlings in 17% of subplots with ponderosa pine seedlings, multiple lodgepole pine in 32% of subplots with lodgepole pine seedlings, multiple Engelmann spruce in 20% of subplots with Engelmann spruce, and multiple subalpine fir in 50% of subplots with subalpine fir. Across the 2 sites with at least 1,000 conifer seedlings per hectare, conifer seedlings were significantly (p<0.05) clustered at all sites (Table S). Sites with at least 1,000 conifer seedlings per hectare were dominated by lodgepole pine, however at one of these sites we observed lodgepole and Engelmann spruce co-occurring in three of the seven subplots where spruce was present.

We also found that conifer seedlings were significantly (p<0.05) more likely to occur in subplots with aspen than expected by complete spatial randomness (Table S). However, only two sites had sufficient densities of both aspen and conifer seedlings to test this hypothesis. At these sites we found most (67 - 89%) of the subplots with conifer seedlings also had aspen seedlings. Most of this co-occurrence included lodgepole pine; 21 out of 26 subplots had both lodgepole pine and aspen seedlings. However, Engelmann spruce co-occurred with aspen seedlings in three of the seven subplots where spruce seedlings were present.

## Site-level drivers of postfire aspen seedling density

Site-level variability in bioclimatic conditions influenced the density of aspen seedlings observed two-years following the Cameron Peak Fire. Our model of aspen seedling density explained a moderate amount of the variation (pseudo R2= 0.42) and correctly predicted 95% of the sites where aspen seedlings were observed (20 of the 21 sites with aspen seedlings), but only 31 % of the absences (4 of the 13sites without aspen seedlings). In the zero component of the model, elevation was the only significant predictor (Table ). Sites without aspen seedlings occurred at lower elevations (standardized β = -0.79; standard error = 0.4; Fig. A). In the count component of the model, elevation and the density of conifer seedlings were significantly related to aspen seedling density (p<0.05; Table ). Fig. B-C). Density increased with elevation (standardized β = 0.73; standard error = 0.25; Fig. B). Aspen seedling density also increased with the natural log of conifer seedling density (standardized β = 1.16; standard error = 0.23; Fig. C).

**Table** **:** Summary of zero-inflated negative binomial model selection results for absence (zero model) and abundance (count model) of post fire aspen stems. χ2 test statistics and p-values are from likelihood ratio tests comparison the full model to a reduced model where the focal predictor is dropped. AIC values are for the reduced model (i.e., higher AIC indicates removing this variable reduces model fit).

| Component | Predictor | AIC | χ2 | p |
| --- | --- | --- | --- | --- |
| Zero | Elevation | 379.6 | 4.96 | 0.026 |
| Zero | Heat load | 375.3 | 0.69 | 0.405 |
| Zero | Herbaceous cover | 375.5 | 0.94 | 0.332 |
| Zero | Conifer seedling density | 374.6 | 0.01 | 0.913 |
| Zero | Seed-source distance | 374.7 | 0.07 | 0.784 |
| Count | Elevation | 374.7 | 6.62 | 0.010 |
| Count | Heat load | 368.5 | 0.46 | 0.500 |
| Count | Herbaceous cover | 369.4 | 1.35 | 0.246 |
| Count | Conifer seedling density | 375.8 | 7.79 | 0.005 |
| Count | Seed-source distance | 368.2 | 0.19 | 0.663 |

![](data:application/octet-stream;base64,)

**Figure** **:** The predicted effects of elevation (A-B) and conifer seedling density (C) on patterns of aspen seedling density across the Cameron Peak Fire. In all panels, the shading illustrates the 95% confidence interval.

## Microsite drivers of postfire aspen seedling establishment

Within sites with aspen seedlings, we found microsite conditions were important predictors of aspen seedling establishment (Fig. ). Notably, aspen seedlings were about 21.5 times more likely to occur on bryophytes than any other substrate type. Seedlings also preferentially established in large and small concavities, with preferences of 13.9 and 6.5, respectively. Finally, aspen seedlings were more likely to establish in microsites near large CWD than microsites with small CWD, small and large CWD, or no CWD (preference for large CWD = 4.1)

![](data:application/octet-stream;base64,)

**Figure** **:** Aspen seedlings preferentially establish within certain microsite conditions. (A) Positive (black) and negative (blue) modeled aspen seedling microsite preferences. (B) Modeled probabilities of seedling occurrence in a given microsite condition (green) and the probability a given microsite condition is available (dark gray). In both A and B, error bars illustrate one standard error. Note that the y-axis in the substrate panel in part A is broken to better show the data.

# Discussion

Here we found widespread, but highly variable establishment of aspen seedlings across conifer forests that burned at high severity during the Cameron Peak Fire, despite the absence of aspen in the pre-fire stand. Aspen seedlings were more likely to occur at higher elevations, where they were often clustered within concave microsites, on bryophyte substrates, and near large coarse woody debris. Collectively, this adds to the support for the idea that cooler and wetter climate conditions promote aspen seedling establishment in recently burned ecosystems ([Quinn and Wu 2001](#ref-quinn2001QuakingAspenReproduce), [Buma and Wessman 2012](#ref-buma2012DifferentialSpeciesResponses), [Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling), [2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)). We also found aspen seedlings were more likely to occur at sites with greater postfire conifer seedling densities, consistent with previous research which has highlighted the potential for aspen to expand into recently burned conifer forests and shape the dynamics of burned ecosystems ([Gill et al. 2017b](#ref-gill2017PopulusTremuloidesSeedling), [Nigro et al. 2022](#ref-nigro2022WildfireCatalyzesUpward)).

## Distribution and abundance of aspen and conifer seedlings

Across the Cameron Peak Fire, we found aspen seedlings at 62% of sites surveyed. This occupancy rate is similar to the highest fire-level occupancy rates reported by Kreider and Yocom ([2021a](#ref-kreider2021LowdensityAspenSeedling)), who sampled 15 fires spanning a latitudinal gradient from northern Arizona to northern Wyoming. Across this sample, they reported aspen seedlings in only 13% of sites, with occupancy rate for individual fires events varying from 0 to 61% ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling)). In the case of the Cameron Peak Fire, the high occupancy rates may be attributed to our sampling design, which included larger sampling area than Kreider and Yocom ([2021a](#ref-kreider2021LowdensityAspenSeedling)) and excluded areas that burned at low or moderate severity, where seedling establishment may be less likely ([Turner et al. 2003](#ref-turner2003PostfireAspenSeedling), [Gill et al. 2017b](#ref-gill2017PopulusTremuloidesSeedling)). The high occupancy rates may also reflect characteristics of the burned landscape (e.g., elevation).

At sites with aspen seedlings, our median density of 300 seedlings ha-1 and maximum density of 7850 seedlings ha-1 is comparable to median (range: 100-900; median across fires: 100 seedlings per ha-1) and maximum (range: 100-9700; median across fires: 150 seedlings ha-1) densities for the 15 fires surveyed by Kreider and Yocom ([2021a](#ref-kreider2021LowdensityAspenSeedling)). However, postfire establishment of aspen from seed can be extremely abundant. For example, following both the 2017 Brian Head and the 1998 Yellowstone fires aspen seedling densities exceeded 150,000 seedlings ha-1 ([Kaye et al. 2003](#ref-kaye2003AspenStructureVariability), [Kreider and Yocom 2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)). Such dense postfire establishment of aspen can have important implications for postfire vegetation trajectories and rates. Yet, within the Cameron Peak fire only 38% (n= 8) of the sites with aspen seedlings surveyed here would meet local USFS requirements for restocking (741 seedlings per ha-1) ([USFS 1997](#ref-usfs19971997REVISIONLAND)), and thus may have more limited effects on the postfire ecology of this system.

Two years following fire, we found relatively low regeneration of subalpine fir, Engelmann spruce and ponderosa pine, and to a lesser extent lodgepole pine despite their presence in the pre-fire community. Compared to local USFS requirements for conifer restocking (370 seedlings per ha-1) ([USFS 1997](#ref-usfs19971997REVISIONLAND)), only six of our sites were stocked - five sites with lodgepole pine and one site where densities of both lodgepole pine and Engelmann spruce exceeded the stocking threshold. These low densities may have been due to the relatively the warm and dry weather in the two years post-fire. Such conditions have emerged as important constraints on postfire germination and initial survival of conifer seedlings in recent decades ([Stevens-Rumann and Morgan 2019](#X621eff21b20d26397ca583d3c14a7b6bd8eb0d1)). Additionally, viable seed may have been limited by pre-fire outbreaks of bark beetles ([Andrus et al. 2021](#ref-andrus2021FutureDominanceQuaking), [Rhoades et al. 2022](#ref-rhoades2022LimitedSeedViability), [2025](#X79a77eb1db2611553612a4aa27b7bf5406b22dc)), which caused widespread and severe mortality across the study area in the beginning of the 21st century ([Rodman et al. 2021](#ref-rodman2021EffectsBarkBeetle)). While infilling of burned areas with conifers seedlings can continue for decades following the fire, seedling densities two-years following fire are often good predictors of recruitment failure in conifer forests across the West ([Stevens-Rumann et al. 2018](#X59b7f55b19137c4c2a7bc304c3b5f88b9e0cece)). Thus in the absence of human intervention, conifers may remain limited across the system. Nonetheless, further research is needed to understand how the forests may develop ([Gill et al. 2017a](#X20c3490eb62bbd0e5e5969f3850fe7d9843e3d5)).

While aspen seedling regeneration was not incredibly abundant across the Cameron Peak burn scar, densities of aspen seedlings often exceeded densities of conifer seedlings. Here we found aspen seedlings were present at 60% of the sites with conifer seedlings and outnumbered conifer seedlings at 83% of these sites. Further aspen seedlings and conifer seedlings were more commonly found clustered together within a site than expected by complete spatial randomness, highlighting shared environmental conditions suitable for regeneration. The co-occurrence of aspen and conifer seedlings both across and within sites suggests that rates and trajectories of postfire recovery may be shaped by competition between aspen and conifers.

## Drivers of aspen seedling establishment

Across our study sites, aspen seedling occurrence and density increased with elevation. Higher elevations are characterized by cooler summers and greater winter snowpack accumulation and retention, which lower moisture stress. Low elevation sites, in addition to already being at the lower bounds of aspen habitat suitability ([Rehfeldt et al. 2009](#ref-rehfeldt2009AspenClimateSudden)), were also characterized by greater herbaceous cover, which may both out compete aspen seedlings and occupy the substrate preferred by seedlings ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling)). Nonetheless, we observed aspen seedlings as low as 2462 m, where annual precipitation averages 463 mm and mean daily temperature averages 5.6°C. In comparison, Kreider and Yocom ([2021a](#ref-kreider2021LowdensityAspenSeedling)) reported postfire aspen seedling establishment was unlikely where average annual precipitation values were less than 750 mm on average. Yet here we found 62 % of sites with aspen seedlings normally received less than 750 mm precipitation annually. At these sites, microsite conditions controlling soil moisture may be particularly important in controlling aspen regeneration.

Contrary to previous research which clearly shows that aspen seedling density is greater when live aspen trees are nearby ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling), [2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)), our modeling did not support this expectation. This may because across the Cameron Peak fire, the distance from live aspen generally increased with elevation, largely because aspen were more common at lower elevations prior to the fire and fire severity increased with elevation (Fig. **S**). As result, sites with higher climatic suitability were typically at a greater distance from live seed sources. Such mismatches between landscape-patterns of climatic suitability and seed availability may limit the potential for aspen to expand into new habitats.

At the microsite scale, seedling occurrence was related to the bryophyte cover, the presence of coarse woody debris, and concave microtopography, which may have connections to the accumulation and retention of seeds, moisture, and nutrients ([Fairweather et al. 2014](#Xa8aa2d86741ffb0df0f55debfdff8f830c78d37), [Kreider and Yocom 2021b](#Xb37a08252af320b6a65b3f38a24c3f3726e41e1)). Fallen logs or other large debris and local concavities may serve to collect both seeds and moisture necessary for successful establishment and survival in the first 1-3 growing seasons ([Fairweather et al. 2014](#Xa8aa2d86741ffb0df0f55debfdff8f830c78d37)). Importantly, microsite conditions favorable to aspen establishment were not evenly distributed across the fire. For example, we identified bryophytes as the dominant substrate at 1% of random samples, while we found 46% of aspen seedlings had established on bryophytes. Understanding how suitable microsites are distributed across landscapes will be central to predicting postfire regeneration of aspen and understanding how forest change following wildfire.

# Conclusions

Increased fire frequency, severity, and extent in western North America may exceed the capacity of some formerly conifer-dominant forests to maintain historical distributions ([Coop et al. 2020](#X11a688d84acc973314350de60810e0a8ca4666e)), while conditions following high severity burns may allow establishment of aspen seedlings both within and outside of pre-fire aspen populations ([Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling), [Nigro et al. 2022](#ref-nigro2022WildfireCatalyzesUpward)). We observed this phenomenon in high severity zones of the Cameron Peak Fire, particularly in areas where topographic heterogenity and post-fire legacies resulted in increased moisture availability. Concurrent with aspen seedling regeneration, we found low overall regeneration of conifer species relative to aspen. Across our system, limited conifer regeneration likely results from warm and dry weather and pre-fire outbreaks of bark beetles that caused extensive tree mortality - conditions that are expected to become more common with climate change ([Bentz et al. 2010](#ref-bentz2010ClimateChangeBark), [IPCC 2021](#ref-ipcc2021ClimateChange2021)). These trends coupled with changing fire regimes and our observations, along with a growing body of research on aspen seedling regeneration in the West (e.g., [Gill et al. 2017b](#ref-gill2017PopulusTremuloidesSeedling), [Andrus et al. 2021](#ref-andrus2021FutureDominanceQuaking), [Kreider and Yocom 2021a](#ref-kreider2021LowdensityAspenSeedling), [Nigro et al. 2022](#ref-nigro2022WildfireCatalyzesUpward)), suggest that aspen may increasing shape post-fire ecology across the West.

# **List of abbreviations**

CWD: coarse woody debris

# Declarations

## Ethics approval and consent to participate

Not applicable

## Consent for publication

Not applicable

## Availability of data and material

The datasets generated and/or analyzed during the current study are available in Dryad Research Data Repository, link (REF)

## Competing interests

The authors declare that they have no competing interests.

## Funding

This work was supported by the Joint Fire Science Program (JFSP, Project ID 21-2-02-29).

## Authors’ contributions

**SVC**: conceptualization (equal); methodology (equal); data collection (lead); methods (supporting); writing - original draft (equal), writing - review and editing (equal); **SJH**: conceptualization (equal); methodology (equal); data collection (supporting); methods (lead); writing - original draft (equal), writing - review and editing (equal). All authors read and approved the final manuscript.

## Acknowledgements

We would like to thank Chuck Rhoades and Monique Rocca for feedback on earlier draft of this work. We thank Preston Young and Samantha Maldonado for help in the field. Additionally, we would like to thank Colorado State University for supporting SVC.

# References

Abatzoglou, J. T., and A. P. Williams. 2016. [Impact of anthropogenic climate change on wildfire across western US forests](https://doi.org/10.1073/pnas.1607171113). Proceedings of the National Academy of Sciences 113:11770–11775.

Allen, C. D., M. Savage, D. A. Falk, K. F. Suckling, T. W. Swetnam, T. Schulke, P. B. Stacey, P. Morgan, M. Hoffman, and J. T. Klingel. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: A broad perspective. Ecological applications 12:14181433.

Andrus, R. A., S. J. Hart, N. Tutland, and T. T. Veblen. 2021. [Future dominance by quaking aspen expected following short-interval, compounded disturbance interaction](https://doi.org/10.1002/ecs2.3345). Ecosphere 12:e03345.

Arel-Bundock, V., N. Greifer, and A. Heiss. 2024. [How to interpret statistical models using marginaleffects for r and python](https://doi.org/10.18637/jss.v111.i09) 111.

Baker, F. S. 1925. [Aspen in the central rocky mountain region](https://doi.org/10.5962/bhl.title.108077). Washington, D.C.

Balch, J. K., B. A. Bradley, J. T. Abatzoglou, R. C. Nagy, E. J. Fusco, and A. L. Mahood. 2017. [Human-started wildfires expand the fire niche across the United States](https://doi.org/10.1073/pnas.1617394114). Proceedings of the National Academy of Sciences 114:2946–2951.

Bentz, B. J., J. Régnière, C. J. Fettig, E. M. Hansen, J. L. Hayes, J. A. Hicke, R. G. Kelsey, J. F. Negrón, and S. J. Seybold. 2010. Climate change and bark beetles of the western united states and canada: Direct and indirect effects. BioScience 60:602613.

Brooks, M. E., K. Kristensen, K. J. van, A. Magnusson, C. W. Berg, A. Nielsen, H. J. Skaug, M. Maechler, and B. M. Bolker. 2017. [glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling](https://doi.org/10.32614/RJ-2017-066) 9.

Buma, B., and C. A. Wessman. 2011. [Disturbance interactions can impact resilience mechanisms of forests](https://doi.org/10.1890/ES11-00038.1). Ecosphere 2:art64.

Buma, B., and C. A. Wessman. 2012. [Differential species responses to compounded perturbations and implications for landscape heterogeneity and resilience](https://doi.org/10.1016/j.foreco.2011.10.040). Forest Ecology and Management 266:25–33.

Carter, S., S. Hart, C. Rhoades, and M. Rocca. 2024. [Occurrence, distribution, and driving environmental factors of quaking aspen regeneration by seed in the cameron peak fire burn scar](https://api.mountainscholar.org/server/api/core/bitstreams/14cc73e5-8e74-42f2-b414-45e877ef91c0/content).

Cook, M., T. Chapman, S. Hart, A. Paudel, and J. Balch. 2024. [Mapping Quaking Aspen Using Seasonal Sentinel-1 and Sentinel-2 Composite Imagery across the Southern Rockies, USA](https://doi.org/10.3390/rs16091619). Remote Sensing 16:1619.

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

Davis, K. T., M. D. Robles, K. B. Kemp, P. E. Higuera, T. Chapman, K. L. Metlen, J. L. Peeler, K. C. Rodman, T. Woolley, R. N. Addington, B. J. Buma, C. A. Cansler, M. J. Case, B. M. Collins, J. D. Coop, S. Z. Dobrowski, N. S. Gill, C. Haffey, L. B. Harris, B. J. Harvey, R. D. Haugo, M. D. Hurteau, D. Kulakowski, C. E. Littlefield, L. A. McCauley, N. Povak, K. L. Shive, E. Smith, J. T. Stevens, C. S. Stevens-Rumann, A. H. Taylor, A. J. Tepley, D. J. N. Young, R. A. Andrus, M. A. Battaglia, J. K. Berkey, S. U. Busby, A. R. Carlson, M. E. Chambers, E. K. Dodson, D. C. Donato, W. M. Downing, P. J. Fornwalt, J. S. Halofsky, A. Hoffman, A. Holz, J. M. Iniguez, M. A. Krawchuk, M. R. Kreider, A. J. Larson, G. W. Meigs, J. P. Roccaforte, M. T. Rother, H. Safford, M. Schaedel, J. S. Sibold, M. P. Singleton, M. G. Turner, A. K. Urza, K. D. Clark-Wolf, L. Yocom, J. B. Fontaine, and J. L. Campbell. 2023. [Reduced fire severity offers near-term buffer to climate-driven declines in conifer resilience across the western united states](https://doi.org/10.1073/pnas.2208120120). Proceedings of the National Academy of Sciences 120:e2208120120.

Dawe, D. A., E. Whitman, M. Michaelian, A. J. Tepley, and M.-A. Parisien. 2025. [Sexual and vegetative recruitment of trembling aspen following a high-severity boreal wildfire](https://doi.org/10.1186/s42408-025-00359-2). Fire Ecology 21:32.

DeRose, R. J., R. S. Gardner, R. L. Lindroth, and K. E. Mock. 2022. [Polyploidy and growthdefense tradeoffs in natural populations of western quaking Aspen](https://doi.org/10.1007/s10886-022-01355-5). Journal of Chemical Ecology.

DeRose, R. J., K. E. Mock, and J. N. Long. 2015. [Cytotype differences in radial increment provide novel insight into aspen reproductive ecology and stand dynamics](https://research.fs.usda.gov/treesearch/47622). Canadian Journal of Forest Research. 45: 18.:1–8.

Elff, M. 2023. [Mclogit: Multinomial logit models, with or without random effects or overdispersion](http://mclogit.elff.eu).

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

Gill, N. S., D. Jarvis, T. T. Veblen, S. T. A. Pickett, and D. Kulakowski. 2017a. [Is initial post-disturbance regeneration indicative of longer-term trajectories?](https://doi.org/10.1002/ecs2.1924) Ecosphere 8:e01924.

Gill, N. S., F. Sangermano, B. Buma, and D. Kulakowski. 2017b. Populus tremuloides seedling establishment: An underexplored vector for forest type conversion after multiple disturbances. Forest ecology and management 404:156164.

Hansen, W. D., W. H. Romme, A. Ba, and M. G. Turner. 2016. [Shifting ecological filters mediate postfire expansion of seedling aspen (Populus tremuloides) in Yellowstone](https://doi.org/10.1016/j.foreco.2015.12.012). Forest Ecology and Management 362:218–230.

Harris, M. P., J. D. Coop, J. A. Balik, J. R. McFarland, S. A. Parks, and C. S. Stevens-Rumann. 2025. [Aspen impedes wildfire spread in southwestern United States landscapes](https://doi.org/10.1002/eap.70061). Ecological Applications 35:e70061.

Hijmans, R. J. 2022. [Terra: Spatial data analysis](https://CRAN.R-project.org/package=terra).

Iglesias, V., J. K. Balch, and W. R. Travis. 2022. [U.s. Fires became larger, more frequent, and more widespread in the 2000s](https://doi.org/10.1126/sciadv.abc0020). Science Advances 8:eabc0020.

IPCC. 2021. Climate change 2021: The physical science basis. Contribution of working group i to the sixth assessment report of the intergovernmental panel on climate change. (V. Masson-Delmotte, P. Zhai, A. Pirani, S. L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M. I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J. B. R. Matthews, T. K. Maycock, T. Waterfield, Ö. Yelekçi, R. Yu, and B. Zhou, Eds.). Cambridge University Press.

Johnstone, J. F., G. Celis, F. S. Chapin III, T. N. Hollingsworth, M. Jean, and M. C. Mack. 2020. [Factors shaping alternate successional trajectories in burned black spruce forests of Alaska](https://doi.org/10.1002/ecs2.3129). Ecosphere 11:e03129.

Johnstone, J. F., T. N. Hollingsworth, F. S. Chapin, and M. C. Mack. 2010. [Changes in fire regime break the legacy lock on successional trajectories in alaskan boreal forest](https://doi.org/10.1111/j.1365-2486.2009.02051.x). Global Change Biology 16:1281–1295.

Kaye, M. W., T. J. Stohlgren, and D. Binkley. 2003. [Aspen structure and variability in Rocky Mountain National Park, Colorado, USA](https://doi.org/10.1023/A:1026065826336). Landscape Ecology 18:591–603.

Kreider, M. R., K. E. Mock, and L. L. Yocom. 2020. [Methods for Distinguishing Aspen Seedlings from Suckers in the Field](https://doi.org/10.1093/jofore/fvaa030). Journal of Forestry:fvaa030.

Kreider, M. R., and L. L. Yocom. 2021a. [Low-density aspen seedling establishment is widespread following recent wildfires in the western United States](https://doi.org/10.1002/ecy.3436). Ecology 102:e03436.

Kreider, M. R., and L. L. Yocom. 2021b. [Aspen seedling establishment, survival, and growth following a high-severity wildfire](https://doi.org/10.1016/j.foreco.2021.119248). Forest Ecology and Management 493:119248.

Kulakowski, D., C. Matthews, D. Jarvis, and T. T. Veblen. 2013. [Compounded disturbances in sub-alpine forests in western Colorado favour future dominance by quaking aspen (Populus tremuloides)](https://doi.org/10.1111/j.1654-1103.2012.01437.x). Journal of Vegetation Science 24:168–176.

Landhäusser, S. M., B. D. Pinno, and K. E. Mock. 2019. [Tamm Review: Seedling-based ecology, management, and restoration in aspen (Populus tremuloides)](https://doi.org/10.1016/j.foreco.2018.09.024). Forest Ecology and Management 432:231–245.

Lloret, F., H. Estevan, J. Vayreda, and J. Terradas. 2005. [Fire regenerative syndromes of forest woody species across fire and climatic gradients](https://doi.org/10.1007/s00442-005-0206-1). Oecologia 146:461–468.

Long, J. N., and K. Mock. 2012. [Changing perspectives on regeneration ecology and genetic diversity in western quaking aspen: Implications for silviculture](https://doi.org/10.1139/x2012-143). Canadian Journal of Forest Research 42:2011–2021.

Lüdecke, D., M. S. Ben-Shachar, I. Patil, P. Waggoner, and D. Makowski. 2021. [Performance: An r package for assessment, comparison and testing of statistical models](https://doi.org/10.21105/joss.03139) 6:3139.

McCune, B., and D. Keon. 2002. [Equations for potential annual direct incident radiation and heat load](https://www.jstor.org/stable/3236745). Journal of Vegetation Science 13:603–606.

McIlroy, S. K., and D. J. Shinneman. 2020. [Post-fire aspen (Populus tremuloides) regeneration varies in response to winter precipitation across a regional climate gradient](https://doi.org/10.1016/j.foreco.2019.117681). Forest Ecology and Management 455:117681.

Mock, K. E., C. M. Callahan, M. N. Islam-Faridi, J. D. Shaw, H. S. Rai, S. C. Sanderson, C. A. Rowe, R. J. Ryel, M. D. Madritch, R. S. Gardner, and P. G. Wolf. 2012. [Widespread Triploidy in Western North American Aspen (Populus tremuloides)](https://doi.org/10.1371/journal.pone.0048406). PLOS ONE 7:e48406.

MTBS Project. 2024b. [MTBS data access: Burned severity mosaics dataset](https://mtbs.gov/direct-download).

MTBS Project. 2024a. [MTBS data access: Burned area boundaries dataset](https://mtbs.gov/direct-download).

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

Nigro, K. M., M. E. Rocca, M. A. Battaglia, J. D. Coop, and M. D. Redmond. 2022. [Wildfire catalyzes upward range expansion of trembling aspen in southern Rocky Mountain beetle-killed forests](https://doi.org/10.1111/jbi.14302). Journal of Biogeography 49:201–214.

Parks, S. A., and J. T. Abatzoglou. 2020. [Warmer and Drier Fire Seasons Contribute to Increases in Area Burned at High Severity in Western US Forests From 1985 to 2017](https://doi.org/10.1029/2020GL089858). Geophysical Research Letters 47:e2020GL089858.

PRISM Climate Group. 2021. [Monthly 30-year climate normals (1981-2010)](https://prism.oregonstate.edu/normals/).

Quinn, R. D., and L. Wu. 2001. Quaking Aspen Reproduce From Seed After Wildfire in the Mountains of Southeastern Arizona.

R Core Team. 2024. [R: A language and environment for statistical computing](http://www.R-project.org). R Foundation for Statistical Computing, Vienna, Austria.

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

Rhoades, C. C., T. S. Fegel, R. M. Hubbard, and M. E. Chambers. 2022. [Limited seed viability in long-dead serotinous lodgepole pine trees in the Southern Rockies, USA](https://doi.org/10.1016/j.foreco.2022.120565). Forest Ecology and Management 526:120565.

Rhoades, C. C., T. S. Fegel, C. Schneider, and A. G. Vorster. 2025. [Post-fire seedling recruitment across a range of stand age in bark-beetle impacted lodgepole pine forests: Informing reforestation needs](https://doi.org/10.1016/j.foreco.2025.123048). Forest Ecology and Management 596:123048.

Rodman, K. C., R. A. Andrus, C. L. Butkiewicz, T. B. Chapman, N. S. Gill, B. J. Harvey, D. Kulakowski, N. J. Tutland, T. T. Veblen, and S. J. Hart. 2021. [Effects of bark beetle outbreaks on forest landscape pattern in the Southern Rocky Mountains, U.S.A.](https://doi.org/10.3390/rs13061089) Remote Sensing 13:1089.

Stevens-Rumann, C. S., K. B. Kemp, P. E. Higuera, B. J. Harvey, M. T. Rother, D. C. Donato, P. Morgan, and T. T. Veblen. 2018. Evidence for declining forest resilience to wildfires under climate change. Ecology Letters 21:243252.

Stevens-Rumann, C. S., and P. Morgan. 2019. [Tree regeneration following wildfires in the western US: A review](https://doi.org/10.1186/s42408-019-0032-1). Fire Ecology 15:15.

Turner, M. G., K. H. Braziunas, W. D. Hansen, and B. J. Harvey. 2019. [Short-interval severe fire erodes the resilience of subalpine lodgepole pine forests](https://doi.org/10.1073/pnas.1902841116). Proceedings of the National Academy of Sciences 116:11319–11328.

Turner, M. G., W. H. Romme, R. A. Reed, and G. A. Tuskan. 2003. [Post-fire aspen seedling recruitment across the Yellowstone (USA) Landscape](https://doi.org/10.1023/A:1024462501689). Landscape Ecology 18:127–140.

USFS. 1997. [1997 REVISION of the LAND and RESOURCE MANAGEMENT PLAN: Arapaho and roosevelt national forests and pawnee national grassland](https://www.fs.usda.gov/Internet/FSE_DOCUMENTS/fseprd641737.pdf).

Walker, X. J., K. Okano, L. T. Berner, R. Massey, S. J. Goetz, J. F. Johnstone, and M. C. Mack. 2023. [Shifts in Ecological Legacies Support Hysteresis of Stand Type Conversions in Boreal Forests](https://doi.org/10.1007/s10021-023-00866-w). Ecosystems 26:1796–1805.

Wickham, H. 2016. [ggplot2: Elegant graphics for data analysis](https://ggplot2.tidyverse.org).

Zeileis, A., and T. Hothorn. 2002. [Diagnostic checking in regression relationships](https://CRAN.R-project.org/doc/Rnews/) 2.

Zuur, A. F., E. N. Ieno, N. Walker, A. A. Saveliev, and G. M. Smith. 2009. [Chapter 11: Zero-Truncated and Zero-Inflated Models for Count Data](https://doi.org/10.1007/978-0-387-87458-6_11). Pages 261–293. Springer New York, New York, NY.

# Supplementary Material

**Table S** **:** Characteristics of study sites sampled within the Cameron Peak Fire. Annual precipitation and temperature values are 1991-2010 normals from PRISM (2021).

| Site | Latitude (°N) | Longitude (°W) | Elevation (m) | Mean annual temperature (°C) | Mean annual precipitation (mm) | Distance to live aspen (m) | Aspen seedling density (no. per ha) | Conifer seedling density (no. per ha) |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
| 10 | 40.588 | -105.856 | 2,898 | 1.8 | 584 | 210 | 0 | 250 |
| 11 | 40.593 | -105.861 | 2,901 | 1.8 | 584 | 783 | 100 | 0 |
| 12 | 40.590 | -105.859 | 2,926 | 1.8 | 584 | 390 | 150 | 0 |
| 13 | 40.564 | -105.876 | 3,051 | 1.3 | 820 | 114 | 0 | 50 |
| 14 | 40.560 | -105.874 | 3,040 | 1.2 | 867 | 141 | 50 | 0 |
| 15 | 40.566 | -105.868 | 3,025 | 1.3 | 761 | 331 | 0 | 0 |
| 16 | 40.564 | -105.867 | 3,012 | 1.3 | 761 | 228 | 0 | 0 |
| 17 | 40.646 | -105.866 | 2,715 | 2.8 | 546 | 104 | 50 | 0 |
| 18 | 40.642 | -105.862 | 2,739 | 2.6 | 488 | 260 | 0 | 50 |
| 19 | 40.637 | -105.861 | 2,751 | 2.4 | 502 | 158 | 100 | 0 |
| 2 | 40.616 | -105.530 | 2,519 | 5.1 | 459 | 349 | 0 | 250 |
| 20 | 40.578 | -105.863 | 2,959 | 1.4 | 723 | 461 | 3,000 | 50 |
| 21 | 40.562 | -105.829 | 3,029 | 1.3 | 804 | 361 | 400 | 300 |
| 22 | 40.548 | -105.886 | 3,206 | 0.6 | 1,000 | 192 | 50 | 0 |
| 23 | 40.549 | -105.890 | 3,259 | -0.2 | 1,069 | 156 | 1,450 | 0 |
| 24 | 40.546 | -105.890 | 3,199 | -0.2 | 1,069 | 156 | 100 | 0 |
| 25 | 40.562 | -105.810 | 3,068 | 1.3 | 712 | 808 | 3,050 | 1,300 |
| 26 | 40.562 | -105.813 | 3,099 | 1.3 | 762 | 660 | 800 | 500 |
| 27 | 40.562 | -105.816 | 3,090 | 1.3 | 762 | 594 | 800 | 250 |
| 29 | 40.612 | -105.541 | 2,599 | 5.1 | 458 | 436 | 0 | 100 |
| 3 | 40.616 | -105.534 | 2,546 | 5.1 | 459 | 358 | 0 | 0 |
| 30 | 40.615 | -105.526 | 2,462 | 5.6 | 463 | 132 | 100 | 50 |
| 31 | 40.724 | -105.584 | 2,574 | 4.9 | 431 | 190 | 0 | 0 |
| 32 | 40.722 | -105.582 | 2,611 | 4.9 | 431 | 379 | 0 | 0 |
| 34 | 40.522 | -105.774 | 3,106 | 0.6 | 786 | 851 | 1,500 | 150 |
| 35 | 40.522 | -105.772 | 3,093 | 0.6 | 786 | 869 | 400 | 0 |
| 37 | 40.523 | -105.781 | 3,119 | 0.6 | 804 | 1,097 | 0 | 150 |
| 38 | 40.524 | -105.777 | 3,154 | 0.6 | 786 | 1,107 | 7,850 | 1,750 |
| 4 | 40.596 | -105.856 | 2,850 | 1.8 | 584 | 721 | 0 | 650 |
| 5 | 40.595 | -105.855 | 2,825 | 1.8 | 584 | 605 | 50 | 450 |
| 6 | 40.594 | -105.855 | 2,835 | 1.8 | 584 | 541 | 300 | 250 |
| 7 | 40.645 | -105.862 | 2,710 | 2.6 | 488 | 201 | 3,800 | 450 |
| 8 | 40.643 | -105.864 | 2,724 | 2.8 | 546 | 63 | 150 | 250 |
| 9 | 40.589 | -105.854 | 2,865 | 1.8 | 584 | 348 | 0 | 50 |

**Table S** **:** Summary of tests of complete spatial randomness for aspen seedlings.

| Site | density (no. per ha) | χ2 | p |
| --- | --- | --- | --- |
| 7 | 3,800 | 601.6 | 0.001 |
| 20 | 3,000 | 586.7 | 0.001 |
| 23 | 1,450 | 441.7 | 0.001 |
| 25 | 3,050 | 502.1 | 0.001 |
| 34 | 1,500 | 86.7 | 0.003 |
| 38 | 7,850 | 237.6 | 0.001 |

**Table S** **:** Summary of tests of complete spatial randomness for conifer seedlings.

| Site | density (no. per ha) | χ2 | p |
| --- | --- | --- | --- |
| 25 | 1,300 | 231.7 | 0.001 |
| 38 | 1,750 | 83.6 | 0.002 |

**Table S** **:** Summary of tests of independence of aspen and conifer seedlings.

| Site | conifer density (no. per ha) | aspen density (no. per ha) | χ2 | p |
| --- | --- | --- | --- | --- |
| 25 | 1,300 | 3,050 | 22.15 | 0.001 |
| 38 | 1,750 | 7,850 | 4.83 | 0.039 |

![](data:application/octet-stream;base64,)

**Figure S****:** Residual diagnostic plots from the zero-inflated negative binomial model of aspen seedling density. Plots were generated using the DHARMa package (Hartig 2024).

![](data:application/octet-stream;base64,)

**Figure S****:** Plots of the simulated residuals from the zero-inflated negative binomial model of aspen seedling density as a function of elevation (left) and conifer seedling density (right). Plots were generated using the DHARMa package (Hartig 2024).

![](data:application/octet-stream;base64,)

**Figure S****:** The relationship between elevation and aspen’s distribution across the area burned in the Cameron Peak Fire. A) Boxplots comparing the elevation range of areas with and without aspen prior to the fire. B) Boxplots comparing the elevation by fire severity. C) The density of observations for areas that burned in the Cameron Peak fire that were greater than 50 m from live aspen and less than 1000 km from live aspen prior to the fire by elevation and distance to seed source.